Forest Governance and Economic Values of Forest Ecosystem Services in the Northwest Region of Vietnam

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STATEMENT OF ORIGINALITY

This is to certify that, to the best of my knowledge, the content of this thesis is my own work. This thesis has not been submitted for any degree or other purpose.

I certify that the intellectual content of this thesis is the product of my own work and that all the assistance received in preparing this thesis and sources have been acknowledged.

Nguyen Minh Duc

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ABSTRACT

Forest ecosystems provide valuable services to humanity but have been continually undervalued and degraded. This problem is a result of inadequate institutional arrangements for forest governance. This study aims to assess the effects of alternative forest governance regimes on the provision and the associated economic values of forest ecosystem services, particularly in the Northwest region of Vietnam. It aims to answer three specific questions:

1. What are the possible, feasible and credible forest governance scenarios that could be applied in this region in the future?
2. What are the likely changes in forest cover as a result of implementing alternative forest governance scenarios?
3. What are the likely changes in the provision and valuation of forest ecosystem services under the alternative forest governance scenarios?

In answering these questions, this study argues for the framework that integrates the ecosystem services' approach with the assessment of forest governance regimes. The framework incorporates mapping land use and land cover (LULC) changes stemming from possible changes in forest governance regimes, quantifying the resulting changes in the provision of forest ecosystem services (FESs) and estimating the associated economic values. In the context of the study site in the north-western uplands of Vietnam, I tested three alternative forest governance scenarios: business-as-usual, with a dominant government role; community-based governance; and private forestry governance. For each forest governance scenario, the changes in forest LULC were mapped based on land suitability analysis and transition likelihood for the period 2010–2020. The resulting maps were used with other variables related to climate conditions and biophysical attributes of forest ecosystems (e.g., precipitation, reference evapotranspiration, plant evapotranspiration coefficient, soil characteristics, digital elevation model, carbon pools, etc.) as inputs into the InVEST model (Integrated Valuation of Environmental Services and Trade-offs). The InVEST model was used to estimate the quantity of three specific FESs: carbon storage/sequestration, provision of water for hydropower production and reduction of sediment load to reservoirs. I applied various valuation methods to value these services: the social cost of carbon was used to estimate the economic value of carbon storage/sequestration; the replacement cost of removing sediment deposited into reservoirs was applied for valuing the reduction of

sediment yield; and the residual value of water supply for hydropower generation was used for valuing water yield.

The study shows that forest governance plays an important role in the provision of FESs and, accordingly, in determining the associated values of FESs. It indicates that forest LULC is likely to improve under the business-as-usual and community-based scenarios, while there is little change under the private forestry scenario. Consequently, the aggregated annual value of the three services of interest is likely to increase under all scenarios, but it is particularly significantly under the business-as-usual and the community-based scenarios. The findings also indicate that the value of the services is highly dependent on both the supply of these services and the benefits that they contribute to society's welfare. Based on the findings, I argue that community-based forest governance is an alternative forest governance regime that can effectively replace the current state-dominant regime.

This study makes contributions to knowledge in three specific ways. Firstly, it integrates the frameworks of forest ecosystem service valuation with the assessment of the effects and effectiveness of forest governance. Secondly, it is the first study that has applied the InVEST model (the combination of ecological production functions and economic valuation approaches) to determine the outcomes of forest governance (in terms of changes in values of FESs) in the context of Vietnam. This is particularly significant as Vietnam is a developing country with limited data availability. In addition, it is one of the few applications of InVEST in the tropics. Thirdly, the findings regarding the provision and the associated values of the FESs could be very useful in terms of policy design and evaluation. This is very important in Vietnam, as it has been a country on the forefront of the application of incentive-based programs for forest conservation. The existing Payments for Forest Ecosystem Services system in Vietnam can be adjusted and calibrated based on the findings provided in this study to achieve better forest conservation outcomes.

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ABBREVIATIONS

5MHRP	Five Million Hectares of Reforestation Programs
ABSCM	Attribute-based Stated-Choice Modelling
CBFM	Community-Based Forest Management
CPCs	Communal People's Committees
CPRs	Common Pool Resources
CVM	Contingent Valuation Method
ESV	Ecosystem Services Valuation
EVN	Vietnam Electricity
FG	Forest Governance
FIPI	Forest Inventory and Planning Institute
FLA	Forest Land Allocation
FMBs	Forest Management Boards
GDEM	Global Digital Elevation Model
GIS	Global Information System
HWSD	Harmonised World Soil Database
InVEST	Integrated Valuation of Ecosystem Services and Trade-offs
IPCC	Intergovernmental Panel on Climate Change
LAI	Leaf Area Index
LULC	Land Use and Land Cover
MA	Millennium Ecosystem Assessment
MARD	Ministry of Agriculture and Rural Development
MONRE	Ministry of Environment and Natural Resources
MW	Mega-Watt
NDVI	Normalised Difference Vegetation Index
NTFPs	Non-Timber Forest Products
O&M	Operation and Maintainance
PFES	Payment for Forest Ecosystem Services
PPP	Purchasing Power Parity
RCFEE	Research Centre for Forest Ecology and Environment
REDD	Reducing Emissions from Deforestation and Forest Degradation
RUSLE	Revised Universal Soil Loss Equation

SCC	Social Cost of Carbon
SDR	Sedimentation Delivery Ratio
SEI	Structured Expert Interviews
SFEs	State Forest Enterprises
SFMI	Research Institute for Sustainable Forest Management and Forest Certification
SOCs	State-Owned Companies
TEV	Total Economic Value
UEI	Unstructured Expert Interviews
USLE	Universal Soil Loss Equation
VAFS	Vietnamese Academy of Forest Science
VNFOREST	Vietnam Administration of Forestry
VNUF	Vietnam National University of Forestry

Chapter 1
INTRODUCTION

1.1 Motivation for the Study

1.1.1 Sustainable Development Problems

Despite the concept of sustainable development gaining prominence in the 1980s (Brundtland *et al.* (1987), human societies continue to struggle with determining suitable ways to account for the environment and natural resources. This section articulates the questions regarding the demand for ecosystem services by human societies; the ongoing loss of ecosystems and the corresponding ecosystem services; and the pertinent reasons for such a loss that can be pinpointed to the development processes of human societies.

Humans need ecosystems and ecosystem services for their survival and wellbeing. Ecosystems provide all the basic materials for human life and wellbeing, such as food, water and fresh air. In addition, human living conditions are closely linked to the condition of ecosystems because our living environment is crucially regulated by the ecosystems. Furthermore, nature enriches our spiritual lives by providing areas for recreation and enjoyment. As a result of rapid progress and development, the demand for ecosystem services by human societies has been growing very quickly. The findings of the Millennium Ecosystem Assessment (2005) pinpointed that the human use of ecosystem services is expanding, corresponding to the growth of human population and the expansion of consumption. As the world population and consumption continues to grow at a fast pace, the demand for ecosystem services will expand more rapidly.

In contrast to the increasing demand for ecosystem services, most of the global ecosystems and the services that they provide have declined and/or degraded rapidly over the past few decades. The review by Balmford *et al.* (2002), covering a large number of studies relating to ecosystems and ecosystem services, showed that loss and degradation of the remaining natural habitats have continued largely unabated. MA (2005) reported that about 60% of the most important ecosystem services evaluated were degraded or used in unsustainable ways. Postel and Thompson (2005) also warn that global forested watersheds have been negatively modified. More than half of the world's overall land area has been converted to agriculture or urban-industrial use. For example, in Europe, at least 90% of 13 major watersheds have lost their primary green cover; in China, some 85% and 78% of forest cover have been lost in the Yangtze and the Yellow River basins, respectively; in the Indus basin, over 90% of forest cover has been cleared and converted for other uses.

The contradiction between the rapid increase in the demand for ecosystem services and the degradation of ecosystems has seriously challenged the notion of sustainable development. The total economic value of primary ecosystems often exceeds that of converting the ecosystem for any other purpose, such as farming, logging or other intensive uses (MA 2005; TEEB Synthesis 2010). This view suggests that human societies have ignored the trade-off between economic growth and the stock of natural capital. We have chosen to prioritise short-term economic development at the expense of the health of our ecosystems. These have far greater value, but those values are often unpriced and realised only over the longer term. As a consequence, economic growth has improved at the expense of ecosystem degradation. The degradation of ecosystems has been negatively influencing the environment, the climate, disturbing the water balance and reducing biodiversity. Climate change, environmental pollution, natural hazards and human health problems are only some examples of the negative effects faced by human societies as a result of ecosystem degradation (Pearce 2001).

In addition, ecosystem degradation widens social inequality, as it particularly worsens the livelihoods of the poor and indigenous people, because they often live in rural and regional areas and their livelihoods depend on natural resources. Furthermore, scientists predict that if we continue using the ecosystem in an unsustainable way, the future development of human societies may become untenable (MA 2005).

The decline and degradation of natural resources in general and the decreased provision of ecosystem services in particular, is a result of human influence over ecosystems. The study of Costanza and Farber (2002) argued that human influence on ecosystems has dramatically increased over the past century. Balmford *et al.* (2002) claimed that human impacts are the crucial reason for the loss and degradation of ecosystems. The findings of MA (2005) showed that over the past half century, in order to meet the expanding demand, we have changed ecosystems at an unprecedented scale. Human societies have overexploited natural resources in order to satisfy their growing demand for food, fresh water, timber and other raw materials. This has not only caused a rapid change in the structure and functioning of the world's ecosystems, but also a substantial and irreversible loss of biodiversity. Furthermore, Pattanayak *et al.* (2010) stressed that in spite of significant efforts to invest in restoring natural ecosystems, global ecosystems are continuing to degrade at a rapid pace. Similarly, Christie and Rayment (2012) showed that global biodiversity continues to decrease in spite of significant conservation efforts, that results from a failure to fully acknowledge the array of ecosystem services delivered by biodiversity.

A large number of publications share the point of view that human institutions, or the lack of appropriate institutions, particularly regarding land use and land management, are to blame for the degradation of ecosystem services. Hardin (1968) argued that it is the human governance mechanisms relating to natural resources that cause the problems of overuse and degradation of natural resources. He pointed out that human societies fail to establish adequate governance mechanisms to sustainably use natural resources. Most decisions relating to resource management are driven by the market mechanism. Therefore, only the ecosystem services that are tradable on markets influence human choices, while the others that are not fully captured in the markets, are often overlooked in policy decisions. Consequently, the non-marketed benefits, which often characterise most ecosystem services, are typically not considered in public decisions (Costanza and Daly 1992; De Groot *et al.* 2010; Ring *et al.* 2010). This has resulted in their gross overuse.

1.1.2 Theoretical Views on the Problem of the Unsustainable Use of Natural Resources

a) Natural Resource Management Perspectives

Over the last four decades, in response to the problems of the unsustainable development in general and the problems of inefficient use and management of natural resources, in particular, the academic community has made attempts to postulate scientific foundations for sustainable natural resource management. The problems of unsustainable use of natural resources are considered to originate from inefficient regimes of natural resources governance. There is much debate on "the tragedy of the commons", as proposed by Hardin (1968). The tragedy of the commons occurs when there are no restrictions to prevent individuals from continuing to exploit the common natural resources, which they do to maximise their self-interest. This often results in depletion or degradation of the common resources. Consequently, the whole societies are negatively affected. According to Hardin, to overcome the tragedy, institutional regimes are needed to create barriers and prevent potential users from overusing the common resources. He argued that there are only two state-established institutional regimes - either centralised governments or private property - that could create restrictions to the free access problem and sustain commons over the long-term period. The notion proposed by Hardin was supported by other theorists. For example, some suggested that government ownership and control is the most efficient form of natural resource governance (Ophuls 1973; Terborgh 1999; Lovejoy 2006). Others suggested that private property is the most effective way to prevent the tragedy of the commons (Smith 1981; Simmons *et al.* 1996).

Other scholars, led by the work of Ostrom (1990), argued for the advantages of community resource management over centralised government and privatisation of natural resources in terms of sustainably managing the commons. The notion of community self-governance of natural resources has been put forward by many authors (Feeny *et al.* 1990; Ostrom 1999; Agrawal and Ostrom 2001; Brown *et al.* 2003; Dietz *et al.* 2003; Ostrom 2005, 2008). They contended that "the tragedy of the commons" had rarely been resolved by the government's efforts of claiming common resources. In addition, they argued that there were many pieces of evidence all over the world to indicate that common resources have been effectively managed by community self-governance systems. They found that individuals do not solely chase their own self-interest, but consider the common interest of the community as well, when making decisions relating to common resources. They also found that community users have the capacity to regulate individual behaviour to fit in with the community benefits by constructing and enforcing their own rules and norms. Thus, a self-organised community governance regime can solve the problem of overuse or destruction of common natural resources.

b) Economic Perspectives

Besides the above points of view regarding the arrangements of natural resources management, economists believe that human societies make effective decisions on the use of natural resources if the values of the natural resources are fully captured and considered in the decision-making processes. Significant contributions have been made from economists in this regard, especially in environmental and natural resource economics and ecological economics. Economists have made substantial efforts to construct methods of valuing environmental and natural resources. Integrating the economic notions with the knowledge of ecology, the field of ecological economics has modelled ecosystem functions, flows of ecosystem services and the valuation of ecosystem services.

In addressing the problem of ecosystem degradation as a result of the failure of market mechanisms (i.e. the environment and natural resources not being reflected or valued within existing markets), environmental and natural resource economists have made efforts to work out the true value of ecosystems by conceptualising the framework of the total economic value (Costanza *et al.* 1997; Costanza 2000; Loomis *et al.* 2000; Farber *et al.* 2002; Turner *et al.* 2010; Freeman *et al.* 2014). According to the total economic value framework, ecosystem's values are divided into two domains, which are 'use values' or 'active use values' and 'non-use values' or 'passive-use values' (Grafton *et al.* 2004; Smith *et al.* 2006). Use values, either direct-use values or indirect use values, are those that can be observed through changes in market behaviour when there is a change in an ecosystem, such as environmental quality or the stock of natural resources. Non-use values are those that are not related to actual use of goods or services. Non-use values cannot be observed via pure market behaviour, and are subcategorised into 'existence values', 'bequest values', and 'option values'. Existence values is the non-use value that people place on ecological goods/services for their existence, even if they will never use these goods/services. Bequest values are defined as an individual may have value for the ecological entity when he wants to reserve it for future generations while option values are the values that an individual place on ecological goods/services if he/she may use them in the future. The use values of ecosystems can be measured by market demand functions. However, the non-use values of ecosystems are much more difficult to measure. In order to capture the values of environmental and natural resources, a number of valuation methods have been developed. They can be categorised into two groups: revealed-preference methods and stated-preference methods. The reveal methods are applied to value ecosystem services that can be observed in the market (valuing use value). Meanwhile, the stated-preference values can be used to measure both use and non-use values of ecosystems (Grafton *et al.* 2004).

c) Ecosystem Service Framework

Ecological economics, as a rapidly emerging research field, brings innovation to valuing ecosystems. The notion of valuing services derived from ecosystems clarifies the benefits of ecosystems to humans. Influenced by the work of Costanza and Daly (1992), Costanza *et al.* (1997), and Costanza and Daly (1992); Daily (1997), who conceptualised ecosystem services and initiated valuing ecosystem services, the knowledge of ecosystems and ecosystem services has been developed at various levels from small-scale catchments to a global scale. In the early 2000s, studies were conducted relating to various facets of ecosystem services. This work culminated in a Special Issue of *Ecological Economics* 41 (2002), where the work at the forefront of ecological economics was published (Boumans *et al.* 2002; Costanza and Farber 2002; De Groot *et al.* 2002; Farber *et al.* 2002; Gustavson *et al.* 2002; Howarth and Farber 2002; Limburg *et al.* 2002; Patterson 2002; Sutton and Costanza 2002); Villa *et al.* (2002). These publications clarified a framework for ecosystem valuation. In addition, and perhaps stemming from this work, the Millennium Ecosystem Assessment 2005 (MA 2005) was the first global assessment of ecosystem services to conceptualise the ecosystem service framework for valuing ecosystems and ecosystem services. This innovative approach analysed the influences of environmental changes on ecosystems and human welfare. Ecosystems were evaluated through their provision of services to human societies and how these services benefit humanity. This framework also described the ways that humans utilise ecosystem services, the human impacts on the ecosystems and their provision of services.

This framework has been widely appreciated and further developed by the scientific and policy-making communities. Researchers have proposed more comprehensive methods for quantifying linkages of ecosystem conditions and the production of ecosystem services by incorporating economic valuation at finer spatial scales and timeframes applicable for ecological management decision making (Farber *et al.* 2006; Fisher and Kerry Turner 2008; Fisher *et al.* 2008; Naidoo *et al.* 2008; Tallis *et al.* 2008; Turner and Daily 2008; Carpenter *et al.* 2009; Bateman *et al.* 2013; UK NEA 2014). In 2012, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services was established to enhance the communication between the scientific community and policy makers and to build a capacity for, and strengthen the use of ecosystem service science and assessments in policy making.

In parallel to the development of conceptual theories on ecosystem services, researchers have developed various models for quantification and valuation of ecosystem services. These models also take advantages of GIS technology to map the multiple services. In addition, these models were designed to be applied in a broad range of biophysical, social and governance systems and to different decision contexts. Among them is the Integrated Valuation of Ecosystem Services and Trade-offs (InVEST), developed by the Natural Capital Project, which is an open-source, spatially-explicit ecosystem service modelling tool that has become popular among researchers (Nelson and Daily 2010; Bagstad *et al.* 2013).[1] This model was developed based on the principles of ecological production functions and economic valuation methods. It applies the framework that links ecological production functions to the benefits flowing to the human society. The framework includes supply, service and value. Supply denotes what the ecosystem can potentially provide; service involves the human usage of the ecological supply; and value reflects social preference. It is designed to aid decision makers in natural resource management. Particularly, it explores how changes in the conditions of ecosystems are likely to result in changes in ecosystem services that benefit humans (Sharp *et al.* 2015). InVEST consists of a set of models that quantify, map and estimate economic values of ecosystem services across landscapes by using land use and land cover (LULC) patterns and other biophysical attributes of landscapes. For example, InVEST can model carbon storage and carbon sequestration, water provision for hydropower generation, soil conservation, sediment reduction, pollination, nature-based recreation and tourism, coastal protection values, etc.,. In addition, this model is flexible in terms of data availability because it can run at different complexity levels that make it more flexible in situations of limited data, especially in developing countries (Nelson *et al.* 2009).

1.1.3 Problems with Applied Research on Ecosystem Services

Although the interest in ecosystem service quantification and valuation has been expanded, there is sparse research that transfers these scientific advances to policy practices. Liu *et al.* (2010) reviewed the practice of ecosystem services valuation studies (ESV) and concluded that while most of the ecosystem service valuation studies were driven by the hope that the outputs from those studies would contribute to ecosystem management, the actual use of these outputs was far below the expectation. An extensive review conducted by Laurans *et al.* (2013) reported that most peer-reviewed valuation papers give very little attention to how and why ecosystem service valuation studies are not included in actual decision making. Rogers *et al.* (2015) were concerned about the actual practices of non-market valuation research in an Australian context. They found that decision makers rarely used the results from non-market valuation studies in their actual decision making.

[1] The Natural Capital Project (NatCap-www.naturalcapitalproject.org) was formed in 2006.

There are obstacles that prevent decision makers from taking ecosystem service valuation into account in environmental management. The results of ecosystem valuation studies are rarely used for cost-benefit analyses in natural resource management decision-making processes. This is because combining the complicated nature of ecosystem functions and services with the difficulties in obtaining consistent and credible valuation for multiple ecosystem services (Carpenter *et al.* 2009; De Groot *et al.* 2010; Ring *et al.* 2010), amplifies the limited knowledge of decision makers on non-market valuation concepts and techniques Rogers *et al.* (2015).

At the same time, there have been calls for the need to evaluate the modes of natural resource governance. Feeny *et al.* (1990) argued that evidence accumulated over recent decades indicated that private, state and communal property right regimes are all potentially viable resource management options. Strongly influenced by the thought of Hardin, many scholars and public decision makers have recommended state-centric control of all common resources. Consequently, in many countries, national legislation has been implemented and state-owned agencies have taken the administrative responsibilities of managing natural resources. Unfortunately, evidence shows that government-based regulation has hardly been as successful as expected. In contrast, a significant body of evidence shows that many common pool resources are successfully governed by community-based regimes (Ostrom 2008). However, those who argue for the community self-governance regime also realise that this is not a panacea on its own to solve the problems of common pool resource management (Ostrom *et al.* 2007). Brown *et al.* (2003), who examined community forestry in Cameroon, concluded that community forestry cannot, on its own, instantly improve forest governance. Pagdee *et al.* (2006) conducted a meta-study that tested the influence of community forestry variables on the conditions of forest cover and concluded that there are certain attributes of community forest management that make this governance regime effective. They consist of "tenure security, clear ownership congruence between biophysical and socio-economic boundaries of the resources, effective enforcement of rules and regulations, monitoring, sanctioning, strong leadership with a capable local organisation, the expectation of benefits, common interests among community members and a local authority." The study of Andersson *et al.* (2013) also supports the conclusions reached by Pagdee *et al.* (2006). Using data from a sample of 200 forest user groups in Bolivia, they argued that the outcomes of community forest governance (e.g., forest cover, forest biodiversity and forest vegetation density) vary with the capacity of the local community, in terms of self-organised rule making, monitoring and sanctioning. In summary, as argued by Dietz *et al.* (2003), there is no straightforward verdict on a governance system for which there is unequivocal evidence in all settings. Therefore, in order to determine which particular governance regime is the most effective in a specific setting of a particular natural resource, it is necessary to apply a diagnostic approach that enables us to articulate how attributes of a resource system, the services provided by that system, the users of that system, and the governance system are jointly influenced and how they are indirectly affected by interactions and consequent outcomes obtained at a particular time and space (Ostrom 2007; Ostrom *et al.* 2007).

1.1.4. Problems of Forest Governance in Vietnam

Forests are some of the most important ecosystems to provide multiple essential services that have substantial benefits for human wellbeing. The work by Costanza *et al.* (1997) estimated the annual value of the world's forests to be 4.706 billion 1994 USD, approximately 969 USD/ha/year. In a recent work of (Costanza *et al.* 2014), the updated annual value derived from the world forest was estimated at 16.2 billion 2007 USD. Brandon (2014) stated that tropical forests provide a wide range of important services to humans: climate regulation, provision of fresh water, food, biodiversity, human health and so on.

In Vietnam, after experiencing a rapid loss of forests and serious forest degradation in the 1980s, and the emerging acknowledgement of the benefits of the forest to society, the government has shifted its focus from forest production to forest protection and development since the early 1990s. Following the Land Law 1993, many legal documents were issued to establish a legal framework for property rights, especially for the process of long term and stable land allocation to private sectors. The establishment of the legal framework is in the hope of achieving more sustainable use of natural resources in general and forest resources, in particular (FSIV 2009). In parallel with the legislation of a legal framework, the government invested in many national programs and projects, such as the Five Million Hectare Reforestation Program (ended in 2011), and the program of forest protection and sustainable development as a continuing program aiming to increase forest coverage, as well as forest quality. Furthermore, the government has been continuously searching for, and experimenting with various alternative approaches to forest management. Although the state-owned agencies are still the dominant players in forest management, communities and individuals have played increasingly important roles in sustaining forest resources (Sikor 2001; Meyfroidt and Lambin 2008a; Meyfroidt *et al.* 2009; Ngai 2009; Phuc and Nghi 2014). Community forest management has existed for a long time in Vietnam. However, this forest governance regime was weakened by the process of centralisation over forest resources before the structural reform of Doi Moi.[2] After the Doi Moi, and during the decentralisation process, the community forest governance regime was reinstated by the government. In the decentralisation process, privatisation of forest resources was also implemented by the allocation of forest land and forests to individuals/households. In addition, and significantly, market-oriented approaches to forest management (e.g. payments for forest ecosystem services), have been introduced by the government since 2008 and are considered to be a potentially very successful regime for sustainable forest management (Pham *et al.* 2013).

In spite of these positive developments, the current forest governance regime in Vietnam has been criticised for its inefficiency, particularly for the still dominant state-controlled regime, the continuous loss of old growth forest and forest degradation, and the neglect of forest rights and livelihoods of the local population (Sikor and Tan 2011; Phuc *et al.* 2013). According to the recent studies of McElwee (2012), Hung *et al.* (2011) and FSIV (2009), during the period from 1999 to 2005, rich natural forested areas nationally reduced by 10–13%. It is therefore required to search for alternative forest governance regimes that can help to manage forests more effectively.

1.1.5 Research Gaps

Based on the above survey of the literature, I argue that there is a lack of knowledge about the link between natural resource governance and ecosystem service valuation. The debates on forest governance mostly focus on the governance regimes that facilitate sustainable forest management. The debates over the efficiency of forest governance are often concerned with

[2] After the Six Party Congress in December 1986, the broad thrust of Doi Moi was officially implemented. Doi Moi can be literally translated as 'renovation', but means 'structural reform'. The structural reform included the improvement of private property rights, increases in macroeconomic stability, and a shift from a centrally-planned economy to a market economy. The most important policy changes were implemented in the agricultural sector. Under the process of Doi Moi, farm households were allocated agricultural lands that were previously managed by state cooperatives and allowed to sell their products in the open markets. Furthermore, after the Land Law was issued in 1993, farm households were granted long and stable agricultural land tenure that includes the right to transfer, exchange, lease, bequeath as inheritance, and use land as collateral for bank loans.

the sustainable use of forests (Ostrom 2005; Pagdee *et al.* 2006; Berkes 2007) as reflected through forest conditions (e.g. forest cover, forest biodiversity or forest vegetable density (Agrawal and Chhatre 2006; Pagdee *et al.* 2006)), social performance (e.g. social equity, livelihood of local dwellers (Ostrom 2007)) for the outcomes of forest governance arrangements. This means that the economic valuation of forest ecosystem services has not been adequately incorporated in the debates over forest governance. Incorporating economic valuation of forest ecosystem services in the assessment of resource governance will provide a more applicable tool for decision makers in natural resource management and, as a result, natural resources can be more sustainably used (Turner *et al.* 2010). At the same time, economic efficiency needs to be introduced into the existing academic debate on the institutional arrangements of natural resources.

1.2 The Statement of the Research Problem

I argue that it is needed to introduce a framework that connects forest governance with the values of forest ecosystem services to fulfil the above-stated research gap. This framework should encompass the effects that forest governance has on forest conditions (i.e. the natural resource management perspective), the provisioning capacity of forest ecosystem services and the economic values of forest ecosystem services (i.e. an ecological economic perspective). This framework could be used for assessing the economic efficiency of alternative forest governance regimes. Since public decisions should be driven by a cost-benefit framework, decisions in forest management also need economic indicators to verify which forest governance regime might be more superior to others in terms of economic efficiency. In addition, in order to aid decision making, it is necessary to propose alternative options of forest governance that will be feasible in the future. These practical options, therefore, align economic valuation studies with public decisions in forest governance.

This study aims to bring an economic valuation argument to the forest governance debate. I apply ecosystem service modelling and economic valuation methods of natural resources to assess economic values of forest ecosystem services among possible forest governance scenarios in the context of Vietnam. I focus on quantifying and valuing the changes of forest ecosystem services in relation to the changes of forest conditions resulting from hypothetical implementation of alternative forest governance scenarios. The importance of forest resources and the existing forest governance regimes in Vietnam make it suitable to conduct research on assessing the economic values of the forest ecosystem services under different forest governance regimes.

1.3 The Research Questions

Given the discussion in the previous sections, this study aims to determine the effects of forest governance regimes on the values of forest ecosystem services. Therefore, this study tries to answer how alternative forest governance regimes would affect the provision in terms of physical quantities and the associated values of forest ecosystem services. Given the context of the research site in the Northwest region of Vietnam, the specific questions of the study include:

1. What are the possible, feasible and credible forest governance scenarios that could be applied in this region in the future?
2. What are the likely changes in forest cover as a result of implementing alternative forest governance scenarios?
3. What are the likely changes in the provision and values of forest ecosystem services under the alternative forest governance scenarios?

1.4 The Research Objectives

The research aims to assess several forest governance alternatives from an ecological economic perspective. It analyses the effects of the forest governance regime on spatially-explicit changes to forest cover. Then it attempts to quantify and evaluate the three main forest ecosystem services (i.e. water yield or water supply service, carbon storage and sequestration service, and sediment reduction service) in relation to the changes in forest cover. Therefore, the specific objectives are:

1. to determine the feasible scenarios of forest governance regimes for this area of Vietnam;
2. to quantify and map the forest cover changes caused by the forest governance regime scenarios; and
3. to quantify and evaluate the changes in the provision and economic values of the forest ecosystem services (i.e. carbon storage/sequestration, water provision for hydropower production, and sediment reduction services) under the different forest governance scenarios.

1.5 Significance of the Study

As mentioned above, most studies relating to sustainable forestry have focused on either valuation or the governance issues, with little consideration for the links between the two. Recently, it has been increasingly recognised that forest governance plays an important role in the capacity of forests to provide ecosystem services and, accordingly, in determining the corresponding values of forest ecosystem services. The extent to which forest governance influences the values of forest ecosystem services depends on the spatial and biophysical conditions of forests and the use of forest ecosystem services. By determining the changes in the values of forest ecosystem services as a result of forest governance regimes in the Northwest region of Vietnam, this study contributes to knowledge by integrating forest ecosystem service valuation with forest governance.

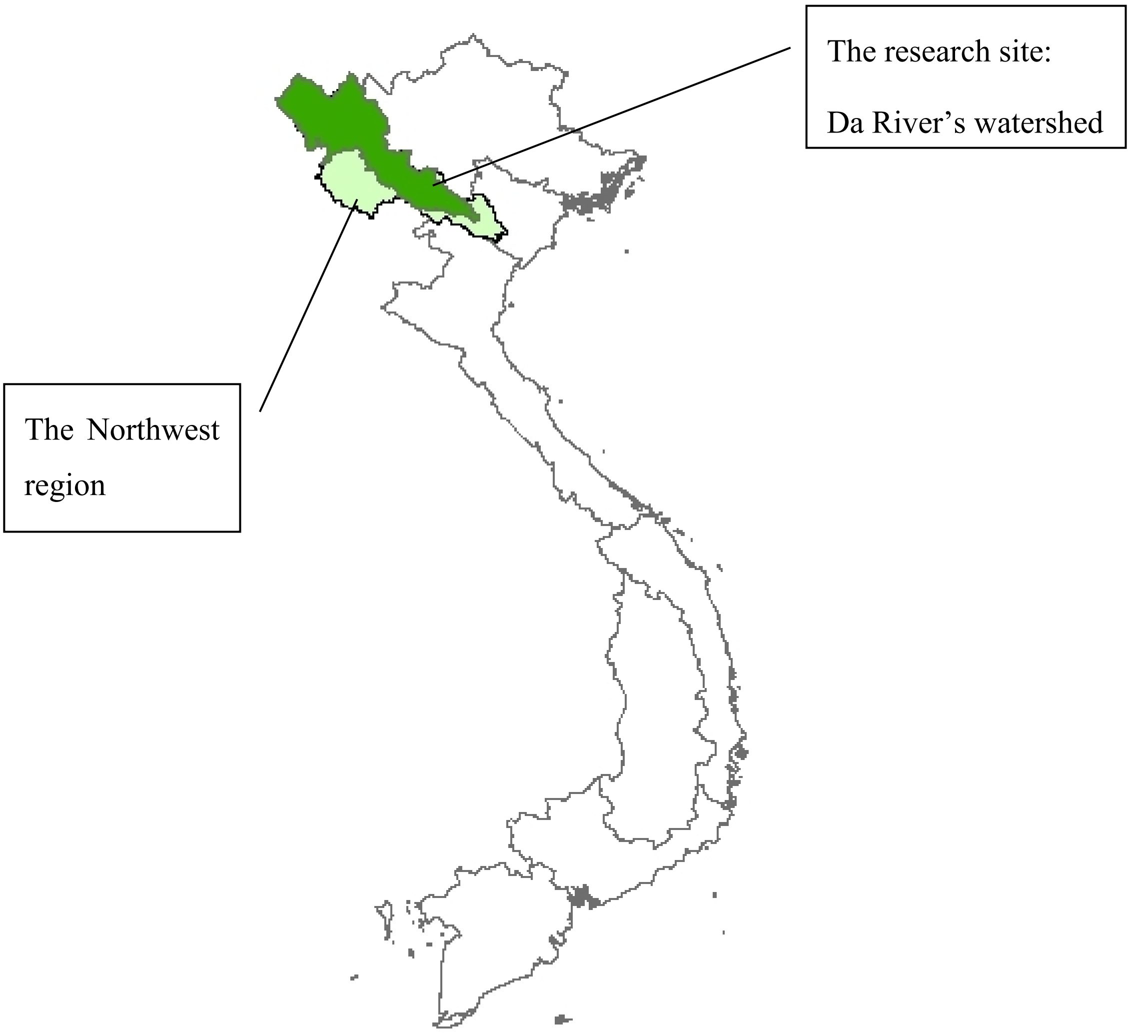
In addition, this study is the first one that has applied the InVEST model (i.e. the model that integrates ecological production functions with an economic valuation approach) to determine the changes in values of forest ecosystem services, particularly for the purposes of assessing the impacts and efficiencies of forest governance in the context of Vietnam. This is particularly significant because it shows that this application is suitable for Vietnam, and potentially for other developing countries that are facing the issues of limited data availability.

Aside from these, the findings regarding the provision and the associated values of the forest ecosystem services could be very useful for policy decision making in forest management. This is especially important in Vietnam where incentive-based programs for forest conservation and reforestation have been implemented. The findings provided in this study can be useful for adjusting and calibrating the existing Payments for Forest Ecosystem Services in the country to reach better forest conservation targets as well as gain the benefits derived from forest ecosystems.

1.6 The Scope of the Study

This study is conducted in the Northwest region of Vietnam. This upland region has a total land area of 3.74 million hectares, of which 41.6% were covered by forests in 2010.[3] The study focuses on forest ecosystem services provided by the Da River watershed (Figure 1.1).

Figure 1.1 The research region



[3] The percentage of forest cover in 2010 is derived from the government database reported in the Decision No. 1828/QĐ-BNN-KL of the Ministry of Agriculture and Rural Development of Vietnam.

Forests can provide many ecosystem services, such as conservation of biodiversity, water supply, carbon storage and sequestration, soil protection, recreation, tourism and so on. This study, however, focuses on three specific forest ecosystem services: water yield, carbon storage and sequestration and sediment reduction. The motivation behind this choice is that they are the three forest ecosystem services of most interest in the literature, as argued by Brandon (2014), Ninan and Inoue (2013) and (Ferraro *et al.* 2012). Moreover, these three forest ecosystem services are considered to be the most important in the context of the research region (the uplands of Vietnam). Forest carbon storage and sequestration are important for their essential role in climate regulation in general, and because Reducing Emissions from Deforestation and Forest Degradation (REDD) projects have been scheduled for implementation in Vietnam. When it comes to ecosystem services related to the provision of water yield, most of the government's projects of payments for forest ecosystem services focus on water supply services (Pham *et al.* 2013). Furthermore, in the study region, water supply for hydropower generation brings substantial value to the overall national economy. The sediment reduction service is worth studying because it provides great benefits in terms of precluding siltation of water reservoirs and downstream waterways. In relation to this, sediment yield reduction is also an important environmental concern in this region.

1.7 Overview of the Thesis

This introductory chapter has presented the rationale for studying the influence of forest governance on the values of forest ecosystem services, the research problems, research questions and objectives. This chapter has also described the significance and scope of the research. In Chapter 2, I review the literature relating to forest ecosystem services, forest ecosystem service quantification and valuation models and forest governance in order to rigorously explain the research gaps. Chapters 3 and 4 present the empirical analyses based on the local characteristics and the current forest governance regime in Vietnam. In Chapter 5, I present a conceptual framework of the study. I define and elaborate concepts and variables of forest governance, forest conditions, forest ecosystem service quantity and values. Forest ecosystem service production functions, land use and land cover changes, values of forest ecosystem services and valuation methodologies are also presented in this chapter. In Chapter 6, I describe the research methods and data. I also present processes and results of the construction of alternative forest governance scenarios. Figure 1.2 summaries linkages between the research questions, objectives and research methods. It shows the flow of the research questions, associate research objectives and research methods used to obtain these objectives. For example, the answers of the question 1 that fit the objective 1 are required to solve the question 2. Similarly, the findings of the objective 2 is the answers for question 2 and are used to solve the question 3. In addition, the specific research methods used to achieve these objectives are shown as well. In Chapter 7, the findings are presented that show the changes in land cover, quantity and values of forest ecosystem services under the three alternative forest governance regimes. Finally, in Chapter 8, I draw conclusions and provide discussion about the contributions of this study and its policy implications for future forest governance.

Figure 1.2 Summary of the Research Questions, Objectives and Methods

Research Questions:	**Research Objectives:**	**Research Methods:**
1) What are the possible, feasible and credible forest governance scenarios that could be applied in this region in the future?	1) To determine the feasible scenarios of forest governance regimes for this area of Vietnam.	Scenario development based on legal framework of forest governance
2) What are the likely changes in forest cover as a result of implementing alternative forest governance scenarios?	2) To quantify and map the forest cover changes caused by the forest governance regime scenarios.	Mapping forest LULC through the combination of expert's perception on the transition likelihood of LULC change and land suitability analysis.
3) What are the likely changes in the provision and value of forest ecosystem services under the alternative forest governance scenarios?	3) To quantify and evaluate the changes in the provision and economic values of the forest ecosystem services under the different forest governance scenarios.	- Applying InVEST model to quantify the provision of forest ecosystem services - Using economic valuation methods to evaluate the values of the forest ecosystem services under these alternative scenarios.

Chapter 2
LITERATURE REVIEW

2.1 Introduction

Many studies have attempted to determine the solutions for of the problem of unsustainable use of forests. Although the literature covers various theories, this chapter reviews three major themes that have recently emerged in the literature. These themes comprise: the ecosystem services framework, the valuation of forest ecosystem services, and the overview on forest governance regimes. The literature mentions these themes in a variety of ways, though this study has primarily focussed on studies that relate their application to assess the effects of forest governance regimes on the provision and economic values of ecosystem services, especially forest ecosystem services. Regarding the ecosystem service framework, this chapter pays particular attention to the definitions and classification systems that can be used for valuation purposes. When it comes to the valuation of forest ecosystem services, the framework of the total economic value is reviewed and the challenges of applying the valuation studies for decision making are assessed. Regarding the recent debates on forest governance, this review has mainly focussed on discussions in the field of natural resources management regarding "the tragedy of the commons" that was proposed by Hardin (1968). By surveying these themes in the literature, I found that they are still being discussed separately. Thus, I argue for the incorporation of these themes in the course of determining effective forest governance regimes in terms of securing the provision and values of forest ecosystem services.

2.2 Ecosystem Services Framework

2.2.1 Ecosystem Services: Links Between Human Welfare and Ecosystems

The concept of ecosystem services reflects links between human welfare and ecosystems. Although the term 'ecosystem service' is very common in the literature related to natural resource management, the concept of ecosystem services is defined quite differently. The commonly cited definitions are as follows:

Daily (1997) defined that "ecosystem services are the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life". According to Daily, ecosystem services sustain biodiversity and the production of ecosystem goods consumed by humans. They also include functions that support human lives, such as cleansing, recycling and renewal. Ecosystem services also provide many intangible aesthetic and cultural benefits.

Costanza *et al.* (1997) pointed out that ecosystem services consist of ecosystem goods and services that bring benefits that humans extract, directly or indirectly, from ecosystem functions (such as the habitat, biological or ecosystem components or processes). By defining it this way, ecosystem services includes flows of materials, energy and information provided by natural resources to combine other forms of capital to support human wellbeing.

De Groot *et al.* (2002) explained that: “ecosystem goods or services are reconceptualised from observed ecosystem functions when human values are implied”. They stressed that the concept of ecosystem services is constructed based on anthropocentric perspectives. For them, complex ecological structures and processes are translated into ecosystem functions that provide ecosystem services valued by humans.

The Millennium Ecosystem Assessment (MA 2005) broadly defined that “ecosystem services are the benefits provided by ecosystems”. They classified ecosystem services into four categories: provisioning services, regulating services, cultural services and supporting services. The MA’s definition has been widely accepted in recent literature of ecosystem services (De Groot *et al.* 2010; TEEB Synthesis 2010; Bateman *et al.* 2011; UK NEA 2011; De Groot *et al.* 2012; Bateman *et al.* 2013; Costanza *et al.* 2014; UK NEA 2014). The significant contribution of the new literature is the revised conceptual framework that link human societies and their wellbeing with the ecosystems. In this framework ecosystem services benefit to humans either indirectly through the interaction with other forms of capital or directly to human wellbeing.

The above definitions share common thought rooted in anthropocentric perspectives when referring to ecosystem services as human benefits derived from ecosystems. However, they differ in several aspects. De Groot *et al.* (2002) and Daily (1997) agreed that ecosystem services are considered to be ecosystem conditions/structures, processes and life-support functions. Costanza *et al.* (1997) described ecosystem services as goods and services derived from ecosystem functions. On the other hand, MA (2005) synthesised the broad and neutral definition brought by the global scientific community, in which ecosystem services are defined as benefits that humans obtain from ecosystems.

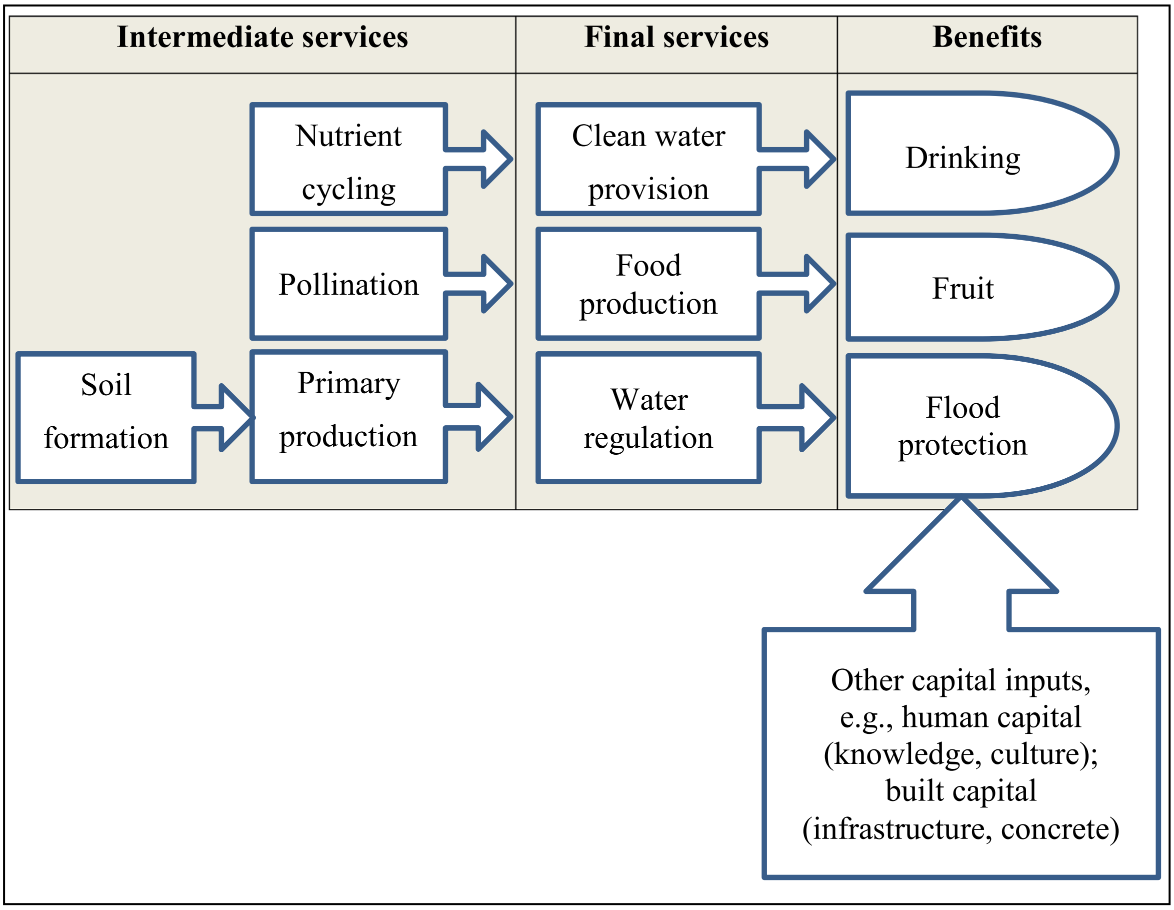
Although being widely accepted by the academic community, these definitions are criticised for being ambiguous and challenging when applied to decision making in natural resource management (Boyd and Banzhaf 2007; Wallace 2007). In order to make it easier to take values of ecosystems into account in decision-making processes, Wallace (2007) and Boyd and Banzhaf (2007) offered alternative definitions of ecosystem services.

While accepting the broad definition provided by MA (2005) that "ecosystem services are the benefits provided by ecosystems", Wallace (2007) argued for an explicitly separated definition of the key terms: ecosystem functions/processes and services. According to him, the terms ecosystem functions and ecosystem processes are synonyms. He defined ecosystem processes as the complex physical and biological cycles and interaction among biotic elements of ecosystems. In broader terms, the processes engage in the transfer of energy and materials. Compared with the definition proposed by De Groot *et al.* (2002) who defined ecosystem functions as "the capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly", Wallace's definition is significantly different. For him, ecosystem functions are a subset of ecological processes and ecosystem structures, and are not ecosystem services. In addition, he agreed that ecosystem services are benefits derived from ecosystems, and he proposed a new term: "ecosystem benefits" for ecosystem services.

Boyd and Banzhaf (2007), on the other hand, aiming to integrate the values of ecosystem services into accounting systems, argued that the definition of ecosystem services should be consistent and compatible with the conventionally defined terms of goods and services currently used in the national accounts. For this purpose, they represented ecosystem services as ecological components (things or qualities) that are directly utilised (either consumed directly or combined with other inputs) by humans. Based on their definition, ecosystem services are end-products, objective rather than qualitative, provided by ecosystems. Therefore, ecosystem services do not include indirect ecosystem processes and functions. In addition, ecosystem services are physically measurable since they refer to components of ecosystems.

The work of Fisher *et al.* (2008) and also Fisher *et al.* (2009) conceptualised the term of ecosystem services by integrating the definitions discussed earlier. On the one hand, in line with the definition of MA (2005), Daily (1997) and Costanza *et al.* (1997), ecosystem services are defined as "the aspects of ecosystems utilised (actively or passively) to produce human wellbeing". Based on this definition, ecosystem services are either ecological elements (structures, components) or processes and functions. These ecological phenomena become ecosystem services only if human societies receive benefits from them. Also, to become services, they do not have to be directly utilised, as argued by Wallace (2007) and Boyd and Banzhaf (2007). In their view, ecosystem services are those that have a direct effect on social welfare and are usually in combination with other capital inputs (e.g., human capital, manufactured capital). The notion that ecosystem services provide benefits to human wellbeing through the interaction with other forms of capital is also proposed by an updated study of Costanza and his colleagues on valuing the global ecosystem services (Costanza *et al.* 2014). Fisher *et al.* (2008) also noted that the definition of ecosystem services should be based on the specific characteristics of the ecosystem of interest. For the purposes of economic valuation and accounting practices, they suggested that only benefits derived from final services can be aggregated to avoid the problem of double-counting (Figure 2.1).

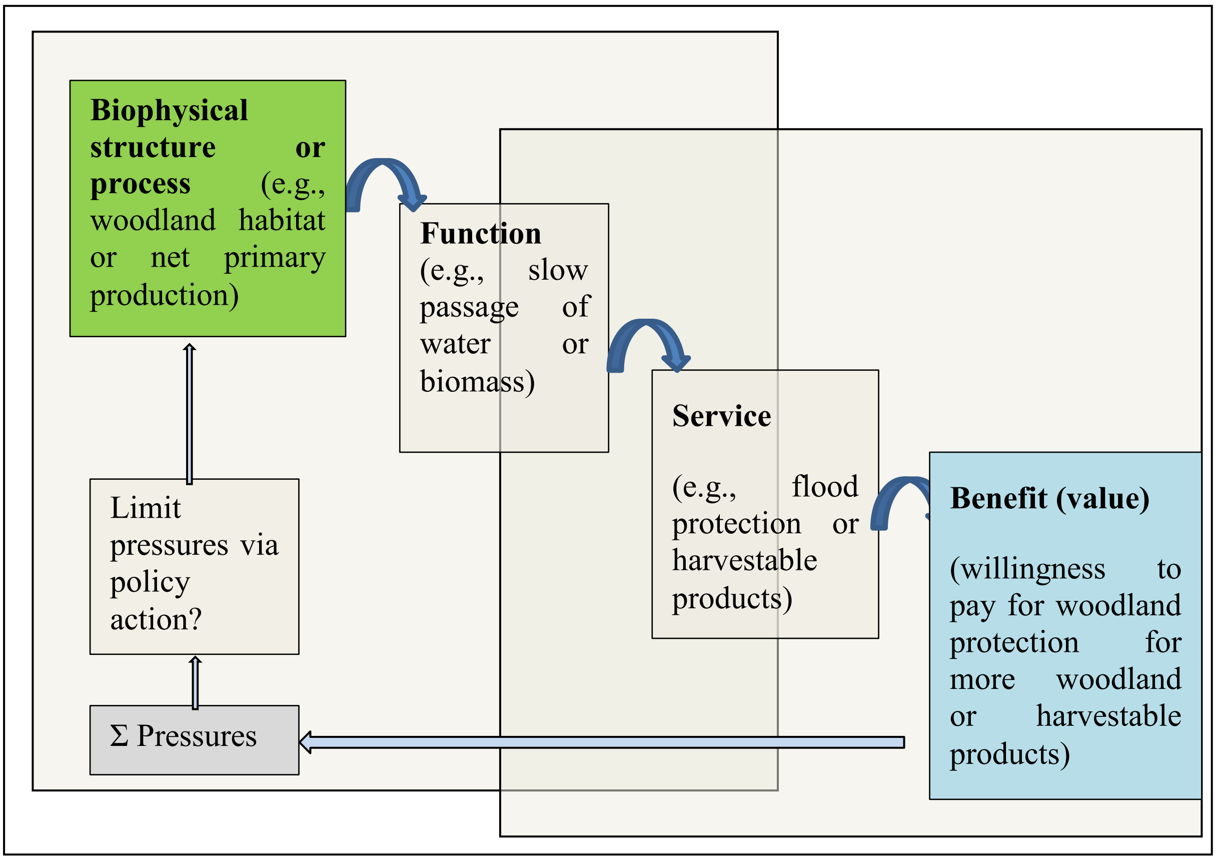
Figure 2.1 Stylised relationships among representative intermediate services, final services and benefits



Source: Fisher *et al.* (2008)

Haines-Young and Potschin (2010) joined the debate by analysing the links between biodiversity, ecosystem services and human wellbeing. They argued that the mere definition of MA (2005): "ecosystem services are the benefits ecosystems provide" makes it difficult to interpret precisely what an ecosystem service is in practice. Although they did not propose their own definition, they provide a diagram that presents the relationship between "biodiversity, ecosystem function, ecosystem services and human wellbeing" (see Figure 2.2). As shown in the figure, there is a stylised distinction between ecological structures and processes and the benefits that people enjoy. It is quite clear that Haines-Young and Potschin (2010) did not consider ecosystem structures or functions as ecosystem services, but rather "ecosystem flows" derived from the ecosystems that benefit humans. The diagram also integrates the ideas of Wallace (2007) and Boyd and Banzhaf (2007) regarding intermediate and final ecosystem services. It shows that an ecosystem service can be either an intermediate or a final one depending on how human societies utilise it, indirectly or directly. This is similar to the view of Costanza (2008).

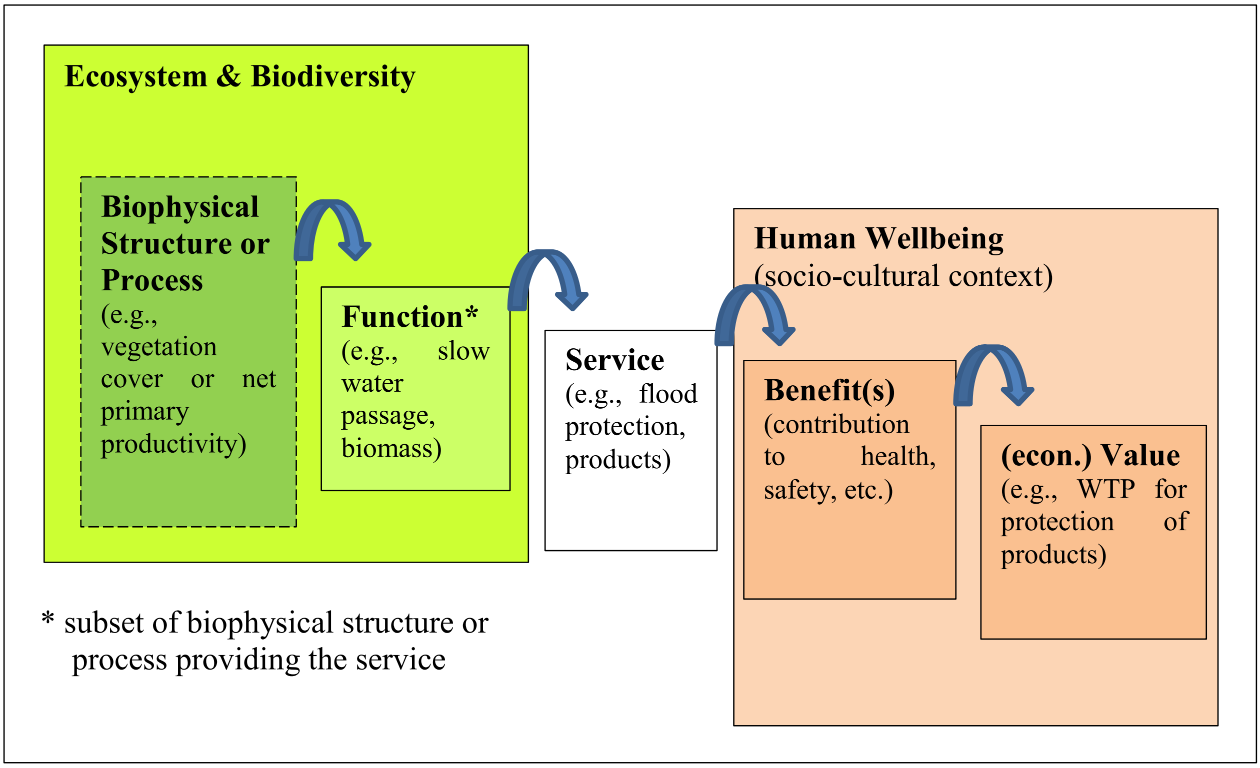
Figure 2.2 The relationship between biodiversity, ecosystem function and human wellbeing



Source: Haines-Young and Potschin (2010)

The work of De Groot *et al.* (2010) on determining the "challenges in integrating the concept of ecosystem services and values in landscape planning, management, and decision making" took the issue of the distinction between ecosystem functions and services proposed by Wallace (2007) and Fisher *et al.* (2009) into account. They then introduced a figure adapted from Haines-Young and Potschin that portrays the link between ecosystems and human wellbeing (see Figure 2.3). The figure illustrates that ecosystem services are generated by ecosystem functions. At the same time, ecosystem functions are established by ecosystem structures and processes. It can be interpreted that ecosystem services are provided by ecosystem structures, and processes that are intermediated by ecosystem functions. The actual usage of ecosystem services brings benefits to human wellbeing.

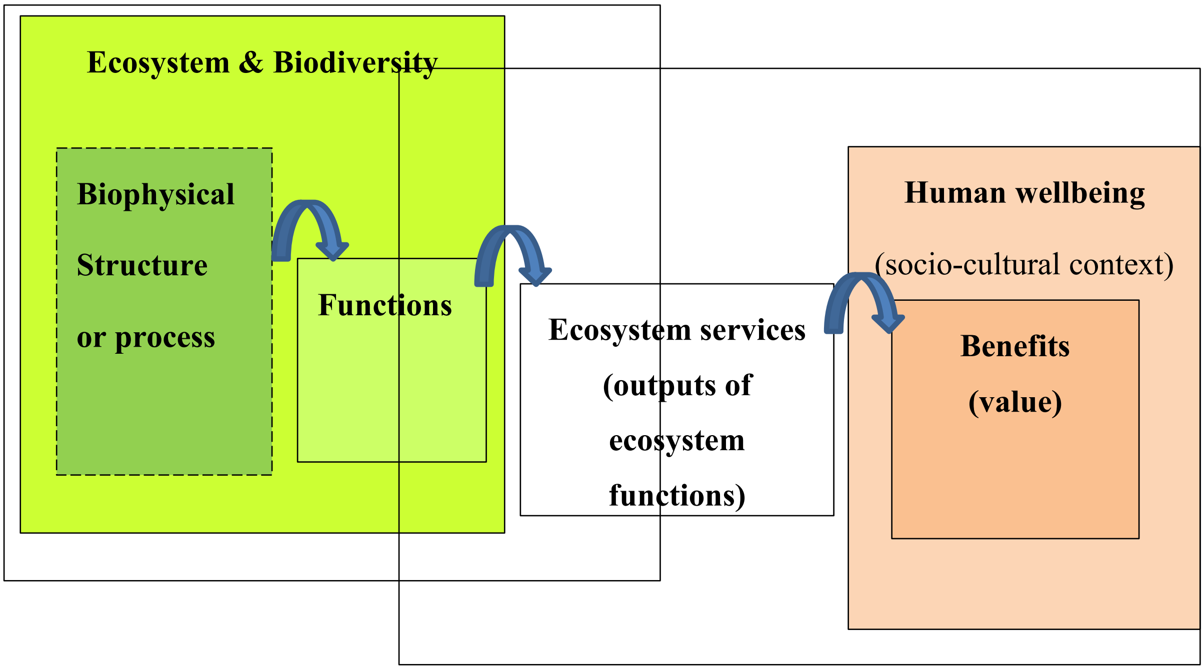
Figure 2.3 Framework for linking ecosystems to human wellbeing



Source: De Groot *et al.* (2010)

In this work, I use the notion of ecosystem services proposed by Fisher *et al.* (2009), De Groot *et al.* (2010), and also the recent studies of De Groot *et al.* (2012) and Costanza *et al.* (2014). Because this notion is generally consistent with common usage in literature. In addition, for the purpose of valuing ecosystem services for decision making in natural resource management, I also concur with the conceptual structure of the link between ecosystems and human wellbeing constructed by Haines-Young and Potschin (2010), as well as the thoughts of Boyd and Banzhaf (2007) and Wallace (2007), in the sense that ecosystem services are physically measurable and provide benefits to humanity. Therefore, in this study, ecosystem services are defined as 'outputs of ecosystem functions' that are indirectly or directly consumed by humans. These outputs, as mentioned by Costanza and his colleagues (Costanza *et al.* 1997; Costanza *et al.* 2014), consist of "materials, energy and information" derived from ecosystems that combine with other types of capital, such as manufactured and human capital, to create human wellbeing. They are physically measurable and directly or indirectly contribute to human welfare. This concept of ecosystem services is depicted in Figure 2.4, which represents the links between the ecosystems and human wellbeing.

Figure 2.4 Ecosystem services: Linking ecosystems to human wellbeing



2.2.2 Ecosystem Services Classification System for Valuation Purposes

In order to integrate various ecosystem service approaches to natural resource management, it is essential to classify ecosystem services properly. There have been several efforts to classify ecosystem services. This section begins with the general lists of ecosystem services proposed by Daily (1997), the list offered by Costanza *et al.* (1997) (which is reused in the updated study (Costanza *et al.* 2014)), and then the mainstream classification in the literature suggested by De Groot *et al.* (2002) (that was revised in the later work publised in 2010 and 2012 (De Groot *et al.* 2010; De Groot *et al.* 2012)), and MA (2005). Subsequently, the alternative typology argued by Wallace (2007), Boyd and Banzhaf (2007) is discussed. The arguments of Fisher *et al.* (2009) and Costanza (2008), regarding the classification of ecosystem services for valuing ecosystem services, are also mentioned.

Daily (1997) provided a list that is considered to be a set of ecosystem functions/processes. By defining ecosystem services as the ecosystem conditions and processes, Daily tried to illustrate the link between ecosystems and human wellbeing. Obviously, this classification comes from a perspective of an ecologist immersed in ecologically underpinned processes. The list was criticised by Boyd and Banzhaf (2007) as being an unsuitable classification scheme for valuation purposes. This is because the list is not practical enough to measure ecosystem services. Firstly, measuring outcomes of ecosystem processes is much easier than measuring the processes. Secondly, a process that provides final services is valuable, but this process is not a final service in an economic sense, meaning that if a process is considered as an ecosystem service, the problem of double-counting might occur (Boyd and Banzhaf 2007).

Costanza *et al.* (1997) offered a comprehensive list of ecosystem services when they tried to measure the economic value of the world's ecosystem services. In this list, they only took renewable ecosystem services into consideration, including 17 main categories (e.g., non-renewable fuels, minerals and the atmosphere are excluded from their list). In each major category of ecosystem services, they showed corresponding ecosystem functions that engage in the provision of the services as well as some key examples of ecosystem services. They also made a cautious note that ecosystem services and functions do not necessarily exhibit one-to-one correspondence, meaning that, in some cases, an ecosystem service can be an output of more than one ecosystem function while, in other cases, one ecosystem function can contribute to more than one ecosystem service. They also recognised the problem of "double-counting" caused by the interdependence between ecosystem functions and ecosystem services. Attempting to solve the double-counting problem, they suggested distinguishing 'joint and addable products' from products that would represent 'double-counting'. This task is not easy because, in some cases, when ecosystem functions and services are interdependent, they can jointly produce benefits to humans. In these cases, their benefits can be added. In some other cases, double-counting would result from one service representing different aspects if they were aggregated.

De Groot *et al.* (2002) aimed to standardise a conceptual framework to help describe, classify and value ecosystem functions and services. For this purpose, they organised ecosystem services based on ecosystem functions. In conceptualising ecosystem functions as a subset of ecosystem structures and ecosystem processes, they proposed four fundamental categories of ecosystem functions including: regulation, habitat, production and information functions that include 23 specific functions. Firstly, they defined regulation functions as the capacity of ecosystems to regulate essential ecological processes and life-support systems. These regulation functions provide many services that provide benefits to humans either directly or indirectly (e.g., clean air, water, soil and biological control services). Secondly, habitat functions are the functions of a natural ecosystem that provide habitat or refuge and reproduction to wild plants and animals. These functions play essential roles in the conservation of biodiversity, genetic resources and evolutionary processes. Thirdly, production functions are those that relate to photosynthesis and nutrient uptake, which typically use the energy from sunlight to convert carbon dioxide, water and nutrients into a wide variety of carbohydrate structures. Fourthly, information functions are defined as ‘reference functions’ that are vital sources of inspiration for science, art and culture. These functions also relate to the provision of opportunities for spiritual enrichment, mental development and leisure.

Generating an entire ecosystem service typology, De Groot *et al.* (2002) present specific functions based on the four categories. For every function, they provided examples of associated ecosystem services and ecosystem components and processes that underpin each function. Similar to Costanza *et al.* (1997), they realised that ecosystem processes and services do not always show a one-to-one correspondence. For example, the function of “gas regulation” maintains air quality based on biogeochemical cycles (e.g., CO_2/O_2 balance), but also contributes to the greenhouse effect and thereby engages in “climate regulation”. They also noted that ecosystem functions and services can overlap, which results in the possibility of economic “double-counting”. For example, gas regulation functions (and related services) have an impact on the climate and can, therefore, be double-counted in valuing the climate regulation service.

In the recent work of De Groot *et al.* (2010; 2012), they revised the ecosystem service classification after the popular publications of MA (2005) and TEEB Foundations (2010). The major revisions include the changes in the four categories of ecosystem services. They renamed the “ecosystem functions” to “ecosystem services” and replaced the “information functions” by “cultural services” and also the specific names of particular services. In addition, they also reordered the four categories as: provisioning, regulating, habitat, and cultural services and focused more on describing indicators used to measure the stock and flows of ecosystem services.

MA (2005) attempted to create an ecosystem service framework that links ecosystem services to human welfare. Similar to the classification of De Groot *et al.* (2002), MA (2005) proposed a genetic typology of ecosystem services’ classification based on ecosystem functions. They classified ecosystem services into four main groups including provisioning, regulating, cultural and support services (see Table 2.1). This way of classifying ecosystem services has advocated ecosystem service measurement and valuation for decision making (Boyd and Banzhaf 2007; Fisher *et al.* 2009).

However, the classification of MA (2005) was criticised by Wallace (2007) and Boyd and Banzhaf (2007) for the vague classification and risk of double-counting in valuing ecosystem services. Although acknowledging the effort of the MA classification scheme, in terms of motivating quantifying ecosystem services, they claimed that the MA’s system is overly generic and confounding. For example, the regulating services relate to ecosystem functions and processes that cause difficulties in practical measurement. Besides this, supporting services are not services in their own right, in the sense that they are not final services, as defined by Wallace (2007) and Boyd and Banzhaf (2007), which leads to the problem of double-counting (Boyd and Banzhaf 2007; Wallace 2007; Fisher *et al.* 2009).

Wallace (2007) criticised the mainstream literature of ecosystem service classification because it is represented by the typology of the ecosystem service offered by MA (2005), which has a risk of double-counting and there are difficulties in applying for natural resource management. He argued that for effective decisions, the classification must be constructed as a coherent set of services that can be evaluated and traded off in a decision system. Examining the classification system of MA (2005), Wallace pointed out that the classification of the MA processes (means) and services (ends) are mixed up, which leads to the categories not being able to be used for decision making. For example, according to Wallace, water regulation is not a service on its own right; instead it is a process to achieve drinking water or to control flooding. Therefore, the regulation services are not ends, but they are means to attain ends. Similarly, the supporting services are not ends in their own right. For example, nutrient and water cycling are not ends; instead, they are means to provide human welfare. Wallace also demonstrates the possibility of double-counting when using the MA (2005) classification system for valuing ecosystem services. For instance, many regulating services underpin several services, and if the services are valued separately, double-counting occurs.

Table 2.1 Categories of ecosystem services and examples of related services

Type of service	Service
Provisioning services	Food
	Fibre
	Genetic resources
	Biochemicals, natural medicines, etc.
	Ornamental resources
	Fresh water
Regulating services	Air quality regulation
	Climate regulation
	Water regulation
	Erosion regulation
	Water purification and waste treatment
	Disease regulation
	Pest regulation
	Pollination
	Natural hazard regulation
Cultural services	Cultural diversity
	Spiritual and religious values
	Aesthetic values
	Knowledge systems
	Educational values
	Inspiration
	Social relations
	Sense of place
	Cultural heritage values
	Recreation and ecotourism
Supporting services	Soil formation
	Photosynthesis
	Primary production
	Nutrient cycling
	Water cycling

Source: Adapted from Box 2.1 (p. 43) in the Millennium Ecosystem Assessment (2005).

Wallace (2007) presented an alternative classification scheme. He first redefined ecosystem services as the structure and composition of particular ecosystem elements. Then he classified the services based on the specific human values they support that include: 1) adequate resources (basic needs), 2) protection from predators, diseases and parasites, 3) benign physical and chemical environments, and 4) socio-cultural fulfilment. He argued that this classification system links values with ecosystem services and can be useful for analysing alternative uses of natural resources so that decisions can be made to sustain human wellbeing. This is because the classification removes the confusion between processes (means) and services (ends), which solves the problem of double-counting and is consistent with a human decision-making framework because the values describe important aspects of human wellbeing.

The classification of Wallace, however, was strongly criticised by Costanza (2008) for its oversimplification of the complex nature of ecosystem services. According to Costanza, Wallace's idea that ecosystem processes are means while ecosystem services are ends is flawed, because human wellbeing is 'the end'. Therefore, by definition, all ecosystem services are means to achieve human wellbeing. In this sense, ecosystem processes can be services.

Boyd and Banzhaf (2007) argued for the standardised units of ecosystem services that can be used in welfare accounting, so they offered a definition of final ecosystem service units (which is rooted in economic principles) that are comparable to the conventional national accounting system. In this manner, they attempted to classify ecosystem services in the way that the values of ecosystem services can be incorporated into the existing national accounts. They illustrated a classification of ecosystem services associated with sources of human wellbeing.

According to Boyd and Banzhaf, the sources of wellbeing consist of harvested elements that satisfy the needs of human consumption of food, fibre and other materials, or the human demand for good health, safety and recreation. When classifying in this way, only end-products derived from ecosystems are taken into account. It is also worth noting that their classification is rooted in their notions that ecosystem services are economic benefit-specific. By benefit-specific, they mean that ecosystem services are benefit-contingent and depend on particular human activities or wants. This means that a particular ecological component is accounted as an ecosystem service if it directly contributes to human welfare and is not accounted as an ecosystem service if it is used as an intermediate input in processes of producing final products consumed by humans (see Boyd and Banzhaf (2007) for an example of this).

Fisher *et al.* (2009) provided a comprehensive assessment of previous ecosystem service classification schemes. They contended that the main theme of ecosystem service classification, such as the one proposed by MA (2005) is not suitable for valuation purposes because it could lead to double-counting. Therefore, for valuation purposes, they argued for a classification scheme that differentiates ecosystem services into intermediate services, final services and benefits, like the one proposed by Boyd and Banzhaf (2007). Recognising that the link between ecosystems and human welfare is complex, the same service can be both intermediate and final depending on its links to human welfare. They suggested that the classification must be clear about whether the service is the final one or not. This classification avoids the risk of double-counting because only final benefits are valued and aggregated.

Recently, TEEB Foundations (2010) proposed a updated typology of ecosystem services that mainly based on the classification system initiated by MA. TEEB provided 22 specific ecosystem services and also categorised them into 4 main groups: provisioning, regulating, habitat and cultural and amenity services. The important revision of TEEB in comparison with the MA is the omission of supporting services and the initiation of habitat services. The habitat services include those that relate to maintenance of life cycles and genetic diversity. The classification of TEEB complies with the notion that ecosystem services and benefits gained by human societies are necessarily differentiated and ecosystem services can benefit humans in multiple ways.

The MA classification system of ecosystem services was also adopted in the UK National Ecosystem Assessment (UK NEA 2011). This is the first attempt to assess the UK natural ecosystem resources that involved about 500 experts from various government, academic, NGO and private sector institutions working in the fields of natural sciences, economics and the social sciences. The UK NEA follow-on reported in 2014 (UK NEA 2014) continued using the MA classification. The important contribution of the UK NEA (2011, 2014) is the incorporation of the post-MA advances focus on 'final ecosystem services' to avoid the double counting of services into the ecosystem service classification.

The study by Baulcomb *et al.* (2015) paid more attention and provided insights on cultural ecosystem services in the context of coastal ecosystems. The paper developed a pathway to explicitly identify and value cultural dimensions of ecosystem services. It suggested that a well-defined ecosystem service classification is required to incorporate cultural-ecological linkages as specific cultural ecosystem services.

In summary, there are different schemes that can be used to classify ecosystem services that fit different purposes of the study (Costanza 2008). For the purpose of valuing ecosystem services that aids decision making in natural resource management, I argue for the notion of dividing ecosystem services into intermediate services and final services, as suggested by Boyd and Banzhaf (2007), Fisher *et al.* (2009), and UK NEA (2011, 2014). This classification system helps to avoid the problem of double-counting. Therefore, when applying the mainstream classification scheme proposed by De Groot *et al.* (2002), MA (2005), and TEEB Foundations (2010) as well, it is essential to carefully examine the degree of the connection of the ecosystem services of interest to human welfare to identify the final services that are valued and aggregated into the total values of the ecosystem services.

2.3 Valuation of Forest Ecosystem Services

2.3.1 The Total Economic Value Framework

From the perspective of economists, the problem of worldwide forest loss and degradation is due to weak economic incentives for forest conservation. Within the existing market mechanism, forest conservation is less profitable than forest exploitation (Pearce 2001). The lack of economic incentives results from many essential ecological services provided by forests not being apparent in market transactions. This means that the market does not provide comprehensive information for effective decision making relating to forest ecosystems (MA 2005). When the total values derived from forests are not captured, decision making relating to forests is likely to be misguided and society then becomes worse off due to the loss or degradation of forests (Pearce 2001; MA 2005; Pascual *et al.* 2010).

Therefore, in order to change human behaviour in relation to forests and to achieve sustainable use of forests, economists attempt to impute economic values to non-marketed benefits provided by forests. For this purpose, the concept of total economic value (TEV) is widely used as a framework of forest ecosystem service valuation (Smith *et al.* 2006; Pascual *et al.* 2010). In this framework, the benefits provided by forests encompass two primary types of value, which are "use values" (also known as "active use value") and "non-use values" ("passive-use values") (Pearce 2001; De Groot 2006; Smith *et al.* 2006). The 'use value' type consists of 'direct-use values' that stem from both consumptive and non-consumptive uses of forests; for example, timber, non-timber forest products, etc., and 'indirect use values'; for example, values derived from forest environmental services, such as freshwater provision, soil protection, carbon storage, sequestration and so on. Non-use values are those values that do not relate to direct or indirect use of ecosystem services. They reflect the desire of individuals to keep ecosystem services available for other people to use (Kolstad 2000). They consist of existence values, altruistic values and bequest values. The existence value of forests is attributed to the existence of forest that can be reflected by individuals' willingness to pay for forest conservation or biodiversity. This willingness to pay is not related to the use of forests. Altruistic value is the value that can be captured by the willingness of an individual to pay to maintain forest ecosystem services so that others may use them. Bequest values reflect the desire to pass on forests to future generations to use. Another type of value is option value. The option value relates to keeping the possibility of the use of forests available in the future (Pearce 2001; Smith *et al.* 2006; De Groot *et al.* 2010; Freeman *et al.* 2014).

Based on the notion of the neoclassical economics, ecosystem services that are not captured by the market can be seen as positive externalities. Because they are not apparent in market transactions, environmental economists either identify links between market goods and environmental goods in order to estimate the welfare changes that are associated with the changes in ecosystem services or, alternatively, they must create hypothetical markets to evaluate the welfare changes. Viewed in this way, a range of valuation methods has been developed and applied to value forest ecosystem services (Grafton *et al.* 2004; Pascual *et al.* 2010). The valuation methods are comprised of three primary groups: (a) direct market valuation approach, (b) revealed-preference methods, and (c) stated-preference methods. Based on the work of several authors (De Groot *et al.* 2002; Farber *et al.* 2002; Grafton *et al.* 2004; De Groot 2006); Pascual *et al.* (2010), the description of each method, its strengths and limitations, are summarised as follows:

a) Direct Market Valuation Approaches

Direct market valuation approaches use data that can be observed in market transactions. They include: (a) market price-based methods, (b) cost-based methods, and (c) production function methods. Market price-based methods are applied to evaluate an ecosystem service when this ecosystem service is traded on the market (e.g., provisioning ecosystem services: food, timber, non-timber forest products, etc.). The market price of this ecosystem service is a good indicator for valuing this service. Cost-based methods involve the estimation of the costs required to obtain the same benefits provided by ecosystem services through human-made technologies. There are several techniques, including the avoided-cost method, replacement-cost method and mitigation or restoration-cost method. The avoided-cost method implies the costs that would be avoided with the presence of an ecosystem service to its economic value. The avoided-damage method measures the benefits of an ecosystem service by either the value of property protected by the ecosystem service, or the cost of actions required to avoid damages without the presence of this ecosystem service. The replacement-cost method estimates the economic values derived from particular ecosystem services through the costs required to replace this service with man-made technologies. The third method refers to the costs relating to mitigating the effects resulting from the loss of ecosystem services or the costs of restoring these services. Production function-based methods apply scientific knowledge on the cause-effect relationship between the ecosystem services and the production of marketed commodities to estimate the value of these ecosystem services. These methods measure the value of the ecosystem services based on the contribution of these services to the value of the marketed goods.

These methods have the advantage of relying on observed information that reflect actual individual preferences or costs. These data are relatively easy to gather (Ellis and Fisher 1987). However, relying on observed data leads to primary limitations of these approaches. This is because of market failure: the lack of markets for most of the ecosystem services or the existing markets being distorted. In the former case, these methods cannot be applied due to some required data not being available. In the latter case, where markets exist but are not perfectly competitive, market prices are not a good sign of individual preferences or marginal costs. As a result, the values of ecosystem services that are estimated from the distorted market information, will be biassed, which means that they are not a useful guide in policy decision making. In addition, each direct market valuation method has drawbacks. For example, production function-based methods rely on the knowledge of the effects of ecosystem services on the production of traded goods. However, as argued by Daily (1997) and Daily *et al* (2000), the existing knowledge on the cause-effect links between ecosystem services and the production of marketed commodities is still in the early stages of development. In the case of the replacement-cost method, there may be situations when the social loss due to the absence of ecosystem services is less than the cost of replacing them with human-made technologies (Farber *et al.* 2002).

b) Revealed-preference Approaches

Revealed-preference methods use the observed behaviour of an individual in the markets that relate to the ecosystem services in question. The two and probably most commonly used revealed-preference methods of environmental valuation are the travel-cost method and hedonic pricing. The travel-cost model mostly involves estimating tourism or recreation values of ecosystem services placed in recreation areas. It is based on the assumption that the demand of ecosystem services requires travel to recreation areas. Therefore, the travel costs (may include both direct expenses and opportunity costs of time) can be accrued for the value of the ecosystem services. For example, as described by Farber *et al.* (2002), "recreation areas attract distant visitors whose value placed on that area must be at least what they were willing to pay to travel to it". The hedonic pricing method (also known as attribute pricing) estimates values of ecosystem services based on the assumption that the demand for these ecosystem services may be reflected in the prices of marketed commodities associated with them. For example, housing prices are relevant to amenity attributes, including ecosystem services, such as house prices at beaches being usually higher than those that are located inland (Farber *et al.* 2002).

These revealed-preference methods also have some limitations. Generally, if the market of the commodities associated with the ecosystem services of interest is imperfect, the estimated values of these ecosystem services will be biassed. In addition, because they rely on observed information, these methods cannot estimate non-use values of ecosystem services. Besides, these methods require large data, complex statistical analysis to determine the relationship between environmental goods and the associated market goods. Consequently, conducting these methods is costly and time-consuming (Pascual *et al.* 2010).

c) Stated-preference Approaches

Stated-preference methods are the only ones that can value environmental goods in cases where there are no markets to provide information on the value of environmental goods. These methods involve developing hypothetical scenarios of the changes of ecosystem services and eliciting individuals' willingness to pay for an improvement or willingness to accept forgoing this improvement or the degradation/loss of ecosystem services in social surveys. The individuals' responses are then modelled to estimate the values of the changes in ecosystem service conditions (Grafton *et al.* 2004). The common types of stated-preference methods include the contingent valuation method (CVM), attribute-based stated-choice modelling method (ABSCM) and group valuation. The CVM often involves rigorous construction of a scenario that offers a hypothetical environmental improvement and asking individuals to state their willingness to pay for the offer. For example, in a survey, individuals may be asked to express their willingness to pay for an environmental improvement project that prevents landslides, which is something that they might benefit from by avoiding damages caused by landslides (Ahlheim *et al.* 2009). The ABSCM also makes use of social surveys to elicit individuals' expressions of their choices among alternative options that are defined by different levels of attributes of ecosystem services and the associated payment that would be required. This method models the responses of individuals regarding the levels of the attributes (i.e. the levels of the ecosystem services and payment) to estimate the value of the ecosystem services (Holmes and Adamowicz 2003; Grafton *et al.* 2004). Group valuation has recently been getting greater attention in the course of ecosystem service valuation. Rooted in social and political perspectives, this valuation method applies the principles of deliberative democracy and the assumption that decision making relating to public good should rely on open public debate rather than an aggregation of individual preferences (De Groot *et al.* 2002). This method is acknowledged for its ability to deal with the issue of social equity relating to the allocation of ecosystem services (Wilson and Howarth 2002).

Although these methods have been widely used in non-market valuation, particularly in ecosystem service valuation, it is worth noting that they have been severely criticised. The criticism primarily relates to the validity and reliability of the results and various sources of errors and bias. The validity and reliability of the stated-preference studies are questioned because of their hypothetical nature. For example, there is an uncertain relationship between individuals' expression of willingness to pay when they are doing the survey and what their actual behaviour might be if they were faced with such a decision (Venkatachalam 2004; Carson and Hanemann 2005). Furthermore, some authors, such as Diamond and Hausman (1994) argued that CVM is a flawed approach for valuing non-use values. The quality of CVM studies is difficult to validate due to the absence of market parallels (Venkatachalam 2004).

2.3.2 Valuing Forest Ecosystem Services

Benefits derived from forest ecosystems have been recognised as critical to human wellbeing. The term 'forest ecosystem services' has quickly emerged since the 1980s (Ferraro *et al.* 2012). The literature relating to forest ecosystem services has grown rapidly, resulting in the diversity of the definition of forest ecosystem services, as well as classification systems. Overall, there are at least two classification schemes that aim to connect forest ecosystem services to human wellbeing (Nasi *et al.* 2002). These classification schemes come from ecologists' and from economists' perspectives. Typically, from ecological perspectives, forest ecosystem services are classified based on the forest ecosystem functions that bring benefits to humans. On the other hand, from economists' point of view, they are classified by the types of economic values that they provide. Recently, the concepts of forest ecosystem services and the classification systems have been discussed in the work of De Groot *et al.* (2002), and especially in the ecosystem framework proposed by MA (2005). This section reviews the literature relating to the valuation studies on forest ecosystem services.

The study of Costanza *et al.* (1997) on "the values of the world's ecosystem services and natural capital" estimated the economic values of the global ecosystem services using a meta-analysis approach. Their findings showed that the forests are one of the most valuable ecosystems. Similar to the general classification of ecosystem services, forest ecosystem services are categorised based on ecological functions. Among the evaluated services, those that are regulated are the most valuable, such as climate regulation, soil erosion control and nutrient cycling, followed by provision services, e.g., raw materials, recreation and food production. Several services were not valued, including gas regulation, pollination, biological control and habitat/refugia, mainly because they had not yet been adequately studied in these ecosystems.

The work of Chomitz and Kumari (1998) was one of the first rigorous reviews of literature regarding the benefits provided by forests. Their focus was on domestic benefits of tropical forests. Based on their reviews, there are two types of services derived from tropical forests that frequently appear in the literature. They consist of hydrological benefits (e.g. erosion control, flood prevention, water supply) and benefits from non-timber forest products. These authors also pointed out that their review was not comprehensive because other potential benefits were not considered, such as ecotourism services, sales of bioprospecting rights and carbon sequestration services.

Pearce (2001) conducted a comprehensive survey of the literature that values forest ecosystem services to examine economic reasons for forest degradation and loss. From an economics point of view, he classified forest ecosystem services based on economic value typology. Therefore, forest ecosystem services are categorised into four groups that consist of direct-use values, indirect use values, option values and non-use values (Table 2.2). He found that the values of carbon storage and sequestration and timber values are dominant in the literature. Tourism and recreation values are also of interest in valuation studies. The values relating to biodiversity and genetic information have not been of much concern. Similarly, benefits derived from forested watersheds have not been adequately considered. For non-use values, several studies have attempted to estimate the existence and bequest values of forests. Pearce argued that more effort should be put into the economic valuation of forest ecosystem services, particularly in valuing option and existence values in order to have more powerful arguments for forest conservation.

Table 2.2 Pearce (2001)'s forest ecosystem service classification

Value category	Definition of value	Example of forest ecosystem services
Direct-use values	Values arising from consumptive and non-consumptive uses of the forest.	- Timber, fuel wood, and non-timber forest products - Extraction of genetic material - Tourism and recreation
Indirect use values	Values arising from various forest services that indirectly produce benefits to humans.	- Protection of watersheds: soil conservation, water flow regulation, water supply, water quality regulation - Storage of carbon (climate benefits)
Option values	Values reflecting a willingness to pay to conserve forests even though there is no likelihood of current usage.	
Non-use values	These values reflect a willingness to pay for conservation or sustainable use of forests even if they are unrelated to the current or planned forest usage.	

Source: Adopted from Pearce (2001)

Similar to the work of Pearce (2001), Nasi *et al.* (2002) sought scientific evidence to solve the problem of forest degradation and forest loss from economic perspectives. According to them, there are two classification foundations of forest ecosystem services found in the literature. These are the ecosystem function-based classification and the economic value-based classification. The former categorises forest ecological services into several major types including: water quantity and quality, climate regulation, carbon storage, pollination, seed dispersion, natural pest control, cultural, aesthetics, recreational and amenity services. The latter is based on the economic perspectives where forest ecosystem services are classified into direct-use values, indirect use values and non-use values, which are similar to those proposed by Pearce (2001). Forest commodities, such as timber, non-timber products and forest non-commodity benefits, such as forest recreation, are examples of direct-use values. Indirect use values include the services of forests that indirectly contribute to human utility through positive externalities. They often relate to watershed protection (hydrological related services) or climate regulation. Non-use values are attributed to values that are attached to the existence of forests (existence values), or values attached to preserve the options for future usage (option values), or as bequests to future generations (bequest values).

Nasi *et al.* (2002) also contended that it is necessary to distinguish the values of forest ecosystem services to human society at different levels: local, regional, national and global. Local values typically relate to forest products and services that are utilised by forest users; for example, timber, fuelwood, non-timber products that are harvested by a community for self-consumption or for sale, or timber that is logged and sold by a logging enterprise, etc. Regional values refer to values captured at the provincial or state level. Benefits of forest ecosystem services go beyond local forest users, such as downstream users of a watershed. National values (or domestic values), as defined by Chomitz and Kumari (1998)), are values that are received by people living within a national boundary. For example, forest ecosystem services that provide national values are wildlife habitat protection or water supply for hydropower generation. The global values refer to those that are obtained by people living outside the nation to which they belong. Carbon sequestration is a typical service that produces global values.

Since 2005, the ecosystem service framework proposed in the Millennium Ecosystem Assessment (MA 2005) by world-leading experts, have been widely acknowledged and increasingly adopted by the studies relating to forest ecosystem services. As previously mentioned, the MA (2005) classified ecosystem services into the categories of provisioning, regulating, supporting and cultural. The concepts and classifications recommended by MA (2005) have been further developed. For example, Ojea *et al.* (2012), joined the debate on the classification of ecosystem services proposed by the MA (2005), by seeking insight into the problem of the MA's classification of ecosystem services for economic valuation. They examined the issue of MA's classification by surveying original studies on valuing forest water-related services. They then compared the values of the original studies with the values of these services after being reclassified, based on an alternative classification, which they named as an output-based classification.[4] Following the MA classification, forest water-related services were categorised into four primary groups: provisioning, regulating, cultural and amenity and support services. At the same time, based on the output-based classification, they placed these services into four groups: 1) improvement of extractive water, 2) improvement of in-stream water supply, 3) water damage mitigation, and 4) provision of water-related cultural services.[5] Their findings showed that the application of the MA classification for economic valuation can create a risk of double-counting due to service overlapping and service ambiguity, and the MA classification is not able to distinguish between final outcomes that forest ecosystem services contribute to human welfare and the intermediate benefits. Therefore, they argued for an output-based classification suggested by Boyd and Banzhaf (2007) for valuing forest ecosystem services.

Reviewing the recent publications on valuing forest ecosystem services in developing countries, Ferraro *et al.* (2012) argued that although the interest in valuing forest ecosystem services has grown dramatically, rigorous studies are still limited. They showed that the current literature on forest ecosystem services valuation has still focussed mainly on the use values derived from forests that include carbon storage and sequestration, ecotourism, hydrological services, pollination, services relating to human health and non-timber forest products. The estimated values of these services are widely varied. For example, the estimated value of carbon storage ranges from 378 USD/ha to 1,500 USD/ha, which is a result of two main factors. The first factor is the stock of carbon per hectare that varies depending on location; for example, African dry forests (72 tC/ha) and south-east Asian rainforests (225 tC/ha). The second factor is the price used for evaluating carbon storage in monetary terms, which also varies widely. There are two types of pricing that are usually used: the market price of carbon credit, or the social cost of carbon, which is measured by the social damages caused by an additional tonne of carbon released into the atmosphere. Regarding hydrological services, few studies have attempted to value multiple services such

[4] The term 'output-based classification' refers to the classification that considers only the services which are directly used by humans, as proposed by Boyd, J. and Banzhaf, S. (2007). What are ecosystem services? The need for standardized environmental accounting units, *Ecological Economics* 63, 616-626.

[5] The four categories of output-based classification are adopted from Brauman, K.A., Daily, G.C., Duarte, T.K. and Mooney, H.A. (2007). The nature and value of ecosystem services: an overview highlighting hydrologic services, *Annu. Rev. Environ. Resour.* 32, 67-98.

as soil erosion prevention, water regulation, freshwater provision. Concerning the values of non-timber forest products (NTFPs), although there are a large number of studies, Ferraro *et al.* (2012) argued that few studies are reliable.

Ninan and Inoue (2013) also reviewed recent studies that estimate the value of forest ecosystem services. Similar to Ferraro *et al.* (2012), they found that the estimated value of forest ecosystem services varies. Estimated total value of forest ecosystem services ranges from $8 to $4,080 per hectare in 2010 PPP US Dollars, with a mean of $753.[6] Regarding the individual value, the value of carbon sequestration is the highest, followed by the value of pollination services, watershed protection/hydrological services, waste treatment and other services. In addition, the authors found that most of the forest valuation studies have tried to estimate only the total benefits rather than the changes in values of forests among alternative usages. They argued that, among other things, future studies should focus on assessing the values of forest conservation versus forest conversion.

Recent papers dealing with valuing forest ecosystem services have placed efforts on various issues of assessing total economic values of forest ecosystem services for decision making. García-Nieto *et al.* (2013) assessed the ecosystem services provided by forests in the Sierra Nevada Mountains (south-east Spain). This study sought ways to integrate ecological and social information into the assessment of ecosystem services and make it spatially explicit. The results of the study showed that mapping both the supply- and demand-sides of forest ecosystem services is essential for decision making in environmental management. They explained that spatial information of the supply and demand of ecosystem services helps to identify high-priority protection areas or to define the suitable institutional scale for managing the services. Abram *et al.* (2014), considered spatially explicit information on ecosystem services is needed to influence decision making in land use planning. This paper attempted to evaluate economic values of the inclusive range of forest ecosystem services (provisioning, cultural/spiritual, regulating and supporting ecosystem services) based on the perceptions of local people. It showed that understanding perceptions of local in dynamic and multi-use landscapes is important for forest conservation. Häyhä *et al.* (2015) assessed the total economic values (TEV) of the ecosystem services derived from a mountainous forest area in North Italy. The authors considered both biophysical and monetary units of ecosystem services are measured. They also made use of GIS techniques to analyse and visualise the

[6] PPP is purchasing power parity.

spatial provision and distribution. Their results showed that regulating and provisioning services of forest ecosystems have a major importance accounting for 49% and 40% of the TEV, respectively, while the cultural services were 11%. They recommended that mapping ecosystem services is an important technique of ecosystem service assessment. This technique is useful for understanding and visualising the spatial distribution of services that enable to identify priority areas as well as analysing trade-offs and synergies among services. They also argued that a combination of biophysical unit measurement and an economic valuation of all forest ecosystem services is essential to persuade policy makers in ecosystem management. Ninan and Kontoleon (2016) also recognised the importance of measuring the full range of forest ecosystem services, attempted to estimate the TEV that derives from forests. They tried to estimate all possible ecosystem services that are provided by the Nagarhole national park in Karnataka, India. The ecosystem services consist of water conservation, soil protection, carbon sequestration, recreation, nutrient cycling,air purification, biodiversity, pollination, NTFPs, and grazing). They also tried to measure the value of disservices of forests including wildlife damages and forest fires. They argued that if the value of forest ecosystem services is considered in decision-making process forests in tropical countries like India can be conserved.

2.3.3 The Use of Economic Valuation of Forest Ecosystem Services in Decision Making

In spite of the large number of valuation studies on ecosystem services in general, and forest ecosystem services, in particular, having been conducted, existing knowledge of the values of forest ecosystem services is often disconnected from policy options (Ferraro *et al.* 2012; Laurans *et al.* 2013; Rogers *et al.* 2015). Ferraro *et al.* (2012), after examining the ecosystem services (including forest ecosystem services) valuation studies, found that recent ecosystem services valuation studies (ESV) usually tried to estimate economic values derived from ecosystems without considering a policy context. For example, many studies attempted to measure the values of ecosystem services rather than articulating how the values would change in different policy settings or under alternative management statuses. Ferraro *et al.* (2012) argued that future research should integrate policy, relating natural resource management to valuation research. For instance, valuation research could be in the form of evaluating impacts of government conservation projects or community-based forest governance regimes on social welfare or the livelihoods of local people.

Laurans *et al.* (2013) also examined how ecosystem service valuation studies are connected with environmental policies. They conducted a comprehensive review of the use of ecosystem services economic valuation studies. They found that even though most of the ESV studies justified that the use of their valuation results for decision making is the ultimate purpose of conducting these studies, the use of the results was not adequately considered. Usually, an ESV represents the estimation of economic values and then makes a suggestion that it should be used for decision making. However, an ESV does not usually provide information on the way the results can be used in the process of decision making. Laurans *et al.* (2013), therefore, argued that the actual use of ESV results should be the core interest of future ESVs.

Rogers *et al.* (2015) presented a precise approach to analyse the use of ESV results in decision making in an Australian environment management context. They surveyed experts who have conducted non-market valuation research about the influence of their studies on policy. They also carried out a large number of interviews with decision makers in Australian environmental management organisations, about how much their decisions are influenced by the results of non-market valuation studies. The authors found that while the researchers were very optimistic regarding the impacts of their studies on decision-making processes, decision makers had rarely used the valuation results in practice. The main reason constraining the application of the ESV results was that most decision makers were not familiar with non-market valuation techniques.

In summary, it can be seen that there has been little use of ESV results in decision-making processes. There are two primary reasons for this. The first comes from the academic community. Scientists pay much attention to estimating economic values of ecosystem services with little consideration for their use in decision making. The second is derived from the decision makers due to their limited knowledge of economic valuation methods.

2.3.4 Payments for Ecosystem Services (PES): Practical Schemes of Managing Ecosystems using Economic Incentives

In general**,** PES can be defined as financial incentives to solve the problem of environmental externalities. In particular, there are distinct perspectives that are in favour of PES. In principle, PES refers to a market based mechanism. As defined by Wunder (Wunder 2005) PES is:

- “(a) a voluntary transaction where
- (b) a well-defined environmental service (or a land use likely to secure that service)
- (c) is being ‘bought’ by a (minimum one) service buyer
- (d) from a (minimum one) service provider
- (e) if and only if the service provider secures service provision (conditionality).”

The idea is that PES should “attempt to put into practice the Coase Theorem” (Engel et al. 2008). On the other hand, ecological economists argue that PES should not be a market based instrument (Farley and Costanza 2010; Muradian *et al.* 2010; Muradian *et al.* 2013). Instead, recognising the complex nature of ecosystem services and the high transaction costs of PES schemes, they propose a more adaptive, transdisciplinary approach. They try to seek a variety of payment mechanisms, both market and non-market to achieve of the goals of conserving ecosystems (Muradian *et al.* 2010; Balvanera *et al.* 2012).

Recently, the PES approach has received considerable attention in developing countries, particularly in Latin America and Southest Asia. The initiated PES programs provides incentives for the conservation of natural resources such as soil, water and forests in these regions. However, there are not many practical schemes that meet all five criteria of Wunder's definition, while most of PES schemes are more in line with the definition of Muradian et al. (2010) (Sattler et al. 2013; Schomers and Matzdorf 2013). The common characteristics of the PES programs in the developing world is that of a state-led intervention. PES policies are in fact a combination of market economic incentives and regulations (Balvanera et al. 2012; McElwee et al. 2014). Thus, institutions and governance play important roles in PES programs (Pham et al. 2013; Suhardiman et al. 2013). In summary, practical PES systems can be an effective way to manage ecosystem services.

2.4 Debates on Forest Governance

In response to the problems of unsustainable use of natural resources (including forests) over the last four decades, scientists in the field of natural resource management argue that the primary cause of these problems originates from the inefficient regimes of natural resource governance. Hardin (1968) launched the term “the tragedy of the commons” to explain that common natural resources have been overexploited. He argued that individuals become trapped in a situation when there are no restrictions to prevent them from using common resources. In order to maximise their own benefits, individuals continuously exploit common resources. This then leads to the problem of common resources being overused to the point of depletion or degradation. Consequently, individuals end up losing those benefits. Therefore, Hardin suggested intervention from the state to create an institutional regime that prevents users from overusing common resources. He argued that either centralised governments or private property could be effective institutional regimes to sustain the commons.

Since Hardin's work, theorists have continued to discuss how to manage common resources efficiently. Several scholars support Hardin's ideas. For example, Terborgh (1999) and Lovejoy (2006) argued that state-led protected areas are an effective way to preserve ecosystem services worldwide. Smith (1981) and Simmons *et al.* (1996), on the other hand, suggested that privatisation of common resources is the only way to resolve the 'tragedy of the commons' and conserve natural resources.

In practice, most of the forests around the world have been claimed as state assets (Art and Visseren-Hamdkers 2012). However, deforestation and forest degradation have continued in many regions around the globe (FAO 2010). This has led to waves of criticism directed at state-based forest governance regimes, particularly after the work of Ostrom (1990).

Ostrom (1990) argued that local communities can effectively manage natural resources. She explained that local communities have the ability to self-organise in terms of regulating individual behaviours relating to common resources. Individuals, as community members, are regulated by community rules and norms, so that when it comes to the use of the community's common resources, they not only consider options that maximise their own benefits, but also those that are in line with the rules and norms of their community (Ostrom 2005). The notion of community self-governance of natural resources has been followed by many authors (Feeny *et al.* 1990; Ascher 1995; Agrawal and Ostrom 2001; Brown *et al.* 2003; Agrawal and Chhatre 2006; Bowler *et al.* 2010). They found that community-based governance regimes, rather than the state-based governance regimes, can fix the problem of "the tragedy of the commons". Ascher (1995) showed that government forestry often leads to forest conditions becoming worse. In contrast, there have been many successful cases all over the world where community self-governance systems have effectively managed common resources (Agrawal and Chhatre 2006; Pagdee *et al.* 2006; Ellis and Porter-Bolland 2008; Somanathan *et al.* 2009; Tan *et al.* 2009; Baland *et al.* 2010; Bowler *et al.* 2010).

Regardless, it is argued that no particular governance regime is a panacea to solve the problems of the depletion and degradation of common pool resources (Dietz *et al.* 2003; Ostrom 2007; Ostrom *et al.* 2007). Instead, any governance system can be a potentially viable resource management option in a particular context (Dietz *et al.* 2003; Ostrom 2008). Regarding community-based forest governance regimes, in order for them to be effective, they must have some key attributes. For example, Pagdee *et al.* (2006) examined 31 articles that include 69 case studies based on community forestry worldwide to determine what characteristics of the community enabled this forest governance to be effective. They found that a large number of community variables significantly contributed to the success of community-based forest management. These included "tenure security, clear ownership, congruence between biophysical and socio-economic boundaries of resources, effective enforcement of rules and regulations, monitoring, sanctioning, strong leadership with capable local organisations, expectations of benefits, common interests among community members, and the local authority". Andersson *et al.* (2013) also shared several similar conclusions. Upon conducting statistical testing using data from 200 forest user groups in Bolivia, they found that local monitoring, self-organised rules and the ability to force sanctions, strongly affects the success of local communities regarding sustaining their forests.

It is worth noting that the debates on the impacts and effectiveness of forest governance are mainly concerned with forest conservation and/or the improvement of forest conditions. The success or failure of a particular forest governance regime is often assessed by how well this regime system preserves forests. This is measured through forest conditions (e.g., forest cover, forest biodiversity or forest vegetable density (Agrawal and Chhatre 2006; Pagdee *et al.* 2006). In addition, other aspects, such as social performance (e.g., social equity, livelihood of local dwellers (Ostrom 2007), poverty alleviation (Tan *et al.* 2009), etc., the forest use rights of the local community, local participants in forest management (Sikor and Tan 2011; Arts 2014) are also considered in the outcomes of forest governance arrangements.

2.5 Conclusion

By conducting a survey of the literature relating to ecosystem service frameworks, forest ecosystem services valuation and forest governance, I found that despite a substantial number of studies on valuing forest ecosystem services, their use in decision making was limited. In addition, there is lack of connection among the forest governance debates, valuation studies and ecosystem service assessment in the literature. The linkage among the three themes is necessary to increase the use of valuation and ecosystem service assessment studies in public choices. Therefore, I argue that it is essential to apply the ecosystem service framework to assess the impacts and effectiveness of alternative forest governance arrangements on the provision and economic values provided by forest ecosystems. I believe this framework can be used for decision making regarding the choice of the most efficient options among alternative forest governance regimes.

Chapter 3

THE NORTHWEST REGION OF VIETNAM: A DESCRIPTION OF NATURAL AND SOCIO-ECONOMIC CHARACTERISTICS

3.1 Introduction

This chapter briefly describes the natural and socio-economic characteristics of the Northwest region of Vietnam. It explains natural conditions regarding topography, hydrology, climate features and forest conditions. Then it explains the socio-economic characteristics, including demography and the livelihoods of local people who live in the forest areas. In order to concisely represent these features, information is presented in the text as well as through tables and figures.

3.2 Natural Characteristics

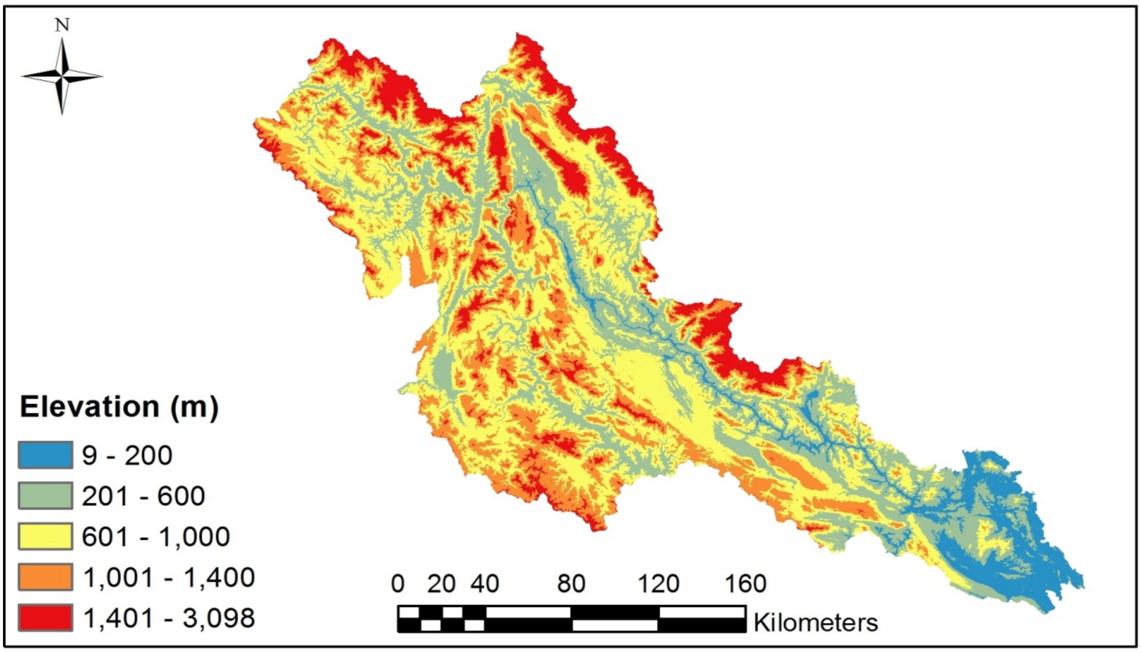
The Northwest region of Vietnam is the remote upland region. According to the Decision 1828/QĐ-BNN-TCLN of the Ministry of Agriculture and Rural Development, this region consists of four provinces: Hoa Binh, Son La, Lai Chau and Dien Bien (VNFOREST 2011). This region is considered to be like a green roof over the large lowland region consisting of many big cities, including Hanoi, the capital (please refer to Figure 1.1 in Chapter 1). The total land area of this region is about 3.7 million hectares, of which roughly 75% of the land area has been allocated for forest land by the government (VNFOREST Statistics).[7] This section provides a brief description of this region's natural characteristics regarding topography, hydrology, climate conditions and forest cover status.

The topography of this region is complex because it is characterised by high elevations in combination with steep slopes. The elevations mostly range from 200 metres to more than 1400 metres above sea level, which is higher in the north-western part and lower in the south-eastern part (Figure 3.1). Two massive high mountain ranges shape the region topography. They include the Hoang Lien Son mountain range on the north-eastern side and the Ma mountain range on the south-western side. These two mountain ranges make up the Da River watershed in the middle, which consists of lower hills and a series of limestone plateaus (e.g., Son La and Moc Chau plateaus), several valleys (e.g., Dien Bien, Lai Chau, Son La and Hoa

[7] The statistical data of VNFOREST, MARD, 1999-2013 is available online at: http://www.kiemlam.org.vn/Desktop.aspx/List/So-lieu-dien-bien-rung-hang-nam/ (accessed on 5/3/2015).

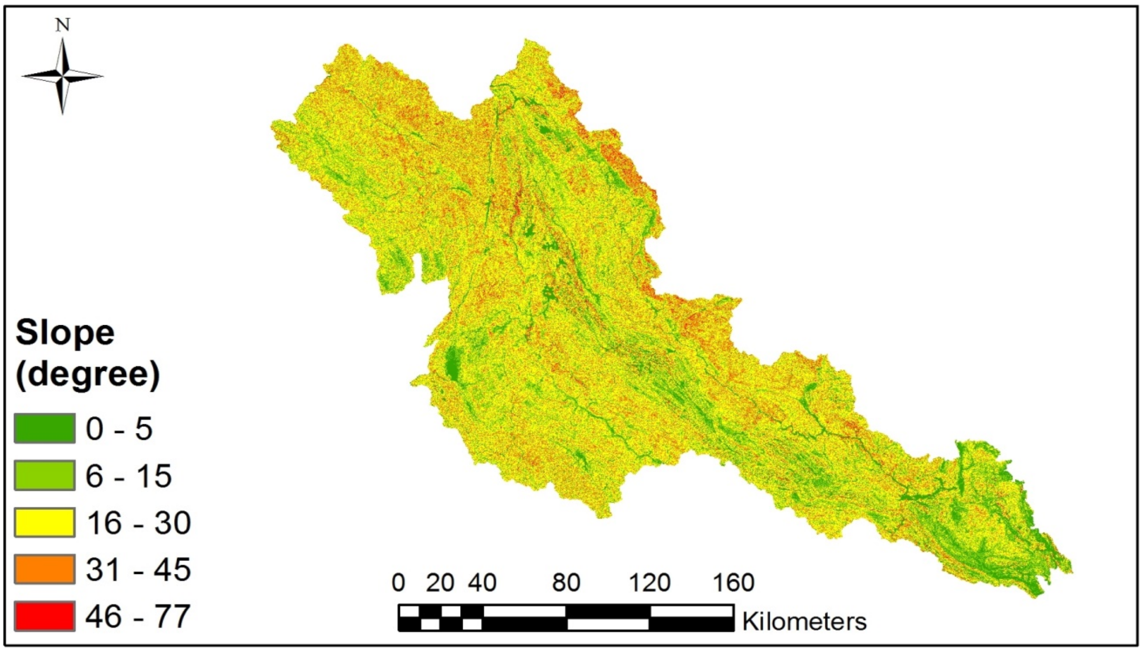
Binh valleys). With its complex and high terrain areas, this region is also characterised by steep slopes. Most of the land surface has slopes ranging from 10 to 30 degrees. Only 20% of the land area has slopes of less than 15 degrees and about the same percentage of the landscape has slopes of more than 30 degrees (Figure 3.2).

Figure 3.1 Elevation map of the Northwest region



Source: Extracted from GDEM V2, 2011.[8]

Figure 3.2 Slope map of the Northwest region



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[8] GDEM V2 is Global Digital Elevation Model Version 2 released by NASA in October 2011, available at: http://asterweb.jpl.nasa.gov/gdem.asp (accessed on 15 August 2014).

Source: Generated based on GDEM V2, 2011.

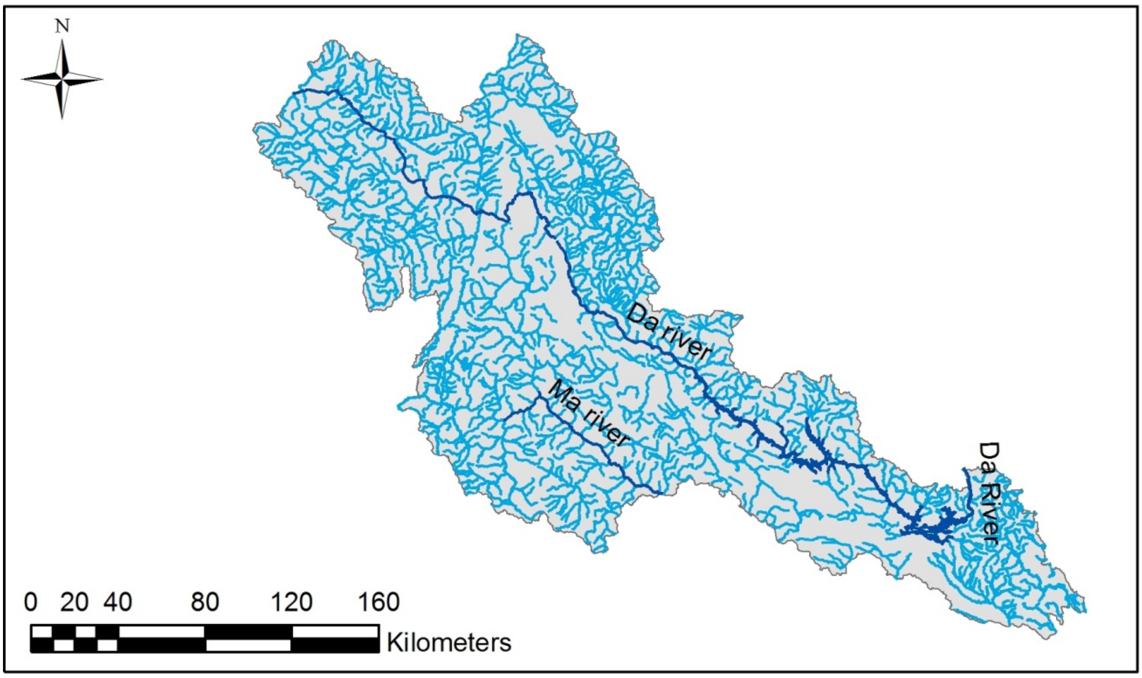
Regarding the hydrological conditions, there is a high density of rivers and streams due to the complex topography. These rivers and streams form two main river systems, including the Ma River and Da River systems, which flow from the north-west in a south-easterly direction (Figure 3.3). The water flows vary during the flooding season (from June to October) and the dry season (from November to May). Within the Da River System, there are some hydropower stations, which include many small hydropower stations, and the three largest hydropower plants, Hoa Binh (1.920 MW), Son La (2400 MW) and Lai Chau (1.200 MW). The construction of these large hydropower plants has created substantial water reservoirs, which have the total storage capacity of 20,337 million cubic metres, of which 7,000 million cubic metres has been designed for flood control, which significantly regulates the water flows of the river system in the lower streams (Table 3.1).

Table 3.1 Storage capacity of water reservoirs in the Da River watershed (million m^3)

	Hoa Binh	Son La	Lai Chau
Total designed storage capacity	9,862	9,260	1,215
Total designed storage buffer for flood control	4,000	3,000	0

Source: Vietnam Electricity, Ministry of Industry and Trade, 2015.

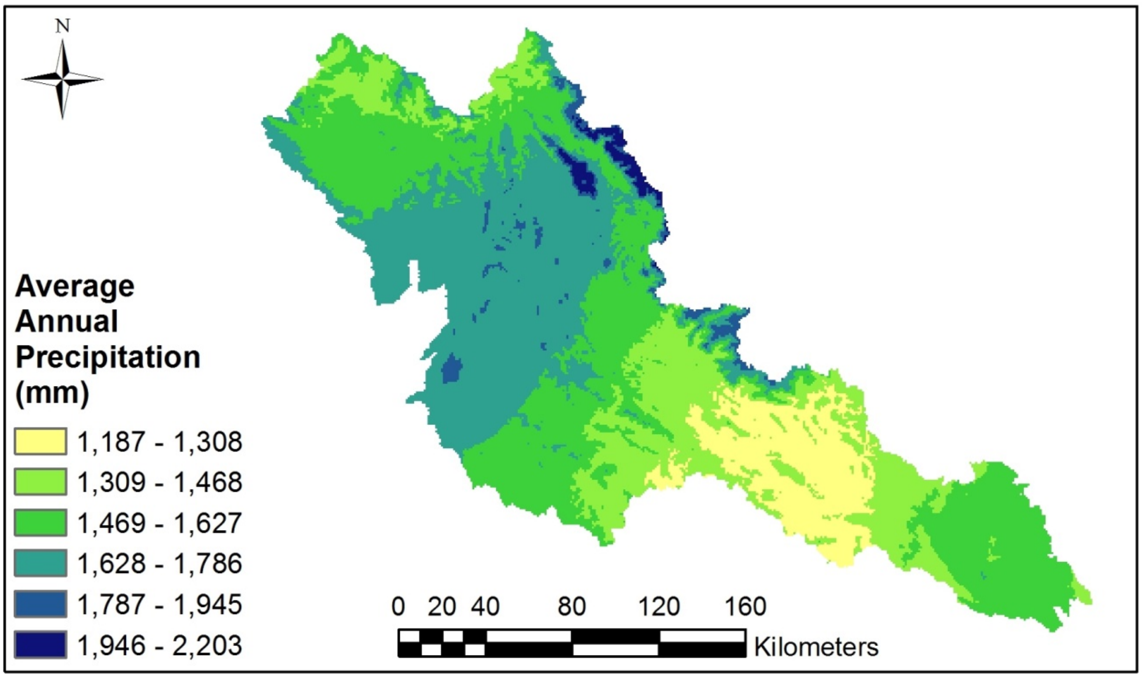
Figure 3.3 The river and stream system of the Northwest region



Source: Vietnam Ministry of Natural Resources and Environment, 2008.

This region experiences tropical monsoon climate conditions. The rainy season occurs during the months from April to September, while the dry season occurs during the months from November to March. The average annual temperature ranges from 22.5 to 23.2^{0}C. In the summer time (May to August), the average temperature is around 25^{0}C, while in the winter months (December to February), it is between 14 to 16^{0}C. The average annual precipitation is quite high, ranging from about 1200 to 2200 mm (Figure 3.4), of which about 80% occurs in the rainy season (April to September).

Figure 3.4 Map of the average annual precipitation of the Northwest region



Source: WorldClim database.[9]

The Northwest region is relatively rich in terms of forest resources. About 43.9% of the total land area was covered by forests in 2013 (Table 3.2), of which more than 90% was natural forests. In addition, there was still an enormous available land area of bareland/shrubland (about 34% of the total land area in 2010, see Figure 3.6) for the forest to expand into. In fact, this type of land use and land cover has been allocated as forest land, and most of the recent increases in forest area are mainly due to the natural regeneration of this land cover type. It is also worth noting that forests are degraded, and are in a recovery process. Since the early 2000s, forest cover has been gradually increasing (Table 3.2). Most

[9] WorldClim database provides the global monthly precipitation averaged over the period from 1950 to 2000 and is available at: http://worldclim.org (accessed on 12 September, 2014).

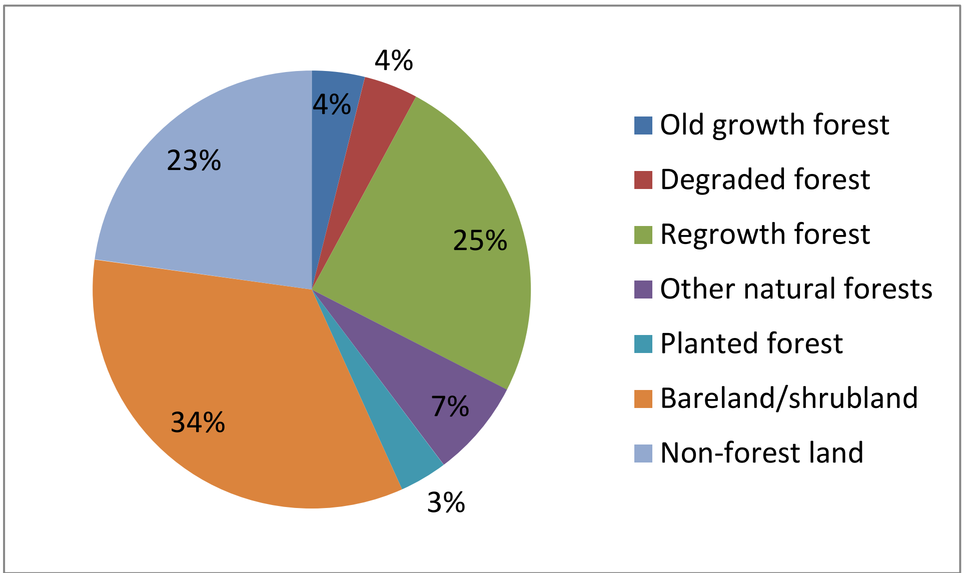
of the forest area consists of regrowth forests. Meanwhile, old growth forests are rare (about 4% of the total land area) and fragmented, and found in high mountainous areas (see Figure 3.5 and Figure 3.6) that have been either zoned as highly restricted protection forests by the government, or conserved by local communities.

Table 3.2 Forest cover changes in the Northwest region, 2002–2013

Year	Total forested area (thousand ha)	Of which		Bareland/ shrubland (thousand ha)	Forest cover percentage (%)
		Total natural forest area (thousand ha)	Total planted forest area (thousand ha)		
2002	1239.2	1157.4	81.8	1,950	34.8
2003	1349.5	1265.1	84.4	1,470	36.1
2004	1402.1	1307.7	94.4	1,407	37.6
2005	1455.0	1377.0	78.0	1,394	39.0
2006	1487.0	1399.2	87.8	1,327	39.7
2007	1492.4	1399.9	92.5	1,300	39.9
2008	1519.6	1420.5	99.1	1,286	40.7
2009	1545.0	1422.4	122.6	1,258	41.3
2010	1555.7	1429.2	126.5	1,230	41.6
2011	1572.0	1442.4	129.6	1,221	42.0
2012	1632.7	1495.5	137.2	1,205	43.9
2013	1637.7	1507.9	129.8	1,141	45.2

Source: VNForest, MARD Statistical Database, 2002-2013 (VNFOREST 2002-2013).[10]

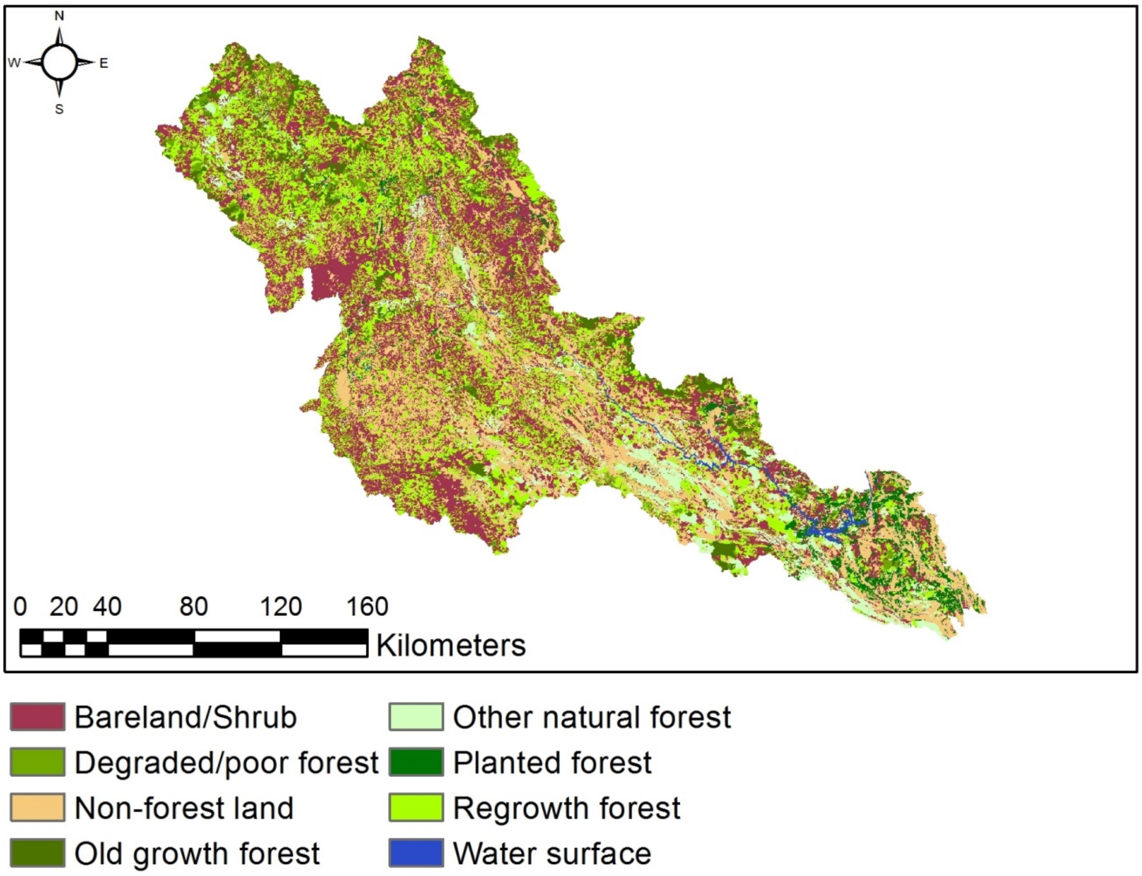
Figure 3.5 Forest cover proportion, 2010



Source: Forest Inventory and Planning Institute (FIPI), Vietnam.

[10] The statistical data of VNFOREST, MARD, 2002-2013 is available online at: http://www.kiemlam.org.vn/Desktop.aspx/List/So-lieu-dien-bien-rung-hang-nam/ (accessed on 5/03/2015).

Figure 3.6 Forest cover map of the Northwest region



Source: Forest Inventory and Planning Institute (FIPI), Vietnam.

3.3 Socio-economic Characteristics

The Northwest region is inhabited by many ethnic minority groups and is also the poorest region of Vietnam. The livelihoods of the local people depend very much on natural resources. This section provides an overview regarding the demography and livelihood sources of the local residents.

Table 3.3 shows some key population characteristics of this region. The total population of this region is relatively small, in comparison with other parts of Vietnam, at about 2.9 million people in 2012 (about 3.2% of the country's population). Most of the people live in rural areas (more than 85%) with a relatively low population density of 76 people per km^2.

Table 3.3 Population characteristics of the Northwest region and the provinces in 2012

	Northwest region	Of which:			
		Dien Bien	Lai Chau	Son La	Hoa Binh
Total pop (thous. people)	2857.2	519.3	397.5	1134.3	806.1
Rural pop (thous. people)	2448.3	441.6	345.0	976.6	685.0
% of rural population	85.7	85.0	86.8	86.1	85.0
Total land area (km^2)	37,451.8	9,569.9	9,112.3	14,174.4	4,595.2
Population density (people/km^2)	76	54	44	80	175

Source: General Statistical Office of Vietnam (2012).

The Northwest region of Vietnam is home to more than 25 ethnic groups and only 20% of the total population are Kinh people (the majority ethnic group of Vietnam). All other groups are considered to be ethnic minorities. In the Northwest region, Thai and Muong people are the two dominant ethnic groups, followed by the H'Mong people. Besides these, there are more than 20 other ethnic groups, including Dzao, Kho Mu, Tay, Xinh Mun, Khang, Giay, Phu La, Mang people, etc., (Table 3.4).

People of ethnic minority often organise themselves in villages with the same ethnic identity. The distribution of these ethnic group villages is strongly related to land elevation. The H'Mong and Dzao people live in the highest parts. Their typical production method is shifting cultivation.[11] Upland cultivation is an indispensable source of their living. It provides a variety of vegetables, such as beans, papaya, sesame and peppers; cereals, such as corn, cassava and rice; and other crops providing materials for their self-subsistence, such as cotton and indigo for clothing. Additionally, they also rely on forests for non-timber forest products (NTFPs), such as a variety of wild meat and vegetables, herbs and medicines, which are also important sources for their livelihood.

The middle elevation comprises of areas where groups that speak the Mon-Khmer language can be found (e.g., the Kho Mu, Xinh Mun, Khang and Mang people). Their main production activity is also shifting cultivation. They often apply primitive production methods: typically slash and burn, dig holes and plant. Upland rice, corn and cassava crops as

[11] Shifting cultivation, which is also known as slash and burn agriculture system, involves clearing an area of vegetated land and cultivating for several years then abandoning for naturally restoring; and shifting to a new fresh area or a previously cultivated one that has been naturally restored.

well as animal grazing and NTFPs are their main livelihood sources. Due to low farming productivity, they still have a nomadic way of life.

Table 3.4 Ethnic structure of the Northwest region in 2009

Ethnic Groups	Population (People)	Proportion (%)
All groups	2,722,080	100.0
Thai	909,902	33.4
Muong	585,057	21.5
Kinh	543,983	20.0
H'Mong	416,521	15.3
Dzao	88,242	3.2
Kho Mu	34,879	1.3
Tay	27,270	1.0
Xinh Mun	23,215	0.9
Ha Nhi	17,565	0.6
Khang	12,815	0.5
Giay	11,496	0.4
Phu La	9,809	0.4
Mang	3,635	0.1
Other Ethnic groups (more than ten minority groups)	37,691	1.4

Source: General Statistical Office of Vietnam (2010).

The Thai, Muong, Kinh and Tay people inhabit the valleys or lower lands that are often near streams and rivers. Irrigated rice production is their main source of food. The Thai and Muong people are well known for their complex irrigation systems that channel water from upland streams to their rice fields. Beside rice production, they are also involved in upland field cultivation which contributes to the diversification of farming systems, of which corn crop are very important sources income. Their other principal livelihood comes from grazing animals, handy crafts and NTFPs.

With the diversity of living conditions and cultural backgrounds, their sources of livelihoods are varied, but they share the same characteristic that all of them depend greatly on natural resources, in particular, forests. With generations of people living in forests, their accumulated knowledge regarding nature has been reflected in their culture, especially in their customary laws that play a vital role in forest conservation (CIRUM 2012).

In general, they consider that the forest is a collective property that belongs to all the people living within their village (Quynh 2009). Therefore, they believe that everyone should be responsible for protecting forests. Particularly, the Thai people are highly concerned about forest protection because they deeply understand the forest functions in terms of soil erosion protection, the prevention of sediment loads to streams and crop fields and flash floods. The customary laws of the Thai people include many of rules on the extraction of forest resources, hunting and protection of head watershed forests. They represent the classification practices of each area to serve the various needs of life, such as exploitation being strongly prohibited in the mountain forest watershed areas, and the mountains that serve their spiritual life, which are known by the generic name of 'sacred forest' . The people are absolutely not permitted to slash and burn on the mountain forests that provide bamboo, wood for building and other material needs of life. The violation of forest protection laws is strictly sanctioned (Quynh 2016).[12]

Because of the leading role of the Thai people in the development history of this region, their culture, particularly in relation to forest management, has been the dominant culture over other ethnic groups living in the Northwest region. Other ethnic groups living in the region, such as the H'Mong and Dzao people in high the mountains, as well as the Kho Mu, Khang and Xinh Mun people living in the middle elevation, also voluntarily comply with the Thai customary laws. This is not only because of the leading role of the Thai people, but also because this customary law of the forest management complies with the long-term interests of all the peoples in the region (Quynh 2016).

Most of the people living in this region are very poor. In 2012, GDP per capita of the Northwest region was only 901 USD/person (Hoa Binh Statistical Office 2013; Lai Chau Statistical Office 2013; Dien Bien Statistical Office 2014; Son La Statistical Office 2014).[13] In 2010, the poverty rate was 60.1%, which was much higher than the average incidence of

[12] Quynh, H.V. (2016). Customary laws protecting natural resources and environment of some ethnic groups in the Northwest and Central Highlands. The internet source of the electronic journal of legislative studies is available at http://www.nclp.org.vn/nha_nuoc_va_phap_luat/luat-tuc-bao-ve-tai-nguyen-thien-nhien-moi-truong-cua-mot-so-toc-nguoi-o-tay-bac-va-tay-nguyen (accessed on 15/6/2016).

[13] GDP was calculated at the prices in 2012 and converted to USD through the average exchange rate in that year.

poverty in the whole country, of about 20.7% (The World Bank Vietnam 2012).[14] Furthermore, 36.5% of the total population in 2010 lived in extreme poverty, most of them belonging to ethnic minority groups living in mountainous rural areas where natural sources, particularly forests, were their critical source of livelihood (The World Bank Vietnam 2012).[15]

3.4 Conclusion

In summary, this chapter has concisely represented the descriptions of the research site. The Northwest region of Vietnam is a remote, highly mountainous area that is relatively rich in forest resources. Its natural conditions are characterised by the complex topography of steep slopes and dense streams and rivers. It has monsoon climate conditions with high average rainfalls, particularly in the rainy season. The forest cover percentage is relatively high, but the forests are mostly degraded and regrown. Regarding socio-economic characteristics, this region is occupied by multiple ethnic minority groups, with the majority of them being quite poor and living on forest resources.

[14] A person was considered to live in poverty if his/her household's average income was less than 2.26 PPP USD in 2005, per person per day as defined by the The World Bank Vietnam (2012). Report on Poverty in Vietnam 2012. The World Bank Vietnam, Ha Noi, Vietnam.

[15] Extreme poverty is defined by the income level that is required to cover critical basic needs (e.g., food, clothes and shelter) that was equivalent to 1.50 PPP 2005 USD/person/day (The World Bank Vietnam, 2012).

Chapter 4
THE CURRENT FOREST GOVERNANCE IN VIETNAM

4.1 Introduction

This chapter aims to provide an overview of the current status of forest governance in Vietnam, and its transition. It begins with a summary of the transition of forest governance, which started when the country began the well-known renovation process (Doi moi). This was the structural reform from a centrally-planned economy to a market economy, which included the improvement of private property rights. Consequently, the current framework of forest governance is described. The framework is represented in terms of the institutional arrangements of forest management. The institutional components include administrative forest classification, forest land owners, the regulation of forest allocation and the property rights of forestland and forests.

4.2 Summary of the Forest Governance Transition and the Current Status of Forest Allocation in Vietnam

4.2.1 Forest Governance Transition

There has been a forest governance transition in Vietnam since 1987. The general direction of this transition has been to move from a strict state-controlled regime to one where governance of forest resources devolves to communities and individuals. The state used to be the sole owner of the nation's natural resources in general, and the forestry sector, in particular. After modern Vietnam was founded in 1954, the government's forest policy aim was a complete nationalisation of forestland and the establishment of State Forest Enterprises (SFEs) to manage these lands. In this governance regime, forests and forestlands were treated as national assets owned by the state. The SFEs focussed on timber exploitation while paying little attention to forest protection and the maintenance of forest stock (Meyfroidt and Lambin 2008b; McElwee 2012). In the early 1980s, there was a need for structural economic reform due to the poor economic performance of the country. The structural reform included the improvement of private property rights and a shift from a centrally-planned economy to a market economy. Followed by the success of the structural reform in the agricultural sector under Doi Moi, forest governance also began transforming in early 1990s. In this process of restructuring, SFEs were changed, and even dissolved, in some cases. The transformation of SFEs saw them turning into more private-oriented business enterprises (named state-owned companies - SOCs), whereas forest management boards (FMBs) were also instituted for the purpose of overseeing forest conservation and protection. Today, SOCs in the forestry sector are continuing to function primarily for timber exploitation, while FMBs are assigned the roles of conserving and protecting forests. The process of transformation is continuing in the Vietnamese forestry sector. In the early 1990s, the government started the Forest Land Allocation (FLA) process as a mechanism of transferring the governance of state-owned forests to local communities. The FLA was, and remains, the major vehicle for forest governance decentralisation in Vietnam. Forestland and forests are no longer solely allocated to, and managed by, the state sector. The non-state sector is represented by communities, households and individuals who are now eligible to be allocated forestland and forests (Phuc and Nghi 2014).

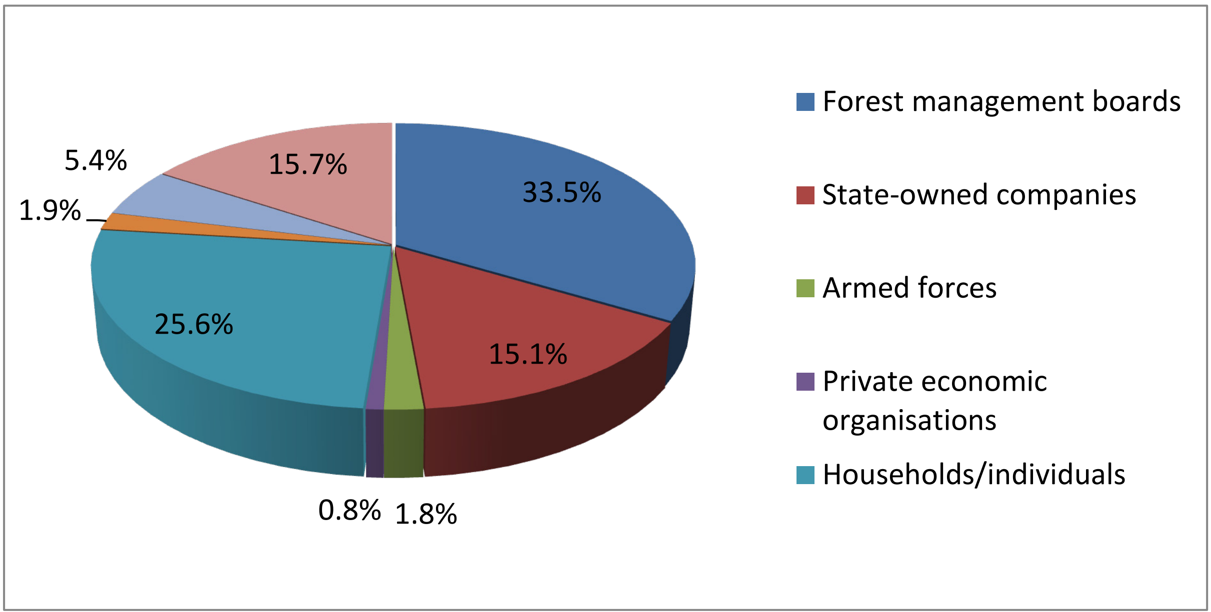
In the 1990s, there was another important shift in government policy from forest exploitation to forest conservation. In this process, the environmental services of forest ecosystems were paid more attention. Besides the direct-use values derived from forests (such as timber and non-timber products), indirect use values and non-use values (such as forest environmental services) started to be taken into account during the decision-making process (Pham *et al.* 2013). For the purposes of forest conservation, the government-zoned special-use and protected forest areas were instituted with the functions of natural preservation and environmental protection.

Since the late 2000s, the government adopted market-oriented approaches to forest ecosystem services with an aim of creating sustainable financial sources for forest protection and conservation. Several pilot PFESs projects were initiated in 2008 and scaled up to a nationwide scheme in 2012. At the end of 2012, PFESs generated a total revenue of 1,782 billion VND (about USD 85 million), accounting for about 0.8% of the national forestry budget (Pham *et al.* 2013). In addition, with international support, the government also began preparing to implement Reducing Emissions from Deforestation and forest Degradation plus ($REDD^+$) programs from 2009 (Hung *et al.* 2011). In this preparation phase, many pilot $REDD^+$ projects were implemented at the national and provincial levels.

4.2.2 The Current Status of Forest Allocation

As reported by the Ministry of Agriculture and Rural Development (MARD) at the end of 2010, the forest cover rate in Vietnam was 39.5% (VNFOREST 2011). In particular, the total area zoned for forestland was 16.25 million hectares (ha), of which some 13.4 million ha were forested areas and 2.9 million ha were available for reforestation and afforestation. Of the 13.4 million ha of forested area, those classified as special-use forests, protected forests and production forests accounted for 2 million ha, 4.8 million ha and 6.4 million ha, respectively. Most of the forested areas were natural forests. For example, 89.9% of forests classified as special-use and protected forests were natural forests, and 64.3% of the production forests were natural forests.

Figure 4.1 The status of forest allocation, in 2010



Source: Decision 1828 dated 11/08/2011 by Vietnam Administration of Forestry, MARD (VNFOREST 2011).

As represented in Figure 4.1, state organisations (state-owned companies and forest management boards) are still the dominant players in the forestry sector. The state sector has been allocated more than half of the total forested area (50.4%). The second largest portion of forestland has been allocated to households/individuals. Meanwhile, village communities have been officially allocated only 1.95% of the total forest area, accounting mostly for natural forests. In addition, about 2.1 million ha (15.7% of the forested area) are authorised to local Communal People's Committees (CPCs) to manage and they are permitted to allocate these to either village communities or households/individuals (see Table 4.1 for details).[16]

[16] Officially, allocation means that the allocation is recognised by the laws.

Table 4.1 Forest allocation by forest owners in 2010

	Forest land (ha)	Natural forest land (ha)	Planted forest land (ha)
Total area	13,388,075	10,304,816	3,083,259
Forest management boards	4,487,813	3,954,911	532,902
State-owned companies	2,018,273	1,491,391	526,882
Armed forces	247,075	195,220	51,855
Private economic organisations	110,528	26,814	83,714
Households/individuals	3,431,555	2,012,653	1,418,902
Communities	258,265	227,506	30,759
Other organisations	726,409	628,686	97,723
Communal People Committee (unallocated)	2,108,159	1,767,636	340,522

Source: Decision 1828 dated 11/08/2011 by Vietnam Administration of Forestry, MARD (VNFOREST 2011).

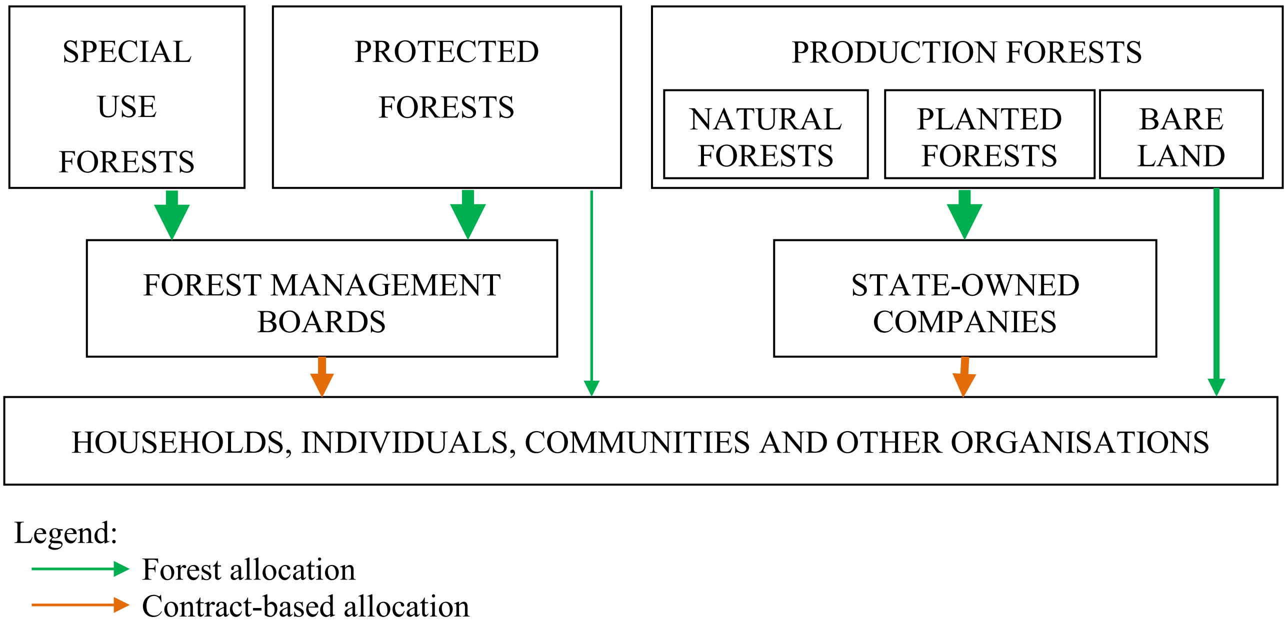
As shown in Table 4.1, forestlands allocated to state sectors are relatively high quality forests. Forests allocated to forest management boards mainly fall in protected and special-use forests. Those allocated to state-owned companies are mostly production forests, of which 73.9% fall under high timber volume natural forest. Similarly, 83.8% of the forested areas currently managed by CPCs are natural forests. In contrast, the non-state sectors, which include households/individuals and private forestry companies, have been allocated mainly production forests (70% are production forests and about 30% are protected forests), which consist of either poor/degraded natural forest for natural regeneration, or bare land for forest plantation.

Local communities have often managed natural forests that surround their villages for long periods of time (Tan *et al.* 2008a; Ngai 2009). The forests are often communal sacred/spiritual/cultural or head watershed forests of these communities. For the local communities, these type of forests are not only their sources of livelihood but attached to their beliefs and culture. They believe that these forests keep their spirits and if they want to harvest, they are required to perform rituals and ask permission before cutting trees down (CIRUM 2012). Although the state has mostly claimed forests as it's asset, these types of forests continue to be managed by the communities with or without official land title (Sikor and Tan 2011). Other types of forests assigned to communities are often remote and difficult to manage by households/individuals or state organisations.

4.3 Overview of the Current Framework of Forest Governance

Since the 1990s, the state has established a legal framework focusing on natural resource management. Vietnam's constitution states that land and natural resources are the common property of people and the state is the representative manager of forestland and forests. The two laws, including the Land Law and the Law on Forest Protection and Development, are core components of the legal framework of forest governance.[17] The Land Law confirms the state's ownership and management rights over forestland. The state implements forest land classification and forestland use planning. The state grants forestland tenure with rights and obligations to forestland users, including state organisations and non-state organisations, such as individuals, households and communities. The Law of Forest Protection and Development asserts that the state has ownership and use rights of natural forests and planted forests that are financed by the state budget. This law stipulates forest regulations and property rights of forest users for different types of forests and forest owners. The brief review of the legal framework of forest governance and what it is based on Hung *et al.* (2011) is summarised in Figure 4.2.

Figure 4.2 Summary of the current legal framework of forest land and forest allocation



Source: Hung *et al.* (2011).

[17] Vietnamese National Assembly (2003). The Law on Land 2003. in Assembly, V.N. (ed.), *No. 13-2003-QH11* Hanoi, Vietnam.

Vietnamese National Assembly (2004). The Law on Forest Protection and Development 2004. in Assembly, V.N. (ed.), *No. 29/2004/QH11*, Hanoi, Vietnam.

4.3.1 Forest Land and Forest Classification

In Vietnam, property rights relating to forests are regulated by the Land Law and the Law of Forest Protection and Development. The Land Law basically regulates land ownership and land use rights, while the Law of Forest Protection and Development focuses solely on the property rights relating to forest resources. The combination of the two laws formulates the classification of forestland and forests. According to the Land Law, forestland is classified into three categories: protected forestland, special-use forestland and production forestland. Similarly, the Law of Forest Protection and Development also classifies forests into three categories, including protected forests, special-use forests and production forests, respectively. In addition, within the production forest category, there are three subclasses: natural forests, planted forests and reproduction forests. Characteristics of these forest lands/forest types are described as follows:

– *Protected forests* have mainly environmental protection functions. Protected forests are those that primarily protect and regulate water resources, protect soil, prevent erosion, limit natural calamities, moderate the climate and ensure ecological balance and environmental security. In the uplands, the primary functions of protected forests are water regulation, consisting of flood prevention, soil erosion reduction, and retention of sediment deposits in riverbeds.
– *Special-use forests* are identified as forests with a high quality of biodiversity and a high timber volume capacity. The functions of special-use forests are to preserve nature, protect the various species in the forest ecosystems, conserve gene sources of forest flora and fauna, provide sites for research and protect historical and cultural values and scenic areas for recreation and tourism.
– *Production forests* have a main function of producing both timber and non-timber products for the market. Production forests also serve to protect the environment and preserve the ecological balance.

4.3.2 Forest Land Users and Forest Owners

Both state sectors and non-state sectors are recognised as forestland users and forest owners. According to the Land Law, forestland users consist of state organisations, households, individuals, communities, economic organisations, Vietnamese citizens residing overseas, and foreign organisations and individuals implementing investment projects in Vietnam. However, among these entities, the three major forestland users are state organisations, households/individuals and communities. At the same time, the Law on Forest Protection and Development recognises seven types of forest users: (1) state forest management boards, (2) economic organisations, (3) households and individuals, (4) army units, (5) forestry-related scientific organisations, (6) Vietnamese citizens residing overseas investing in Vietnam, and (7) foreign organisations and individuals investing in Vietnam. Although village communities are not considered to be a legal entity under Civil Law, they are also granted forest use rights and considered to be forest owners.[18] The main forest owners are also state organisations (state forest management boards and state-owned enterprises), households/individuals and communities. In addition, a special notice should be considered to the users who are not domestic stakeholders, such as Vietnamese citizens residing overseas, foreign organisations and individual. They can only lease (not be allocated) production forestland for forest plantation with the approval of the Prime Minister on the case by case basic.

4.3.3 Regulations of Forest Allocation and Management by Forest Types

For the classified forest types, the state sets different rules for forest allocation and regulation of forest use rights. For the protected forests, most areas are assigned and managed by state-mandated forest management boards. The state finances the management boards for forest protection and development. The management boards can arrange their own labour resources or subcontract to local households, individuals and village communities for implementing forest protection and development. Some fragmented protected forests are assigned or leased to households, individuals, communities and economic organisations. The forest users are allocated state budget funds for forest protection and development. In order to maintain environmental protection functions for this type of forest, timber exploitation is very limited. Only dead trees are allowed to be harvested in these forests.

When it comes to special-use forests, these are very strictly protected areas. These types of forests are mainly allocated to, and managed by, state forest management boards. There

[18] As defined in the Law of Forest Protection and Development, a village community that is eligible to be allocated forest land, has to have the same customs, practices and traditions of close community association with forests in their production, life, culture and beliefs; be capable of managing forests; and have demand and file applications for forest assignment. However, the Civil Law does not identify the village community as a legal identity.

are few areas assigned to other organisations or village communities as a precaution, in case they do not have forest management boards. To protect these areas, the state annually allocates budgets to the forest management boards for forest protection and development purposes. In addition, timber logging is strictly prohibited in these forests.

The state subclassifies production forests into natural production forests and planted production forests. These two subclasses of production forest are regulated differently. The natural production forest areas are mostly assigned to state-owned enterprises. Those with scattered natural forests (often degraded forests or bare forest land) are assigned or leased to households, individuals or private companies for protection and regeneration. With natural production forests, forest owners used to have rights to exploit timber and non-timber products. However, since 2013 the state has stopped allowing timber logging in natural production forests. With regard to planted production forests, the forest owners can exploit all kinds of forest products, but must follow the technical guidance mandated by the state. Forests must be replanted right after exploitation, or natural regrowth measures must be implemented during the course of exploitation.

4.3.4 Property Rights Relating to Forest Land and Forests

Property rights for forestland and forests are complicated. Forestland users do not have forestland ownership rights, but rather forestland tenure. The Land Law states that forest land belongs to the Vietnamese people, with the state as the representative owner. The forestland users have property rights over forest land in the form of forestland tenure. Regarding the property rights of forests, the state only recognises private ownership rights over planted production forests that are established using the forest user's own budget. For other types of forests, including special-use forests, protected forests, natural production forests and planted production forests that are supported by the state budget, the forest recipients are only granted forest use rights and obligations (not forest ownership rights) by the state. The property rights of forestland and forests for the three major forest users are described as follows:

a) Forest Land Tenure and Forest Use Rights of the State Sector

Under the Land Law, the state sector has restricted forestland tenure. Special-use forest management boards and protected forest management boards are allocated special-use forests and protected forests, respectively. The state-owned companies are allocated production forests, which are usually natural production forests and state-funded planted production forests. While being allocated, these state organisations are granted land use right certificates that guarantee stable and long-term use. However, they do not have the right to exchange, assign, donate or lease the land use right, or to mortgage, guarantee or contribute capital using the land use rights.

With regard to forest use rights generally, they are limited and differ among the different types of forests. For production forests, state-owned companies have stable and long-term use rights. They can enjoy the added value of the forests to exploit forest products regulated by law and to make contracted allocations to households, individuals and local communities for forest protection, regeneration and afforestation. They are able to lease forests to organisations, households or individuals for combined forestry/agricultural/fishery production, according to the forest management regulations. Generally, they are not allowed to convert, transfer or donate these forest use rights, or to use them as a mortgage. However, they can use the increase in the value of forest as mortgage, or provide a guarantee for their financial loans. The increase in the value of forests is defined as the value added by the forest development resulting from the investment made by the state-owned companies. In practice, measuring this increasing value is difficult, so the rights relating to this value are usually not feasible.

With *special-use forests*, the forest management boards have stable, long-term forest use rights. They obtain benefits as a result of their efforts in protecting and growing forests. The benefits obtained from forest product exploitation are limited, as they must comply with strict forest management regulations. For example, they are only permitted to exploit dead or felled trees and non-timber forest plants, except for endangered, precious and rare forest plant species. They can receive economic benefits from leasing forestland to economic organisations for commercial ecotourism under projects approved by competent state bodies. They are also funded by the state to carry out forest management, protection, new forest plantation and facilitation of natural regeneration. They are allowed to make contracted allocations to households, individuals, and local communities for forest protection, regeneration and afforestation of bare land zoned for special-use forest development.

With regard *to protected forests*, forest use rights are very similar to those that apply to special forest management boards (the owners of special-use forests). They also have stable, long-term forest use rights and they receive state budget financing for their efforts in forest protection, regeneration and afforestation. They can contract local households, individuals, and communities for these purposes. They are permitted in a limited capacity to exploit timber and non-timber products. For example, they can cut down dead and diseased trees, or trees standing in areas with a density higher than that prescribed in the forest management regulations, with the exception of endangered, precious and rare forest plant species.

b) Forest Land Tenure and Forest Use Rights of Households/Individuals

As previously mentioned, households and individuals are allocated production forests and protected forests. Production forest areas allocated to households and individuals are poor, scattered natural production forests or bare land planned for setting up production forests. Protected forest areas are rarely allocated to households and individuals. Protected forests may be allocated to households if the area is less than 5000 ha, which is not large enough for the establishment of a state forest management board. The forest land tenure and forest use rights of households and individuals for different types of forests are summarised as follows:

Production forests. When being allocated production forestland, households and individuals have the right to be granted forest land tenure in the form of land use certificates. The land use tenure is valid for 50 years with extension conditions. Households and individuals are allowed to sell, transfer, donate and lease their land use tenure. They can also mortgage the tenure, use it for a capital contribution to business purposes, and they can inherit/bequest forest land tenures.

According to the Law of Forest Protection and Development, together with the land tenure, the households and individuals are granted forest use rights. They are allowed to undertake agro-forestry and silvopastoral practices in the allocated production forests, but not in excess of 30% of the total allocated area. The forest use rights are granted in a different way for natural production forests and planted production forests. When natural production forests are allocated, their users are allowed to collect timber and non-timber products, subject to certain limitations. They can collect dead or diseased trees. They have rights to mortgage, guarantee and contribute capital on business equivalent to the value of the forest use rights. They can also inherit/bequest their forest use rights. However, they are not allowed to sell, lease or donate their forest use rights on natural production forests. The forest use rights granted to households and individuals over planted production forests are more than those granted to natural production forests. Households/individuals have the right to harvest and decide on the usage of all the products derived from forests. They are also permitted to sell, transfer, lease, inherit and donate their forest use rights and can use them as collateral for loans.

Protected forests. Forestland tenure granted to households and individuals for protected forests are almost similar to those that are granted for production forests except for the duration of the tenure. The timeframe of the land tenure is vague and is often mentioned as long-term and stable. The forest use rights over protected forests are limited. Since the main purpose of allocating protected forests to households and individuals involves forest protection, timber logging and non-timber product collection are strictly regulated. Households and individuals obtain financial compensation for their efforts in forest protection and new forest plantation. They are allowed to transfer or inherit their forest use rights, mortgage them and use them to contribute capital to business.

Contract-based allocations of forests to households and individuals. In addition to directly allocated forestland and forests by the state, households often sign contracts with state organisations for forest protection (special-use and protected forests) or reforestation of production forests. The relationship between the households and the state organisations is of a casual nature. The duration of the contracts are short-term and can be renewed annually. Households get paid for their efforts in forest protection, regeneration and afforestation. With this type of contract-based forest allocation, households do not have forest land tenure and their forest use rights are very limited and specified by the contracts. Usually, they are allowed to harvest non-timber forest products as defined by the contract. In addition, since the PFES scheme was introduced in 2011, households can obtain payment from the PFES fund (Pham *et al.* 2013). Under the current PFES scheme, individuals/households receive payment for their efforts in forest protection and development. Their payment is based on the contracted areas and on the results of their performance. If their performance meets the contract agreement, they will get 90% of the total PFES allocation paid for their contract-based protection and development areas. The remaining 10% is used for the administration fees of the state-owned organisations, which is high, due to a large number of forest owners (Pham *et al.* 2013). The state organisations use the 10% of the total funding for monitoring and appraisal of the contract performance.

c) Forest Land Tenure and Forest Use Rights of Communities

Even though communities are formerly recognised as one of the legal users of forest land and forest, forests allocated to communities are specifically defined. The three kinds of forests that can be allocated to communities include: (1) forests that have been communally managed and used efficiently for a long period of time, (2) forests which are the head watershed of the communities or other common communal interests that cannot be assigned to organisations, households or individuals, and (3) forests which are on the borders of villages, communes or districts that cannot be assigned to organisations, households or individuals and must be assigned to village communities for the sake of communal interests.

Forestland tenure and forest use rights are granted to communities with some restrictions. When it comes to forestland tenure, communities can be granted land use certificates over a long period of time. The land use rights of communities are protected by law. However, they are not allowed to convert, transfer, donate, lease, mortgage, guarantee or contribute capital to business equivalent to the value of the land tenure.

With forest use rights, communities are granted these for long-term use in accordance with an allocation period. Communities have rights to harvest forest products for communal and household member use. In natural forests, communities are allowed to collect dead and diseased trees, and to thin trees out in high density forest areas, with the exception of endangered or rare species. In plantation forests, communities are allowed to exploit supplemental trees and to thin trees out in high density forest areas. They are permitted to carry out agro-forestry, complying with forest regulations and management rules. Communities can receive technical and financial support for forest protection and management, which is provided by the state, and be compensated in case the state withdraws the allocated forests. However, communities are not allowed to divide forests among community members, or to convert transfer, donate, lease, mortgage, guarantee or contribute capital on business equivalent to the value of the forest use rights.

Besides being directly allocated by the state, communities can be allocated forestland/forests by state organisations through contracts. The state forest management boards can sign with local communities for the protection of their allocated forests (i.e. special-use and protected forests). The state-owned enterprises can subcontract their tasks of forest plantation and forest regeneration, or secure forest protection through local communities. When obtaining forests via contracts with state organisations, communities do not have forest land tenure over the contracted forest areas. Their benefits are payments, including PFES, for their efforts in forest protection, regeneration and plantation (Pham *et al.* 2013). In addition, they may also derive some benefit from forest products, but must follow the terms and conditions of the signed contracts.

In summary, the current legal framework has allowed to a shift away from a state-led forest management system to individual/community-based forest management regimes. In this decentralisation process households and communities have better access to forest land either directly from the state or indirectly through contract-based allocation from state forest companies or state forest management boards.

4.4 Conclusion

This chapter shows that although the forest governance arrangement in Vietnam is largely still a state-based regime where the state dominates the forestry sector, there has been a shift towards community-based and individual-based forest governance arrangements. The state-owned organisations still manage most of the forestland and forests. In this regime, local communities and households have more recently been involved in managing forests. The current legal framework relating to the forestry sector has improved the property rights of local communities and households and enabled the process of allocating forestland and forests to communities and households. However, local communities and households have also played subordinate roles in protecting and developing state forests through subcontracts with the state-owned organisations.

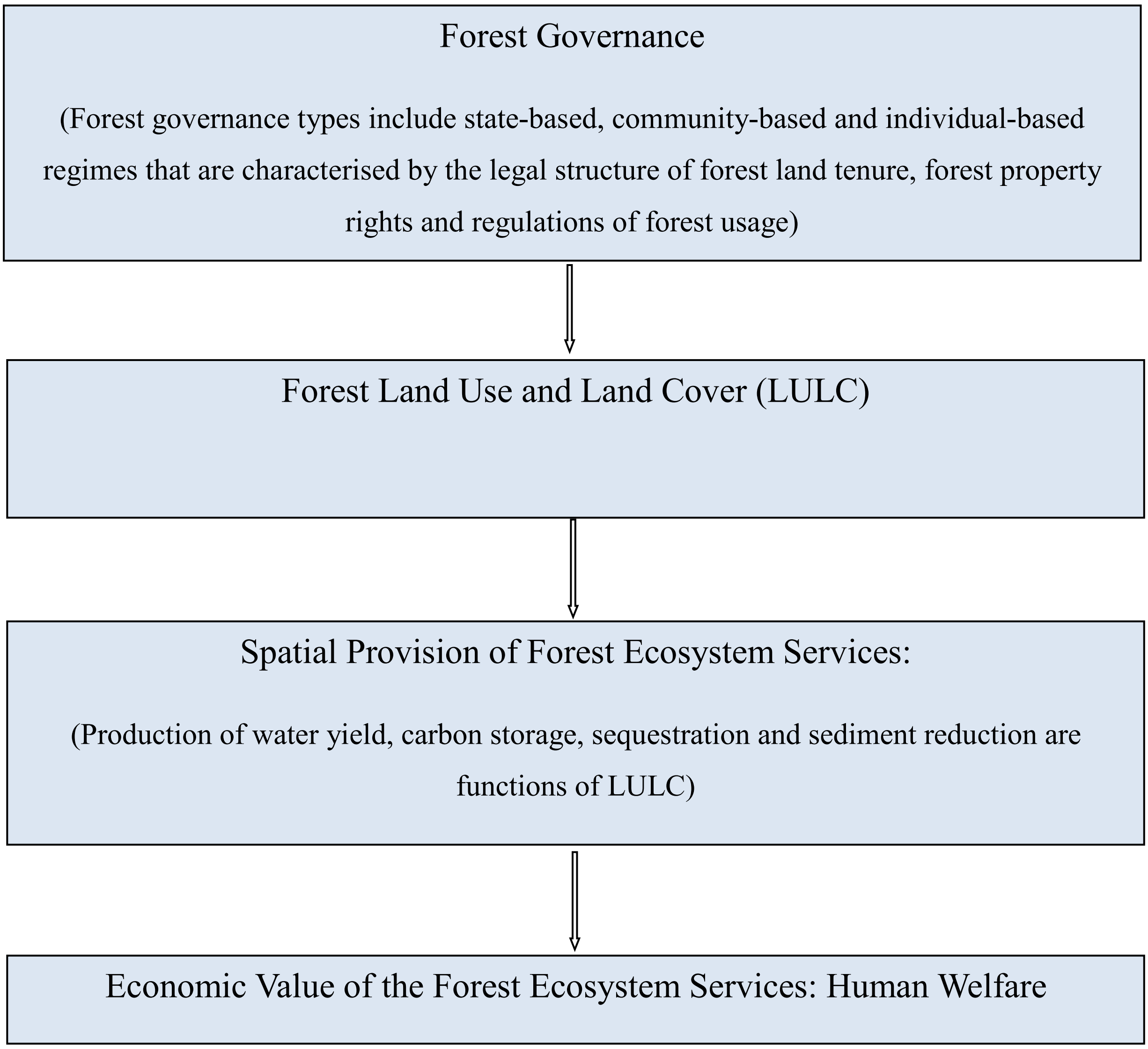
Chapter 5

CONCEPTUAL FRAMEWORK: QUANTIFYING, MAPPING AND VALUING FOREST ECOSYSTEM SERVICES TO ASSESS THE IMPACTS AND EFFECTIVENESS OF FOREST GOVERNANCE

5.1 Introduction

Based on the argument of Chapter 2 that reviews the literature relating to forest ecosystem services, forest ecosystem services valuation and forest governance, this chapter describes the conceptual framework of the study. I argue for the application of the ecosystem service framework approach to determine links between forest governance and values of forest ecosystem services. The conceptual framework has been adopted from the ecosystem services framework proposed by (MA 2005), as depicted in Figure 5.1. Human governance of natural resources affects the ecosystems as well as the provision of services by those ecosystems that, in turn, contribute benefits to human wellbeing. The framework includes four components: (1) forest governance, (2) forest land use and land cover, (3) provision of forest ecosystem services, and (4) values of the forest ecosystem services. Forest governance refers to human institutional regimes governing forests, which are characterised by the legal structure of forest land tenure, forest property rights and regulations of forest usage (Dietz *et al.* 2003). Forest land use and land cover (LULC) denotes the purposes of forest land usage and biophysical characteristics of the forest land surface, respectively (Fisher *et al.* 2005). Forest ecosystem services are defined as 'the outcomes of ecosystem functions' derived from forest ecosystems that are indirectly or directly utilised by humans. The value of forest ecosystem services can, therefore, be envisaged as a benefit that contributes to human welfare. This chapter outlines the concepts of forest governance and its effects on LULC changes. Then it presents the production functions that estimate the provision of forest ecosystem services based on LULC. Subsequently, it discusses the applicability of several valuation methods to value forest ecosystem services.

Figure 5.1 The conceptual framework of the study: Links between forest governance and economic values of forest ecosystem services



Source: Adopted from MA (2005): Conceptual framework of interactions between biodiversity, ecosystem services, human wellbeing, and drivers of change.

5.2 Forest Governance

5.2.1 The Concepts of Forest Governance

In the broadest meaning, the concept of governance can be defined as “the many ways in which public and private actors from the state, market and/or civil society govern public issues at multiple scales, autonomously or in mutual interaction” (Art and Visseren-Hamdkers 2012). This definition of governance includes various forms of governing mechanisms that appear in the literature, which consist of state-centric governance and new alternative modes of governance. State-centric governance refers to conventional state-led, top-down, or command-and-control ways of steering public affairs. The alternative governance modes vary from governing with the state (public-private partnerships) (Agrawal *et al.* 2008) to without the state (self-organisation) (Ostrom 1990).

By the broadest definition of governance, in this study, the concept of forest governance can be defined as the informal and formal institutions that regulate forest property rights, forest utilisation, benefit-sharing mechanisms of forest resources, interactions among forest stakeholders and forest protection and development. The definition use in this thesis is adopted from the idea of Ostrom (1990), relating to governing the common resources that relate the governance to the institutions that guide behaviour. This is also in line with the broad definition proposed by Giessen and Buttoud (2014) who argued "forest governance comprises: a) all formal and informal, public and private regulatory structures; i.e. institutions consisting of rules, norms, principles and decision procedures concerning forests, their utilisation and their conservation, b) the interactions between public and private actors therein and c) the effects of either on forests".

In accordance with the definition of forest governance, and following the recent discussions of environmental/forest governance (Lemos and Agrawal 2006; Art and Visseren-Hamdkers 2012; Andersson *et al.* 2013; Arts 2014; Giessen and Buttoud 2014), decentralisation of forest governance (Visseren-Hamakers and Glasbergen 2007; Agrawal *et al.* 2008; Capistrano 2008; Larson and Soto 2008; Hysing 2009) and community-based forest governance (Ostrom 2005, 2007; Capistrano 2008; Ellis and Porter-Bolland 2008; Ngai 2009), the types of forest governance are categorised into: state-based forest governance, community-based forest governance and individual-based forest governance. Attributes of these forest governance modes are described in the following section.

5.2.2 Alternative Forest Governance Regimes

a) State-based Forest Governance

State-based governance of common pool resources (CPRs) emerged in response to "the tragedy of the commons" as argued by Hardin (1968). He used common grasslands as an example for his argument. On common grasslands, each herdsman tries to maximise his benefits by raising as many cattle as possible. This is because extra animals provide the herdsman with additional benefits. However, the carrying capacity of the grasslands is undermined by the addition of each animal. This leads to a negative impact on all herdsmen. Each herdsman gains marginal benefits for each additional animal, which outweighs the negative impacts faced by him, so he continues to increase his herd. At some point in time, when the grasslands are doomed to be depleted, the tragedy of the commons, as named by Hardin, occurs. To overcome the tragedy, Hardin argued to restrict access to the CPRs through either state regulation or privatisation. In practice, state regulations (i.e. state-based governance) are commonly applied worldwide (Arts 2014).

In this study, state-based forest governance refers to formal institutions set by the state (e.g., national forest laws) concerning forests. Under the formal institutions, forest ownership often belongs to the state. Forest reserves, especially rich and highly valuable timber forests, are declared state property (Scott 1998) and are managed by state organisations (Phuc *et al.* 2013; Phuc and Nghi 2014). To a certain degree, this system also accepts private ownership of forest land and forests (Art and Visseren-Hamdkers 2012). Local people are excluded from their ancestor's forests, and "customary laws" of local communities relating to forests are ignored or suppressed (Jewitt 1995).

b) Community-based Forest Governance[19]

The concept of community-based forest governance is strongly influenced by the work of Elinor Ostrom (1990, 2005). Ostrom criticised Hardin's ideas of rationalism by showing that people do not solely make decisions based on rational reasoning (e.g., cost-benefit analysis and utility maximising) and that they are also influenced by institutions (i.e. rules, norms, beliefs and values) established by the community to which they belong. This means that individuals do not only consider options that maximise their utility, but also those that are appropriated in a given community. Using many successful cases worldwide, she argues that CPRs can be very successfully governed by local community institutions. According to Ostrom, community-based forest management institutions can be defined as the set of community's rules and norms that determine decisions about forest resources by community members. Her school of thought has been strongly influential in researching new modes of forest governance that go beyond state forestry (Arts 2014).

In this study, I follow the concepts of community-based forest management discussed in recent studies, particularly those that were conducted in Asia (including Vietnam). In these

[19] In this study, the two terms, 'community-based forest governance' and 'community-based forest management', are used interchangeably.

studies, community-based forest management (CBFM) is conceptualised as a mechanism of governing forests that integrates local institutions into forest management (Springate-Baginski *et al.* 2003; Capistrano 2008; Heilmrich 2010; Sikor and Tan 2011). The key attributes of this mechanism are that the community have legitimately long and secure rights to forest resources and tenure, and they themselves set the rules for forest utilisation, management, and conservation. Sunderlin and Huynh (2005) and Ngai (2009) identified two type of CBFMs: traditional and introduced. The traditional CBFM is a form that has existed for a long time. It is based on indigenous knowledge with regard to forest benefit-sharing and management systems of local communities. The introduced CBFM is mainly advocated by governments and non-government organisations to rebuild the traditional CBFM.

Practically, a substantial number of local communities have already managed significant areas of forest in practice with or without formal recognition by the government (Sikor and Tan 2011). In addition, with recent emerging recognitions by the Government of Vietnam, there are thousands of pilot projects of community-based forest management that have been implemented nationwide (Tan *et al.* 2008b; Tan *et al.* 2009; Wode and Huy 2009). In the upland areas of Vietnam, the majority of the population belongs to a specific ethnic group. Their institutional arrangements of forest use and management are in forms of kinship organization, regulations on ancestral domains and local rules or village customs (Bien 2001). According to (Ngai 2009), these community-based forest regimes are often consistent with the local consensus on their practical demands, usage of forest resources and compatible with the state laws/regulations.

c) Individual-based Forest Governance

Decentralisation in the forest governance system or a shift from state-based forest governance to other forms of governance has recently occurred as a result of the practical shortcomings of state-based forest governance. Decentralisation through privatisation of state forests is one of the dominant trends around the world, particularly in the Asia-Pacific region (Ferguson and Chandrasekharan 1999; Agrawal *et al.* 2008; Capistrano 2008). According to Ferguson and Chandrasekharan (1999) and Capistrano (2008), decentralisation through privatisation has been a means of gaining efficiencies through liberalising markets. Although the paths of decentralisation through privatisation vary, they generally involve the legitimisation of private property rights concerning forests, the establishment of new markets for forest property rights, forest products and market-based instruments for forest ecosystem services (e.g., payments for forest ecosystem services). The outcome of this process is that individuals (households) have rights and entitlements over land and forest resources and they receive the benefits derived from the forest products and services. This provides an effective solution for the exclusion problem of the common pool resources. In addition, these rights and entitlements for individuals have expanded and, with the establishment of new markets, some of these rights and entitlements can be exchanged (Ferguson and Chandrasekharan 1999; Capistrano 2008).

In parallel with decentralisation, there is a market-based forest governance regime that is called payments for forest ecosystem services (PFES) (McElwee 2012; Phuc 2012; Fauzi and Anna 2013; Sattler *et al.* 2013). The core idea is that forests provide ecosystem services to society, which are typically not captured in the economic accounts or in policies relating to forests. Thus, the basic functions of PFES are to account for the external, non-market values of the environmental services and introduce financial incentives for the owners of these environmental services. PFES mechanisms relate to the creation of market instruments, particularly with regard to privatisation and the hands-off approach of the state (McElwee 2012). Successful PFES projects have close links to secure land tenure. Insecure long-term tenure leads landowners (farmers) to avoid making long-term investments, while the success of PFES requires the long-term investments and commitment of providing FESs by the landowners (Grieg-Gran *et al.* 2005; Adhikari 2009).

In line with these transitions in forest governance practices, in this study, the individual-based forest governance is defined as the institutions that govern forests based on the privatisation of state forests, market-led mechanisms of forest property rights and market-based instruments for PFES. Under this governance regime, individuals have long and secure forest property rights and tenure. These property rights and tenure can be exchanged in markets. In addition, forest owners also receive benefits from providing forest ecosystem services through the PFES mechanism.

5.3 Land Use and Land Cover

5.3.1 Land Use Versus Land Cover

Land use and land cover are distinct, in spite of the two terms usually being used interchangeably (Fisher *et al.* 2005). Land cover is defined as the biophysical characteristics on the surface of the land. Land cover includes, for example, forests, scrub land, grass land, bare land, construction areas, bodies of water, etc. On the other hand, land use relates to the purposes for which humans make use of the land cover (Riebsame *et al.* 1994; Turner *et al.* 1994; Lambin *et al.* 2000; Fisher *et al.* 2005). Land use is often categorised as urban land, agricultural land, forest land, etc., because these terms reflect human purposes or intentions.

Land use and land cover are very much related. Land cover is strongly influenced by land use. Land use determines land cover change through modifications and conversions. The modification is when conditions within a cover type change; for instance, changes in the quality of forest cover. In contrast, the conversion means one type of cover is completely converted to another, such as a conversion of forests to agricultural land (Riebsame *et al.* 1994; Turner *et al.* 1994; Lambin *et al.* 2000).

5.3.2 LULC Changes in Relation to Forest Governance

The role of forest governance in determining forest conditions (e.g., LULC patterns) has been recognised as significant. Revisiting the previously-discussed modes of forest governance, it can be seen that since Hardin proposed the tragedy of the commons, governments and the academic community have recognised that common pool resources, including forests, would be degraded or lost if the governance arrangement were ineffective. Hardin suggested that either state regulations or privatisation of the common pool resources could sustain the commons in the long-term. For the purposes of sustaining forests, many governments in the 1970s and 1980s all over the world applied the ideas of Hardin that advocated state regulations on forests (Agrawal *et al.* 2008). In contrast, being strongly influenced by the thoughts of Ostrom, the new mode of forest governance, which is called community-based forest governance, has recently arisen (Art and Visseren-Hamdkers 2012). A large number of scholars show that community-based forest governance successfully sustains forests in many countries around the globe, especially in developing countries (Long and Zhou 2001; Conroy *et al.* 2002; Klooster 2002; Gibson *et al.* 2005; Pagdee *et al.* 2006; Ellis and Porter-Bolland 2008). Local communities can effectively organise and manage their resources because they have the capacity, which is developed through their accumulated knowledge of resources and cultural norms to construct and enforce rules and norms that coordinate collective actions that harmonise individual and communal rationality. The other emerging mode of forest governance based on the privatisation of state forestry, the market mechanism of forest property rights and forest ecosystem services, also achieves good outcomes for the condition of forests in some countries, such as China (Capistrano 2008). Private ownership and market mechanisms provide incentives for the rational usage of forests. This also helps to overcome the exclusion problem leading to the overuse of forests.

Although the property rights regime of forests is important, it can only weakly determine the changes to forest cover (Feeny *et al.* 1990; Agrawal *et al.* 2008). Instead, there are many other facets of forest governance that influence forest conditions. As argued by Dietz *et al.* (2003) and Ostrom *et al.* (2007), no one simple mode of governance has always been successful in every context. Any forms of ownership could succeed or fail. State forestry's practical shortcomings mainly come from the limitations in solving the exclusion problem, due to the weakness of law/policy enforcement. There are many cases where forests are claimed to be state-owned forests but, in fact, they are open-access forests to local users (Feeny *et al.* 1990; Bien 2001). Similarly, community-based forest governance is not a panacea for forest sustainability. There are many factors that influence the success of this new mode of forest governance. Among these, tenure security, clear ownership, effective local enforcement of rules and regulations, monitoring, sanctioning and strong leadership with capable local organisations are strongly related to forest sustainability (Pagdee *et al.* 2006; Chhatre and Agrawal 2008; Andersson *et al.* 2013). With regard to the privatisation regime, both costs and benefits relating to forests are the responsibility of the owner of the forests. The net benefits are reflected in the markets which, in turn, give the owner an incentive to sustain the forest. However, these incentives do not always positively ensure the sustainable use of forests, particularly when the markets fail to capture the external values of forest ecosystem services.

5.3.3 Mapping LULC Changes: Scenario Analysis

Scenarios describe events and states of the future that challenge current assumptions and broaden perspectives (Duinker and Greig 2007; Sharp *et al.* 2015). Instead of trying on predicting one certain outcome, they present the stories of alternative futures, from the expected to the unpredictable events and states (Swart *et al.* 2004; van Vliet *et al.* 2010). They must be rooted in the present, plausible (not impossible), analytically coherent and internally consistent (MA 2005; Bishop *et al.* 2007). Scenario analysis is the useful tool for future studies. They often provide two functions: 1) creativity and sparking new ideas when they enable us to think deeply and creatively about futures, 2) risk management when they enable us to test alternative plausible outcomes in the conditions of the uncertain future (Bishop *et al.* 2007; Duinker and Greig 2007).

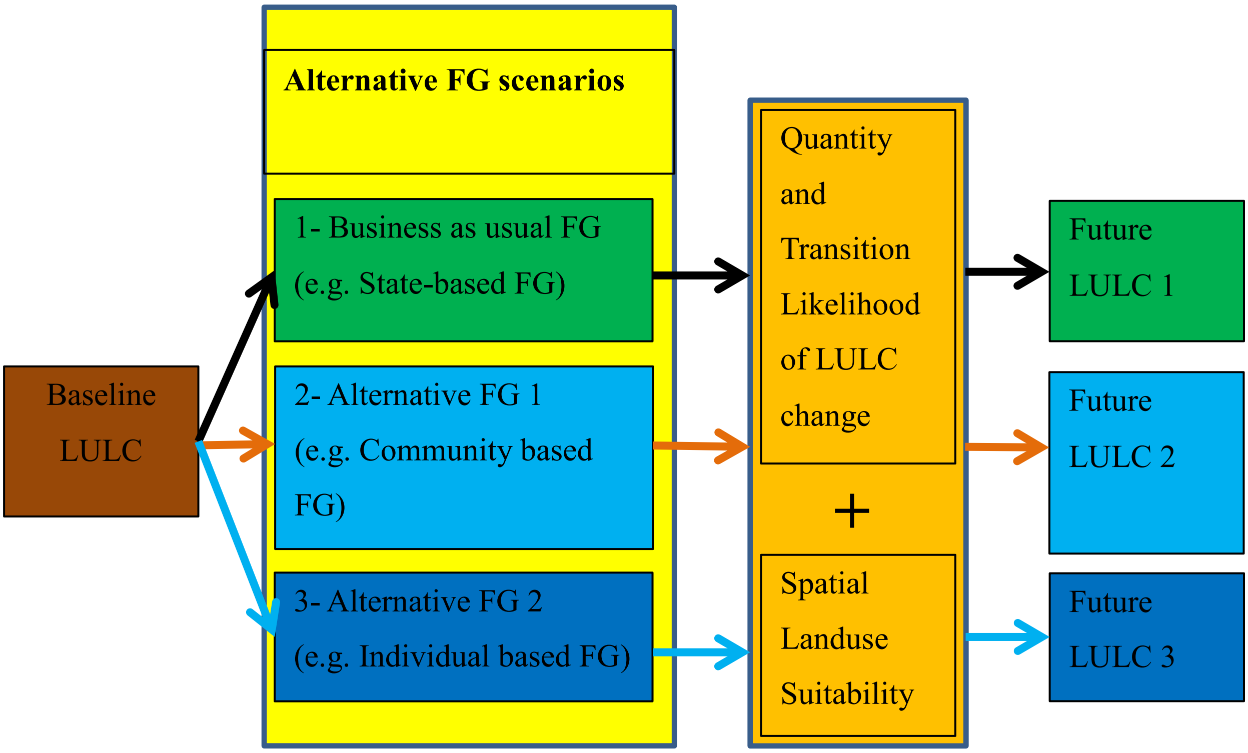
Scenario development in LULC studies often employs participatory approach (Patel *et al.* 2007; Shaw *et al.* 2009; Plieninger *et al.* 2013; Reed *et al.* 2013). According to Kok *et al.* (2015) involving stakeholders in scenario development can bring significant advantages. It makes developed scenarios are more consistent and robust. In environmental studies, scenarios can not be plausible without reference to the those who use, value and determine the environment. The four main advantages of participatory scenario development are: "1) make scenarios more relevant to stakeholder needs and priorities, 2) extend the range of scenarios developed, 3) develop more detailed and precise scenarios through the integration of local scientific knowledge, and 4) move beyond scenario development to facilitate adaptation to future change". Modern scenario development tools often combine qualitative scenario development outcomes, that are results of participatory scenario development, with quantitative modelling techniques. Quantitative models can provide detailed information that can be used as a consistency check of qualitative stories (Kok and van Vliet 2011).

In order to aid decision making in environmental management, it is essential to examine possible changes in ecosystems and evaluate these associated changes in the provision of ecosystem services under alternative scenarios. Recently, dealing with the complexity of driving forces of LULC change including cultural, social, political, and economic factors, land suitability analysis with the support of GIS application has been widely used in LULC change analysis (Carver 1991; Pereira and Duckstein 1993; Antoine *et al.* 1997; Joerin *et al.* 2001; Baja *et al.* 2002; Malczewski 2004; Greene *et al.* 2011; Nyeko 2012). The GIS-based land suitability analysis, which uses multi-criteria evaluation – a method of evaluating various criteria in decision-making, identifies and maps the most appropriate spatial explicit for future land uses according to land use demand and specify requirements, preferences This enables to incorporate expert knowledge into land suitability analysis (Malczewski 2004). The InVEST scenario generator employs this approach by combining land suitability analysis and expert knowledge. The output is a GIS map of LULC that can be used as an important input into other ecosystem service production models (Sharp *et al.* 2015).

Particularly, this scenario generator incorporates the modern GIS, multi-criteria evaluation methods, and expert opinions to map alternative future LULC. This tool applies the principle of land suitability analysis that is the changes in LULC occur in areas that are relatively more suitable. It is designed to incorporate stakeholders in terms of identifying and estimating impacts of physical and environmental factors that influence the suitability of land parcels for conversion and the likelihood of transition as well to map LULC change. There are three major components of the tool's inputs: 1) the transition likelihood, 2) the factors that influence change and 3) the quantity of anticipated change under given scenario (Sharp *et al.* 2015).

In this paper, by using the InVEST scenario generator tool, scenarios of LULC changes are developed in the way that is plausible and straightforward to decision makers. The alternative regimes of forest governance are incorporated into the scenario generation process to generate alternative LULC change scenarios (Figure 5.2). The figure shows that forest governance determines the quantity of LULC change and location preferences as well. Especially, the LULC scenarios are shaped by spatial land suitability, the quantity and transition likelihood of LULC change which are the results of a particular governance regime. The suitability of LULC is analysed based biophysical factors such as elevation, slope, soil type, cover conditions, rainfall distribution, distance to roads, distance to rivers, distance to villages. Meanwhile, the transition likelihood and the quantity of LULC change can be estimated through the historical trend of LULC change and experts' opinions. The methods will be presented in detail in section 6.2.4.

Figure 5.2: Mapping the Effects of Forest Governance (FG) on LULC Changes

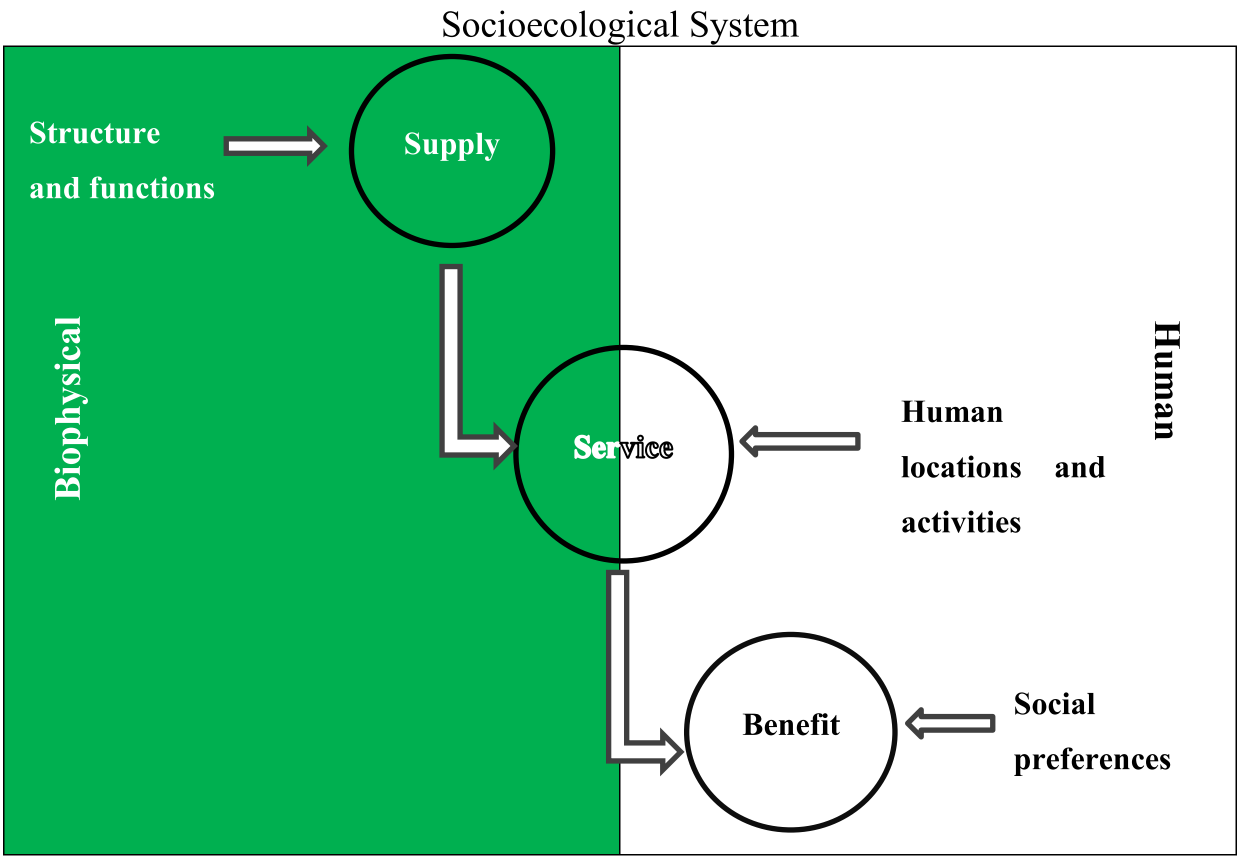


5.4 Production Models of Forest Ecosystem Services: The InVEST Models

As outlined in Chapter 2, forest ecosystem services are defined as 'outputs of ecosystem functions' that are indirectly or directly utilised by humans. By this definition, forest ecosystem services are physically measurable, and their provision can be modelled. This section presents the theoretical foundation of the three separate ecosystem service production models (i.e. InVEST model package that consists of water yield, carbon storage, sequestration and sediment delivery ratio models) that were used to estimate the quantity of the forest ecosystem services of interest. As earlier mentioned in section 1.1.2 (pages 6-7), InVEST is a tool set for inquiring how changes in the conditions of ecosystems result in changes in flows of benefits to humans. It is spatially-explicit ecosystem service modelling tools that have been developing by Natural Capital Project (http://www.naturalcapitalproject.org). At the beginning, these modelling tools required ArcGIS to run, but the new interface (from InVEST version 2.3.0) can run independently and the results can be explored by any GIS software. InVEST often employs ecological production functions that quantify the provision of ecosystem services using LULC patterns and other biophysical variables (Nelson *et al.* 2009; Tallis and Polasky 2009; Tallis 2013). InVEST links production functions to the benefits flowed to humans through a framework presenting "supply, service, and value" (Figure 5.3). "Supply" represents what an ecosystem potentially provides. "Service" takes human activities and usages of these services into account. "Value" involves social preferences (Sharp et al. 2015). InVEST can provide results either in biophysical or monetary terms, depending on the availability of data and the purpose of analysis.

InVEST toolset is grouped into three categories: 1) supporting services, 2) final services, and 3) tools to facilitate ecosystem services. Examples of ecosystem services and commodity production that InVEST can model include water quality, water yield for hydropower, sediment load, carbon storage and sequestration, pollination, recreation and tourism, timber and non-timber forest products, coastal protection, wave energy production, marine aquaculture production, fishery and so on . In this thesis, I only use three modelling tools that belong to the final service toolset: carbon storage and sequestration, sediment delivery ratio, and water yield models. These are suitable for exploring ecosystem services provided by terrestrial systems (Nelson *et al.* 2009; Tallis and Polasky 2009). The data requirements, processes, and outputs of these models are presented in Appendix 5.1. In addition, I also make use of the supporting tool of scenario generator, which was described in more detail in the previous section 5.3.3.

Figure 5.3: InVEST Models: Linking Ecological Function to Ecosystem Services and Human Benefits



Source: Tallis et al. 2012 cited in Sharp *et al.* (2015)

Particularly, based on the ecological processes, the three models used in this study employ the patterns of LULC as a main model variable in combination with other biophysical factors of ecosystems (e.g., soil, slope, etc.) and climate factors (e.g., precipitation, evapotranspiration) to estimate the provision of the forest ecosystem services of interest (Sharp *et al.* 2015). In the following subsections, firstly, the conceptual framework of the carbon storage and sequestration model is presented. Subsequently, the theoretical foundation of the sediment delivery ratio model and the water yield model are outlined.

5.4.1 Carbon Storage and Sequestration Model

a) The Conceptual Framework of the InVEST's Carbon Storage and Sequestration Model

The fundamental background of the carbon storage and sequestration measurement is the carbon cycle, as expressed by IPCC (2006). The carbon cycle is depicted in Figure 5.2. This figure shows the five carbon pools and the processes relating to carbon sequestrated by the growth of living plants (i.e. the above ground and below ground pools), fluxes out from the pools to the atmosphere, as well as all transfers among the pools. The five carbon pools include above ground biomass, below-ground biomass, soil-organic matter, dead organic matter and harvested wood products. Above ground biomass includes all parts of living plants above the ground. Below ground biomass comprises all living roots of the above ground biomass. Soil-organic matter is the organic matter stored in soil. Dead organic matter consists of both litter and dead wood. Optionally, the model includes the fifth carbon pool – harvested wood products – such as furniture, fuelwood and other timber products.

Based on the carbon cycle, IPCC (2006) proposed a comprehensive framework for estimating the changes in carbon stock in a landscape. In the landscape of interest, the changes of carbon stock equal the sum of the changes of carbon stock in all LULC classifications. The estimation is presented by the equations below:[20]

$$\Delta C = \Sigma \Delta C_{LUi} \tag{5.1}$$

where ΔC represents carbon stock change for the whole area; ΔC_{LUi} represents carbon stock change in a specific LULC classification i (e.g., forest land, crop land, shrub land, etc.); $i = 1$ to n (i.e. n is the number of LULC classifications).

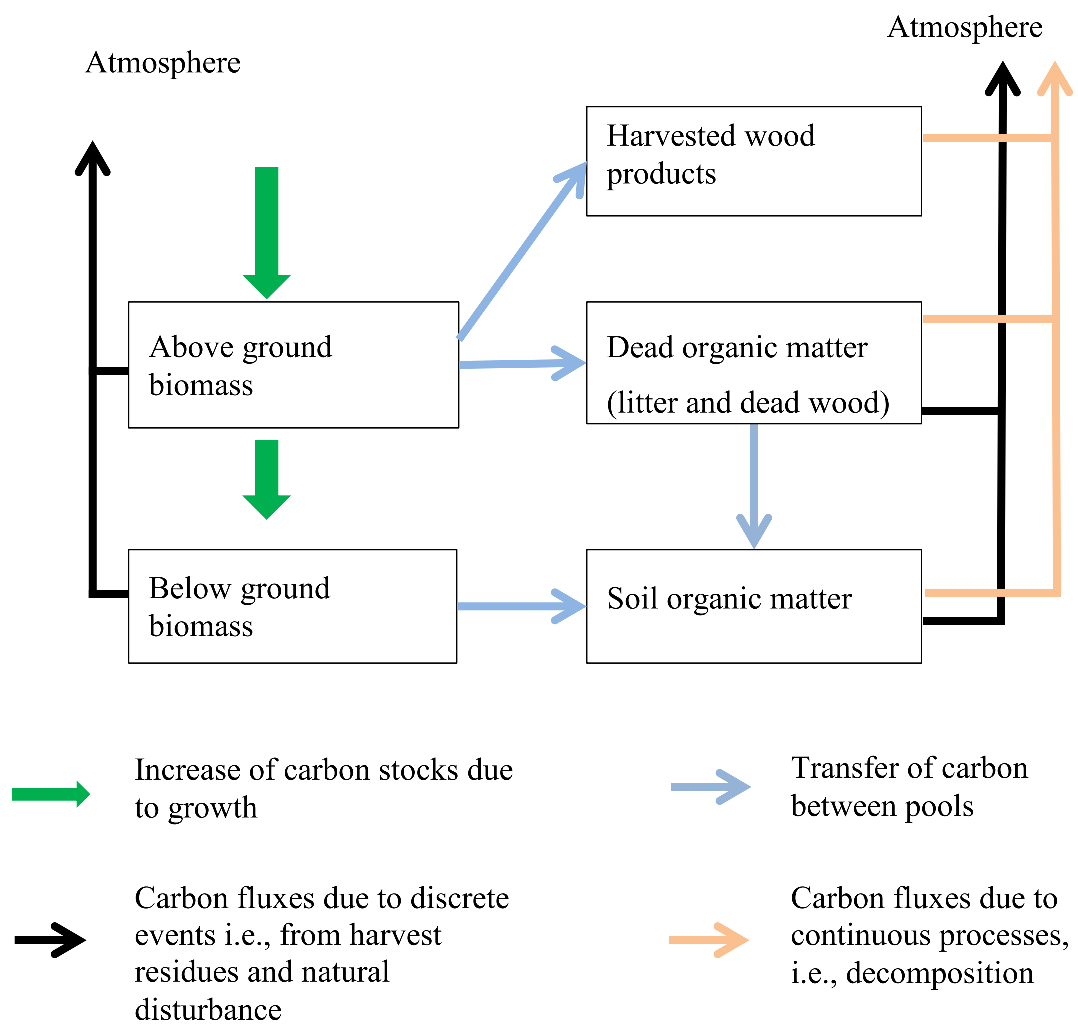
The changes in carbon stock for a specific LULC classification are aggregated from the changes in all carbon pools as given:

$$\Delta C_{LUi} = \Delta C_{ABi} + \Delta C_{BBi} + \Delta C_{SOi} + \Delta C_{DOMi} + \Delta C_{HWPi} \tag{5.2}$$

where ΔC_{LUi} represents carbon stock change in the LULC classification i; ΔC_{ABi}, ΔC_{BBi}, ΔC_{SOi}, ΔC_{DOMi}, ΔC_{HWPi} represent above ground biomass, below-ground biomass, soil-organic matter, dead organic matter, and harvested wood product change in the LULC classification *i,*respectively.

[20] The equations are adopted from IPCC (2006): Equation 2.1, Equation 2.2 and Equation 2.3.

Figure 5.4 Carbon cycle



Source: Adapted from IPCC (2006), and Sharp *et al.* (2015).

The changes in the carbon pools are associated with flows of carbon in the carbon cycle. As previously mentioned, the flows include carbon removal from, and carbon emission to, the atmosphere. Carbon removal from the atmosphere results from the growth of above ground and below ground living biomass. Meanwhile, carbon emission is a consequence of both continuous processes (e.g., decomposition) and discrete events (e.g., LULC conversion: fire, harvest, etc.). Disturbance to LULC, such as forest fires, disease or harvest can release a substantial amount of carbon into the atmosphere. In contrast, reforestation or afforestation can accumulate a large amount of carbon in plants and soil. Therefore, the LULC changes significantly impact carbon storage and carbon sequestration.

The conceptual framework of the InVEST carbon storage and sequestration model, as described by Sharp *et al.* (2015) is also founded on the carbon cycle. The model includes four major carbon pools that consist of above ground biomass, below-ground biomass, soil and dead organic matter. The fifth pool, harvested wood products, is optional.

The model simplifies the carbon cycle to estimate carbon stock in a parcel of land. It assumes that there is no gain or loss of carbon in any LULC classification over time. In addition, it also ignores the transfers between the carbon pools. This means that it only takes LULC changes (e.g.,. the changes of one LULC classification to another) and the harvest of wood products into account regarding the changes in carbon stock over time. The model uses maps of LULC types and stocks in the four carbon pools to calculate the amount of carbon stored in a parcel of land. Besides its use on wood harvest rates and harvested product degradation rates, it estimates the total biomass removed from a harvested area on the parcel of land. The net change in carbon storage over time (carbon sequestration or loss) is estimated by the difference in storage of carbon that is due to the LULC changes and the harvested wood products at different points in time.

b) Strengths and Limitations of the Model

There are two main strengths of this model. Firstly, with the critical assumptions simplifying the carbon cycle, the model requires relatively little information relating to carbon processes. This is very important because it can deal with the problem of the lack of data in carbon processes, especially in developing countries. Secondly, the model precisely shows how much and where carbon is stored, how much carbon is sequestered or lost over time and how LULC changes influence the volume of carbon stored and sequestered over time. This information is critical for effective decisions that can drive the provision of these ecosystem services.

However, by oversimplifying the carbon cycle, the model has some important limitations. For example, it assumes that the carbon stock level in all LULC types is fixed over time. This means that if any parcel of land does not experience LULC change, carbon stock in this land parcel will remain unchanged over time. This assumption ignores the fact that many areas are under the natural processes of gain and/or loss of carbon, or are recovering from past LULC. Another limitation is that the model does not consider the transfers between the carbon pools. For example, if trees (above ground and below ground biomass) die, much of the carbon stored in the above ground part of the trees becomes carbon stored in dead organic matter, and the below ground part becomes carbon stored in soil-organic matter.

5.4.2 Sedimentation Delivery Ratio Model

One of the main objectives of this study is to quantify the capacity of forests that protect reservoirs from being filled by sediment. The total quantity of soil erosion that is delivered to reservoirs needs to be estimated. Given the fact that the total amount of sediment delivered to reservoirs is a fraction of the total soil loss of upstream watersheds, this section argues that the InVEST Sedimentation Delivery Ratio Model (SDR Model) is, therefore, appropriate for this purpose. The InVEST SDR model is founded on the combination of: 1) the revised universal soil loss equation (RUSLE), which is a revision of the well-known universal soil loss equation (USLE) that was developed by Wischmeier and Smith (Wischmeier and Smith 1960; Wischmeier and Smith 1965; Wischmeier and Smith 1978), proposed by Renard *et al.* (1997), and 2) the sediment delivery ratio approach developed by Borselli *et al.* (2008), Vigiak *et al.* (2012), López-Vicente *et al.* (2013) and Cavalli *et al.* (2013). The model's outputs include the annually exported sediment that is delivered to the stream, the potential amount of eroded sediment, as well as sediment retention by vegetation and topographic features (Sharp *et al.* 2015). These two theoretical foundations are presented below.

a) The Revised Universal Soil Loss Equation (RUSLE)

As explained by Renard *et al.* (1997) in the USLE, the expected average annual soil loss is estimated as:

$$A = R \cdot K \cdot L \cdot S \cdot C \cdot P \qquad (5.3)$$

where A represents the computed spatial average soil loss and the temporal average soil loss per unit of area. A is expressed in tonnes · ha^{-1} · $year^{-1}$; R represents the rainfall runoff erodibility factor; K represents the soil erodibility factor – the soil loss rate per erosion index unit for a specified soil type; L represents slope length factor; S represents slope steepness factor; C represents the cover-management factor ; P represents the support practice factor – the ratio of soil loss with a support practice, like contouring, strip-cropping and terracing, to soil loss with straight row farming up and down the slope.

As discussed by Sharp *et al.* (2015), the InVEST SDR model, which is a spatially-explicit model, adopts the RUSLE to compute the amount of annual soil loss at the pixel level. The equation used by the model to estimate the amount of annual eroded sediment is:

$$usle_i = R_i \cdot K_i \cdot LS_i \cdot C_i \cdot P_i \tag{5.4}$$

where $usle_i$ represents the amount of soil loss at pixel i (tonne · ha^{-1} · year^{-1}); R_i represents rainfall erosivity (MJ · mm (ha · hr)$^{-1}$); K_i represents soil erodibility (tonne · ha · hr (MJ · ha · mm)$^{-1}$); LS_i represents the slope length-gradient factor; C_i represents the crop management factor; and P_i represents the support practice factor.

The LS_i factor replaces L and S in the USLE, as mentioned above. It is calculated by the method proposed by Desmet and Govers (1996), which is:

$$LS_i = S_i \frac{(A_{i-in} + D^2)^{m+1} + A_{i-in}^{m+1}}{D^{m+2} \cdot x_i^m \cdot 22.13^m} \tag{5.5}$$

where S_i represents the slope factor for grid cells calculated as the function of slope radians θ (S = 10.8 · sin (θ) + 0.03 where θ < 9%; S = 16.8 · sin (θ) – 0.50 where θ ≥ 9%); A_{i-in} represents the contributing area (m^2) at the inlet of a grid cell, which is computed from the d–infinity flow direction method; D represents the grid cell linear dimension (m); $x_i = |sin\alpha_i| + |cos\alpha_i|$ where α represents the aspect direction for grid cell i; m represents the RUSLE length exponent factor. The value of m is based on the classical USLE (m = 0.2 for slope ≤1%; m = 0.3 for 1% <slope ≤3.5%; m = 0.4 for 3.5% <slope ≤5%; m = 0.5 for 5% <slope ≤9%; m = β/(1+ β), where β = sin θ/0.0986/(3sin θ$^{0.8}$ + 0.56) for slope >9%) (Sharp *et al.* (2015). In addition, to avoid overestimation of the LS factor, as suggested by (Renard *et al.* 1997), in heterogeneous landscapes, long slope lengths are limited to a value of 333 m.

b) The Sediment Delivery Ratio Approach

After the amount of annual soil loss is computed, the amount of sediment delivered to the streams is calculated. Based on Borselli *et al.* (2008), the SDR model calculates the sediment exported to the stream from a given pixel i, E_i (tonne · ha^{-1} · yr^{-1}) as per the equation below:

$$E_i = usle_i \cdot SDR_i \tag{5.6}$$

where SDR_i represents the sediment delivery ratio for a pixel i. Then, the total watershed sediment load E (tonne · ha^{-1} · yr^{-1}) is given by:

$$E = \sum_i E_i \tag{5.7}$$

The model takes advantage of the work of Borselli *et al.* (2008), Cavalli *et al.* (2013) and Vigiak *et al.* (2012) to estimate the sediment delivery ratio. For a pixel i, the SDR ratio is calculated by the formula introduced by Vigiak *et al.* (2012), as follows:

$$SDR_i = \frac{SDR_{max}}{1+\exp(\frac{IC_0 - IC_i}{k})} \tag{5.8}$$

where SDR_{max} represents the maximum theoretical sediment delivery ratio; average value of SDR_{max} is set to 0.8 (Vigiak *et al*. 2012); IC_i represents the connectivity index developed by Borselli *et al.* (2008); IC_0 and k are calibration parameters that influence the shape of the SRD-IC relationship.

Borselli *et al*. (2008) proposed the equation to calculate the connectivity index, *IC*, as follows:

$$IC = log_{10}(\frac{D_{up}}{D_{dn}}) \tag{5.9}$$

where D_{up} represents the upslope component that is defined as:

$$D_{up} = \bar{C} \cdot \bar{S}\sqrt{A} \tag{5.10}$$

where $\bar{C}$ and $\bar{S}$ are the average *C*-factor and the average slope gradient of the upslope contributing area (m/m), respectively, and *A* denotes the upslope contributing area (m^2); and D_{dn} is the downslope component that is given by:

$$D_{dn} = \sum_i \frac{d_i}{C_i S_i} \tag{5.11}$$

where d_i represents the length of the steepest downslope flow path that goes through the i^{th} cell (m); C_i is the *C*-factor and S_i is the slope gradient of the i^{th} cell. To avoid infinite values for *IC*, Cavalli *et al.* (2013) suggested that the values of *S* be set to a minimum of 0.005 m/m, if they are less than this value, and an upper limit of 1 m/m to control very high values of *IC* on steep slopes.

c) Limitation of the InVEST SDR Model

Due to the theoretical foundation of the model being based on the RUSLE, one of its critical limitations is the RUSLE's limited scope. Although the equation is commonly accepted, it only represents sheet, rill/inter-rill erosion processes (Renard *et al.* 1997). Other sources of sediment, such as gully erosion, streambank erosion and mass erosion are not taken into account (Sharp *et al.* 2015).

In addition, because the model attempts to run with a relatively low number of parameters, outputs are very sensitive to input parameters. Therefore, small errors in estimating empirical parameters of the RUSLE (i.e. rainfall erosivity, soil erodibility, slope length-gradient factor, C-factor and P-factor) can lead to large errors in estimations. Besides, the model is very sensitive to the calibration parameters, IC_0 *and* k. Although the work of Vigiak *et al.* (2012) and Cavalli *et al.* (2013) provided instructions to set these parameters, the cautious interpretation of the model absolute values is necessary.

5.4.3 Water Yield Model

a) The Conceptual Framework of the InVEST Water Yield Model

The InVEST water yield model estimates the annual average water yield at watershed and subwatershed levels. Inputs of the model include climate and biophysical factors, and particularly LULC patterns and vegetable evapotranspiration coefficients associated with the LULC patterns. The climate factors consist of average annual precipitation and reference evapotranspiration. The biophysical factors comprise the depth-to-root restricting layer, root depth and plant-available water fraction. The output of the model is sensitive to the changes of LULC and this is because LULC changes can influence hydrologic cycles. The changes in LULC determine evapotranspiration patterns, infiltration processes and water retention. These changes also affect the amount of water and time delivery of water to reservoirs (World Commission on Dams, 2000 cited in Sharp *et al.* (2015)). The InVEST water yield model is, therefore, a useful tool to measure the effects of LULC changes on annual water yield. The theoretical foundation and mathematical expression of this model are described below.

InVEST captures and models the average annual water yield from a watershed. The model defines water yield as the amount of water provided by the watershed, which is the difference between the precipitation and the actual evapotranspiration. By this definition, water yield includes overland flows, subsurface flows and also base flows. At the pixel level, the model also assumes that water yield is derived from pixel channels to the point of interest through either one of the three flows, or all of them. At the subwatershed/watershed level, water yield is the sum of all pixels' water yield in the landscape. The mathematical expression of the model framework is presented in Equation 5.12 below.

When applied, the modified water balance equation proposed by Chapin III e*t al*. (2002) (cited in Carvalho-Santos *et al.* (2014)), is written as:

$$Y = P - AET \tag{5.12}$$

where Y represents annual water yield (overland flow, subsurface flow and base flow); AET represents the annual actual evapotranspiration; P represents the annual precipitation.

At the pixel level, according to Sharp *et al.* (2015), the model computes water yield as follows:

$$Y(x) = \left(1 - \frac{AET\ (x)}{P(x)}\right) \times P(x) \tag{5.13}$$

where $Y(x)$ represents the annual water yield on pixel *x*; $AET\ (x)$ represents the annual actual evapotranspiration on pixel *x*; $P(x)$ represents the annual precipitation on pixel *x*; $\frac{\mathrm{AET\ (x)}}{\mathrm{P(x)}}$ represents the evapotranspiration portion of the water balance.

The model differentiates vegetated LULC and non-vegetated LULC landscapes to calculate actual evapotranspiration. For non-vegetated LULC, actual evapotranspiration is measured as follows:

$$AET(x) = Min(K_c(l_x) \cdot ET_0(x), P(x)) \tag{5.14}$$

where $ET0(x)$ represents the reference evapotranspiration; $K_c(l_x)$ represents the evapotranspiration factor of the LULC that can be estimated based on the guidance provided by Allen *et al.* (1998); $P(x)$ represents the precipitation.

For the vegetated LULC, the model indirectly measures actual evapotranspiration via the evapotranspiration portion. Based on the work of Fu (1981) and Zhang *et al.* (2004), that estimated the mean of annual evapotranspiration based on Budyko's curve, this evapotranspiration portion is expressed as:[21]

$$\frac{AET\ (x)}{P(x)} = 1 + \frac{PET\ (x)}{P(x)} - \left[1 + \left(\frac{PET(x)}{P(x)}\right)^w\right]^{1/w} \tag{5.15}$$

where $PET\ (x)$ represents the potential evapotranspiration; w represents a non-physical parameter that characterises the natural climatic soil properties; $PET\ (x)$ is estimated from the vegetable evapotranspiration coefficient that is associated with the LULC and the

[21] The Budyko's curve was named after the water balance model proposed by Budyko, M.I. (1961). The heat balance of the earth's surface, *Soviet Geography* 2, 3-13.

reference evapotranspiration on pixel *x*. The vegetable evapotranspiration coefficient. $PET\ (x)$, is given as:

$$PET\ (x) =\ K_c(l_x) \cdot\ ET_0\ (x) \tag{5.16}$$

where ET_0 represents the reference evapotranspiration of pixel *x*; $K_c(l_x)$ represents the vegetable evapotranspiration coefficient of the LULC and l_x of pixel *x,* which is largely driven by the LULC patterns on the pixel (Allen *et al.* 1998); *w(x)* represents an empirical parameter that can be estimated based on the work of Donohue *et al.* (2012). *w(x)* is computed as follows:

$$w(x) = Z\ \times \frac{AWC(x)}{P(x)} + 1.25 \tag{5.17}$$

where *Z*, known as the Zhang parameter, refers to the seasonality factor. Z-parameter reflects the local precipitation pattern and other hydrogeological characteristics that can be estimated by the number of rain events per year (Donohue *et al.* (2012); $AWC(x)$ represents the volumetric (mm) available water content of plants that establish the amount of water available for use by a plant; the 1.25 is the minimum value of $AWC(x)$ (the minimum value of AWC(x) is assumed to occur when root depth is 0, as explained by Donohue *et al.* (2012).

$AWC(x)$ is determined by the soil texture and effective rooting depth, and can be estimated by the product of the plant-available water capacity (PAWC) and the minimum of restricting layer depths and root depths of vegetation:

$$AWC(x)\ = \text{Min}\ (rest.layer.depth,\ root.\ depth) \cdot PAWC \tag{5.18}$$

where *rest.layer.depth* is the soil depth at which root penetration is inhibited because of physical or chemical characteristics; *root.depth* of vegetation is considered to be the depth at which 95% of the vegetation type's root biomass occurs; *PAWC* represents the available water capacity of plants.

b) Limitations of the InVEST Water Yield Model

The main limitation of the water yield model is that it is based on annual averages, which ignore the seasonal variation of water flows. The time to delivery of water during the year determines the values of the water supply (e.g., for hydropower production). Changes in land use land cover patterns possibly affect the timing of flows as much as the water yield. However, this limitation is acceptable in this empirical research context because there are four large reservoirs in the research site that can regulate the temporal flow variations.

5.5 The Economic Values of Forest Ecosystem Services

In order to aid decision making in forest management, these values have to be stated in monetary terms because public choices are usually based on a cost and benefit analysis. This section conceptualises the values of the three forest ecosystem services of interest and argues for appropriate methods of valuing them. It begins with an outline of the theoretical foundations of the economic values of carbon storage and carbon sequestration, then sediment yield reduction, and, lastly, water supply for hydropower production. It is also worth noting that this study calculated the unit values of the three FESs then put these unit values into the InVEST models as economic parameters after the physical outputs were generated.

5.5.1 Valuing Carbon Storage and Carbon Sequestration

Generally, carbon storage and carbon sequestration provide benefits to human societies through the reduction of carbon dioxide emissions that cause climate change (Patton *et al.* 2015). In other words, social values of carbon storage and carbon sequestration can be interpreted as social damage avoided as a result of the reduction of carbon dioxide emissions to the atmosphere. The social values of carbon storage and sequestration services are estimated through the marginal damage costs of carbon dioxide emissions that contribute to climate change, which is often termed as the 'social cost of carbon' (Pearce 2003; IPCC 2007; Stern 2007; Tol 2009; Nordhaus 2011).

The 'social cost of carbon' (SCC) is an important concept in climate change economics that is defined as the value in monetary terms of the social damage caused by emitting an additional tonne of carbon, 1 tC, (in the form of carbon dioxide - CO_2) into the atmosphere at some point in time (Pearce 2003; Nordhaus 2011). The estimation of the SCC emissions is complicated and required to model essential links: emissions to the atmospheric concentration of CO_2, the atmospheric concentration of CO_2 to temperature changes and temperature changes to social damages. The last one also involves the link between temperature changes and sea level rises. The following equations, which are adapted from Pearce (2003) and Nordhaus (2011), illustrate these links in a simple way.

The first equation links carbon emission (E) to atmospheric concentrations of CO_2 (C), given as:

$$C_t = \left(1 - \frac{1}{L}\right) \cdot C_{t-1} + \beta \cdot E_t \tag{5.19}$$

where L represents the residence time of carbon in the atmosphere, and β is a factor that converts emissions into concentrations. The first term on the right hand side models the decay process that captures the rate at which carbon is removed from the atmosphere.

The second equation links the relationship between atmospheric concentrations of CO_2 (C) to radiative forcing (F), which then causes warming, as shown in Equation 5.20, which is derived from empirical measurements and climate models (Nordhaus 2011).

$$F_t = \eta\{log_2[C_t/C_{1750}]\} + F_t^{EX} \tag{5.20}$$

where F_t refers to the change in total radiative forcing of greenhouse gases from carbon emissions, F_t^{EX} represents exogenous forcing, and the first term on the right hand side is the forcing result from carbon emissions.

Radiative forcing has impacts on the global temperature. The third and the fourth equations (Equations 5.21 and 5.22), which are the essence of the complex climate change models, show the impact of radiative forcing (F) on the global warming (T), given as:

$$T_t^U = T_{t-1}^U + \frac{1}{R^U}\left[F_t - \lambda\, T_{t-1}^U - \frac{R^L}{\theta}\,(T_{t-1}^U - T_{t-1}^L)\right] \tag{5.21}$$

$$T_t^L = T_{t-1}^L + \frac{1}{R^L}\left[\frac{R^L}{\theta}\,(T_{t-1}^U - T_{t-1}^L)\right] \tag{5.22}$$

where T denotes temperature, U and L represent upper and lower ocean layers, respectively, R represents the thermal capacity of the ocean layers, θ is the transfer rate between upper and lower ocean layers, and λ is a parameter presenting the change in temperature for a given increase in radiative forcing. Equation 5.22 captures the process of radiative forcing warming up the lower ocean through warming up the atmosphere, the upper ocean.

Based on (Nordhaus 2011), the fifth equation captures the impacts of temperature, T, on annual damage, D, as follows:

$$D_t = f_1(T_t^U) + f_2(SLR_t) + f_3(C_t) \tag{5.23}$$

$$\approx \varphi_1 T_t^U + \varphi_2\,(T_t^U)^2$$

Equation 5.23 tries to capture damages to major sectors including damages from temperature change (T), damages from sea level rise (SLR), impacts of CO_2 fertilisation on agriculture production, which are functions of atmospheric concentrations of CO_2 (C). In the second line of Equation 5.23, the approximation equation shows that damages can be estimated from temperature.

Alternatively, according to Pearce (2003), the link between annual damage, *D*, and temperature, *T*, can be expressed by Equation 5.24 below.

$$D_t = k_t \left(\frac{T_t^U}{\Lambda}\right)^{\gamma} \cdot (1 + \Phi)^{t^* - t^\Lambda} \qquad (5.24)$$

where k_t is the estimated damage due to a doubling of carbon dioxide concentration (doubling in comparision with pre-industrial levels); parameter Λ is the amount of temperature increase (in °C) associated with a doubling of carbon dioxide concentrations; t^* refers to the expected year in which that doubling will occur (usually taken to be by 2050). γ links temperature and damage, indicating that damage, *D,* will rise by γ percent if the temperature rises by 1%; Φ accounts for damage that is related to speed of the change in temperature. If the doubling occurs before t^*, it will make a greater impact and this will be lower if the doubling occurs after t^*.

The final equation (Equation 5.25) models the marginal social cost of carbon, which is the economic impact (social damage) caused by one tonne of carbon emission, given as:

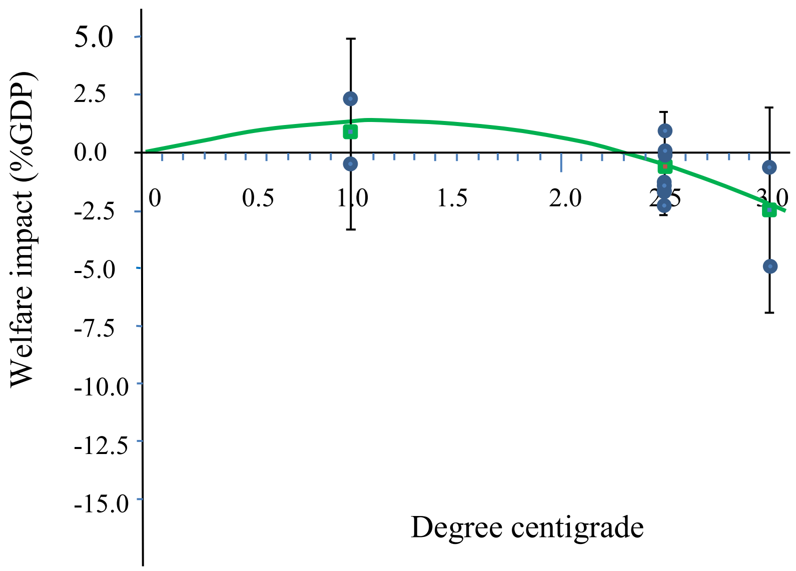
$$V = \sum_0^T \frac{\partial D_t}{\partial E_t} \cdot (1 + s)^{-t} \qquad (5.25)$$

where *V* represents the total social damage resulting from emitting one tonne of carbon; the right hand side term, $\sum_0^T \frac{\partial D_t}{\partial E_t} \cdot (1 + s)^{-t}$, is the present value of all incremental damages expected to occur in the future; *t* refers to time; *s* represents the social discount rate.

A number of studies have tried to estimate the impacts of global warming on global welfare. Based on 14 previous estimations of the global economic impacts of climate change, Tol (2009) expressed the relationship between global welfare change (%GDP) and an increase in global mean temperature (degree centigrade) in comparison with today. This relationship is depicted in Figure 5.3. As shown in the figure, the green line represents the least squares to fit the 14 estimations: $D = 2.46\ (1.25)T - 1.11\ (0.48)\ T^2$, $R^2 = 0.51$, where *D* refers to the economic impact (%GDP), and *T* refers to temperature (degree centigrade); standard deviations are in parentheses. The circular dots represent the previous estimates and the green squares are the means of the previous estimates for the specific global warming assumption. The lines are twice the sample standard deviation from the sample mean.

Regarding the marginal damage costs of carbon dioxide emissions or the SCC, many recent studies have attempted to estimate the SCC, resulting in very diverse findings. The estimated SCC ranges from 31.4 USD/tC (Nordhaus 2007) to 326 USD/tC (Stern 2007) in 2010 USD.[22] Some peer-reviewed studies estimated that the SCC is around 43 USD/tC. For example, Hope (2006) determined that the SCC is 43 USD/tC and Nordhaus (2011), using the RICE 2011 model, obtained the 2015 estimated global social cost of carbon emissions (the damages experienced all over the world) in the business-as-usual scenario, to be 44 USD/tC (in 2005 USD, which is equivalent to 49.3 USD/tC in 2010 USD). A meta-analysis of Tol (2005) concludes that the SCC are unlikely to exceed $50/tC. In an updated meta-analysis in 2008, Tol (2008), using 211 estimates of the social cost of carbon in 47 countries, argued that the social cost of carbon emissions is rarely greater than $78/tC. The estimation by Stern (2007) that SCC is 326 USD/tC is criticised as an outlier in the literature (Tol 2008). This overestimate of Stern was based on an assumption of a near zero discounting rate, which is not reasonable, as argued by Nordhaus (2007).

Figure 5.5 Global welfare impact of the climate change



Source: Adopted from Figure 1 in Tol (2009).

The above paragraphs have presented the theoretical basIc of the social values of carbon storage and sequestration as well as practical estimation results of these values through the

[22] One tonne of carbon, 1 tC, is approximately equivalent to 3.67 tonne of CO_2 emissions.

SCC approach. This study instead of estimating the SCC uses the mean of the previously estimated SCC for valuing the economic value of carbon storage and carbon sequestration.

5.5.2 Valuing Sediment Reduction: Off-Site Approach

In the presence of a dam, sediment accumulated from soil erosion causes a reduction in water storage for hydropower production and the flood control capacity of the reservoir, as well as the lifespan of the hydropower plant and of the dam (Pimentel *et al.* 1995; Phuong 2009a; Lee *et al.* 2011; Pradhan *et al.* 2011; Nguyen *et al.* 2013). Therefore, the social values of soil erosion reduction provided by forests can be measured by the social damages avoided by the reduction of sediment deposited into reservoirs or the replacement cost of sediment removal. Social damages caused by sediment deposits into reservoirs include the loss of hydropower production and the social loss resulting from a decrease in flood control capacity, due to the reduction of water storage capacity in the reservoirs. These damages can be avoided by frequent removal of the deposited sediment or maintenance of the forests (Gunatilake and Gopalakrishnan 1999; Hansen and Hellerstein 2007; Pradhan *et al.* 2011).

Adopting the analytical framework proposed by Pradhan *et al.* (2011), the benefit from the reservoir services in year t, $t = 1,2,\ldots$, are a function of water storage capacity, $f(WSC(t))$, where $WSC(t)$ represents the water storage capacity at the end of year t. $WSC(t)$ is a function of the initial water storage capacity of the reservoir, represented by $WSC(0)$, and the total volume of sediment accumulated in the reservoir at the end of year t, denoted by $SED(t)$. $SED(t)$ is a function of the sedimentation rate (SR) that is defined as the proportion of the initial water storage capacity of the reservoir that is filled with sediment each year. Based on the assumption that a certain watershed's LULC pattern is determined by a given watershed governance (i.e. the forest governance regimes in this study), this proportion is constant over time (i.e. there is a fixed amount of sediment delivered to the reservoir each year). In other words, the working assumption used in the present study is that the sedimentation rate varies among different forest governance regimes but, under a given forest governance regime, it is constant over time. The benefits derived from reservoir services decrease over time with sediment accumulation and the consequent decline in water storage capacity, so that:

$$WSC_i(t) = WSC(0) - SED_i(t) \tag{5.26}$$

where $WSC_i(t)$ represents the water storage capacity in year t under the forest governance regime i; $SED_i(t)$ represents the total volume of sediment accumulated in the reservoir at the end of year t under the certain LULC pattern resulted from forest governance regime i (state-based forest governance, community-based forest governance or individual-based forest governance).

Given the assumption of the constant rate of sedimentation *(SR)*, Equation 5.26 can be rewritten as:

$$WSC_i(t) = WSC(0) - WSC(0) \cdot SR_i \cdot t \qquad (5.27)$$

By frequently removing accumulated sediment, the storage capacity of the reservoir can be preserved. In this study, I assume that the amount of sediment accumulated each year is completely removed. Therefore, I apply the replacement-cost method for measuring the values of the sediment reduction services. This means that the benefit from sediment reduction, as the service of a forested watershed, is treated equivalently to the costs of sediment removal from the reservoir.

5.5.3 Valuing Water Supply for Hydropower Generation

Water yield or water quantity derived from forest ecosystems benefits human societies in many different ways (Young and Loomis 2014). For example, values of water yield are reflected by fresh water for municipal usage, fresh water for agricultural irrigation, fisheries and industrial usage (e.g., hydropower generation) (Guo *et al.* 2000; De Groot *et al.* 2002; Torcellini *et al.* 2003; Pattanayak 2004; MA 2005; Postel and Thompson 2005; Young 2005; MacLeod *et al.* 2006; Núñez *et al.* 2006; Moran and Dann 2008; Thanh 2008; Biao *et al.* 2010). In this study, I focus on estimating the benefits of water supply for hydropower generation. This is due to the limitation of available data for valuing other benefits of water yield. In addition, in the research region, there are many hydropower stations, including the three largest in Vietnam, which produce substantial benefits to the whole economy of Vietnam.[23] It is worth noting that this estimation is an undervaluation with regard to the actual social values of water yield provided by forest ecosystems. In this research region, there are other values of water provision that are considered, such as the water provision for agricultural irrigation, municipal use, etc.

This study applies the economic valuation of water yield for hydropower generation. According to (Young 2005), the values of water yield for hydropower generation can be

[23] The aggregated installed capacity of the three largest hydropower stations located in the research region are 5520 MW.

measured through the marginal product resulting from one unit of water volume (i.e. one cubic metre of water) used for electricity generation. According to Young, the marginal product of water can be measured by the residual valuation method. The residual technique determines the part of the total value of the production of electricity that is assigned for water use for the production of electricity. The combination of main inputs, including capital investment are: costs of operating, maintenance and repair, and water used to generate hydro-electricity. The residual method is applied in valuing water supply for hydropower generation and can be expressed by the following equations:

$$TVP = \sum P_i Q_i + R_w Q_w \tag{5.28}$$

$$R_w = \frac{TVP - \sum P_i Q_i}{Q_w} \tag{5.29}$$

where TVP represents the total value of the energy produced; $P_i Q_i$ represents the total costs of non-water inputs to production, including: 1) investment costs, 2) operation and maintenance costs; R_w represents the value of water (its marginal product); and Q_w represents the cubic metres of water used in electricity generation.

In summary, this section presents the different methods of valuing the three forest ecosystem services. It is worth noting that although these valuation methods are the most appropriate for measuring the economic values of carbon storage and sequestration, sediment yield reduction and water yield in the research region, the estimated values of these forest ecosystem services are approximations of the actual values.

5.6 Conclusion

In conclusion, this chapter has provided a comprehensive conceptual framework for assessing the effects of forest governance on the values of ecosystem services derived from forests. Firstly, it conceptualised forest governance as human institutions that govern forests. These include state-based forest governance, community-based forest governance and individual-based forest governance. In order to link LULC to the provision of forest ecosystem services, the InVEST models were introduced. The theoretical foundations of the InVEST model (i.e. water yield, carbon storage and sequestration, and sediment delivery ratio models) were explained, and the limitations of the models were discussed. Finally, the values of forest ecosystem services and valuation methods were conceptualised. The values of forest ecosystem services, that are reflected by their contribution benefits to human welfare, are captured by appropriate valuation methods. In brief, this theoretical framework is an elaboration of the ecosystem service framework proposed by MA (2005). Although there are some limitations, this is a useful and practical tool for assessing the impacts of forest governance systems on LULC changes, the provision of the forest ecosystem services and the associated economic values of forest ecosystem services as well.

Chapter 6
RESEARCH METHODS AND DATA

6.1 Introduction

Following the conceptual framework established in Chapter 5, this chapter presents a combination of different datasets and methods for determining the effects of forest governance regimes on the forest land use and land cover (LULC) changes that result in the changes in the provision and values of forest ecosystem services. The chapter begins with describing the data, data collection, data processing and analysis that are required to research the effects of the forest governance arrangement on the provision and the economic values of forest ecosystem services. In accordance with the four components of the research framework, there are four data sets, including: a data set for developing hypothetical forest governance (FG) scenarios, a data set for mapping forest land use and land cover, a data set for mapping and quantifying the provision of forest ecosystem services, and a data set for valuing the forest ecosystem services. In order to gather these sets of data, three data collection techniques, including desktop research, unstructured expert interviews and structured expert interviews were carried out. Once these data were gathered, they were pre-processed to meet the technical requirements of the InVEST models, and the other tool (i.e. the scenario generator that is attached to the InVEST package) that was used to map the future patterns of forest LULC. For the purpose of analysis, this study applied both qualitative and quantitative methods. Qualitative methods were used to synthesise published documents regarding the legal framework of forest governance regimes and forest governance practices and to summarise expert perceptions of forest LULC changes under the proposed FG scenarios. The results of the qualitative research were the identification of feasible hypothetical alternative scenarios for forest governance arrangements and the changes of forest LULC under the alternative FG scenarios. On the other hand, the quantitative ecological production models (the InVEST models), were used to quantify the production of forest ecosystem services. Furthermore, this chapter also presents valuation methods used to evaluate these forest ecosystem services.

6.2 Data Sets, Data Sources, Data Collection Techniques and Data Pre-processing

6.2.1 Data Sets and Data Sources

As described in the theoretical framework in Chapter 5, in order to examine the effects of forest governance on the provision and values of forest ecosystem services, this study first attempts to develop feasible future forest governance scenarios and then seeks to determine how forest LULC would change under these proposed scenarios. The changes of the provision of the three forest ecosystem services (i.e. carbon storage and sequestration, sediment reduction, and water yield) are then estimated. Finally, this study evaluates the economic values of the forest ecosystem services under each FG scenario, as well as analyses how the values change over time and how they would differ among different FG scenarios.

For the purposes of this research, it was necessary to collect four separate data sets. They included a data set for developing FG scenarios, a data set for mapping forest LULC change under the generated FG scenarios, a data set for mapping and quantifying the provision of the three forest ecosystem services and a data set for valuing these forest ecosystem services.[24] The descriptions of these data sets and their sources are given below:

a) The Data Set for Developing FG Scenarios

Following the concept of forest governance that was defined as 'the informal and formal institutions that regulate forest property rights, forest utilisation, benefit-sharing mechanisms of forest resources, interactions between forest stakeholders and forest protection and development', the information relating to the legal framework that shapes forest governance arrangements and the forest governance practices was collected. In relation to the legal framework, the formal rules that regulate forest property rights, benefit-sharing mechanisms, forest utilisation regulations, forest protection and development planning, policies for payment of forest ecosystem services were gathered and synthesised. The information was mainly obtained through a desktop research of legal documents and government policies relating to the forestry sector. Also, expert opinions on future feasible forest governance arrangements were obtained. These were gathered via various unstructured interviews during the author's field trip in Vietnam (e.g., provincial experts in Hoa Binh, Son La, Lai Chau, and Dien Bien provinces and national experts in Hanoi) in 2015.

[24] The summary of data sets and data sources are presented in Appendix 6.1.

b) The Data Set for Mapping Forest LULC

In order to map the changes of forest LULC patterns under each generated forest governance regime, I used the scenario generator tool provided by the Natural Capital Project that is attached to the InVEST model.[25] This tool requires the following information: quantity of forest LULC changes, priority of forest LULC, transition likelihood of forest LULC and factors that determine the land suitability of forest LULC. The quantity of forest LULC changes was estimated from the historical trend of forest LULC changes, while other information was acquired from expert interviews. The original information gathered was pre-processed before the data it was used for the scenario generator tool. Data pre-processing will be explained in detail in section 6.2.4.

c) The Data Set for Mapping and Quantifying Forest Ecosystem Services: Carbon Storage and Sequestration, Water Yield and Sediment Reduction Services

The InVEST model was used to map and quantify the provision of the three forest ecosystem services (i.e. carbon storage and sequestration, sediment delivery and water yield). The specific input data needed for modelling each ecosystem service are described in following paragraphs.

Inputs for all models. All three models require GIS maps of baseline LULC, future LULC, DEM (digital elevation model), watersheds and sub-watersheds.[26] The GIS maps of baseline LULC in 2010 were collected from the Vietnamese Forest Inventory and Planning Institute (FIPI). The future LULC maps were projected by the author for each FG scenario and the mapping procedure is explained in section 6.3.2. The DEM was extracted from ASTER GDEM2.[27] The maps of the watershed and sub-watersheds were delineated from the DEM by using ArcGIS hydrology tools. The watershed and sub-watersheds were defined by the outlet points that coincide with the locations of the hydropower stations. In addition to these inputs, each particular model requires additional specific data.

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[25] This tool can be found at the website of the Natural Capital Project: http://www.naturalcapitalproject.org/invest/ (accessed on 3/4/2015).

[26] GIS is Global Information System.

[27] ASTER GDEM2 is the Advanced Spaceborne Thermal Emission and Reflection Radiometer Global Digital Elevation Model developed jointly by the U.S. National Aeronautics and Space Administration (NASA) and Japan's Ministry of Economy, Trade, and Industry (METI), which is available at http://gdex.cr.usgs.gov/gdex/ (accessed on 27/08/2014).

Carbon storage and sequestration model. This model requires information on carbon storage in four major carbon pools, including carbon stored in: soil, the aboveground biomass, the underground biomass and the dead organic matter. Information on carbon stored in the aboveground biomass and in the underground biomass was gathered from a study by Phuong (2009a). This study estimated forest carbon stocks for various forest LULC types in the north of Vietnam. Some LULC types which were not estimated by Phuong (2009) were extracted from the database provided by Saatchi *et al.* (2011) that mapped the carbon stocks of forests in tropical regions, including Vietnam.[28] The information on carbon stored in soil was extracted from the work of Hiederer and Köchy (2012), which represents the global stocks of soil-organic carbon in the topsoil (0-30 cm) and in subsoil (30-100 cm) layers.[29] Information regarding the fourth pool, carbon stored in dead organic matter was assigned to be 10% of the biomass pools for natural forests, as concluded by Phuong (2009) and only 3.5% for planted forests, as summarised by (Que *et al.* 2006; Khoa and Hai 2008). The average carbon stored in each pool for each LULC type was then estimated and is shown in Table 6.1.

Table 6.1 Average carbon stored in different carbon pools for different forest LULC (Tonne/ha)

Forest LULC	**C_ABOVE**	**C_BELOW**	**C_SOIL**	**C_DEAD**
Old growth forest	94.21 (8.92)	25.05 (2.37)	72.42 (7.37)	13.14 (1.24)
Degraded forest	69.89 (4.54)	18.58 (1.21)	72 (10.46)	9.71 (0.63)
Regrowth forest	58.58 (5.98)	15.57 (1.59)	72.94 (9.65)	8.21 (0.84)
Bamboo forest	43.35 (19.51)	11.70 (5.27)	70.56 (10.55)	6.07 (2.73)
Mixed forest	50.97 (16.31)	13.76 (4.40)	74.27 (6.81)	7.14 (2.28)
Rocky mountain forest	54.30 (21.72)	14.66 (5.86)	62.36 (11.48)	7.60 (3.04)
Planted forest	23.61 (11.33)	6.38 (3.06)	65.78 (10.05)	3.31 (1.59)
Shrub/grass (bare land)	14.65 (5.86)	3.95 (1.58)	73.17 (9.22)	2.05 (0.82)

Note: Standard deviation in parentheses.

Sources: Derived from Phuong (2009a) and the database attached to Saatchi *et al.* (2011) and Hiederer and Köchy (2012).

[28] The database of forest carbon stocks in Asia is available online at http://carbon.jpl.nasa.gov/data/data_asia.cfm (accessed on 3/10/2014).

[29] The database of the global soil organic carbon stocks is available online at http://esdac.jrc.ec.europa.eu/content/global-soil-organic-carbon-estimates (accessed on 26/09/2014).

Sediment Delivery Ratio Model. This model requires GIS maps including soil erodibility (K-factor) and rainfall erosivity (R-factor). These factors were estimated from the data that are globally available. I estimated the K-factor from the soil structure provided by the Harmonised World Soil Database (HWSD V1.2), with guidance from the OMAFRA factsheet, Order No.12-051 (Hilborn 2012) and Renard *et al.* (1997) for the conversion from US customary units into the International System (IS).[30] The R-factor was estimated through monthly rainfall data published by the WorldClim database. According to Renard and Freimund (1994), Bagherzadeh (2014), Prasannakumar *et al.* (2012) and Fu *et al.* (2011), in tropical regions, R-factor can be estimated by monthly rainfall using the formula developed by Wischmeier and Smith (1978) as follows:

$$\mathrm{R} = \sum_{i=1}^{12} 1.735 \cdot 10^{\left(1.5\log\left(\frac{Pi^2}{P}\right)-0.08188\right)} \tag{6.1}$$

where R represents the rainfall erositivity factor (MJ mm ha^{-1} h^{-1} y^{-1}), P_i represents monthly rainfall (mm) and P represents annual rainfall (mm).

In addition, this model requires information on the cover-management factor (C-factor) and field support practices (P-factor). The C-factor was estimated from the normalised difference vegetation index (NDVI). According to Van der Knijff *et al.* (2000) C-factor can be calculated as:

$$C_{factor} = exp(-\alpha \frac{NDVI}{\beta - NDVI}) \tag{6.2}$$

where α and β are parameters that determine the shape of the NDVI-C curve. The NDVI was estimated from LandSat 5TM images captured in 2009 using EDRAS Imagine 2013 by the author. The procedure of the NDVI calculation is based on Chander *et al.* (2009). On the other hand, the P-factor - a soil loss ratio determined by a particular support practice (Renard *et al.* 1997), was set to the maximum value of 1, except for the water surface. This is because, in Vietnam, particularly in the research area, the value for P has not been measured for any LULC. The maps of these factors are given in Appendix 6.2. The estimated values of C-factor and P-factor for various LULC types in the research area are shown in Table 6.2.

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[30] HWSD V1.2 is the Harmonized World Soil Database version 1.2 (HWSD V1.2) developed and issued in 2012 by FAO, Rome, Italy and IIASA, Laxenburg, Austria, available online at http://www.fao.org/soils-portal/soil-survey/soil-maps-and-databases/harmonized-world-soil-database-v12/en/ (accessed on 19/08/2014).

Other required parameters of the model were set by using the model's default values. Threshold flow accumulation is the number of upstream cells that must flow into a cell before it is considered part of a stream, such that retention stops. The default values of the threshold flow are 1000. The two calibration parameters, k_b and IC_0, determine the shape of the relationship between hydrologic connectivity and the sediment delivery ratio. The default values are k_b = 2 and IC_0 = 0.5. SDR_{max}, which is the maximum sediment delivery ratio that a pixel can reach. The model's default value for this parameter is 0.8.

Table 6.2 Values of C-factor and F-factor for various LULC

LULC desc	RUSLE C	RUSLE P
Old growth forest	0.038	1
Degraded forest	0.041	1
Regrowth forest	0.046	1
Bamboo forest	0.1	1
Mixed forest (Timber and bamboo)	0.044	1
Rocky mountain forest	0.068	1
Planted forest	0.084	1
Bare rocky mountain	0.131	1
Shrub/grass (bare land)	0.062	1
Water surface	0	0
Residential area	0.181	1
Agricultural and **other land**	0.142	1

Water Yield Model. The water yield model requires additional GIS maps. These GIS maps include: average annual precipitation, average annual reference evapotranspiration, depth-to-root restricting layer, and plant-available water fraction. These maps were extracted from various sources of the global database. The map of precipitation was acquired from the WorldClim database that averaged precipitation over the period from 1950 to 2000.[31] The map of average annual reference evapotranspiration was derived from the work of Trabucco

[31] WorldClim average annual precipitation information is available at http://www.worldclim.org/current (accessed on 24/10/2014).

and Zomer (2009).[32] The information of depth-to-root restricting layer and plant-available water fraction was derived from the HSWD V1.2 database.

Besides these GIS maps, the model requires other biophysical information that consists of the plant evapotranspiration coefficient, the root depth for each LULC and seasonality factors. The plant evapotranspiration coefficient, K_c, for each LULC type was estimated through LAI (Leaf Area Index).[33] For the vegetated LULC, the K_c can be estimated as a function of LAI using the equation below proposed by Allen *et al.* (1998).

$$K_c = (1 - e^{-0.7*LAI}) \tag{6.3}$$

For the non-vegetated LULC, according to Allen *et al.* (1998), the values of K_c for a specific non-vegetated LULC type are as follows: K_c = 1 for less than 2 m open water; K_c ranges from 0.7 to 1.1 for more than 5 m open water; K_c for wetlands can be assumed in the range of 1 to 1.2; for bare soil, it can be estimated at K_c=0.5; K_c for built areas can be set to $f\cdot 0.1 + (1-f)\cdot 0.6$ where f is the fraction of impervious cover in the area. Root depth information of each LULC type was assigned based on Canadell *et al.* (1996). The values of K_c and root depth (mm) for the LULC types in the research area are summarised in Table 6.3.

The seasonality factor (Z-parameter) was estimated through the formula suggested by Sharp *et al.* (2015) as follows:

$$Z = \frac{(w-1.25)P}{AWC} \tag{6.4}$$

where w is empirically estimated by Xu *et al.* (2013), P represents annual precipitation and AWC represents available water capacity. In this research region, w is approximately equal to 1.8 (Xu *et al.* 2013); P, which was extracted from the average annual precipitation WorldClim database, is 1617.4 mm; and AWC, which is derived from HWSD V1.2, is 73.08 mm. Therefore, the value of Z for this research region was estimated at 12.17.

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[32] Average annual reference evapotranspiration was derived from Trabucco, A. and Zomer, R. (2009). Global aridity index (global-aridity) and global potential evapo-transpiration (global-PET) geospatial database, *CGIAR Consortium for Spatial Information. Published online, available from the CGIAR-CSI GeoPortal at: http://www. csi. cgiar. org/(2009). Global Aridity Index (Global-Aridity) and Global Potential Evapo-Transpiration (Global-PET) Geospatial Database. In. CGIAR Consortium for Spatial Information*. This is available online at http://www.cgiar-csi.org/ (accessed on 30/08/2014).

[33] LAI is extracted from GLCF_MODIS_LAI (MOD09A1) which is available at http://glcf.umd.edu/data/lai/ (accessed on 30/09/2014).

Table 6.3 The values of K_c and root depths of various LULC types in the research area

LULC description	K_c	**Root depth (mm)**	**Vegetable LULC (1: Yes, 0: No)**
Old growth forest	1	7300	1
Degraded forest	1	7300	1
Regrowth forest	1	7300	1
Bamboo forest	0.9997	2000	1
Timber and bamboo mixed forest	1	7300	1
Rocky mountain forest	0.9998	7300	1
Planted forest	0.9997	20000	1
Bare rocky mountain	0.5	250	1
Shrub/grass (bare land)	0.398	5000	1
Water surface	0.9849	0	0
Residential area	0.5	7000	0
Agricultural and **other land**	0.7	2000	1

d) The Data Set for Evaluating the Forest Ecosystem Services

This study aims to evaluate the economic values of three forest ecosystem services, and three sub-sets of data were collected. Firstly, as mentioned in Chapter 3, the social value of carbon storage and sequestration was measured by the marginal damage costs of climate change (IPCC 2007; Stern 2007; Tol 2009; Nordhaus 2011). The global SCC was collected from the literature and ranged from 31.4 to 80.9 2010 USD per tonne of carbon.[34] Secondly, sediment reduction was evaluated by the replacement cost of removing sediment from reservoirs. The replacement cost in the research region was collected from a forest research report conducted by Phuong (2009a).

Thirdly, in order to value water supply for hydropower production, technical economic information from the hydropower plants of interest and the price of electricity were collected. The technical information from the hydropower plants was mainly collected from Vietnam

[34] The lower limit of SCC is estimated by Nordhaus, W. (2007). Critical assumptions in the Stern Review on climate change, *Science Magazine's State of the Planet 2008-2009: With a special section on energy and sustainability*. The upper limit is suggested by Tol, R.S. (2009). The economic effects of climate change, *The Journal of Economic Perspectives* 23, 29-51.

Electricity (EVN), the Ministry of Industry and Trade. Economic information, such as the total investment and average annual energy production of the hydropower plants, was collected from the Vietnamese Government and various organisations, including EVN and other companies engaged in the construction and operation of the hydropower plants. The electricity wholesale price in 2010 (P_e) was obtained from government documents, in particular, Decision number 588/QD-BTC of the Ministry of Finance, issued on 22/3/2010. The price was 1,058 VND per kWh in 2010, which is equivalent to 0.056842 US dollars per kWh. [35] The technical and economic information of the hydropower plants is summarised in Table 6.4 and Table 6.5.

Table 6.4 Technical information of the hydropower plants of interest

Hydropower plant	Time span (years)	Turbine type	Efficiency of turbine (β)	Fraction of water used for hydropower generation (y_d)	Water height (m)	Installed capacity (MW)
Hoa Binh	50	Francis	0.925	0.86	88	1920
Son La	50	Francis	0.910	0.94	78	2400
Lai Chau	50	Francis	0.925	0.85	80.5	1200
Nam Na 3	50	Kaplan	0.929	0.96	20	84

Source: EVN, Ministry of Industry and Trade, 2015.

[35] The price in VND was converted to USD though the official exchange rate and MUV Index published by the World Bank. Official exchange rates proposed by the World Bank are available at http://data.worldbank.org/indicator/PA.NUS.FCRF, and the MUV index is available at http://econ.worldbank.org/WBSITE/EXTERNAL/EXTDEC/EXTDECPROSPECTS/0,,contentMDK:20587651~menuPK:5962952~pagePK:64165401~piPK:64165026~theSitePK:476883,00.html

Table 6.5 Economic information of the hydropower plants of interest

	Hoa Binh	Son La	Lai Chau	Nam Na 3
- Total Investment (current price in reported years, bil. VND)	-	60,195.928	35,700	2,800
- Total Investment (constant USD in 2010, million USD)[36]	-	2,686.135	2,168.718	150.433
- Average Annual Energy Production (mil. KWH)	8,160	10,246	4,670	361.12
- Sources of total investment		Government document - Decision number 668 of the Prime Minister, signed on 5/6/2012	Government document - Decision number 819 of the Prime Minister, signed on 7/6/2010	Song Da 4 Joint Stock Company[37]
- Sources of average annual energy production	Hydropower plant's construction and operation company[38]	EVN[39]	EVN[40]	Song Da 4 Joint Stock Company[41]

[36] The total investment was converted from VND to USD via the official the exchange rate at the year of report, then converted to 2010 constant USD via MUV index proposed by the World Bank (MUV 15).
[37] The information is available at http://songda4.vn/?show=viewtt&ic=2&list=5_103&id=149 (accessed on 1/03/2015).
[38] The information is available online at http://www.songda.vn/info/Chitiet/tabid/181/ItemID/4882/View/Details/Default.aspx (accessed on 3/3/2015).
[39] The information is available at http://evnboimb.vn/vi-VN/c63/267/Mot-so-thong-tin-chinh-ve-du-an--Thuy-dien-Son-La.aspx (accessed on 3/3/2015).
[40] The information is available at http://www.evnfc.vn/tin-tuc-amp-su-kien/tin-evnfinance/thu-xep-von-cho-thuy-dien-lai-chau.html (accessed on 3/3/2015).
[41] The information is available at http://songda4.vn/?show=viewtt&ic=2&list=5_103&id=149 (accessed on 1/03/2015).

6.2.2 Data Collection Techniques

In order to obtain the mentioned above data sets, three major data collection techniques were used. These include desktop research, unstructured expert interviews, and structured expert interviews. The use and information gathered by these collection methods are explained as follows:

a) Desktop Research (DR) to Gather and Synthesise Secondary Data

Secondary data collection techniques were utilised to gather information from databases that were published by official organisations. Different types of data were collected from different data sources, both globally and nationally. For example, DEM, climate variables, soil databases, maps of factors for land suitability analysis etc., were gathered from various published databases, such as that of the U.S. National Aeronautics and Space Administration (NASA) and Japan's Ministry of Economy, Trade, and Industry, WorldClim and the FAO-Harmonised World Soil Database. The maps of forest cover, forest land use planning at national and provincial levels, legal documents relevant to Vietnamese forestry, forest conservation and development programs/projects were collected from various government organisations in charge of forest management and land management (e.g., Ministry of Agriculture and Rural Development and Ministry of Environment and Natural Resources). Technical and economic information regarding the hydropower plants was collected from various reports from Vietnam Electricity, the Ministry of Industry and Trade and the Ministry of Finance.

b) Unstructured Expert Interviews (UEI)

Unstructured expert interviews were conducted during my field trip from 2/5/2015 to 20/7/2015 in Vietnam. During the field trip, I arranged a number of meetings with experts and forest managers. The experts were professionals working for government organisations ranging from administrative organisations relating to the governance of forest land, such as the Vietnam Administration of Forestry (VNFOREST), the Forest Inventory and Planning Institute (FIPI), the Ministry of Agriculture and Rural Development (MARD), the Ministry of Environment and Natural Resources (MONRE), research institutions including the Research Centre for Forest Ecology and Environment (RCFEE), the Research Institute for Sustainable Forest Management and Forest Certification (SFMI), the Vietnam National University of Forestry (VNUF), the Vietnamese Academy of Forest Science (VAFS) and forest management organisations at the provincial level. I also had several conversations with other researchers from non-government organisations (NGOs) concerned with forest conservation, such as The Centre for People and Forests and PanNature. The number of experts I met and their respective organisations, are summarised in Table 6.6.

Various topics were discussed in my informal conversations with these experts. From the conversations with experts of the administrative organisations, I focussed on the topic of feasible forest governance arrangements for the future. This consisted of strategic planning for forest protection and development until 2020 (e.g., objectives of forest cover, the patterns of forest cover changes, forest land use planning, physical and environmental factors determining land cover changes), legal and practical barriers of the decentralisation process in forest governance, particularly the process of forest land and forest allocation to individuals, households and local communities, and the implementation of payments for forest ecosystem services. In the meetings with the experts from research institutes and NGOs, I concentrated on the shortcomings of current forest governance and possible alternative FG scenarios. These issues were mainly related to the state rules about the use of forest land and forest property rights, the benefit-sharing mechanisms between state forest management organisations and local people, forest utilisation regulations, and the local communities' rules and norms relating to forest utilisation and conservation. At the provincial level, my conversations with forest managers were related to the practical problems of forest protection and development, forest LULC planning, physical and environmental factors that drive forest LULC changes, forest land and forest allocation to individuals and households and the implementation of Payments for Forest Ecosystem Services. The information gathered from these interviews was utilised to generate alternative FG scenarios as well as to design the structured questions for the follow-up structured expert interviews (SEI).

c) Structured Expert Interviews (SEI)

After conducting the unstructured expert interviews with 29 people, a structured questionnaire was developed, which was used to carry out telephone interviews with key experts that I met during my field trip. These interviews were successfully conducted with 12 provincial forest managers and five national researchers (see Table 6.6).

The experts who accepted the telephone interviews were sent a questionnaire that presented to them the alternative governance scenarios and a series of structured questions asking for their perceptions of the LULC priorities and LULC changes under the three alternative forest governance arrangements. The questionnaire was sent to them in advance so that they had time to think before the phone interview took place. A couple of days after sending the questionnaire, I contacted them by telephone to conduct the interviews. The data collected from these interviews was used to generate a matrix representing LULC transition patterns to estimate the changes of LULC under the alternative FG scenarios. The structured questionnaire is given in Appendix 6.3.

Table 6.6 Types, organisations and number of key informants

Key informants (KIs)	**Organisations**[42]	**Number of KIs engaged UEIs**	**Number of KIs engaged SEIs**
Experts involved in forest governance	VNForest, FIPI, SFMI, NGOs	4	2
Experts specialising in forest ecology and economics	RCFEE, VNFU, VAFS	4	2
Experts involved in land use planning	MONRE	2	0
Experts specialising in PFES	VAFS	2	1
Provincial forest managers			
- Hoa Binh	Department of Forestry, Hoa Binh province	5	2
- Son La	Department of Forestry, Son La province	4	3
- Dien Bien	Department of Forestry, Dien Bien province	4	3
- Lai Chau	Department of Forestry, Lai Chau province	4	4
Total		29	17

[42] VNForest - Vietnam Forest Administration; FIPI - Forest Inventory and Planning Institute; RCFEE - Research Centre for Forest Ecology and Environmental; VNUF - Vietnam National University of Forestry; VAFS - Vietnamese Academy of Forest Science ; MONRE - Ministry of Natural Resources and Environment.

6.2.3 Data Pre-processing: Preparing Inputs for the InVEST Models

Before being ready for use as inputs for the InVEST models, many of the collected map layers had to be pre-processed because some specific inputs, including carbon in soil (Carbon Storage and Sequestration Model), R-factor, K-factor, C-factor (SRD Model), depth-to-root or soil depth and plant-available water fraction (water yield model) were estimated or extracted from the gathered available databases. Aside from this, the inputs were collected from the various database sources because they were required to be re-projected and transformed to the same coordinate system. The procedure of pre-processing is presented below.

Firstly, some of the required inputs that were not available were estimated from the available databases. For example, from the HWSD V1.2, many inputs, including soil depth, plant-available water fraction and soil erodibility were extracted. Based on DEM (i.e. GDEM2), the watersheds and sub-watersheds were delineated using the ArcGIS 10.1 hydrology tools. In order to delineate and identify the four watersheds associated with the four hydropower plants of interest (i.e. Hoa Binh, Son La, Lai Chau and Nam Na3), I used the positions of the four hydropower plants as the outlet points. Using the ArcGIS 10.1 raster calculator tool, other inputs, such as R-factor and C-factor, which can be estimated by the equations 6.1 and 6.2, were generated from WorldClim precipitation data and the NDVI was derived from LandSat 5TM images that were captured in 2009, respectively. Finally, all the map layers that were required by the InVEST models were transformed and re-projected to WGS 1984 - UTM_Zone 48N coordination systems using ArcGIS 10.1.

6.2.4 Data Pre-processing: Preparing Inputs to Map Forest LULC Changes

The forest LULC changes were mapped through a scenario approach. Particularly, the 'scenario generator' supporting tool attached to the InVEST model package was used. The process of mapping forest LULC changes under each FG scenario is depicted in Figure 6.2 (please refer to the next section). In order to map the changes of forest LULC under the alternative FG scenarios, there are several inputs required that include the quantity of change, the priority of LULC, transition likelihood matrix, as well as factors that determine land suitability for LULC types and intended changes. This information was collected from secondary data and from the structured expert interviews. The following sub-sections describe how the gathered information was pre-processed to prepare the required information for mapping of the forest LULC changes.

a) Transition Likelihood Matrix

The transition likelihood represents the possibility that a current forest LULC will be converted to another cover type. The higher transition likelihood of a forest LULC moving to another type of cover, the more likely the transition will actually occur. As mentioned earlier in this section, the transition likelihood matrix that describes the possibility of changes in forest LULC patterns was computed from the experts' perception of forest LULC transition possibility on an ordinal scale ranging from 1 to 10 (1 = not expected to occur at all; 10 = will absolutely occur).

Table 6.7a Transition likelihood of forest LULC under the business-as-usual (state-based) FG scenario

State-based FG scenario	**To**						
From	Old growth forest	Degraded forest	Regrowth forest	Planted forest	Bare land	**Total lost**	**Net gained**
Old growth forest	-	4.35 (2.55)	0	0	2.35 (1.58)	**6.7**	**-6.7**
Degraded forest	0	-	0	3.24 (2.46)	3.41 (2.06)	**6.64**	**2.47**
Regrowth forest	0	4.76 (2.54)	-	3.06 (2.49)	3.18 (1.94)	**11**	**-3.59**
Planted forest	0	0	0	-	2.24 (1.64)	**2.24**	**7.7**
Bare land/ shrub land	0	0	7.41 (2.93)	3.64 (1.71)	0	**11.05**	**0.12**
Total gained	**0**	**9.11**	**7.41**	**9.94**	**11.17**		

Note: Standard deviation in parentheses

Table 6.7b Transition likelihood of forest LULC under the community-based FG scenario

Community-based FG scenario	To						
From	Old growth forest	Degraded forest	Regrowth forest	Planted forest	Bare land	**Total lost**	**Net gained**
Old growth forest	0	4.38 (2.83)	0	0	2.44 (1.90)	**6.82**	**-6.82**
Degraded forest	0	0	0	2.94 (2.43)	2.44 (1.36)	**5.38**	**2.88**
Regrowth forest	0	3.88 (2.83)	0	3.00 (2.68)	2.68 (1.49)	**9.56**	**-2.56**
Planted forest	0	0	0	0	4.75 (1.36)	**4.75**	**7.69**
Bare land/ shrub land	0	0	7.00 (2.58)	6.50 (2.83)	0	**13.5**	**-1.19**
Total gained	**0**	**8.26**	**7.00**	**12.44**	**12.31**		

Note: Standard deviation in parentheses

Table 6.7c Transition likelihood of forest LULC under the individual-based FG scenario

Individual-based FG scenario	**To**						
From	Old growth forest	Degraded forest	Regrowth forest	Planted forest	Bare land	**Total lost**	**Net gained**
Old growth forest	0	5.06 (2.49)	0	0	3 (1.58)	**8.06**	**-8.06**
Degraded forest	0	0	0	3.82 (2.56)	3.29 (1.86)	**7.11**	**2.77**
Regrowth forest	0	4.82 (2.63)	0	3.59 (2.58)	3.11 (1.78)	**11.52**	**-6.82**
Planted forest	0	0	0	0	2.82 (1.74)	**2.82**	**9.59**
Bare land/ shrub land	0	0	4.7 (1.76)	5.0 (2.42)	0	**9.7**	**2.52**
Total gained	**0**	**9.88**	**4.7**	**12.41**	**12.22**		

Note: Standard deviation in parentheses

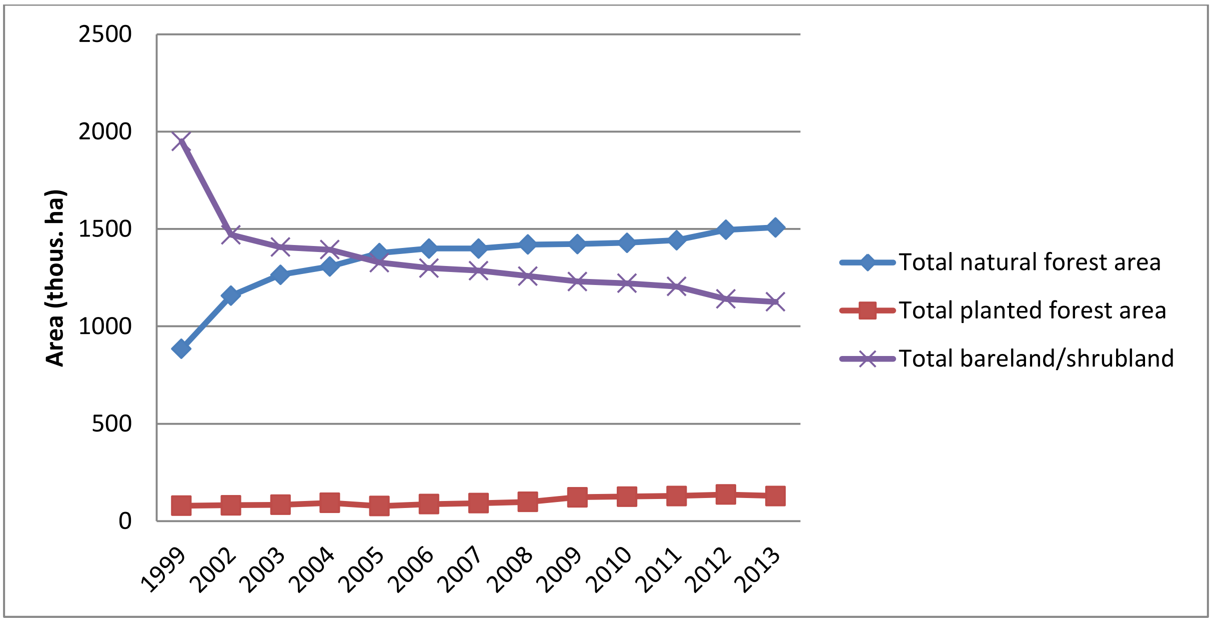
Under each FG scenario, the experts' perception was averaged to produce a transition likelihood matrix. Tables 6.7a, b, and c show the summaries of transition likelihood of forest LULC under the three alternative FG scenarios. Being designed to incorporate the experts' perception, InVEST scenario generator tool only considers the relative values of the transition likelihood instead of the absolute values. The transition likelihood matrices presented in these tables are converted to ordinal scale before being used to map future LULC changes. In addition, the InVEST scenario generator tool only concerns the tranistion likelihood of the gain and loss of a LULC. Take Table 6.7a as an example: reading the columns, old-growth forest will not increase, degraded forest will increase due to a contribution from old growth forest and regrowth forest. In this case, when transition probability of regrowth forest to degraded forest is higher than that of old growth forest to degraded forest, degraded forest will only contribute to the increase in old growth forest after all the available parcels in regrowth forest are exhausted. Similarly, planted forest will increase, and its increase will be first due to bareland/shrubland, then degraded forest if all of bareland/shrubland is converted, and lastly from regrowth forest if all of the two land covers is completed used up. Bareland/shrubland will also expand to degraded forest, then to regrowth forest, old growth forest, and finally planted forest. Additionally, the information of net gained of each LULC type calculated from the transition likelihood matrices will be used to estimate the quantity of changes of each LULC type.

b) Quantity of Changes

The quantities of forest LULC change were estimated for each FG scenario. Under the business-as-usual (state-based FG) scenario, assuming that there will not be any social, economic or natural shocks to significantly influence forest LULC, the recently observed trend of forest LULC indicates that it will remain unchanged in the near future. With this assumption, the historical data of forest LULC changes (from 2006 to 2013) was used to project the forest LULC in 2020. This period was selected because the current forest governance arrangement has remained unchanged since 2006 (see Figure 6.1).

Based on the historical observed data and using a linear regression model, I estimated the time trends for natural and planted forests under the business-as-usual (state-based FG) scenario. The regression equation is given as: projected area = $\alpha + \beta t + e$, where t represents the projected year with the year 2006 equal to 1, and e is error term. The results of the regression models for each type of forest LULC are summarised in Table 6.8.

Figure 6.1 Historical trend of forest LULC changes in the Northwest region



Sources: VNForest, MARD statistical database, 1999-2014.[43]

Table 6.8 Estimation of the area of natural forest and planted forest for 2020

	Area of natural forest (thousand ha)	Area of planted forest (thousand ha)
α	1369.34*** (109.9)	82.82*** (13.43)
β	15.6*** (6.33)	7.3*** (5.98)
R square	0.87	0.86
F-test	40.07***	35.71***

Note: OLS regressions. t-statistic values are in parentheses. ***, ** and * denote significance at the 1, 5 and 10 percent levels, respectively.

Based on the results of the regression models, under the business-as-usual scenario (i.e. the state-based FG scenario) the areas of natural and planted forests were projected to be 1603.6 and 192.3 thousand hectares in 2020, respectively. The total forest land area is equal to forested areas and the areas of bare land/shrub land that are allocated for forest expansion.

[43] The statistical data of VNFOREST, MARD, 1999-2013 is available online at http://www.kiemlam.org.vn/Desktop.aspx/List/So-lieu-dien-bien-rung-hang-nam/ (accessed on 5/3/2015).

Therefore, the total area of bare land/shrub land is equal to the total area of forest land, subtracted by the total forested areas (i.e. natural and planted forests). Consequently, in 2020 the projected area of bare land/shrub land would be 980.9 thousand hectares.

The changes in old growth forests, degraded/poor forests and regrowth forests were estimated as follows. Historically observed data showed that the areas of old growth forest slightly decreased because of illegal timber logging, by 186 hectares annually (provincial forestry sector reports, 2005-2014), while degraded and regrowth forests increased over time. For the old growth forest, assuming this decreased trend continues, the total loss of this forest would be about 1,860 hectares during the period from 2011 to 2020. Because, both degraded and regrowth forests increased over time, I assumed that the ratio between the degraded and regrowth forests would remain the same. Relying on this assumption, the projected areas of the old growth, degraded and regrowth forests would be 138.54, 168.34, and 1038.08 thousand hectares, respectively. The changes of forest LULC up to 2020 under the business-as-usual scenario are represented in Table 6.9.

Table 6.9 Forest LULC changes under the business-as-usual (state-based FG) scenario

	2010	**2020**	
	State-dominant FG	**(1) State-dominant FG**	
	Area (ha)	**Area (ha)**	**Net change (ha)**
Natural forest	1,429,237	1,603,637	174,400
_Old growth forest	140,395	138,535	-1,860
_Degraded forest	143,667	168,343	24,676
_Regrowth forest	886,493	1,038,076	151,584
_Other natural forests	258,682	258,682	0
Planted forest	126,510	192,245	65,735
Bare land/shrub land	1,221,084	980,949	-240,135
Non-forest land	820,408	820,408	0

Source: VNFOREST, MARD Statistical Database 2010.

After quantifying the changes of forest LULC under the state-based FG scenario, quantity changes of forest LULC under the two alternative FG scenarios were projected based on the Markov transition analysis. Expert's perceptions of forest LULC transition probability were derived from the structured expert interviews and utilised to create a transition probability of forest LULC. In the interviews, experts were presented with alternative FG scenarios and then their perceptions were elicited regarding the forest LULC transition possibility (i.e. the possibility of one forest LULC type to transform into another forest LULC type). The transition possibility was elicited on an ordinal scale from 1 to 10 (1 denoting 'not expected to occur' and 10 denoting 'will absolutely occur'). The expert opinions regarding transition possibility were averaged and finalised with a cross-tabulation transition matrix for each FG scenario (see Tables 6.7 a, b and c). Applying the method of analysing land use change proposed by Pontius Jr *et al.* (2004), I calculated the possibility of the net change of each forest LULC that equals the difference between the total possibility of gain and the total possibility of loss. By comparing the results of the possibility of net change for the alternative FG scenarios against the results of the business-as-usual scenario, I calculated the relative scores of the LULC net changes for each forest LULC. Table 6.10 shows the likelihood of net changes of forest LULC under the three FG scenarios, and the relative scores of the transition likelihood of net changes of forest LULC. The relative scores are defined as the ratio of likelihood of the net changes under the two alternative FG scenarios as compared to those of the business-as-usual scenario.

Table 6.10 Relative score of transition likelihood of net changes for forest LULC

Forest LULC	**Likelihood of Net Changes of LULC**			**Relative Score**	
	State Centralised FG	**Community-Based FG**	**Individual-Based FG**		
	(1)	**(2)**	**(3)**	**(2)/(1)**	**(3)/(1)**
Old growth forest	-6.706	-6.813	-8.059	1.0159	1.2018
Degraded forest	2.471	2.875	2.765	1.1637	1.1190
Regrowth forest	-3.588	-2.563	-6.824	0.7141	1.9016
Planted forest	7.706	7.688	9.588	0.9976	1.2443

The relative scores were then utilised to estimate the quantity of forest LULC changes under the two alternative FG scenarios. Let us denote the net changes of a particular forest LULC under the state-based FG scenario, the community-based FG scenario, and the individual-based FG scenario as S, C, and I, respectively; and represent the relative score between the community-based FG and the state-based FG as CR, and the relative score between the individual-based FG and the state-based FG as IR. Then:

- C = S*CR and I = S*IR if the sign of S is similar to the sign of the likelihood of the net changes of this particular forest LULC, or
- C = S – S*(CR - 1) and I = S – S*(IR - 1) if the sign of S is opposite to the sign of the likelihood of the net changes in this particular forest LULC.

For example, the net change of degraded forest in the community-based FG scenario equals 24,676 (the net change of degraded forest for the state-based FG scenario) multiplied by 1.1637 (the relative score of the transition likelihood of degraded forest in the community-based FG scenario against the one in the state-based FG scenario). In this case, the sign of the net change and the sign of the likelihood of the net change of degraded forest are both positive. At the same time, the net change for regrowth forest under the community-based FG scenario equals 151,584 – 151,584*(0.7141 - 1) where 151,584 is the net change of regrowth forest under the state-based FG scenario, and 0.7141 is the relative score of the transition likelihood of regrowth forest in the community-based FG scenario compared to that in the state-based FG scenario. In this case, the sign of the net change is positive while the sign of the likelihood of the net change is negative. Once the net changes of forest LULC were computed, the projected percentage of forest LULC changes in the period from 2010 (baseline year) to 2020 (projected year) under each FG scenario, were calculated. The results of the net changes and the percentages of each forest LULC under the alternative FG scenarios are represented in Table 6.11 and Table 6.12, respectively.

Table 6.11 The net changes of forest LULC under the alternative FG scenarios

	Sign	Relative score		Net change (ha)		
		(2)/(1)	(3)/(1)	State-based FG scenario (business-as-usual) (1)	Community-based FG scenario (2)	Individual-based FG scenario (3)
Old growth forest	+	1.0159	1.2018	- 1,860	- 1,890	- 2,235
Degraded forest	+	1.1637	1.1190	24,676	28,716	27,614
Regrowth forest	-	0.7141	1.9016	151,584	194,915	14,910
Planted forest	+	0.9976	1.2443	65,735	65,578	81,792

Table 6.12 Projected percentage of forest LULC changes, 2010 – 2020

Forest LULC	Percentage of Changes (%)		
	State-based FG	Community-based FG	Individual-based FG
Old growth forest	- 1.3	- 1.3	- 1.6
Degraded forest	17.2	20.0	19.2
Regrowth forest	17.1	22.0	1.7
Planted forest	52.0	51.8	64.7

c) Priority

When there is competition between different forest LULC objectives for a single parcel of land, the forest LULC type that is more prioritised will be considered to be allocated before those that are less prioritised. In order to obtain the priority score of each forest LULC type, in the structured expert interviews, for every FG scenario, I asked the experts to rank the importance of the forest LULC types in an ordered scale from 0 to 10 (0 = the least prioritised, 10 = the most prioritised). The answers were then averaged and translated in an ordinal scale. The results are shown in Table 6.13.

Table 6.13 Priority of forest LULC under the alternative FG scenarios in an ordinal scale

FG scenarios	Priority of forest LULC in an ordinal scale				
	Old growth forest	**Degraded forest**	**Regrowth forest**	**Planted forest**	**Bare land/ shrub land**
State-based	7	6	8	9	2
Community-based	9	7	8	6	2
Individual-based	5	6	7	9	2

d) Factors: Land Suitability Analysis

While the transition likelihood drives the quantity of land cover change, physical and environmental factors determine the susceptibility of a land parcel to conversion. This means that these factors determine where the land cover changes are more likely to occur. Conducting a land suitability analysis is essential in order to map LULC changes, (Pereira and Duckstein 1993; Antoine *et al.* 1997; Joerin *et al.* 2001; Malczewski 2004; Nyeko 2012). The principle of land suitability analysis is that changes in landscape occur in areas that are relatively more suitable (Sharp *et al.* 2015). This section presents the procedure for preparing maps of suitability scores for forest LULC types that are likely to expand under the FG scenarios. The procedure includes: defining factors that determine land suitability for each forest LULC type, creating a suitability score of these factors and indexing the combination of the factors' suitability score to create a single suitability score index for each forest LULC type.

Defining factors. In this study, I applied the GIS rule-based approach (Briassoulis 2000) to define the factors. In order to define the factors determining land suitability under the state-based FG scenario, I first conducted desktop research of the various forestry reports in the research region; for example, the final reports of the Five Million Hectare Reforestation Program, the provincial annual forestry sector reports and the provincial forestry sector planning reports. The unstructured interviews with the forest management experts also helped to identify the physical and environmental factors that determine forest LULC change under the state-based FG scenario. For example, factors that define land suitability for forest regrowth include slope, proximity to streams, proximity to villages and the current cover status of bare land/shrub land. Regarding the community-based FG scenario, I also carried out desktop research of reports on various development projects that implemented community-based forest governance in this study site, in order to define the factors that influence the land suitability of forest LULC changes. Thorough research on the reports of the Social Forestry Development Project (SFDP) and the project of forestry development in

Hoa Binh and the Son La province (KfW7) was conducted.[44] For individual-based FG scenarios, I utilised the government reports that relate to decentralisation in the forestry sector through privatisation. Details of the factor definitions under each FG scenario are summarised in Tables 6.14a, b, and c.

Generating factor suitability score. Once the factors were defined for each forest LULC, I generated each factor's suitability score using fuzzy logic (Baja *et al.* 2002). ArcGIS 10.1 provides fuzzy membership tools that reclassify the input data on a 0 to 1 scale. The values are assigned based on the possibility of being a member of a specified set. A location that has a higher possibility of being a member of a specified set will be assigned a higher value. At extreme levels, locations that are definitely not a member of the specified set will be assigned the value of 0, while those that are definitely a member of the specified set will be assigned the value of 1. Based on the above factor definitions, I used appropriate fuzzy membership functions to create the suitability score map for all of the factors (Jiang and Eastman 2000; Baja *et al.* 2002; Malczewski 2002, 2004). For example, to create a suitability score of the 'distance to roads' factor, I used the fuzzy large membership function. This function is used when the larger input values are more likely to be a member of the set. For the 'proximity to roads' factor, I used the fuzzy small membership function, because this function represents that the nearer to roads a forest is, the more likely a location is to become a member of a specified set.

Indexing factor suitability scores. For the purpose of combining the effects of all factors that determine land suitability for a particular forest LULC, I applied the fuzzy overlay function to create a single indexed suitability score shaped by all factors. The fuzzy overlay function is advanced by fuzzy logic and multi-criteria evaluation in GIS (Jiang and Eastman 2000), and is a useful tool for land suitability analysis (Malczewski 2004). The specific fuzzy overlay sum function, which is an increased function that combines multiple effects of all the factors, was used for indexing the factors' suitability scores. The maps of land suitability for each type of forest LULC expansion under the alternative scenarios are represented in Appendix 6.4.

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[44] These two long projects apply community-based forest governance for forest conservation and development. KfW7 - The project forestry development in Hoa Binh and Son La, 2006-2016. The project has been supported by the German Government, implemented by the Department for Forestry Development, MARD. SFDP - The Social Forestry Development Project (1993-2004) was the first technical cooperation project in the Vietnamese forestry sector. The project was supported by the GTZ (German Technical Cooperation Agency) and implemented by the Department for Forestry Development, MARD.

Table 6.14a Definition of factors determining land suitability for forest LULC changes (State-based FG scenario)

Forest LULC	Factor Description	Explanation
Old growth forest	Distance to roads	The further they are from roads, there is a higher risk of them being illegally cut
	Distance to villages	The further they are from villages, there is a higher risk of them being illegally cut
Degraded forest	Current cover status of land surface reflected by NDVI	Regrowth forest areas with a higher NDVI score will have a higher possibility of becoming degraded/poor forests
Regrowth forest	Current cover status of land surface reflected by NDVI	Bare land/shrub land areas with a higher NDVI score will have a higher possibility of being zoned for regrowth
	Slope	Bare land/shrub land on steeper surface areas have a higher possibility of regrowing
	Proximity to Da River	The nearer they are to the Da River, the higher the possibility that bare land/shrub land areas will be zoned for regrowth
	Distance to villages	The further they are from villages, the higher the possibility that bare land/shrub land areas will regrow
	Elevation	Bare land/shrub land on higher elevation areas will have a higher possibility of regrowth
Planted forest	Current cover status of land surface reflected by NDVI	Bare land/shrub land that has a lower NDVI score is more likely to be converted to planted forest
	Slope	Bare land/shrub land on a slope ranging from 15 to 35 degrees is more likely to be converted to planted forest
	Proximity to roads	The nearer to roads they are, the higher the possibility that bare land/shrub land will be converted to planted forest

Table 6.14b Definition of factors determining land suitability of forest LULC changes (Community-based FG scenario)

Forest LULC	**Factor Description**	**Explanation**
Old growth forest	Distance to roads	The further they are from roads, the higher the risk of being illegally cut
	Distance to villages	The further they are from villages, the higher the risk of being illegally cut
Degraded forest	Current cover status of land surface reflected by NDVI	Regrowth forest areas with a higher NDVI score will have a higher possibility to degrade
Regrowth forest	Current cover status of land surface reflected by NDVI	Bare land/shrub land areas with a higher NDVI score will have a higher possibility of being zoned for regrowth
	Slope	Bare land/shrub land on a steeper surface has a higher possibility to regrow
	Proximity to rivers	The nearer they are to rivers, the higher the possibility that bare land/shrub land areas will be protected for forest regrowth
	Proximity to villages	The nearer they are to villages, the higher the possibility that bare land/shrub land areas will regrow
	Distance to roads	The further they are from roads, the higher the possibility that bare land/shrub land areas will regrow
Planted forest	Current cover status of land surface reflected by NDVI	Bare land/shrub land that has a lower NDVI score is more likely to be converted to planted forest
	Slope	Bare land/shrub land on a slope ranging from 15 to 35 degrees is more likely to be converted to planted forest
	Proximity to villages	The nearer they are to villages, the higher the possibility that bare land/shrub land will be converted to planted forest
	Proximity to roads	The nearer they are to roads, the higher the possibility that bare land/shrub land will be converted to planted forest
	Elevation	Bare land/shrub land on lower elevation areas will have a higher possibility of becoming a planted forest

Table 6.14c Definition of factors determining land suitability of forest LULC changes (Individual-based FG scenario)

Forest LULC	Factor Description	Explanation
Old growth forest	Distance to roads	The further they are from roads, the higher the risk of being illegally cut
	Distance to villages	The further they are from villages, the higher the risk of being illegally cut
Degraded forest	Current cover status of land surface reflected by NDVI	Regrowth forest areas with a higher NDVI score will have a higher possibility of degrading
Regrowth forest	Current cover status of land surface reflected by NDVI	Bare land/shrub land areas with a higher NDVI score will have a higher possibility of being zoned for regrowth
	Slope	Bare land/shrub land on steeper surface areas have a higher possibility to regrow
	Distance to villages	The further they are from villages, the higher the possibility that bare land/shrub land areas will regrow
	Distance to roads	The further they are from roads, the higher the possibility that bare land/shrub land areas will regrow
Planted forest	Slope	Bare land/shrub land on a slope ranging from 15 to 35 degrees are more likely to be converted to planted forest
	Proximity to villages	The nearer they are to villages, the higher the possibility that bare land/shrub land will be converted to planted forest
	Proximity to roads	The nearer they are to roads, the higher the possibility that bare land/shrub land will be converted to planted forest
	Elevation	Bare land/shrub land on lower elevation areas will have a higher possibility of becoming planted forest

6.3 Data Analysis and Interpretation

This section explains the procedure of data analysis and the interpretation of the results. First, the three alternative FG scenarios were developed. Then the land suitability approach was used to project the 2020 forest LULC under each FG scenarios with the forest patterns in 2010 as a baseline. Subsequently, I used the InVEST model to estimate the provision of the three forest ecosystem services of interest for each FG scenarios. The economic values of these forest ecosystem services were also calculated. The provision and values of the three forest ecosystem services that were under the alternative FG scenarios (community-based and individual-based FG scenarios) were then compared to those that are likely derived from the business as usual (the state-based FG) scenario and also to those at the baseline in 2010.

6.3.1 Development of Alternative FG Scenarios

The current legal framework of forest governance and the recent process of decentralisation in the forestry sector, as previously described in Chapter 4, open the door for the community-based and individual-based forest governance arrangements to become possible future forest governance regimes in the research region. These forest governance regimes are developed based on the governance framework defined in Chapter 5 as both formal and informal institutions (e.g., laws, rules, norms, etc.,) that shape forest property rights, forest utilisation, benefit-sharing mechanisms of forest resources and interactions among forest stakeholders. The steps of collecting and analysing data of forest governance were followed as per the practical guideline proposed by Cowling *et al.* (2014). The information collected from secondary data sources and from my conversations with the experts relating to forest governance legal frameworks and practices, was analysed using several criteria consisting of: forest property rights (forest land tenure and forest use rights), formal forest regulations set by the state, informed rules of forest utilisation set by the local community (e.g., customary laws), the benefits of sharing forest protection and forest plantation (including the payment for forest ecosystem service mechanisms), and especially the role of the state, local communities and individuals in forest land use planning. These criteria helped to define the key attributes of the forest governance arrangement as described in Appendix 6.5. As a result, the three forest governance regimes are defined as follows:

The key characteristic of community-based forest governance is that local communities take over the main responsibility in the forestry sector.[45] The forest property rights in the form of forest land tenure and forest use rights are legally granted to the local communities on a long-term basis. The local communities self-organise the use of their allocated forests and forest land use planning based on their customary laws as long as their rules do not conflict with the state laws. They gain benefits from using forest resources as well as from the efforts of forest protection and development through the state financial support, and from

[45] Community is defined here as the village community or local household group that has the same customs, practices and traditions of close community association with forests in their production, life, culture, and beliefs.

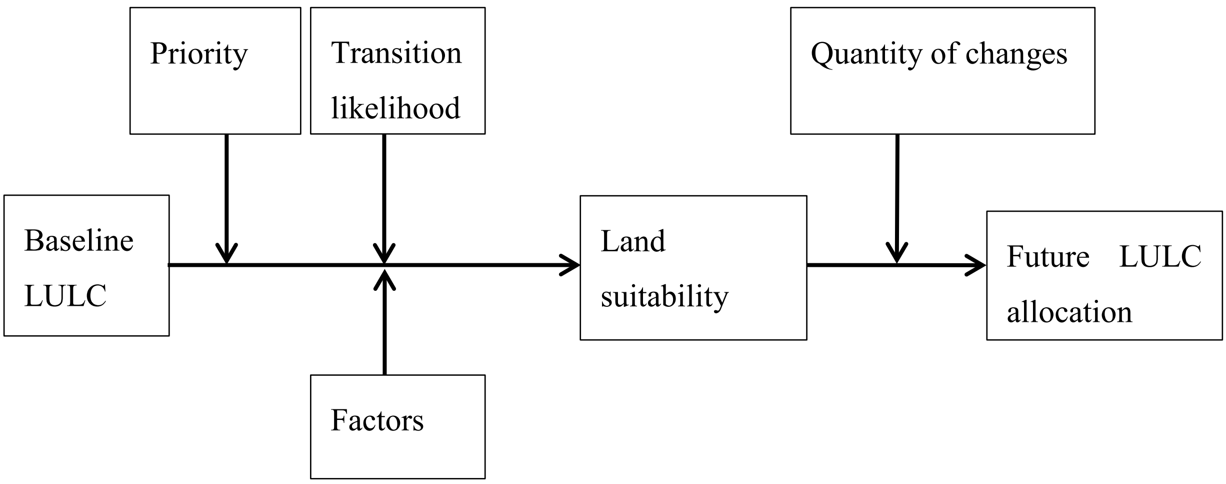
the payment for the forest ecosystem services mechanism. In this forest governance arrangement, the government takes a hands-off approach, no longer influencing matters directly. It retains a supervisory role to ensure that the communities can carry out their functions, especially regarding technical matters and the enforcement of legal claims, such as sanctions for violations, which go beyond communal authorities. In summary, in this forest governance regime, the forest property rights and responsibility for forest management are divulged to local communities.

On the other hand, the individual-based forest governance is characterised by previously state-managed forests and forest land, and the forests previously managed by commune's people committees being allocated to local individual households. The households have long-term land use tenure of forest land and forest use rights legally guaranteed by land titles. The individual households are able to register their allocated forest land and forests on the state cadastral map as well as on the field. The households self-organise land use planning based on their own demands and in accordance with the state land use planning and strategy of forest protection and development. For planted production forest areas, the households self-manage their forestry production on their planted production forest, get benefits from timber exploitation, can get access to financial support from the state funds for forest protection and development and obtain technical support provided by state organisations. The households can also practice agro-forestry production on their allocated planted production forest. For natural production forests, they can obtain benefits from timber exploitation and other non-timber products allowed by the law. They also receive benefits from payments for forest ecosystem services for forest protection and development and directly from the state budget devoted to forest protection and development. Regarding protection forests, households mainly obtain benefits from the payments for forest ecosystem services for forest protection. In addition, they can also receive benefits from state funds for forest protection and forest development and from extracting forest resources (e.g., non-timber forest products) allowed by the law. In this forest governance regime, the government also takes a hands-off approach and retains only a supervisory role to ensure that the households follow the legal framework in the forestry sector, resolve possible conflicts in land tenure and provide financial and technical services.

6.3.2 Mapping and Analysing LULC Changes Under the Alternative FG Scenarios

As mentioned earlier in section 6.2.4, the future LULC changes were mapped by the scenario generator tool that is included in the InVEST model package. Each forest governance arrangement was characterised by the variables of LULC changes (i.e. the quantity of changes, priority, transition likelihood and suitability factors that determine land suitability). This tool projects future LULC based on a baseline LULC and several drivers that were mentioned in section 6.2.4. As shown in Figure 6.2, the information on priority was first considered. Subsequently, the combined effects of transition likelihood and the physical/environmental factors were used to determine the possibility and land suitability for the conversion. When this first LULC conversion reached its quantity of change, the future allocation of this forest LULC was completed. A similar process was repeated for the next forest LULC with a lower priority value. The mapping process was completed when all the forest LULCs subject to change were allocated. Using the prepared inputs, the future forest LULC was mapped for each FG scenario. Once the mapping was completed, the comparison of the future LULC patterns under the three FG scenarios was carried out. The comparison includes both the quantity of changes and spatial LULC patterns of the changes among the scenarios of FG.

Figure 6.2 Flow of mapping forest LULC changes



Source: Adapted from InVEST scenario generator (Sharp *et al.* 2015).

6.3.3 InVEST Models

All outputs of the three separate InVEST models used in this study are in the form of GIS maps and tables. Outputs of the carbon storage and sequestration model include the total amount of carbon stored in the baseline LULC, the total amount of carbon that will be stored in each of the future LULC scenarios and the total amount of carbon that will be sequestrated during the period from 2010 (baseline year) to 2020 (projected year). For the sediment delivery ratio model, the total amount of sediment exported to the stream, the potential sediment retention that represents the difference in the amount of sediment delivered by the examined LULC and a hypothetical situation where all land cover has been converted to bare soil, were considered. For the water yield model, the total volume of water yield per subwatershed and watershed was utilised.

The level of analysis was at watershed levels. The values of the above-mentioned outputs at the baseline LULC were compared with the ones of the future LULC scenarios. Comparisons were also carried out between the future LULC scenarios. These comparisons examine synergy and trade-offs of the forest ecosystem services if the current FG scenario were to be replaced by any of the alternative FG scenarios.

6.3.4 Valuation Methods of the Forest Ecosystem Services

In addition to the quantitative analysis of forest ecosystem service provision, for the purposes of this study, I evaluated the economic values of the forest ecosystem services and articulated the different changes in the values of these services over time and among the alternative FG scenarios. Firstly, I valued the forest ecosystem services derived from the forested watershed in the baseline forest LULC and in the projected future LULC under each FG scenario. Subsequently, for each FG scenario, I articulated the changes in the value of each service over time. Finally, the changes in the value of each service and the aggregated value of all the services were compared among the scenarios. The methods that were used for evaluating the economic values of each service are described in the next sub-section.

a) Calculating Values of Carbon Storage and Sequestration

There has been a lack of studies on the social cost of carbon emissions that are specific to Vietnam. Although there were several studies in Vietnam that attempted to evaluate economic values of carbon storage and sequestration services derived from forests, such as the studies by Phuong and Ha (2007), Phuong (2009b) and Quynh (2010), no study estimated the social cost of carbon emissions specifically for Vietnam. They used the global market price of carbon credit for valuing carbon storage and sequestration services. However, as argued by Sharp *et al.* (2015), the prices of carbon credits largely result from the market rules and regulations and do not truly reflect the contribution to the human welfare of carbon sequestration.

Therefore, in this study, I used the social cost of carbon (SCC) for all carbon sequestration. The global SCC estimated by the previously-published literature was used for valuing carbon storage and sequestration services derived from forests. Due to the uncertainty of the estimated global SCC, this study used the minimum value estimated by Nordhaus (2007) as the lower limit of the SCC and the maximum value proposed by (Tol 2008) as the upper limit of the SCC. By converting to a common value expressed in 2010 USD, this range of SCC varies from 31.4 to 80.9 USD per tonne of carbon.[46] I assumed that an approximate value of 50 USD per tonne of carbon is a mean of the SCC as estimated by (Tol 2009) for the discount rate of 3% and also by Nordhaus (2011). I also assumed that the present values of the SCC of carbon remain constant in the period of my research (2010 – 2020). In fact, as argued by (Nelson *et al.* 2009), whether the SCC will increase, decrease or remain unchanged in the future, is uncertain.

b) Calculating Values of Sediment Reduction

To estimate the value of sediment retention services provided by forests, I assumed that the sediment accumulated each year in the dams was to be completely removed. This allowed the use of the replacement-cost method for measuring the values of sediment reduction services. This effectively means that the benefit from sediment reduction as the service of a forested watershed is represented by the saved cost of sediment removal from a reservoir. Based on Phuong (2009a), who measured the values of sediment reduction services in the north of Vietnam, the total cost of sediment dredging is calculated via the cost of various physical inputs and labour. As shown in Table 6.15, most of the cost comes from specific expenses for hiring machinery and the wages of skilled workers. In total, the estimated average cost of

[46] The Manufactures Unit Value Index (MUV) that is issued by the WB from 1960-2013 and projected to 2025 was used for the conversion purpose. MUV is a composite index of prices for manufactured exports from the fifteen major developed and emerging economies to low- and middle-income economies, valued in U.S. dollars, The MUV Index is available online at http://econ.worldbank.org/-WBSITE/EXTERNAL/EXTDEC/EXTDECPROSPECTS/0,,contentMDK:20587651~menuPK:5962952~pagePK:64165401~piPK:64165026~theSitePK:476883,00.html (accessed on 5/9/2015).

sediment dredging was 1997.15 VND per cubic metre of sediment deposited in the reservoirs (see Table 6.15). This is equivalent to 1.592 USD/m^3 or 1.45 USD/tonne in 2010 USD.[47] The calculation of Phuong was based on the guideline of construction expenses issued by the Ministry of Construction, which has been widely used in the context of Vietnam.

Table 6.15 Estimated cost of dredging 100 m^3 of sediment deposited in reservoirs by (Phuong 2009a)

Item	Description	Unit	Quantity	Unit price (VND)	Amount (VND)
Skilled workers and machinery equipment	Dredging machine 2.3M^3.	shift	0.26	4,094,809	1,064,650
	Barge 250T	shift	0.26	713,465	185,501
	Barge 200T	shift	0.26	593,023	154,186
	Small boat	shift	0.13	201,636	26,213
	Other machine	%	5		71,527
Other labour	Skilled workers	Working (8 hours/day)	1.5	72,945	109,418
Other expenses	Equal to 1.5% of the sum of machinery and labour expenses				24,172
Management costs	Equal to 5% of the total direct costs				89,962
Tax					271,521
Total cost					1,997,150

[47] USD to VND exchange rate based on the World Bank official exchange rate database. Conversion of 2007 USD to 2010 USD is based on MUV methods.

c) Calculating Values of Water Supply for Hydropower Production

I applied the residual valuation method to value the marginal product of water used for hydropower production as suggested by Young (2005). The residual technique determines the part of the total value of the production of electricity that is assigned for water use for the production of electricity. For a specific hydropower plant, the marginal product of water is calculated through the following steps:

- Investment costs: I = total cost of constructing a hydropower plant, compensation for local community resettlement and other related costs for construction.
- The annualised investment costs or equivalent annual costs (*EAC*) are calculated as:

$$EAC = \frac{I \cdot r}{1-(1+r)^{-t}} \tag{6.5}$$

  where r represents the discount rate, and t represents the lifespan of the hydropower plant (years).
- When Q_{ae} (kwh) represents the total annual average electricity production by the hydropower plant, the investment cost per kwh (USD/kwh), C_i, is a ratio of the annualised investment cost and the total annual average electricity production.

$$C_i = EAC/Q_{ae} \tag{6.6}$$

- The annual operation and maintenance (O&M) costs, C_{om} are assumed equal to 2.2% of C_i (IEA 2010), so that:

$$C_{om} = 0.022 \cdot C_i \tag{6.7}$$

- The levelised cost of electricity generation that is the net present value of the unit cost of electricity over the lifetime of a generating asset (International Renewable Energy Agency 2012) from hydropower plants in (USD/kwh), denoted by C_{lev}, is shown as:

$$C_{lev} = C_i + 0.022 \cdot C_i \tag{6.8}$$

- When R_w represents the value of a cubic metre of water (USD/m^3) and Q_w is the annual volume of water used for hydropower production (m^3), then the total value of water used for hydropower production is given as $R_w \cdot Q_w$.
- When P_e represents the price of electricity (USD/kwh) and Q_e is the total annual electricity production (kwh), then the total value of electricity production is shown as:

$$TVP = P_e \cdot Q_e \tag{6.9}$$

where Q_e is determined by the technical characteristics of the hydropower plant and the volume of water used for electricity generation. Q_e can be estimated by the equation adopted from Sharp *et al.* (2015).

$$Q_e = 0.00272 \cdot \beta \cdot y_d \cdot h_d \cdot Q_w \tag{6.10}$$

where β represents the turbine efficiency coefficient (%), γ_d represents the percentage of water volume used for electricity generation in the total water yield flowing into the reservoir of the dam d (%), and h_d represents the water height behind the dam at the turbine (m).

- Applying the residual method, the value of one cubic metre of water used for hydropower generation at a specific hydropower plant, R_w (USD/m^3), can be written as:

$$R_w = \frac{P_e \cdot Q_e - C_{lev} \cdot Q_e}{Q_w} \tag{6.11}$$

In this research region, there are several hydropower plants that are located at different elevation levels in the Da River System. This construction design allows water volume from an upper reservoir to enter into the lower reservoirs. This means that water yield derived from upstream watersheds can be used multiple times for generating energy at multiple hydropower plants. Therefore, the total value of one cubic metre, denoted by R_{w-tot}, equals the sum of the value of water used for hydropower generation at every hydropower plant in the downstream hydropower plants. The total value of one cubic metre of water yield, thus can be shown as:

$$R_{w-tot} = \sum_{j=1}^{n} R_{wj} \tag{6.12}$$

where R_{wj} represents the value of one cubic metre of water used for hydropower generation at hydropower plant j, n is the number of hydropower plants in the cadastral watershed. In particular, four large hydropower plants are considered in this study: Hoa Binh, Son La, Lai Chau and Nam Na 3. [48] Table 6.16 shows key results of the calculations described above, specifically the value of one cubic metre of water at each hydropower plant.[49]

[48] Technical and economic information of the hydropower plants are represented in Tables 6.4 and 6.5.

[49] The calculations are for a discount rate of 10 percent, and the hydropower's lifespan of 50 years.

Once the marginal product of water is determined, the total value of water supply for hydropower generation is estimated. The total value of water in a specific watershed equals the value of one cubic metre of water multiplied by the total volume of water used for hydropower generation derived from this forested watershed.

Table 6.16 Marginal product of water supply for hydropower generation[50]

Hydropower plant	Investment cost per kWh, I (USD/kWh)	O&M cost per kWh (USD/kWh)	Levelised cost of electricity (USD/kwh)	Value of water per KWH (USD/kWh)	Marginal product value of water at a specific hydropower plant, R_{wj} (USD/m3)
Hoa Binh			0.0374461	0.0193961	0.0034792
Son La	0.026441668	0.00058172	0.0270234	0.0298189	0.0054135
Lai Chau	0.046838358	0.00103044	0.0478688	0.0089734	0.0015448
Nam Na3	0.042015292	0.00092434	0.0429396	0.0139026	0.0006745

6.4 Conclusion

In summary, this chapter has described the data and methodology that applied the theoretical framework proposed in Chapter 3 to determine the effects of forest governance on the provision and values of the three forest ecosystem services. This chapter begins with the explanation of required data sets and the sources for collecting these data. The data includes four sets that correspond to the four components of the framework. These sets consist of data for developing FG scenarios, mapping forest LULC changes under the generated FG scenarios, quantifying and mapping the changes of forest LULC, and for valuing the three forest ecosystem services. Various data collection techniques were used, including desktop research, unstructured expert interviews, and structured expert interviews which were utilised to gather the required data. The procedure of data pre-processing was described and the specific data pre-processing steps that were needed to prepare the inputs for the InVEST models, as well as the scenario generator tool, were represented. Finally, the methods of data analysis were described and consisted of both qualitative methods, applied for generating FG

[50] All the values are measured in 2010 constant US dollars.

scenarios to determine the changes of forest LULC, and quantitative methods of estimating the provision of the forest ecosystem services. This study also utilised several valuation methods, including the method of avoided social damages, replacement-cost method and the residual method for valuing carbon storage and sequestration, sediment reduction and water supply for hydropower generation services.

Chapter 7

CHANGES IN FOREST LAND USE AND LAND COVER, PROVISION AND VALUE OF FOREST ECOSYSTEM SERVICES UNDER THE THREE ALTERNATIVE FOREST GOVERNANCE SCENARIOS: FINDINGS AND DISCUSSIONS

7.1 Introduction

Chapter 6 described the data and methods that were utilised to investigate the postulated research questions. This chapter reports on the results of the study and discusses how these results can be interpreted as well as how they contribute to the existing literature. The collected data were processed and analysed to answer the research questions of how the provision and values of forest ecosystem services would vary with the alternative forest governance (FG) arrangements. Based on the theoretical framework that links FG to the values of forest ecosystem services through the effects of FG on the forest LULC changes and the provision of forest ecosystem services, this chapter presents the findings. These findings consist of the patterns of forest LULC changes under each FG scenario, comparisons of the provision of forest ecosystem services determined by the LULC changes and values of these services. These results are then interpreted and are discussed in light of the existing literature. The significance of the study and limitations are also examined in this chapter.

7.2 Forest Cover Changes Under the Alternative Forest Governance Scenarios

As mentioned earlier in Chapter 6, forest LULC changes were mapped based on the transition likelihood of LULC changes derived from expert opinions and land suitability analysis. The estimation of forest LULC changes is summarised in Table 7.1 and depicted in Figure 7.1. As shown in Table 7.1, the area under bare land/shrub land can be expected to decrease in all scenarios. At the same time, forest cover is likely to increase in all scenarios in which natural forest will still be the dominant forest cover type. It is also worth noting that the areas of old growth forest are likely to decrease slightly in all scenarios. This decrease would be the greatest under the individual-based forest governance scenario.

The table also shows that the trend of the changes in forest LULC is likely to be different in these scenarios. Under the state-based (business-as-usual) and community-based FG scenarios, natural forests (e.g., degraded/poor forests and regrowth forests) are likely to expand moderately. The overall increase in natural forest cover is estimated to be 12.2% and 15.5% under the state-based FG and community-based FG scenarios, respectively. Planted forest would also increase rapidly under these scenarios. However, the proportion of planted forest in the overall forested land would still be quite small, at about 5%. Meanwhile, the increase of forest cover under the individual-based FG scenario would not be much. Under this scenario, planted forests are likely to increase by about 65% and reach a proportion of nearly 6% of the overall forested cover in 2020. On the other hand, natural forests are likely to expand slightly. The overall growth of natural forests would be less than 3%, of which degraded/poor forests show the largest expansion. As the proportion of degraded/poor forests are limited to about 4% in 2010, its increase would not contribute much to the overall expansion of natural forest cover.

Table 7.1 Forest LULC patterns and changes under the alternative FG scenarios, 2020

LULC	**Baseline 2010 (%)**	**LULC patterns, 2020 (%)**			**Percentage of LULC change**		
		State-based FG	**Com.-based FG**	**Indi.-based FG**	**State-based FG**	**Com.-based FG**	**Indi.-based FG**
1. Natural forest	39.7	44.6	45.9	40.9	12.2	15.5	2.8
1.1 Old growth forest	3.90	3.85	3.85	3.84	-1.3	-1.3	-1.6
1.2 Degraded/poor forest	4.0	4.7	4.8	4.8	17.2	20.0	19.2
1.3 Regrowth forest	24.6	28.9	30.1	25.1	17.1	22.0	1.7
1.4 Other natural forests	7.2	7.2	7.2	7.2	0.0	0.0	0.0
2. Planted forest	3.5	5.3	5.3	5.8	52.0	51.8	64.7
3. Bare land/shrub land	33.9	27.3	26.0	30.6	-19.7	-23.5	-10.0
4. Non-forest land	22.8	22.8	22.8	22.8	0.0	0.0	0.0

My findings regarding the trend of forest LULC changes under the three scenarios are supported by empirical studies that were previously conducted in Vietnam. Regarding the state-based scenario, forest cover has been expanded gradually in forest regeneration and forest plantation as the results of the government's policies relating to reforestation and forest plantation implemented since the 1990s (Sikor 2001; Meyfroidt and Lambin 2008b; Meyfroidt *et al.* 2009; VNFOREST 2013; MARD 2015). The primary causes of forest cover expansion have been found to be due to a combination of forestry policies, including strict logging bans in natural forests (Tuynh and Phuong 2001), the achievement of the ambitious programs in forest conservation, reforestation and afforestation, such as the Five Million

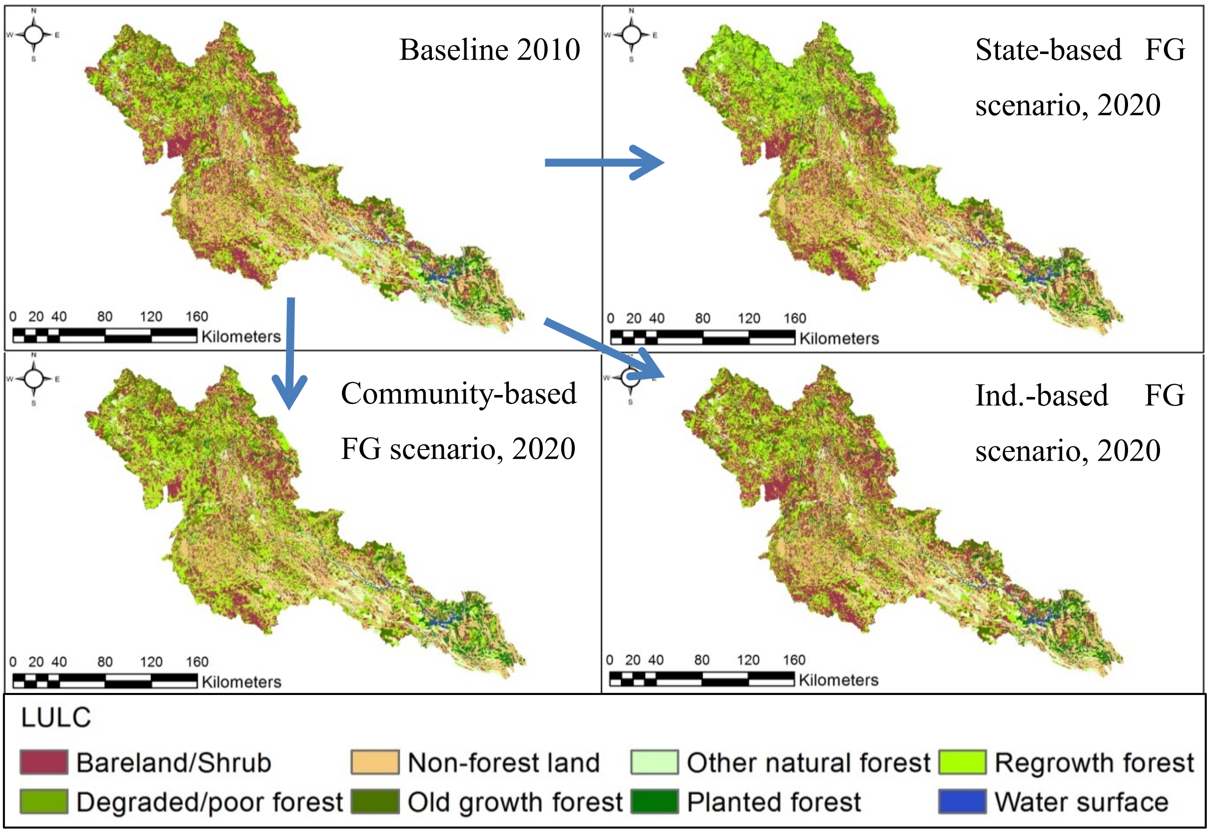
Hectares of Reforestation Programs (5MHRP) from 1998 to 2010 (Government of Vietnam 2011), the followed-up program of forest protection and development for the periods of 2011 to 2020 after the 5MHRP (MARD 2014, 2015), and other indirect drivers, such as the increase in agricultural productivity in mountainous regions that help local people to be less reliant on forests for their livelihoods (Sikor 2001; Meyfroidt and Lambin 2008a), and displacement of the demand for timber to illegal imports from other countries (Meyfroidt *et al.* 2009). This trend of forest cover expansion is expected to continue in the period from 2015 to 2020 since the state made a commitment through its goal of forest protection and development (MARD 2007, 2015). Nonetheless, deforestation and degradation of old growth natural forests due to illegal logging has continued even though many efforts have been made by the government (Government of Vietnam 2011; McElwee 2012; MARD 2014, 2015).

With regard to the improvement of forest LULC, under the community-based FG scenario, this study's results are also consistent with previous publications in Vietnam. Although, in Vietnam, there has been limited forest land legally titled to local communities (VNFOREST 2011), a substantial number of local communities have already managed significant areas of forest in practice with or without formal recognition by the government (Sikor and Tan 2011), especially in the uplands (Tan *et al.* 2008b). In addition, with recent emerging recognitions by the Government of Vietnam, there are thousands of community-based forest management projects that have been implemented nationwide, particularly in the Northwest region (Tan *et al.* 2008b; Tan *et al.* 2009; Wode and Huy 2009). In the uplands, traditional forest management systems are still very active (Wode and Huy 2009; CIRUM 2012). For example, the customary law regarding forest resource usage and management of ethnic minority groups in the Northwest region are still very strong. The report conducted by CIRUM (2012) found that there is a strong link between local people's beliefs and forest conservation. They consider forests to be the community's common resources, which all members have obligations to protect and equally obtain benefits from. In daily practices, they show their respect and nurturing attitude towards trees and forests. For instance, when they need wood from forests, they are required to perform rituals and to ask permission from trees and forests before cutting them down. Given that there are strong customary laws and knowledge that local communities have developed for generations living with forests, they are capable of managing forest resources in a sustainable way (Tan *et al.* 2009; Wode and Huy 2009; Sikor and Tan 2011; CIRUM 2012). In addition, if community-based FG is promoted, particularly through legal and secure forest land title, the efficiency in terms of forest conservation and reforestation of the community-based FG regimes, is likely to be higher. This is because, with the legal title of forest land, local people have a legal basis to exclude outsiders from encroaching into their forests, obtain benefits from external sources such as PFES, or receive compensation for forest land if it is reclaimed by the state. The additional benefits brought by the legal title are likely to create more motivation for them to conserve forests (Tan *et al.* 2009; Sikor and Tan 2011). In summary, under the community-based FG, the forest is likely to be well-protected and secured for regeneration.

When looking at the forest LULC changes under the individual-based FG scenario, the results are also supported by the existing literature. The previous studies conducted in Vietnam showed that allocating forests and forest land to local individual households, and securing land tenure is not likely to be efficient in terms of forest conservation and reforestation (Sikor 2001; Clement and Amezaga 2008; Meyfroidt and Lambin 2008a, b). Privatisation of forests and securing tenure rights is not likely to result in sustainable forestry (McElwee 2004), and giving farmers and securing their forest land tenure may increase more pressure on forests (Angelsen and Kaimowitz 1999) and boost deforestation (Angelsen 1999). Recently, there has been hope that PFES programs, which have been implemented nationwide since 2010, will create more incentives for forest conservation and regeneration. However, the current setting for PFES in Vietnam has generated little motivation for forest conservation. This is due to the very low payments obtained from PFES, which is not enough to compensate for the high opportunity costs of protecting forests or securing fallows for the forest to regrow and not be converted for crops (McElwee 2012; Pham *et al.* 2013).

These findings regarding the changes in forest LULC under the state-based and community-based FG scenarios, therefore, support the current argument regarding the effects of these FG regimes on forest conservation and reforestation. The state-based FG arrangement has shown to be effective in terms of improvement of forest cover (Ferraro *et al.* 2012). The situation has not become worse as pointed by authors of several studies, such as Ascher (1995) and Ostrom (1999). The community-based FG regime is a good alternative because it is as efficient, if not better, in terms of improving forest cover. This result is supported by the arguments of many scholars that community-based FG is more effective in forest conservation (Ostrom 1990, 1999; Agrawal and Ostrom 2001; Brown *et al.* 2003; Ostrom 2005; Somanathan *et al.* 2009). There are also several credible pieces of evidence to show that community-based FG has had more positive environmental effects than state-based FG (Ellis and Porter-Bolland 2008; Baland *et al.* 2010; Bowler *et al.* 2010).

Figure 7.1 Maps of LULC patterns for the baseline in 2010 and under the three alternative FG scenarios for 2020



The spatial distribution of the forest LULC changes under the FG scenarios for 2020 are projected based on land suitability.[51] As shown in Figure 7.1, the maps of forest LULC under the state-based and community-based FG scenarios appear greener, indicating that forest cover is likely to increase significantly under these scenarios. Natural forest would tend to expand in high elevation areas, such as in the north-western part of this region, while planted forests would tend to increase in lower areas located in the south-eastern part. At the same time, the forest LULC map under the individual-based scenario looks almost unchanged compared to that of the baseline. These maps also suggest that the pattern of forest LULC change under the state-based scenario and under the community-based scenario is quite different. The figure depicts that under the state-based FG scenario, the new forest LULC is

[51] Maps of land suitability factors for each type of forest cover are depicted in Chapter 6 (Appendix 6.4a, b and c).

likely to develop more densely at the upper north-west part of the watershed, while under the community-based FG scenario, the new forest cover would expand more fragmentedly.

In summary, forest cover is likely to increase under all scenarios, but through different patterns. Of the three FG scenarios, community-based FG seems to be the best scenario for forest cover to expand, especially when it comes to regrowth forest and degraded/poor forest. In contrast, individual-based FG is likely to facilitate the increase in planted forest, while the increase in natural forest is likely to be very limited. However, the results also show that the quality of forest would be reduced, as the area of old growth forest is likely to decline under all scenarios.

The patterns of the LULC changes at the Da River watershed, where this study quantifies the provision of forest ecosystem services and estimates their values, are captured based on the changes of the whole region. Table 7.2 shows the patterns of forest LULC at the watershed. Natural forests are the main forest cover types, and bare land/shrub land is the second largest forest LULC.

Table 7.2 Forest LULC patterns of the Da River watershed at the baseline and under the alternative scenarios for 2020

Forest LULC	**Baseline 2010 (%)**	**Future Scenarios, 2020 (%)**		
		State-based FG	**Com.-based FG**	**Indi.-based FG**
1. Natural forest	44.39	50.22	49.72	44.96
1.1 Old growth forest	4.91	4.85	4.85	4.83
1.2 Degraded/poor forest	5.03	5.86	5.94	5.85
1.3 Regrowth forest	26.99	32.06	31.48	26.82
1.4 Other natural forests	7.46	7.46	7.46	7.46
2. Planted forest	1.95	4.69	3.89	4.18
3. Bare land/shrub land	35.53	26.96	28.27	32.73
4. Non-forest land	18.13	18.13	18.13	18.13

Similar to the whole region, the overall forest cover of the watershed is likely to increase under all scenarios. Under the state-based FG and community-based scenarios, all types of forest cover, except for old growth forests, are estimated to increase significantly. In particular, regrowth and degraded/poor forests are likely to transform to bare land/shrub land areas. Under the individual-based FG scenario, planted forests would grow rapidly, while natural forest cover is not likely to expand much. However, the quality of natural forest under this scenario would decline because old growth forests would be lost while the gain would come mainly from the expansion of degraded/poor forests.

7.3 The Provision of the Forest Ecosystem Services

7.3.1 The Provision of the Forest Ecosystem Services at the Baseline 2010

This watershed provides a substantial quantity of ecosystem services. With about 46.3% of the land being covered by forest in 2010, the estimated amount of carbon stored in this area was very large, varying from 257.29 to 278.55 million tonnes. On average, about 131 tonnes of carbon was contained in one hectare of forest land. My estimations are consistent with other studies conducted in tropical regions of Asia. Regarding carbon storage, our average carbon density in 2010 was higher than the average of 108 tonnes of carbon per hectare (tC/ha) for Vietnam forest land in 2005 (Meyfroidt and Lambin 2008b), 113 tC/ha in 2010 (Saatchi *et al.* 2011), 113 tC/ha for continental tropical Asia in 1980 (Brown *et al.* 1993) and 108 tC/ha for tropical Asia in 1995 (Houghton and Hackler 1999). The increase in carbon stock can be explained by the forest transition that started in the 1990s, in which both forest cover and carbon stock have gradually increased (Meyfroidt and Lambin 2008b; Clement *et al.* 2009; Meyfroidt *et al.* 2009).

This region is also an important upland watershed in the north of Vietnam. It reduces sediment load to reservoirs and provides water supply for hydropower generation that is critical for the development of the Northwest region in particular, and the country in general. The findings show that sediment yield was predicted at 18.26 tonnes/ha/year in 2010. Given the fact that the Northwest region has very high potential soil erosion caused by high average precipitation ranging from 1300 to 2200 mm/year and very steep upland areas (Tran and J Laituri 2011; Nguyen *et al.* 2013), this relatively low sediment yield is explained by the high proportion of forest cover. For example, the estimation of the sediment yield for different types of forest land in the Northwest region in 2009 by Nguyen *et al.* (2013) shows that the mean amount of sediment yield for forested areas is significantly lower than other types of cover. The average sediment yield of natural forests is 6.47 tonnes/ha/year, while that of grassland is 24.88 tonnes/ha/year, shrub land (20.67 tonnes/ha/year), and plantation forest (10.19 tonnes/ha/year).

When it comes to water provision, the volume of water derived from this watershed was estimated to be about 22,882 million cubic metres per year. On average, one hectare of land provides about 9,271 cubic metres of water a year. This high amount of water volume can be attributed to the high rainfall in this region.

7.3.2 The Provision of Services Under the Alternative FG Scenarios for 2020

This section compares the changes of the stock and the provision of the forest ecosystem services under the alternative FG scenario for 2020. As shown in Table 7.3, of the three FG scenarios, the state-based FG scenario is likely to produce the largest amount of carbon storage and sequestration, mainly because forest cover is likely to increase the most under this scenario. Carbon storage and sequestration would also increase substantially under the community-based FG scenario. Under the individual-based scenario; however, carbon storage would not increase much. These findings are expected because, under both the state-based FG and the community-based FG scenarios, areas of natural and planted forests are likely to increase significantly. Comparing these two scenarios, the total amount of carbon sequestration under the state-based FG scenario would be slightly higher than that of the community-based FG scenario. It is estimated that until 2020, the total amount of carbon sequestered would be about 9.47 million tonnes under the state-based regime, and about 8.63 million tonnes under the community-based scenario. On average, about 0.95 million and 0.86 million tonnes of carbon would be sequestered per annum, respectively. Under the individual-based FG, only a small percentage of forest cover would increase. Consequently, only 1.3 million tonnes would be sequestered by 2020, meaning that about 0.13 million tonnes of carbon would be captured per year.

The trend recorded for the sediment retention service is likely to be similar to that of the carbon sequestration service. Sediment retention is likely to be improved under the state-based and community-based scenarios. In contrast, under the individual-based scenario, the capacity of sediment retention is likely to reduce. In comparison with the baseline in 2010, under the state-based and the community-based scenarios, the capacity of sediment retention would increase by about 0.69 and 0.63 million tonnes per year in 2020, respectively. Conversely, under the individual-based scenario, the sediment retention ability would reduce by more than 1 million tonnes a year.

Table 7.3 The average provision of forest ecosystem services under the three FG scenarios

FES	For 2020				Compared to baseline in 2010			Comparison of the other alternative scenarios to the state-base scenario	
	(0) Baseline 2010	(1) State-based FG	(2) Com.-based FG	(3) Indi.-based FG	(1)-(0)	(2)-(0)	(3)-(0)	(2) - (1)	(3) - (1)
Total carbon storage (mil. tC)*	267.92 (10.63)	277.39 (10.12)	276.55 (10.18)	269.22 (10.20)	9.47	8.63	1.3	-0.840	-8.170
Average carbon storage (tC/ha)	130.93	135.55	135.11	131.56	4.62	4.18	0.63	-0.444	-3.989
Total carbon sequestration (compared to baseline) (mil. tC)	n/a	9.47 (2.39)	8.63 (2.03)	1.30 (0.88)	n/a	n/a	n/a	-0.840	-8.170
Average annual carbon sequestered (mil. tC/yr)	n/a	0.95	0.86	0.13	n/a	n/a	n/a	-0.08	-0.82
Sediment retention (mil. ton./yr)	1884.38	1,885.06	1,885.00	1,883.34	0.69	0.63	-1.05	-0.060	-1.733
Average sediment retention (ton./ha/yr)	760.42	760.70	760.67	760.00	0.28	0.25	-0.42	-0.024	-0.699
Sediment yield (mil. ton./yr)	45.24	44.55	44.61	46.29	-0.69	-0.63	1.05	0.060	1.733
Average sediment yield (ton./ha/yr)	18.26	17.98	18.00	18.68	-0.28	-0.25	0.42	0.024	0.699
Total annual volume of water (mil. m^3/yr)	22,882.40	22,137.60	22,241.80	22,603.11	-744.80	-640.60	-279.29	104.20	465.51
Average annual volume of water (m^3/ha/yr)	9,270.87	8,967.54	9,009.78	9,156.12	-303.33	-261.09	-114.75	42.24	188.59

Note: Standard deviation in parentheses

The expansion or loss of sediment retention capacity leads to the reduction or the rise of sediment exported to streams. On average, the sediment yield is likely to decline from 18.26 tonnes/ha/year in the baseline to 17.98 and 18 tonnes/ha/year in 2020 under the state-based and community-based scenarios, respectively. Under the individual-based scenario; however, the sediment exported would rise to 18.68 tonnes/ha/year in 2020. Comparing the quantity of this service between the state-based scenario (i.e. the business-as-usual scenario) and the alternative scenarios, the provision of sediment reduction service under the community-based scenario would be half a million tonnes per year lower, while the one under the individual-based scenario would be more than 1.7 million tonnes per year. Given that the estimated annual sediment yield is around 45 million tonnes, these differences in sediment reduction among the alternative scenarios range between 1.3% to 3.7%, which is not negligible, but it is not highly significant either.

On the other hand, the provision of water yield is projected to have a different trend. Water yield is likely to decrease under all scenarios. It is estimated that in 2020, the annual amount of water yield will range from 22,137.6 million cubic metres under the state-based scenario to 22,603.1 million cubic metres under the individual-based scenario. Compared to the baseline, the individual-based FG scenario would result in the least reduction of about 279.3 million cubic metres of water per year by 2020. The loss under the state-based scenario would be the greatest with 744.8 million cubic metres per year. Considering the differences between the state-based FG scenario and the alternatives, water yield provided under the community-based and individual-based scenarios would be about 0.5% and 2.1% higher than the yield under the state-based scenario, respectively. Although these differences in percentage are relatively small, the absolute differences are quite significant. Water yield under the community-based and individual-based scenarios would be 104.2 and 465.5 million cubic metres, which is greater than under the state-based scenario.

In summary, these findings are in line with those previously reported in the literature. They show that an increase in forest cover does not always lead to positive effects for all types of forest ecosystem services and that there may be a trade-off among them (De Groot *et al.* 2010). In addition, the changes in provision of forest ecosystem services depends not only on the sum of forest cover changes but also on the types of forest expansion (e.g., either through natural regrowth or plantation) and spatial patterns of forest cover changes (Meyfroidt and Lambin 2011). For example, forest expansion provides good environmental impacts, such as increased carbon storage and sequestration (Silver *et al.* 2000; Paul *et al.*

2002; Que *et al.* 2006; Khoa and Hai 2008; Chisholm 2010), but it often reduces the water yield derived from forests (Calder 2004; Phuong and van Dam 2005). Planted forests, especially of a monocultural type that are often the case in Vietnam (Van San and Gilmour 1999), usually bring negative environmental effects, such as reduced stream flows and increased soil erosion, particularly when they replace natural forests (Jackson *et al.* 2005; Sun *et al.* 2006; van Dijk and Keenan 2007; Clement and Amezaga 2008).

Given the patterns of forest LULC changes under the state-based and community-based FG scenarios, the forest cover expansion results are mainly due to the increase in regrowth and plantation forests, so the findings regarding the changes in the three forest ecosystem services can be expected. With the significant increase in natural forests, carbon storage, carbon sequestration and the reduction in sediment yield expected to increase, water yield is likely to decrease. Meanwhile, under the individual-based FG scenario, plantation forests are the major type likely to expand and more sediment yield is reasonable, with a slight increase in forest cover, while carbon storage, carbon sequestration and water yield are not expected to change much. This analysis of the changes in the provision of forest ecosystem services under different changes in forest LULC patterns stems from the alternative FG scenarios and therefore provides useful information for decision makers when considering options for managing forests and supplying forest ecosystem services.

7.3.3 Mapping Forest Ecosystem Services

This section presents maps for the provision of forest ecosystem services. As can be seen in Figures 7.2 to 7.5, the provision of the ecosystem services is clearly spatially differentiated. The provision of services is an output of the ecological functions underpinning the interaction between forest cover and the other biophysical components of forest ecosystems, so the spatial changes in forest cover strongly influence the provision of forest ecosystem services over the landscape.

Figures 7.2 and 7.3 depict the maps for carbon storage and carbon sequestration under the baseline in 2010, as well as the three future scenarios in 2020. Figure 7.2 shows the changes in carbon storage in the landscape between the baseline and the alternative future scenarios. The changes result from the process of carbon sequestration is mapped in Figure 7.3. It is easily seen in Figure 7.3 that the maps of carbon sequestration under the state-based and community-based scenarios depict that more carbon is captured under the two scenarios.

The maps also indicate that the patterns of carbon sequestration between the state-based scenario and the community-based scenario are quite different. Under the state-based scenario, carbon tends to be captured in the upper north-western part, while it is sequestered more in the lower north-western part, but is also spread more broadly within the region under the community-based scenario. These patterns of carbon sequestration are matched with the changes of the forest LULC as previously mapped in Figure 7.1.

Figures 7.4a and 7.4b display the key output maps of the sediment delivery ratio models. For each forest LULC scenario, the maps of potential soil loss, sediment delivery ratio and sediment yield, which is the total sediment exported from a pixel that reaches the stream, are presented. As shown in the maps, the potential soil loss under the baseline forest LULC as well as under the future forest LULC scenarios, is very high in this region. However, with a relatively low sediment delivery ratio, which is mostly less than 0.1, the sediment yield is relatively low in comparison with the total potential soil loss. These maps also show that in some areas, both the potential soil loss and the sediment yield are very high (i.e. the dark red areas represented in the maps are of potential soil loss and the bright red areas represented in the maps are of sediment yield). This information is useful for targeting areas to control soil erosion, as well as to reduce sediment yield.

The water yield represented in Figure 7.5 is highly and spatially differentiated, as water yield varies widely within the landscape. In general, the middle part of the region provides more water than the other areas, and the north-western part provides more water than the south-eastern part. It can also be seen that among the future scenarios, the patterns of water provision are slightly different. The maps show that under the state-based scenario, the north-western part of the region would provide less water, which possibly relates to more regrowth forest expanding in this part of the region.

Mapping the provision of forest ecosystem services provides useful information for decision making in forest LULC planning. This spatial information helps to incorporate the ecosystem services into policy and decision making (Tallis and Polasky 2009; Maes *et al.* 2012). For example, the information can be used to analyse synergies and trade-offs between the forest ecosystem services in a particular landscape (Nelson *et al.* 2009; Chisholm 2010; Raudsepp-Hearne *et al.* 2010) to compare the provision of these ecosystem services with the usage demand (Burkhard *et al.* 2012; Willemen *et al.* 2012), and can be utilised in monetary valuation on these forest ecosystem services (Gascoigne *et al.* 2011; O'Farrell *et al.* 2011).

Figure 7.2 Carbon storage at the baseline in 2010 and under the three alternative FG scenarios for 2020

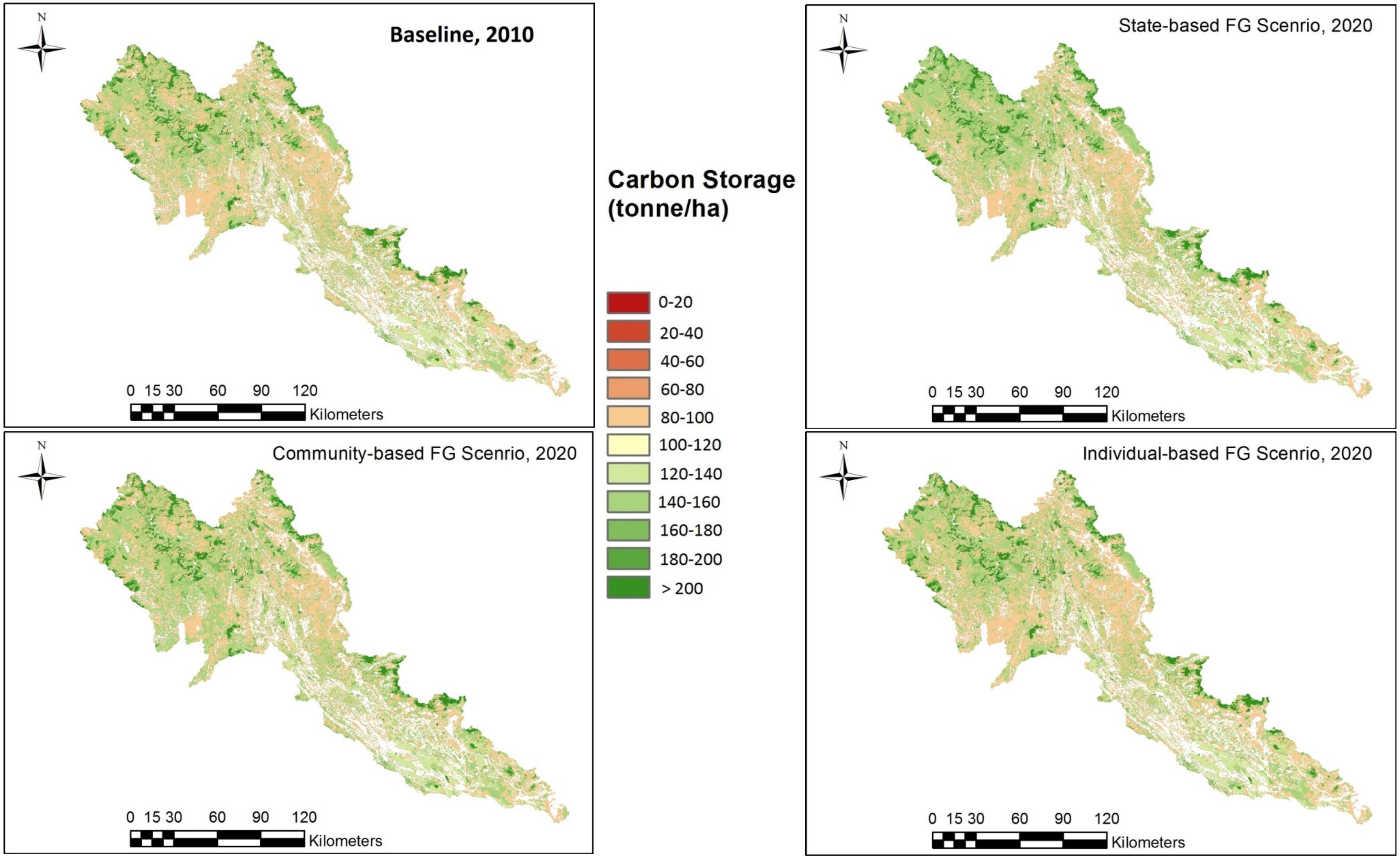


Figure 7.3 Carbon sequestration under the three alternative FG scenarios for 2020

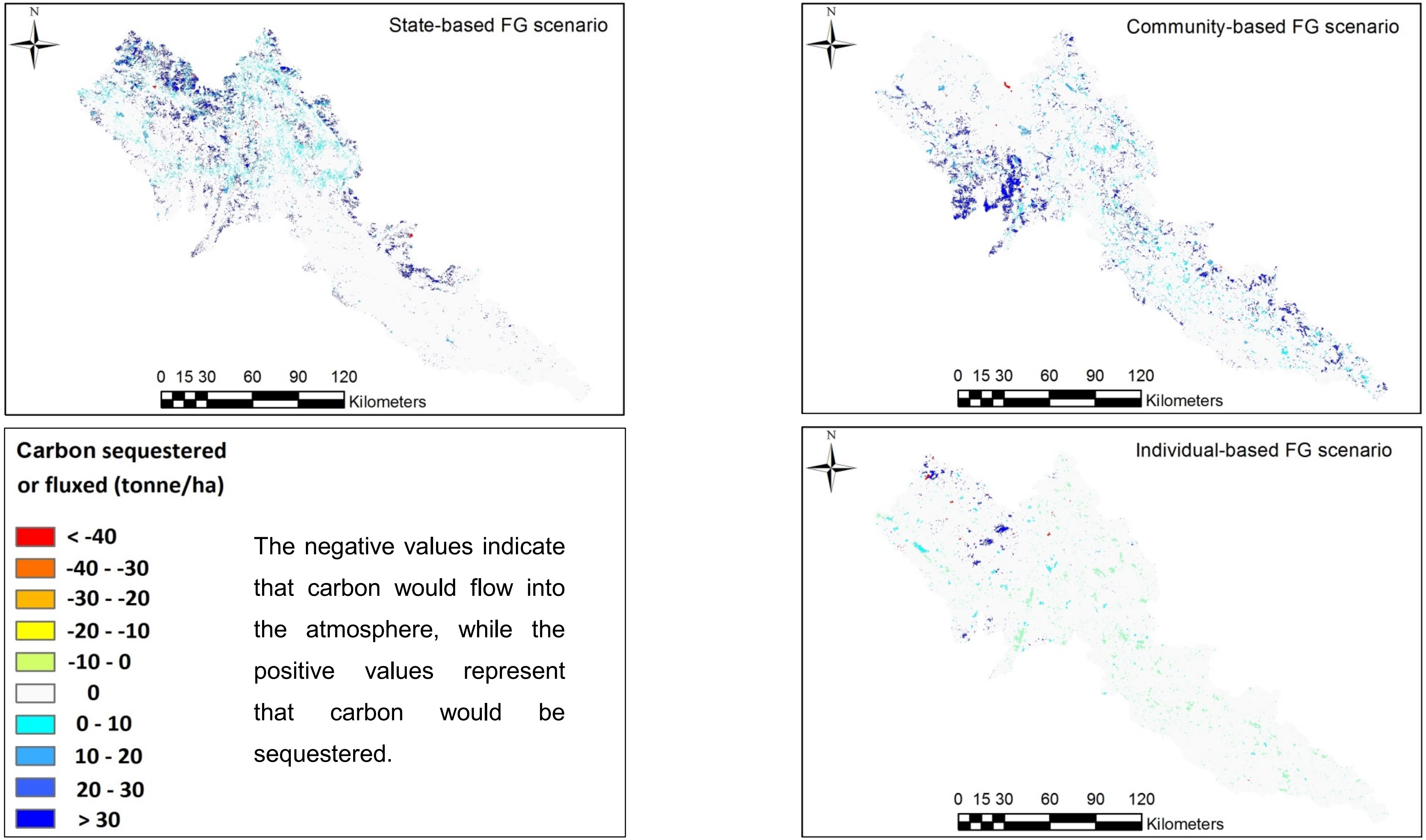


Figure 7.4a The total potential soil loss, sediment delivery ratio and the total sediment exported that reaches the stream (sediment yield*)* at the baseline in 2010 and under the state-based FG scenario for 2020

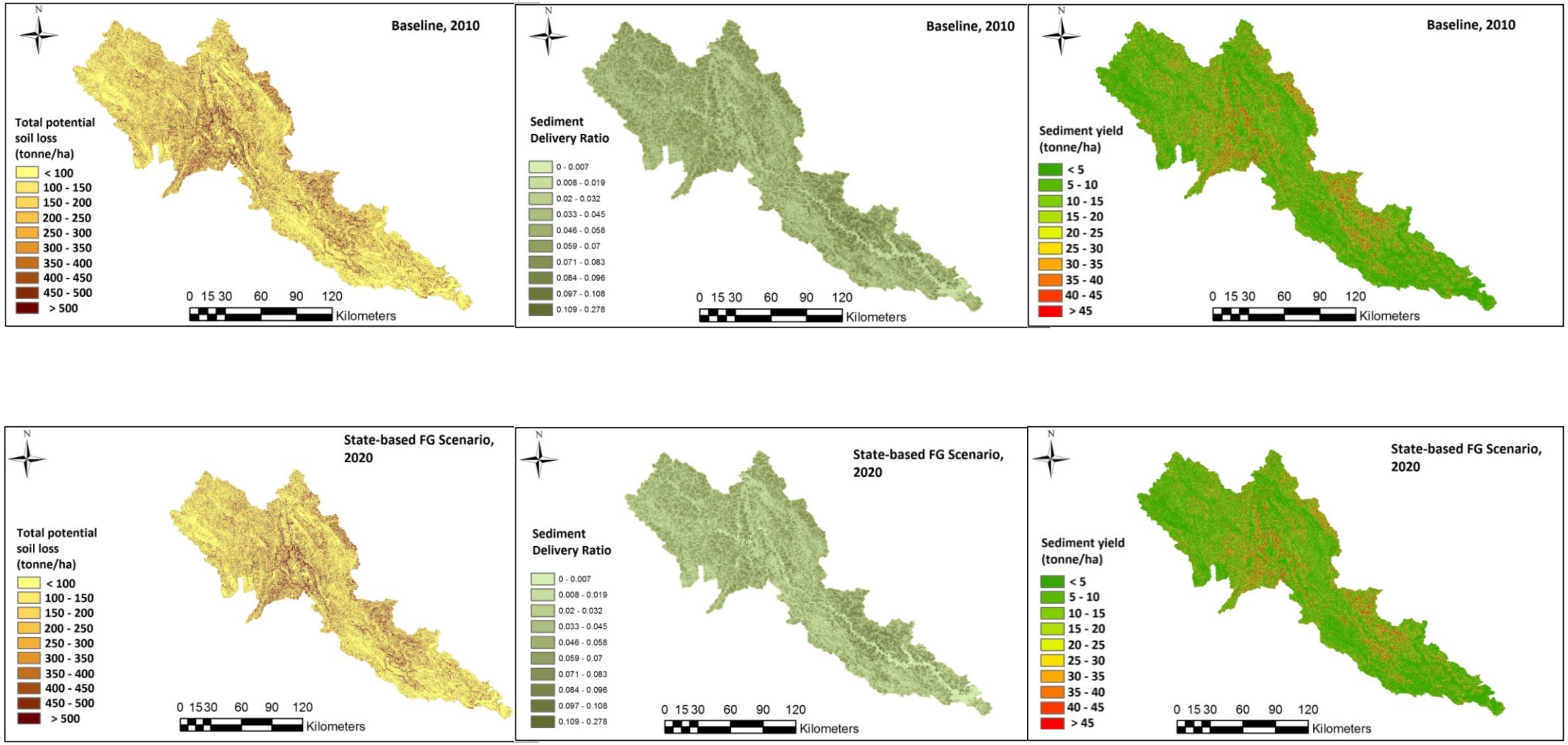


Figure 7.4b The total potential soil loss, sediment delivery ratio and the total sediment exported that reaches the stream (sediment yield) under the community-based FG and the individual-based FG scenarios for 2020

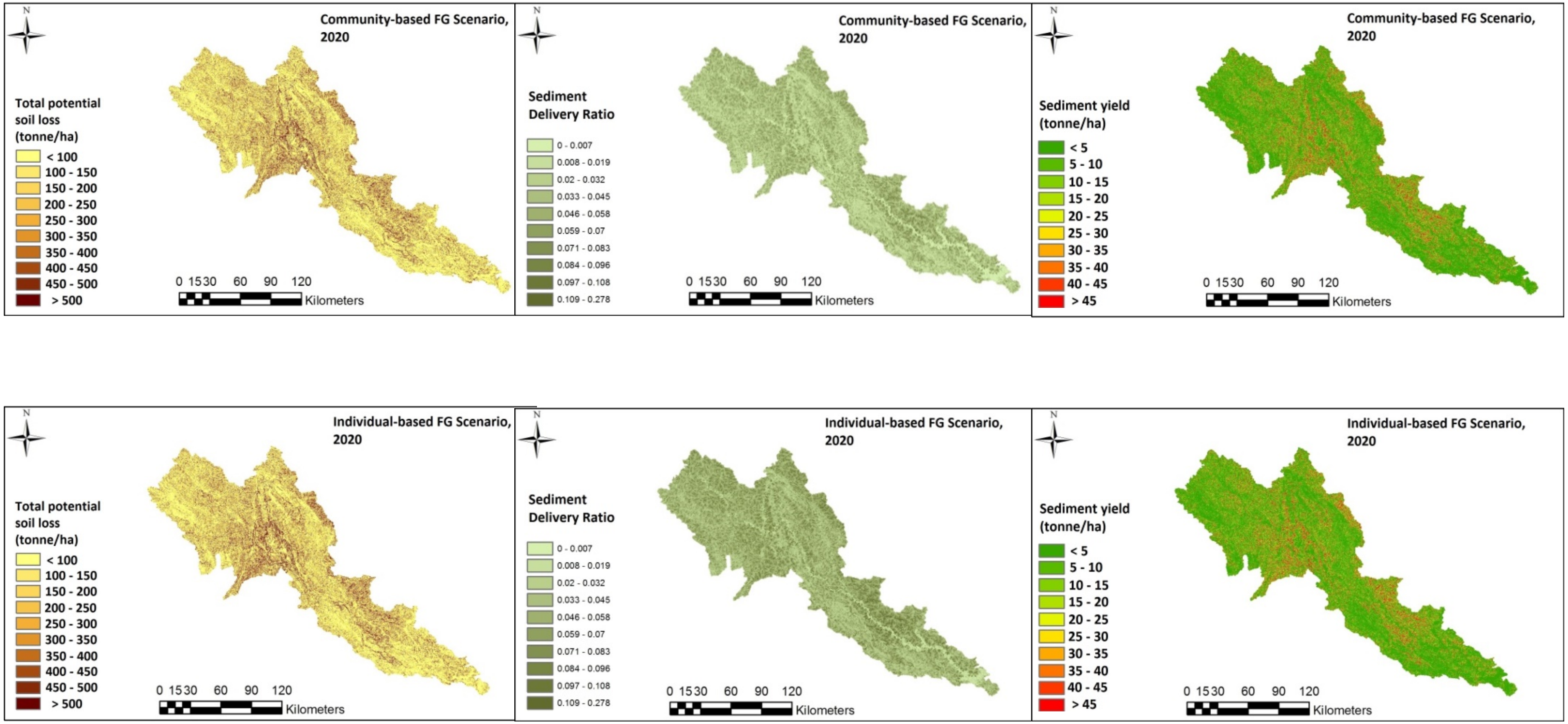
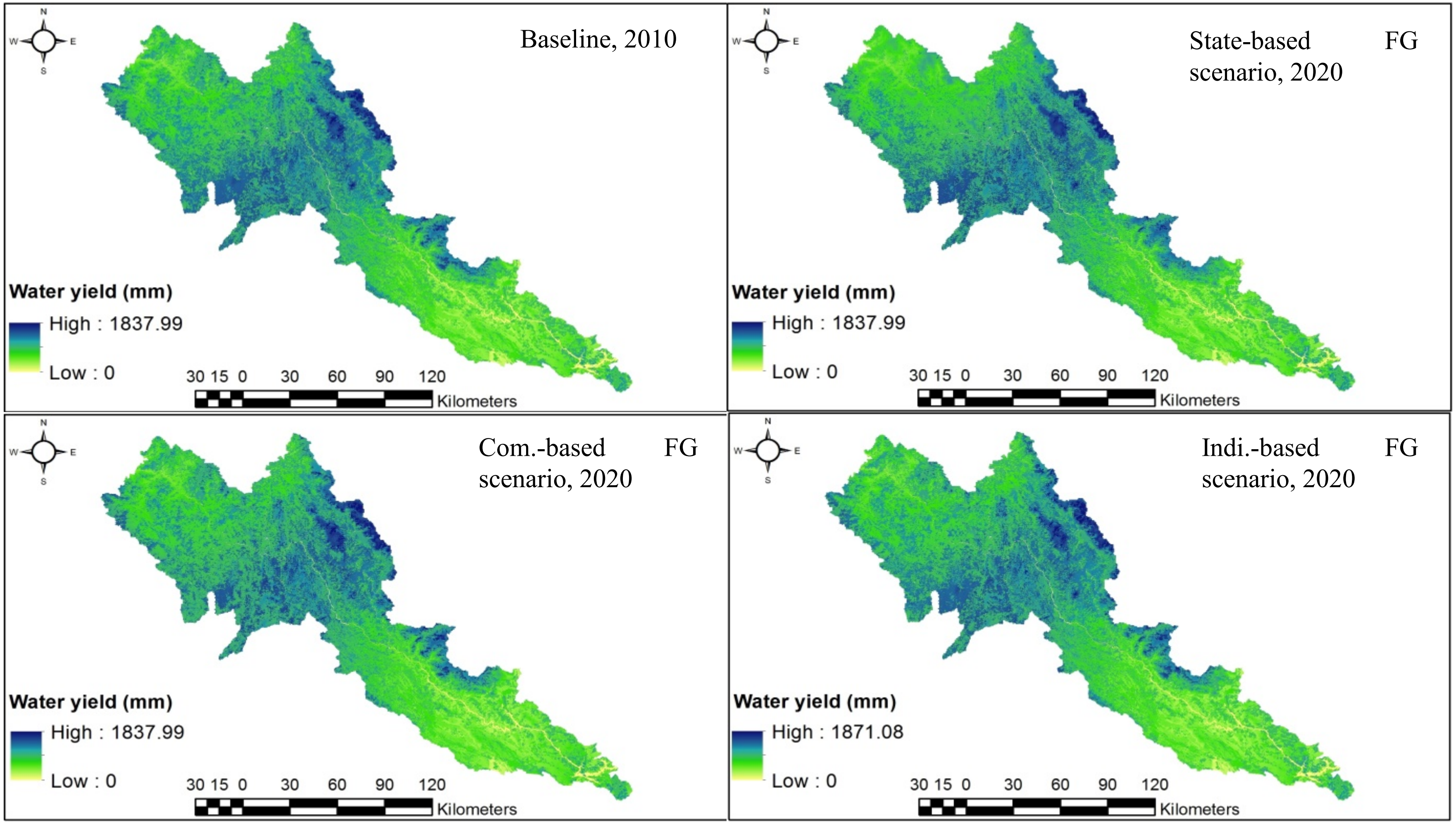


Figure 7.5 Water yield at the baseline in 2010 and under the three alternative FG scenarios for 2020



7.4 The Value of Forest Ecosystem Services Under Alternative Forest Governance Scenarios

7.4.1 The Value of Carbon Storage and Carbon Sequestration

The estimated value of carbon storage and sequestration in this region is substantial and is likely to vary under different FG scenarios in the future. The total value of carbon storage in the baseline of 2010 and for 2020 under the three alternative FG scenarios, is estimated based on the assumption that the social cost of carbon (SCC) will remain constant in the future. The values of carbon sequestration under these scenarios are also calculated. Table 7.4 shows the valuation results calculated at three levels of the SCC that are the lower level of 31.4 USD/tC, the upper level of 80.9 USD/tC, and the average level of 50 USD/tC. As shown in this table, if the average level of the SCC (i.e. 50 USD/tC) were applied, the average value of carbon storage would be around 13.396 million USD with a standard deviation of 531.5 million USD in the baseline 2010. As the carbon storage is likely to increase under all scenarios, the value of carbon storage in 2020 under all scenarios also increases. On average, the total value of carbon storage in 2020 would be 13,869.5, 13,827.5, and 13,461.0 million USD under the state-based, community-based and the individual-based scenarios, respectively. It looks like the values under these scenarios are not much different. However, considering the 10-year period from 2010 to 2020, the difference between the state-based FG scenario and the community-based FG scenario is about 42 million USD, and that between the state-based FG scenario and the individual-based scenario is 408.5 million USD and worth considering, particularly in the context of the developing country.

Regarding the total value of carbon sequestration in the period from the 2010 baseline to 2020, this is equal to the difference between the estimated quantity of carbon stored in 2020 and the quantity of carbon stored in 2010, multiplied by the social cost of carbon. As depicted in Table 7.4, the value of carbon sequestration is also calculated based on three levels of the SCC, including the lower level of 31.4 USD/tC, the upper level of 80.9 USD/tC and the SCC mean of 50 USD/tC.

The value of carbon sequestration also varies among the FG scenarios. using the mean value of 50 USD per tonne of carbon, the value of carbon sequestration under the state-based scenario would vary around the mean of 473.5 million USD, with a standard deviation of about 119.5 million USD. Under the community-based scenario, the mean would be 431.5 million USD (the standard deviation is 77.1 million USD), while under the individual-based scenario, it would be around the mean of 65 million USD.

Table 7.4 The value of carbon storage and sequestration

Unit: mil. 2010 USD

FG scenario	**SCC (USD/tC)**		
	Lower level of 31.4	**Average level of 50**	**Upper level of 80.9**
Total value of carbon storage			
(0) - Baseline, 2010	8,412.7	13,396.0	21,674.7
	(333.8)	(531.5)	(860.0)
(1) - State-based FG, 2020	8,710.0	13,869.5	22,440.9
	(317.8)	(506.0)	(818.7)
(2) - Com.-based FG, 2020	8,683.7	13,827.5	22,372.9
	(319.7)	(509.0)	(823.6)
(3) - Indi.-based FG, 2020	8,453.5	13,461.0	21,779.9
	(320.3)	(510.0)	(825.2)
Total value of carbon sequestration in the period from 2010 to 2020			
(1) – (0): State-based FG	297.4**	473.5**	766.1**
	(75.0)	(119.5)	(193.4)
(2) – (0): Com.-based FG	271.0**	431.5**	698.2**
	(63.7)	(101.5)	(164.2)
(3) – (0): Indi.-based FG	40.8**	65.0**	105.2**
	(27.6)	(44.0)	(71.2)
Differences in the value of carbon sequestration between the state-based scenario and the two alternative scenarios			
(2) - (1)	-26.4**	-42.0**	-68.0**
(3) - (1)	-256.5**	-408.5**	-661.0**

Note: Standard deviation in parentheses; ** indicates the differences that are statistically significant at 0.01 confidence level under a t-test.

Obviously, the difference in the value of carbon sequestration between the state-based FG scenario and the community-based FG scenario would be not be very much, while the difference between the state-based FG scenario and the individual-based FG scenario would be quite large. Assuming that carbon is linearly sequestered annually, the value of carbon sequestration under the state-based scenario would be about 4.2 million USD higher than that under the community-based FG scenario. Compared to the state-based FG scenario, with the individual FG scenario, the value of carbon sequestration under the individual-based scenario would be 40.85 million USD per annum lower. In the context of a developing country, particularly in the context of this research region, these differences cannot be neglected in forestry sector decision making.

Table 7.5 shows the estimated values of carbon sequestration per annum (i.e. in 2020), at various levels of the SCC under the three alternative FG scenarios. Under the assumption that carbon is sequestered at the same rate over time, in 2020 the total carbon sequestration would be equal to the average annual carbon sequestration during the period from 2010 to 2020. Using the mean of 50 USD/tC as the social value of carbon sequestration, in 2020 the total values of carbon sequestration under the state-based and community-based scenarios would be similar at about 47.35 and 43.15 million USD, respectively. Compared to those of 6.5 million USD under the individual-based scenario, the values under the state-based and the community-based scenarios would be significantly higher.

Table 7.5 The value of carbon sequestration in 2020

Unit: mil. 2010 USD

FG scenario	SCC (USD/tC) Lower level of 31.4	Average level of 50	Upper level of 80.9
(1) - State-based	29.74	47.35	76.61
	(7.5)	(12.0)	(19.3)
(2) - Com.-based	27.10	43.15	69.82
	(6.4)	(10.2)	(16.4)
(3) - Indi.-based	4.08	6.50	10.52
	(2.8)	(4.4)	(7.1)

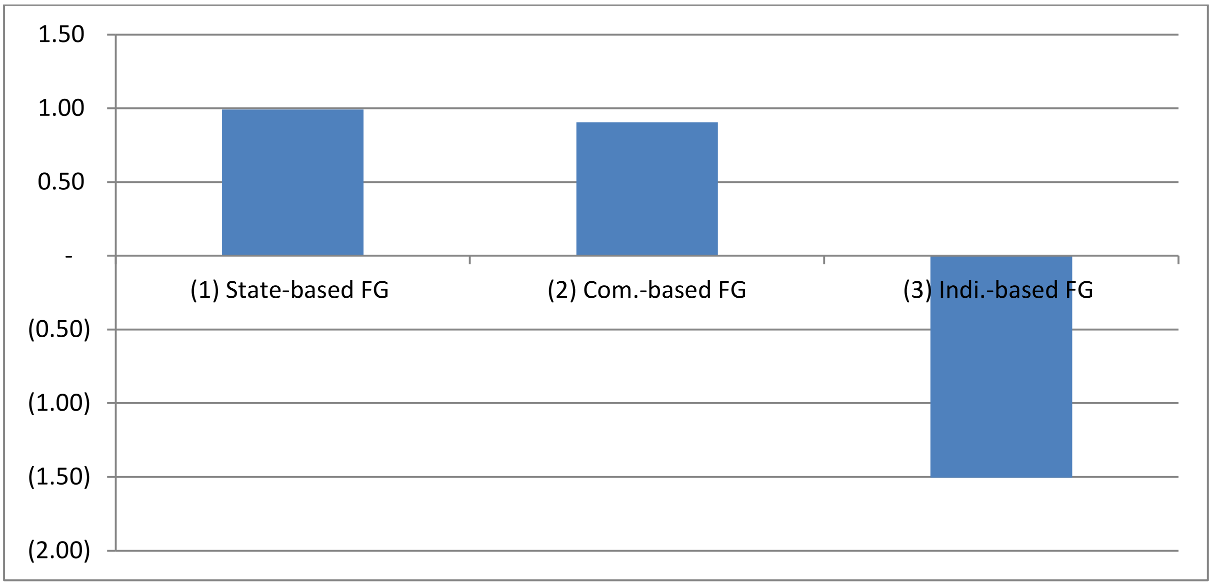
Note: Standard deviation in parentheses

7.4.2 The Value of Sediment Reduction Service

This section represents and compares the economic value of reducing sediment deposited in reservoirs under the alternative FG scenarios. The economic value of the sediment reduction service is estimated based on the replacement costs of removing sediment out of the reservoirs (i.e. 1.45 USD/tonne (Phuong 2009a)) and the changes in the quantity of sediment yield under the three alternative FG scenarios.

Compared to the cost of sediment removal at the baseline, the value of the reduction of sediment deposited to reservoirs is estimated to increase under the state-based and community-based scenarios that result from less sediment being exported to the reservoirs under these scenarios. As shown in Figure 7.6, in 2020, the annual benefits derived from reducing sediment yield under the state-based FG scenario would be about 1 million USD/year, and those under the community-based scenario, would be approximately 0.9 million USD/year. In contrast, under the individual-based scenario, which is likely to have more sediment deposited into the reservoirs, the value of sediment yield reduction would decrease. In other words, the cost of removing sediment exported to reservoirs would increase somewhat. It is estimated that about an additional 1.5 million USD/year will be required for removing the sediment under this scenario in 2020, in comparison with the costs required in the baseline.

Figure 7.6 The annual value of the reduction of sediment yield under the alternative scenarios (mil. 2010 USD)



In summary, regarding the value of the reduction of sediment yield, the positive gains of about 1 million USD per annum are expected under the state-based scenario and under the community-based scenario. In contrast, under the individual-based scenario, society is likely to be worse off as a result of the higher costs of about 1.5 million USD per year that would be required for sediment removal.

It is worth noting that, for simplicity, in the present study, the value of the reduction of sediment load to the reservoirs was estimated under the assumption that the amount of sediment accumulated each year is completely removed. However, in the complicated reality, totally dredging sediment deposited out of the reservoirs may not be a technically sound solution (Gunatilake and Gopalakrishnan 1999). Besides, in the cost-benefit analysis framework, there may be an optimal combination of dredging deposited sediment and reducing sediment load through watershed management (Palmieri *et al.* 2001; Pradhan *et al.* 2011). In other words, an optimal level of sediment removal is where the marginal costs of sediment removal equal the marginal benefits obtained from the reservoir services (Hansen and Hellerstein 2007). The benefits of the reservoir services include hydropower production (Nguyen *et al.* 2013), flooding mitigation and water regulation (Castelletti *et al.* 2012). Due to the lack of data relating to watershed management, as well as the benefits relating to reservoir services, this study has to rely on the proposed assumption. If this assumption does not hold, my results may underestimate or overestimate the true value of the sediment yield reduction service.

7.4.3 The Value of Water Supply for Hydropower Production

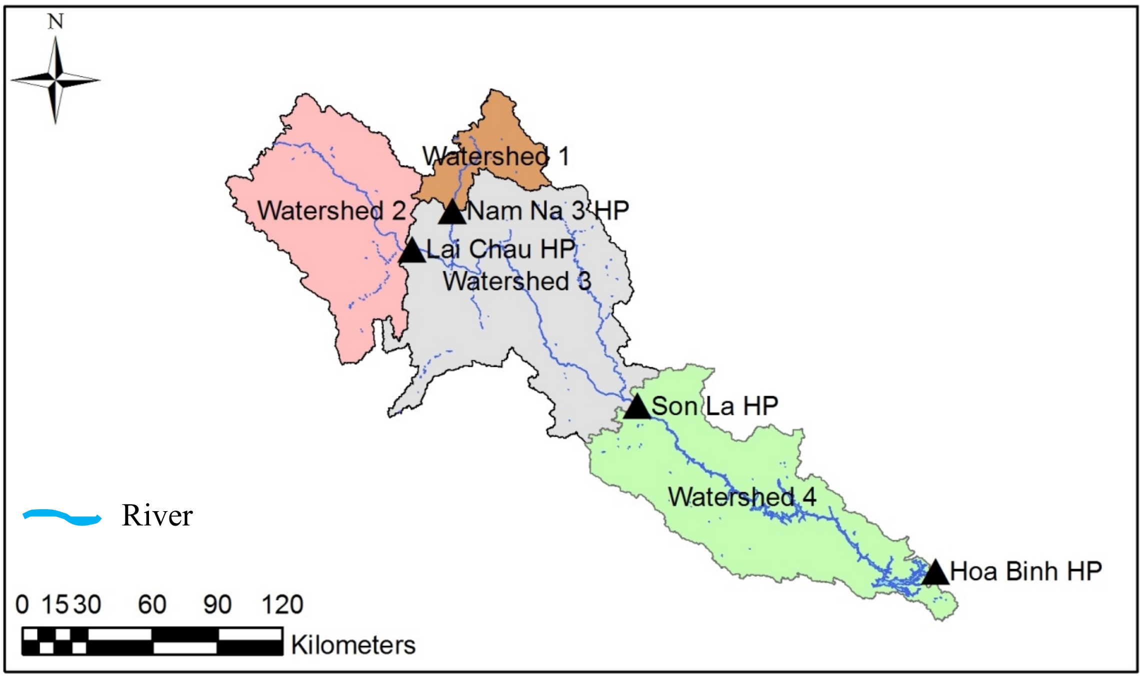
In the context of the research site, the value of water supply for hydropower production is dependent on how efficiently that water is used. The efficiency of using water is not only determined by the efficiency of a particular hydropower plant, but also by the number of times that water is utilised for generating electricity in plants downstream. The efficiency of water usage is determined by the proportion of the total water volume that is used for generating electricity, and the efficiency of a hydropower plant is reflected by how many cubic metres of water are necessary to produce one kWh of electricity. Since there are four large hydropower plants operating in this region (see Figure 7.7), water that is derived from the upper watersheds can be used multiple times for producing electricity at different hydropower plants downstream.[52]

By applying the residual valuation method explained in Chapter 6, the value of the marginal product of water for energy production was calculated. Given the technical and

[52] In fact, there are many other hydropower stations that have been built in this region. However, most of them are small stations and their technical economic information is difficult to obtain. Therefore, this study only focuses on the four largest stations.

economic characteristics of the hydropower plants, the value of the marginal product of one cubic metre of water used for hydropower generation is 0.067 US cents, 0.154 US cents, 0.541 US cents and 0.348 US cents at Nam Na 3, Lai Chau, Son La and Hoa Binh hydropower plants, respectively. Since not all of the total water volume is used for hydropower generation, the value of one cubic metre of water is measured by the fraction of the water used. In addition, as described above, water derived from the upstream watersheds is used multiple times at the downstream hydropower stations. In particular, as presented in Figure 7.7, the water yield derived from watershed 1 is used for hydropower generation three times at Nam Na 3, Son La and Hoa Binh hydropower plants. Water yield that comes from watershed 2 is also used for generating electricity three times, but at the different hydropower plants: Lai Chau, Son La, and Hoa Binh. The water yield from watershed 3 is used at Son La and then Hoa Binh hydropower plants. On the other hand, water yield coming from watershed 4 is only used at the Hoa Binh hydropower plant. The aggregate value of one cubic metre of water derived from the different watersheds was therefore calculated based on the fraction of the water used for hydropower generation, the value of the marginal product of water for generating electricity and the number of times that water was used at different plants. The estimated values are given in Table 7.6. For example, the value of one cubic metre of water derived from watershed 1 was 0.872 US cents, while the values derived from watershed 2, watershed 3 and watershed 4 were 0.939 US cents, 0.808 US cents and 0.299 US cents, respectively.

Figure 7.7 The four large hydropower plants (HP) in the Watersheds



At the baseline in 2010, only the Hoa Binh hydropower plant was operating and, therefore, the value of the marginal product of water for the entire watershed was equal to 0.299 US cents per one cubic metre, as calculated for watershed 4 shown in Table 7.6. Consequently, the total value of water at the baseline was 68.42 million USD for the total volume of water of 22,882.4 mil. m^3 derived in that year (see Table 7.7).

Table 7.6 Value of the marginal product of water provided by different watersheds

Watershed	**Marginal value of water at different hydropower plants, (US cent/m^3)**				**The aggregated value of one cubic metre of water (US cents/m^3)**
	Nam Na 3	Lai Chau	Son La	Hoa Binh	
Fraction of water used	0.96	0.85	0.94	0.86	
Watershed 1	0.067	-	0.541	0.348	0.872
Watershed 2	-	0.154	0.541	0.348	0.939
Watershed 3	-	-	0.541	0.348	0.808
Watershed 4	-	-	-	0.348	0.299

The situation in the future is going to be very different because the four large hydropower plants, Hoa Binh, Son La, Lai Chau, and Nam Na 3, will be fully operational in 2020.[53] As presented in Table 7.7, of the three FG scenarios, under the individual-based scenario, the value of water would be the greatest. The water value would be 161.5 million USD a year under this scenario. Under the state-based and the community-based scenarios, the value would be 157.3 and 159 million USD, respectively. The differences in the value of water between the state-based scenario and the other two scenarios would not be very large, but are worth considering. Compared to the community-based scenario, the annual value of water under the state-based scenario would be nearly 1.7 million USD lower. The difference in the annual value of water between the state-based and the individual-based scenarios would be somewhat greater, and the annual value under the individual-based scenario would be some 4.2 million USD higher. These findings are included in the discussion regarding the impacts of forest expansion on water supply for hydropower generation. Some authors find that reforestation provides better water regulation services to generate more electricity in

[53] All the four hydropower plants are operating by the end of 2016.

China (Guo *et al.* 2000; Guo *et al.* 2007). Other studies, such as Beck *et al.* (2013) conducted in Puerto Rico, and Phuong and van Dam (2005) argue that there is no significant impact of change in forest cover on the stream flow in the tropical watersheds. In the context of this research region, with the presence of several large reservoirs for hydropower generation and water regulation, less water yield caused by forest expansion (Calder 2002; Calder 2004; Sun *et al.* 2006; van Dijk and Keenan 2007) likely reduces the production of the hydropower plants. Therefore, I argue for the position that forest regeneration and plantation lead to the reduction of benefits in terms of water supply for hydropower generation. However, this argument should be cautiously interpreted with the linkages with other hydrological services, such as soil erosion and sediment exportation. As previously represented in section 7.3.2, forest expansion, particularly through natural regeneration, often helps to prevent soil erosion and sediment deposits in the reservoirs, which results in lower costs in operating the hydropower plants, particularly those associated with the costs of sediment removal.

In comparison with the baseline in 2010, the results show a significant increase in the value of water supply for hydropower generation. The value of water in 2020 would be more than two times higher than the value of 2010 under all scenarios. This increase in the value of water would mainly result from the fact that the water in 2020 would be used more efficiently with four hydropower plants being operational, versus only one in 2010. The findings confirm the argument in the literature that the economic values relating to hydrological services are highly specific over the landscape and over time (Ferraro *et al.* 2012; Ninan and Inoue 2013). That is, the economic benefits derived from water yield depend on the relationship between the provisions and human activities, which is downstream hydropower generation, in the case of this study. As depicted in Figure 7.8, different areas provide different values of water. The maps show that in 2020, the upstream areas in the north-western part of the region are darker green, indicating that these areas will provide more value in terms of water supply for hydropower generation. In particular, the value of water derived from one hectare in the upstream areas would be higher than 40 USD/ha, with considerable areas expected to create more than 80 USD/ha. On the other hand, one hectare of land in the downstream area would create less than 40 USD/ha.

Figure 7.8 The value of water supply for hydropower production under the baseline, 2010 and under the alternative FG scenarios for 2020

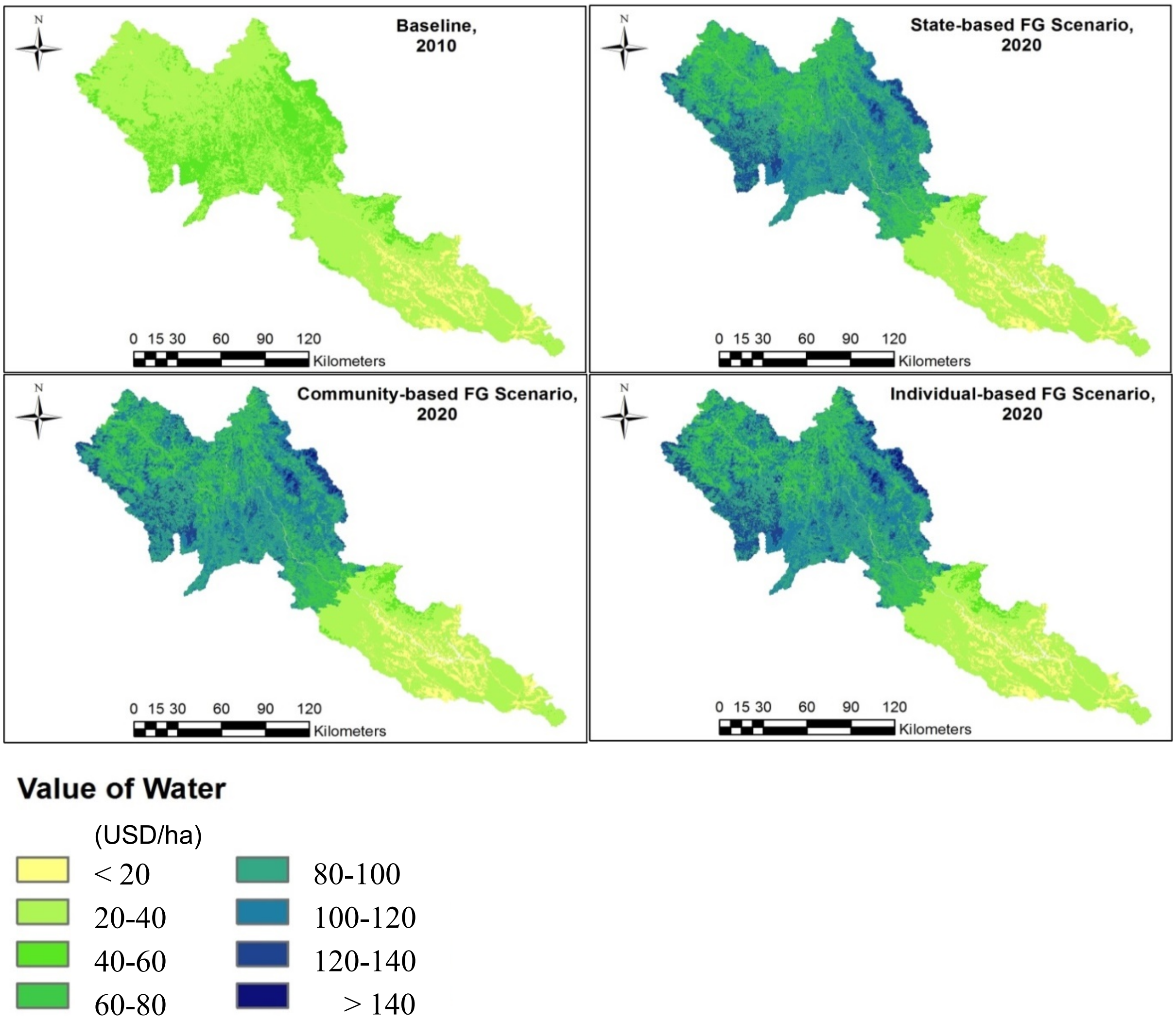


Table 7.7 The total value of water for hydropower production under the three FG scenarios for 2020

Watershed	Total water yield (mil.m^3)				Total value of water supply for hydropower production for 2020 (mil. 2010 USD)				Comparison of alternative scenarios to the state-based scenario (mil. 2010 USD)	
	Baseline	State-based FG	Com.-based FG	Indi-based FG	(0) Baseline	(1) State-based FG	(2) Com.-based FG	(3) Indi-based FG	(2) - (1)	(3) - (1)
Watershed 1	-	1,377.70	1,453.70	1,464.50	-	12.0	12.7	12.8	0.66	0.76
Watershed 2	-	5,246.50	5,389.40	5,469.10	-	49.2	50.6	51.3	1.34	2.09
Watershed 3	-	9,755.30	9,758.20	9,934.83	-	78.8	78.8	80.3	0.02	1.45
Watershed 4	-	5,758.10	5,640.60	5,734.68	-	17.2	16.9	17.2	-0.35	-0.07
Total	22,882.40	22,137.60	22,241.90	22,603.11	68.48	157.3	159.0	161.5	1.68	4.23

7.4.4 Summary of the Differences in the Value of the Forest Ecosystem Services Between the State-based FG Scenario and the Two Alternative FG Scenarios

For the purpose of aiding decision makers in making an effective choice between the alternative FG scenarios, it is essential to summarise the differences in the values of all of the forest ecosystem services of interest. This is because it would be easier for policymakers to choose the most effective possible option, given the specific context of the research region, when all the benefits of these forest ecosystem services are measured with a single indicator.

Table 7.8 The differences in the annual value of forest ecosystem services under the alternative FG scenarios in 2020

Unit: mil. USD

Differences in:	**Com.-based Vs State-based Scenario**	**Indi.-based Vs State-based Scenario**	**Indi.-based Vs Com.-based Scenario**
The value of carbon sequestration*	-4.2	-40.9	-36.6
The value of reducing sediment yield	-0.1	-2.5	-2.4
The value of water supply for hydropower generation	1.7	4.2	2.5
Aggregate difference	**-2.6**	**-39.2**	**-36.5**

Note: * The value of carbon sequestration was calculated by using the SCC of 50 USD/tC

Table 7.8 summarises the differences in the annual value of the three services of interest in this study. These differences were calculated from three components, including the values of carbon sequestration, sediment yield reduction and water supply for hydropower generation. As shown in this table, the annual value of forest ecosystem services in 2020, under the two alternative scenarios, would be lower than those under the state-based scenario (the business-as-usual scenario). Particularly, the annual value under the individual-based scenario would be 39.2 million USD less than that under the state-based scenario, which is a significant difference. The estimated gap between the forest ecosystem service value under the state-based scenario and under the community-based scenario would be much narrower, with only a 2.6 million USD difference. Comparing the individual-based scenario with the community-based scenario, the values obtained under the former scenario would be considerably lower at 36.5 USD per year. The results also show that the value of carbon sequestration and sediment reduction under the state-based scenario would be higher than those under the two alternative scenarios, while the value of water under the two alternative scenarios would be higher than that under the state-based scenario.

It is worth noting that these differences in the value of the services were calculated based on the assumption that the unit values of these services would remain constant in the future, and the SCC would be 50 USD/tC. In fact, whether these unit values will increase, decrease, or remain unchanged is uncertain, and the exact SCC is very difficult to estimate (Tol 2008; Nelson *et al.* 2009). Therefore, it is crucial to understand the assumptions and the modelling of the changes in the unit values. For example, if the SCC and the cost of dredging sediment dropped very sharply and the marginal product of water increased very significantly, it could be possible to observe reverse results, meaning that the aggregate values of these services under the community-based scenario and under the individual-based scenario would be higher than those under the state-based scenario.

7.5 Contributions and Limitations of the Study

7.5.1 Contributions: Extending the Debate on Forest Governance

The present findings that consist of mapping the changes in forest LULC, quantifying and valuing forest ecosystem services under the alternative forest governance scenarios, extend the debate on the impacts and effectiveness of FG regimes. Existing discussions over the efficiency of FG regimes are mostly concerned with forest conservation or sustainable use of forests (Ostrom 2005; Berkes 2007) that are measured through forest conditions (e.g., forest cover, forest biodiversity or forest vegetable density (Ascher 1995; Agrawal and Chhatre 2006; Pagdee *et al.* 2006)). Some recent studies added other aspects, such as social performance (e.g., social equity, livelihood of local dwellers, poverty reduction (Ostrom 2007; Tan *et al.* 2009; Persha *et al.* 2011; Lambini and Nguyen 2014)), local rights, participation, democracy (Sikor and Tan 2011; Phuc *et al.* 2013), and cost effectiveness regarding implementing REDD (Skutsch and Ba 2010; Palmer Fry 2011) in the outcomes of forest governance arrangements. Therefore, these findings of the provision and value of forest ecosystem services under alternative forest governance provide additional and crucial information for decision making in forest management and, consequently, the sustainable use of forests.

Firstly, the results of mapping the changes in forest LULC under the alternative FG arrangements more deeply articulate the outcomes of the FG regimes in terms of forest conditions. My findings show that the improvement in forest cover does not ensure the increase in the provision of all forest ecosystem services. There are trade-offs among the services. For example, the expansion of forest cover leads to an increase in carbon storage and sequestration. During the period from 2010 to 2020, with the better expansion of forest cover under the the state-based and the community-based scenarios, on average, about 0.95 million and 0.86 million tonnes of carbon would be sequestered per annum, respectively. Meanwhile, under the individual-based FG, forest cover is not likely to improve much, about 0.13 million tonnes of carbon would be annually captured. In contrast, water supply to hydropower generation is likely to reduce as the forest cover increases. For instance, compared to the baseline, the loss under the state-based scenario would be the greatest with 744.8 million cubic metres per year while the loss under the individual-based FG scenario would be the smallest of about 279.3 million cubic metres of water per year by 2020. Regarding the service of reduction of sediment yield, the findings show that the total amount of sediment deposited into the reservoirs is likely to reduce under the state-based and community-based FG scenarios, but to grow under the individual-based scenario, although forest cover is likely to increase under all scenarios. Particularly, on average, the sediment yield is likely to decline from 18.26 tonnes/ha/year in the baseline to 17.98 tonnes/ha/year under the state-based and 18 tonnes/ha/year and community-based scenarios in 2020. Meanwhile, under the individual-based scenario, the sediment exported would rise to 18.68 tonnes/ha/year. By mapping the pattern of the forest LULC, this study provides the predicted changes in the provision of forest ecosystem services over the landscape in accordance with the particular changes in forest LULC patterns. This is critical for spatial planning in forest LULC (Tallis and Polasky 2009; Maes *et al.* 2012). The information of spatial provision is also essential for valuing these services because this is determined by the interaction between the provision and the human utilisation of the services over time and space (Chisholm 2010; Raudsepp-Hearne *et al.* 2010; Burkhard *et al.* 2012; Willemen *et al.* 2012). For instance, my results demonstrate that the value of one cubic metre of water is spatially different as shown in Figure 7.8. Specifically, the value of one cubic metre of water derived from watershed 1 is estimated at 0.872 US cents, while the values derived from watershed 2, watershed 3 and watershed 4 are about 0.939 US cents, 0.808 US cents and 0.299 US cents, respectively.

Secondly, by estimating the provision of the forest ecosystem services and the economic values of these services, the study sheds new light on the effectiveness of the state-based FG, as well as its alternatives. The findings show that individual-based FG regimes, which are based on the privatisation of state forests, market-led mechanisms of forest property rights and market-based instruments for FPES, are not likely to be successful in comparison with the state-based and community-based FG regimes. More precisely, the annual value of the three services under the individual-based scenario would be 39.2 and 36.5 million USD lower than that under the state-based and community-based scenarios, respectively. Comparing the community-based FG regime to the state-based FG regime shows that the provision and the economic value of the forest ecosystem services under the two FG regimes are likely to be almost similar. The difference between these two scenarios is only a 2.6 million USD per year.

Integrating the findings of the present study with the previous argument regarding the impacts and effectiveness of the state-based FG and the community-based FG provides sufficient information for decision making regarding forest sustainability. The current state-dominated FG has been criticised for a number of shortcomings. The major drawbacks of the state-based FG include: too much bias in the state sector, a high financial burden on the state budget for forest protection and reforestation (Clement and Amezaga 2008; Phuc 2009; Phuc and Nghi 2014), stress on forest conservation at the expense of the livelihoods of local people (Sikor and Tan 2011), lack of participation of local people in forest management (Clement and Amezaga 2008). As a consequence, there are a large number of conflicts between government policies and local perceptions of forest land tenure and use rights (Clement and Amezaga 2008) that lead to further conflicts and disputes between local communities and state organisations regarding forest use rights in the uplands (Phuc *et al.* 2013). These shortcomings may put sustainable forest management at risk. In contrast, community-based forest FG regimes have been argued to provide many advantages. They can improve the livelihoods of local people, particularly the poor (Tan *et al.* 2008b; Sikor and Tan 2011), make use of local participation in forest protection (Phuc 2009) and often result in a reduction of the financial burden on the state budget for forest protection and development (Clement and Amezaga 2009). Therefore, I argue that community-based FG regimes can be an alternative to the current state-dominant regime because they can provide livelihoods to local people, forest conservation and environmental values all at the same time and at a lower cost.

7.5.2 Limitations of the Study

The main limitations of this study relate to the limitations of the InVEST models that were used to estimate the provision of forest ecosystem services. Although this model package is one of the most advanced modelling tools and has become popular in the academic community (Nelson *et al.* 2009; Tallis and Polasky 2009; Nelson and Daily 2010; Bagstad *et al.* 2013), it has some shortcomings. The models are designed to deal with the problem of the lack of data in ecological structures and processes, especially in developing countries; for example, in the case of Vietnam. With this present design, the model faces some limitations. For example, by oversimplifying the carbon cycle, the carbon storage and sequestration model ignores the natural processes of gain and/or loss of carbon that may result in the underestimation or overestimation of carbon storage and sequestration. The major drawback of the sediment delivery model is that it relies on the USLE equations that only capture rill/inter-rill erosion process. Other sources of sediment such as gully erosion, streambank erosion, and mass erosion are not taken into account in this model. Besides, this model is very sensitive to the variation in the k and IC_0 parameters. The default values of these parameters suggested by (Vigiak *et al.* 2012) are used in this study. Meanwhile, the main shortcoming of the water yield model is that it is based on annual averages. This means that it ignores the seasonal variation of water flows that may influence the value of water. These limitations will be minimised with further development of the models to suit future conditions.

Additionally, due to the limited resources of people and time, this study still values the three major use value types provided by the forest ecosystems (i.e. carbon storage/sequestration, water supply for hydropower and reduction of sediment load to the reservoirs) that represent a significant proportion of the forest ecosystem services (Ferraro *et al.* 2012; Ninan and Inoue 2013; Brandon 2014). However, it certainly does not cover the total economic value derived from forest ecosystems. Particularly, some important services such as conservation of biodiversity, water purification, recreation, and tourism are missing.

In addition, the limitations of the study are associated with the uncertainty of the estimation of transition livelihood of LULC change that relies on the experts' perceptions. In spite of being indispensable and effective to deal with the complex problems of LULC change (Kok *et al.* 2015), and able to link stakeholders and scientists, story and simulation (Shaw *et al.* 2009; Reed *et al.* 2013), the participatory scenario development approach used in the study has faced several drawbacks. That is because the transition livelihoods derived from experts' knowledge are the outcomes of individual expert's perception. In addition, the expert's knowledge associated to the transition possibilities reflects their understanding, beliefs, and dreams regarding the future of LULC change. Therefore, there is the possibility of biases. Particularly, since the majority of the experts have worked for the government organizations at provincial level which are in charge of forest management and the state research institutions, they may prefer the state-based forest governance arrangement to the alternatives.

The last and maybe the most difficult limitation to be solved in this present study is the uncertainty of economic variables used for estimating the economic values of the forest ecosystem services. Specifically, the social cost of carbon (SCC) is very uncertain (Tol 2008, 2009). Estimates of the SCC are very complex and faced with a number of uncertainties (Gillingham *et al.* 2015; Nordhaus 2017). The complexity of the SCC estimation comes from the fact that an attempt to calculate the SCC must involve in the understanding of the full array of emission impacts, through carbon cycle, climate change, and the economic damages caused by the climate change (Nordhaus 2014, 2017). The uncertainties range from those relating to economic and population growth to the carbon cycle, climate change, damages, and the assumption of the discount rate used (Anthoff and Tol 2013; Gillingham *et al.* 2015). Consequently, the estimated values of the SCC vary widely (Tol 2009; Nordhaus 2011, 2014; National Academies of Sciences 2017). For instance, the most up-to-date estimation of US Interagency Working Group for the global SCC for 2020 using DICE-2010, PAGE, and FUND models and the discount rate of 3 percent, are \$40, \$74, and \$22 per ton of in 2010 USD, respectively, that result in average of \$45/t$C0_2$ (equivalent to \$165/tC). Meanwhile, the estimate by the latest DICE-2016R model is approximately twice of about \$87/t$C0_2$ (equivalent to \$319/tC) (Nordhaus 2017). The difference in the estimated values of the SCC is even much wider if different discount rates are used (Nordhaus 2007; Tol 2008; Anthoff and Tol 2013; Gillingham *et al.* 2015). The present study, instead of attempting to estimate the SCC, used the average SCC of \$50/tC, that is derived from Tol (2009). This estimate of the SCC is lower than those estimated by recent studies (Nordhaus 2017). Regarding the economic value of water yield, the real price of electricity that was used to estimate the marginal product of water supply for hydropower generation can also change over time. This may lead to the variation of the value of water yield in the future. The uncertainty of these economic variables would result in variations in the estimated economic values of the forest ecosystem services under the alternative scenarios.

7.6 Conclusion

In conclusion, this chapter has presented the findings and discussions regarding the changes in forest cover patterns and the provision and values of the three forest ecosystem services: carbon storage/sequestration, provision of water for hydropower generation and reduction of sediment exported to reservoirs under the three alternative forest governance scenarios. It shows that of the three FG arrangements, the individual-based FG is not likely to be as effective as the other two in terms of improving forest cover, the provision and economic values of the forest ecosystem services. On the other hand, under the community-based FG scenario, the expected outcomes are likely to improve as much as those under the state-based scenario. In combining the other advantages of the community-based FG with the shortcomings of the current state-dominant FG (e.g., the improvement of local people's livelihoods, poverty reduction, participation and the rights of local people regarding forest management and usage etc.) this study argues that the community-based FG regime could be a good alternative to replace the current state-based FG regime. Therefore, although there are still some shortcomings relating to methodology issues, the results shown in this chapter significantly contribute to the debate in the literature regarding the impacts and efficiencies of the FG arrangement.

Chapter 8
CONCLUSIONS AND POLICY IMPLICATIONS

This study aimed to determine how the provision and the values of forest ecosystem services would be affected by alternative forest governance regimes in the Northwest region of Vietnam. Specifically, it aimed: (1) to determine and describe feasible alternative forest governance regimes for this region; (2) to quantify and map the forest cover changes under the alternative regimes, and (3) to quantify and evaluate the changes in economic values of the three major forest ecosystem services: carbon storage/sequestration, the provision of water for hydropower production and the reduction of sediment deposited in reservoirs under these regimes. This chapter summarises the study's major findings and draws conclusions. In addition, it discusses implications for the design and evaluation of forest conservation and reforestation policies, and offers recommendations for further research.

8.1 Summary of the Findings

This summary describes the determined feasible forest governance regimes in the Northwest region of Vietnam. It then summarises the patterns of forest LULC changes under alternative FG regimes and the associated provisions of the three forest ecosystem services. Finally, it sums up the economic values of these forest ecosystem services under the alternative scenarios of forest governance.

8.1.1 The Alternative Scenarios of Forest Governance Regimes

The state-based forest governance regime scenario (the business-as-usual scenario). This state-based governance regime is characterised by the state dominating the forestry sector. Although there has been a shift towards decentralisation through community-based and individual-based FG arrangements, the state-owned organisations have been continuously in charge of most of the forest land and forests. Local communities and households have kept playing subordinate roles in forest conservation and reforestation through subcontracts with the state-owned organisations.

The community-based forest governance regime scenario. The key characteristic of this regime is that the forest property rights and responsibilities for forest management are delegated to local communities. The local communities are legally granted forest property rights (forest land tenure and forest use rights) on a long-term basis. Under this scenario, the local self-organised institutional arrangements, such as rules and regulations, particularly their customary laws relating to the use, responsibility for forest protection and development, play critical roles in forest conservation and reforestation. Communities can obtain benefits from extracting timber and non-timber products that are allowed by the state laws, from payment for forest ecosystem service and from the subsidies for forest conservation and reforestation.

The individual-based forest governance regime scenario. The regime of individual-based forest governance involves further decentralising the forestry sector to replace the state-dominant regimes with private forestry governance regimes through the privatisation of forest land and forests. Under this scenario, local households are legally granted long-term land use tenure of forest land and forest use rights. Individual households receive benefits from agro-forestry production practices on some proportion of their production forest, timber harvest (of production forest), other non-timber products that are allowed under the state laws and subsidies for forest protection and development. Furthermore, under this regime, the benefits that they can obtain from payments for forest ecosystem services are considered to be their primary motivation for forest conservation and reforestation.

8.1.2 The Patterns of Forest LULC Changes Under the Alternative Scenarios

In general, under all regimes, forest cover is likely to increase, but in different patterns during the period from 2010 to 2020. The expansion of forest cover, particularly through forest conservation and natural regrowth, is likely to increase moderately under both the community-based and the state-based FG scenarios. Meanwhile, under the individual-based FG scenario, forest cover is likely to improve very slightly, which is mostly due to the increase in planted forests. Concerning the spatial patterns of the changes, the new forests are likely to grow more densely at the upper north-western part under the state-based FG scenario, while under the community-based FG scenario, the new forest cover is likely to expand more fragmentedly throughout the research region.

8.1.3 Provision of Forest Ecosystem Services Under Alternative Forest Governance Regimes

Generally, delivery of the forest ecosystem services is the expected outcome from the ecological functions underpinning the interaction between forest LULC and the other biophysical components of forest ecosystems. The different patterns of forest LULC changes under the alternative FG scenarios will lead to differences in the provision of the forest ecosystem services. Under the state-based and the community-based FG regimes, forest cover expansion results mainly from the significant increase in natural forests, where carbon storage/sequestration is likely to increase considerably and sediment deposited into reservoirs and water yield is likely to decline. On the other hand, under the individual-based FG scenario, forest cover mostly remains constant with some increase in planted forests and the provision of the forest ecosystem services likely to remain unchanged. For example, carbon storage/sequestration and sediment load into reservoirs are likely to increase somewhat, while water yield is likely to decrease a little. In addition, it is worth noting that there are trade-offs among the three ecosystem services. Carbon storage/sequestration and the reduction of sediment exported into reservoirs are likely to improve with the expansion of forest cover, particularly through the natural regeneration of forests. Meanwhile, water yield is likely to decline when forest cover increases.

8.1.4 The Economic Values of the Forest Ecosystem Services Under the Alternative Scenarios

The economic values of carbon storage and carbon sequestration. The economic values of carbon storage and carbon sequestration derived from the forest ecosystems are substantial. The values under the state-based FG scenario and those under the community-based scenario for 2020 would be almost the same and significantly higher than those under the individual-based FG scenario. Using the mean of 50 USD/tC for the social cost of carbon (SCC) (Tol 2009) and the assumption that the SCC remains unchanged, on average, the total value of carbon storage in 2020 under the state-based, community-based and individual-based FG scenarios would be about 13,870, 13,828, and 13,461 million USD, respectively. Although the differences in the values of carbon storage under the three alternative FG scenarios are relatively small, the differences in the values of carbon sequestration for the 10 year period from 2010 to 2020 would be significant. The values of carbon sequestration would be about 474, 432, and 65 million USD under the state-based, community-based and the individual-based FG scenarios, respectively. On average, the annual values of carbon sequestration under the state-based and community-based scenarios would be about 47.4 and 43.2 million USD/year, respectively. These values are significantly higher than those of 6.5 million USD/year under the individual-based scenario.[54]

The economic values of reducing sediment deposited into reservoirs. The economic value of the reduction of sediment deposited into reservoirs is estimated through the amount of sediment yield prevented and the average replacement costs of sediment removal of $1.45 in 2010 USD (Phuong 2009a), which is assumed to remain unchanged. Compared to the replacement costs of sediment removal at the baseline in 2010, in 2020, the increase in economic values obtained from preventing sediment loaded into reservoirs would be about 1 million and 0.9 million 2010 USD, under the state-based and community-based FG scenarios, respectively. In contrast, about 1.5 million 2010 USD would be lost because of increased sediment exported into reservoirs under the individual-based scenario.

The values of water provision for hydropower production. The economic value of water provision for hydropower production is measured via the marginal product of one cubic metre of water used for hydropower generation. In 2020, the value of water would be the greatest at about 161.5 million USD per year under the individual-based scenario. Under the state-based and the community-based scenarios, the value would be slightly smaller at about 157.3 and 159 million USD per year, in 2020, respectively. In comparison with those of about 68.4 million USD at the baseline in 2010, the value of water supply for hydropower generation is likely to grow significantly under all scenarios. This increase mainly results from the fact that the water in 2020 would be used more efficiently with four hydropower plants being operational, versus only one (i.e. Hoa Binh hydropower plant) in 2010.

The differences in the value of the forest ecosystem services among the alternative FG scenarios. In 2020, the economic value aggregated from the individual values of the three forest ecosystem services under the state-based FG scenarios would be the greatest. The aggregated value under the state-based FG would be about 39.2 million USD per year higher than those under the individual-based FG. However, the estimated gap regarding the aggregated value between the state-based and community-based FG scenarios would be small, with only 2.6 million USD per year difference.

[54] The economic value is measured in constant 2010 USD price.

8.2 Conclusions

This study aimed to apply the ecosystem service valuation framework to the assessment of the impacts and effectiveness of forest governance regimes. In the literature, the debates on the outcomes of forest governance have not included the provision and the associated economic values of ecosystem services, particularly those provided by forest ecosystem services. By integrating the ecosystem service valuation framework to examine the outcomes of the three alternative forest governance arrangements in the Northwest region of Vietnam, this study presents significant results that advance the current state of knowledge in this domain.

The findings show that forest governance plays an important role in the provision of forest ecosystem services. The delivery of the three forest ecosystem services is different among the alternative FG regimes. Regarding the quantity of the forest ecosystem services, the capacity of providing carbon storage and sequestration and the reduction of sediment exported to reservoirs under the state-based FG and the community-based FG regimes, are likely to be higher than those under the individual-based scenario. At the same time, under the individual-based scenario, the capacity of forest ecosystems to provide water yield used for hydropower production is likely to be greater than for the other two alternatives.

This study also finds that forest governance determines the economic values of the individual forest ecosystem services and, consequently, the aggregated values of all the services. When it comes to aggregated values, the findings show that individual-based FG regimes are not likely to be as effective as the state-based and community-based FG regimes. At the same time, the benefits provided by the forest ecosystems under the community-based FG regimes are similar to those under the state-based FG regimes. Concerning the economic value of the particular service, values derived from carbon storage and carbon sequestration would be the greatest under the state-based FG regimes, followed by those under the community-based FG and then the individual-based FG regimes. Meanwhile, the value of the water provision for hydropower production would be highest under the individual-based FG scenario, followed by those under the community-based and state-based scenarios.

It is also worth noting that the economic values of the forest ecosystem services are determined by both the supply capacity of the forest ecosystems and the benefits that human society obtains from them. Particularly, the economic values of these forest ecosystem services vary due to the changes in the provisioned capacity of the forest ecosystems and the variations in the unit value of these services.

Based on the above-mentioned concluding remarks and the current argument on the effectiveness of forest governance (please refer to the discussions in Chapter 7), I argue that community-based FG would be a good alternative to the current state-dominant regime. This is because, under the community-based FG regime, forest conservation, economic values of the forest ecosystem services and improvements in the livelihoods of the local people are all likely to be achieved at the same time.

The results of this study suggest that it is essential to integrate the frameworks of forest ecosystem service valuation to assess the effects and effectiveness of forest governance regimes. This approach enables evaluation of the changes in social welfare stemming from forest conservation policies and programs. This is especially important for developing countries like Vietnam that have implemented an incentive-based approach for forest conservation and reforestation. With support from ecosystem production function models like the InVEST model package, this research design is applicable in the context of the limited data availability in developing countries.

8.3 Policy Implications

With the recent emergence of incentive-based approaches for forest conservation policy, such as Payments for Forest Ecosystem Service (PFES), it is required that the design and evaluation of these policies and projects be built based on research evidence. The evidence must be derived from the incorporation of the economic values of forest ecosystem services as well as the effectiveness of these policies and projects regarding the provision of these associated services. Therefore, based on the results provided by this study, some policy implications can be drawn:

- Among other things, community-based forest governance should be considered as an effective regime for the success of programs aimed at forest conservation and reforestation. This regime makes use of local institutions for the sustainable use of forests, which often leads to the improvement of society's welfare in terms of securing the provision and values derived from forest ecosystems.
- The value of carbon storage and sequestration among other services provided by forest ecosystems should be added in the existing PFES mechanism or the PFES should be combined with REED^{+}, because its value is relatively high in comparison with those of other forest ecosystem services. The current PFES mechanism,

particularly in the case of Vietnam, has only covered a few forest ecosystem services relating to hydrological services, recreation and tourism. This means that the current PFES is capturing only a small proportion of the values provided by forests and, consequently, probably does not provide adequate incentives for forest conservation. Ideally, the social value of carbon storage and sequestration should be fully captured and incorporated into the PFES payment mechanism. Practically, the potential financial sources that could be obtained from the REDD$^+$ mechanism that Vietnam is preparing, should be integrated into PFES. Therefore, the findings of this study should be used as a reference source for adjusting and calibrating the country's existing PFES, as well as for designing policies relating to REDD$^+$ intended to be implemented in the future. These findings provide useful information to identify the spatial provision of forest ecosystem services in relation to forest cover conditions that are crucial for incentive-based forest conservation policies like PFES and REDD$^+$.

- The application of ecosystem service valuation models such as the InVEST models should be used to assess the environmental and economic effects of environmental projects, such as those that relate to forest conservation and development. The results of the application, that precisely provide both physical provisions of ecosystem services and their associated economic values, enable decision makers to take the economic values of natural capital into account when they make public choices relating to the uses of natural resources.
- For Vietnam and other developing countries, the policies in the field of natural resource management in general, and forest resource management, in particular, should place greater emphasises on natural source inventory, database development, and sharings. The advanced of GIS and information technologies should be utilised for this purpose. The development, assessment, and use of the databases bring benefits to societies. For instance, the benefits include: 1) better opportunities for researchers to study and further develop the models of ecological production functions, and test the effectiveness of alternative management options, 2) more precise, effective, and timely recommendations, that based on scientific results, for public decision makers in natural resource management. These benefits could contribute to the more sustainable use of natural resources, particularly forest resources.

8.4 Recommendations for Further Research

There are several issues that have been identified in the course of the current study that are prime candidates for further study. These are associated with the limitation of data availability in developing countries, particularly those that relate to ecological processes and functions, the application of the total economic value framework to evaluate the benefits derived from forest ecosystem services, and the collaboration of economists and natural scientists to reveal the economic values of forest ecosystem services, the empirical estimation of the social cost of carbon and the test for the consistency of LULC transition likelihoods derived from experts' knowledge. These research opportunities include:

- Studies on the development of the database regarding the information of climate conditions, hydrological cycles and other attributes of forest ecosystems, such as forest cover conditions, soil attributes, carbon pools, etc. A better database will improve the accuracy of the model's estimation on the provision of forest ecosystem services as well as provide opportunities for including more ecosystem services provided by forest ecosystems into a single study.
- Studies on other types of values derived from forest ecosystems should be conducted to capture more adequately the total economic values of forest ecosystem services. For example, water regulation, flood control, water provision for agricultural production, water purification, recreation, ecotourism, biodiversity, etc.
- Studies on forest ecosystem services valuation, particularly those that incorporate ecosystem service framework into valuation research, are recommended. These types of studies should take advantage of the collaboration between economists and ecologists to uncover the economic values of ecosystem services and, consequently, provide better research outcomes.
- Studies on the development of integrated assessment framework and models that specify the estimation of the social cost of carbon for developing countries in general and Vietnam in particular. These studies will benefit both academic communities and decision-making groups who are interesting in measuring and managing the impacts of emissions to developing countries.
- Studies that aim to evaluate the effectiveness of forest conservation policies regarding the capacity of the provision of ecosystem services and the economic values produced by these services. The results of these studies would provide better evidence for the

design and evaluation of forest conservation programs and target the improvement of the delivery of these services.

- Studies that test the consistency and robustness of the participatory scenario development in LULC change. Particularly, the studies that engage in the structured tests of the outcomes of scenarios by stakeholders' background and characteristics are needed. The results of these studies will give insights and recommendations for efficient applications of participatory scenario development to LULC change.

To conclude, understanding the links between forest governance and forest ecosystems, the service provision capacity of the forest ecosystems and their associated benefits that humans enjoy is essential for decision making in the sustainable use of forest resources. Integrating the ecosystem service framework to quantify and evaluate forest ecosystem services will help to develop better knowledge regarding these links. In addition, it will enable assessment of the provision and economic values resulting from different forest conservation and management policies and, subsequently, help design effective governance systems needed for sustainable use of forests.

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Appendixes

Appendix 5.1 Summary of the Three InVEST Models used

	Step	Data requirements	Process	Outputs
Carbon storage and sequestration				
Required	Service	LULC	Looks up carbon stock(s) in pixel	Total carbon stock (mg/pixel)
		Carbon pools:		
		- Aboveground biomass		
		- Belowground biomass		
		- Dead organic matter		
		- Soil		
Optional	Service	Carbon removed via timber harvest	Calculates carbon stored in harvested wood products per pixel	Total carbon stock, including that in HWP (mg/pixel)
		First year of timber harvest		
		Half life of harvested wood products		
		Carbon density in harvested wood		
		Biomass conversion expansion factor		
Optional	Service	Future LULC	Calculates difference between carbon stocks	Carbon sequestration rates (Mg/pixel/year)
Optional	Value	Value of sequested carbon	Calculates value of carbon	Value of sequestered carbon (currency/pixel/yr)
		Discount rate		
		Timespan		
		Annual rate of change in carbon value		
Water yield: Reservoir hydropower production				
Required	Supply	LULC	Calculates pixel level yield as difference between precipitation and actual evapotranspiration	Mean annual yield (mm/watershed/yr, mm/pixel/yr)
		Mean annual precipitation (mm)		
		Mean annual reference evapotranspiration (mm)		
		Plant available water content		
		Evapotranspiration coefficient		
		Root depth (mm)		
		Effective soil depth (mm)		
		Seasonality factor		
Required	Service	Consumptive use by LULC	Substracts water consumed forby different LULC	Mean annual water yield available for hydpower production (mm/watershed/yr)
		Subwatershed and watershed shapefiles		
Optional	Value	Calibration coefficient	Estimates power for a given volume of water	Energy production (kwh/watershed/yr. Kwh/pixel/yr)
		Turbine efficiency (0.7-0.95)		
		Inflow volume for hydropower (fraction)		
		Hydraulic head (m)		
		Operation cost (currency)	Calculates net present value of energy produced over lifetime of dam	Net present value (currency/watershed/yr, currency/pixel/yr)
		Hydropower price (currency)		
		Life span of the hydropower station (years)		
		Discont rate (%)		

SDR model				
Required	Service	LULC Digital elevation model (DEM) Rainfall erosivity index (R-factor) Soil erodibility (K-factor) Cover-management factor (C-factor) Support practice factor (P-factor)	Calculates total amount of potention soil loss from USLE equation	Total potension soil loss per pixel (tons/pixel)
		Threshold flow accumulation Calibration parameters (*k_b, IC_0*: the default values are $k_b = 2, IC_0 = 0.5$) SDR_{max} (the default value is 0.8)	Calculate sediment delivery ratio and sediment load Calcualte	Total amount of sediment exported from each pixel that reaches the stream (tons/pixel)
		Watershed(s) shapefiles	Calculate total amount of potention soil loss and total amount of sediment exported to the stream per watershed	Total amount of potention soil loss, total amount of sediment exported to the stream per watershed (tons/watershed)

Source: Sharp *et al.* (2015)

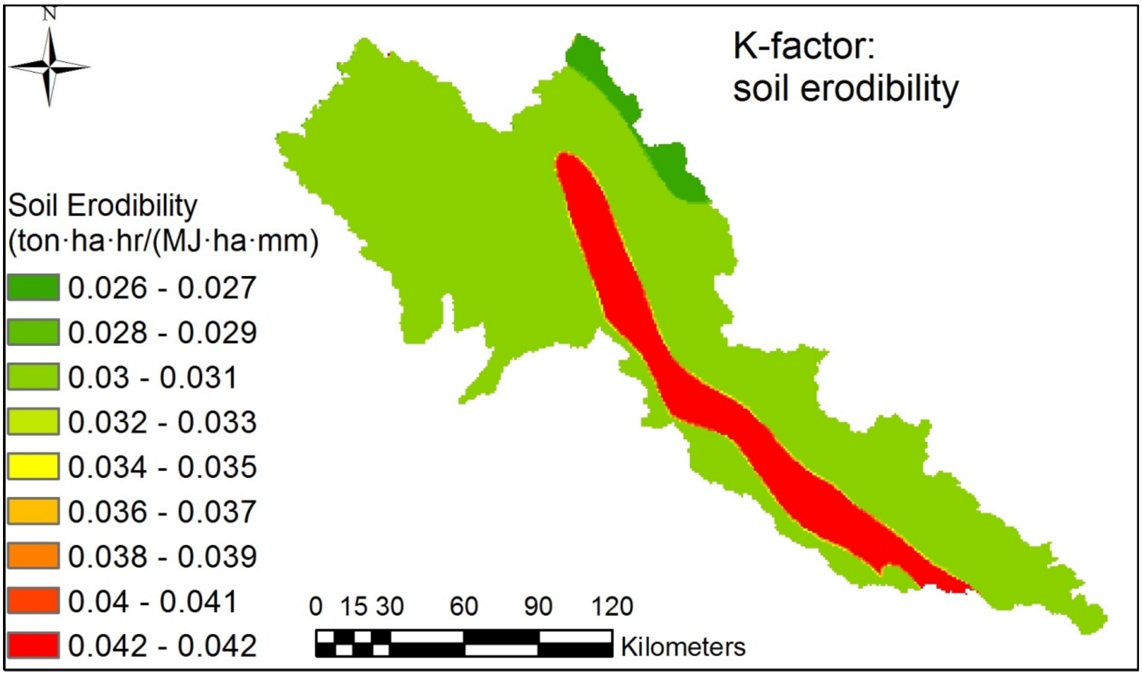
Appendix 6.2 Data sets, key information, and data source schemes

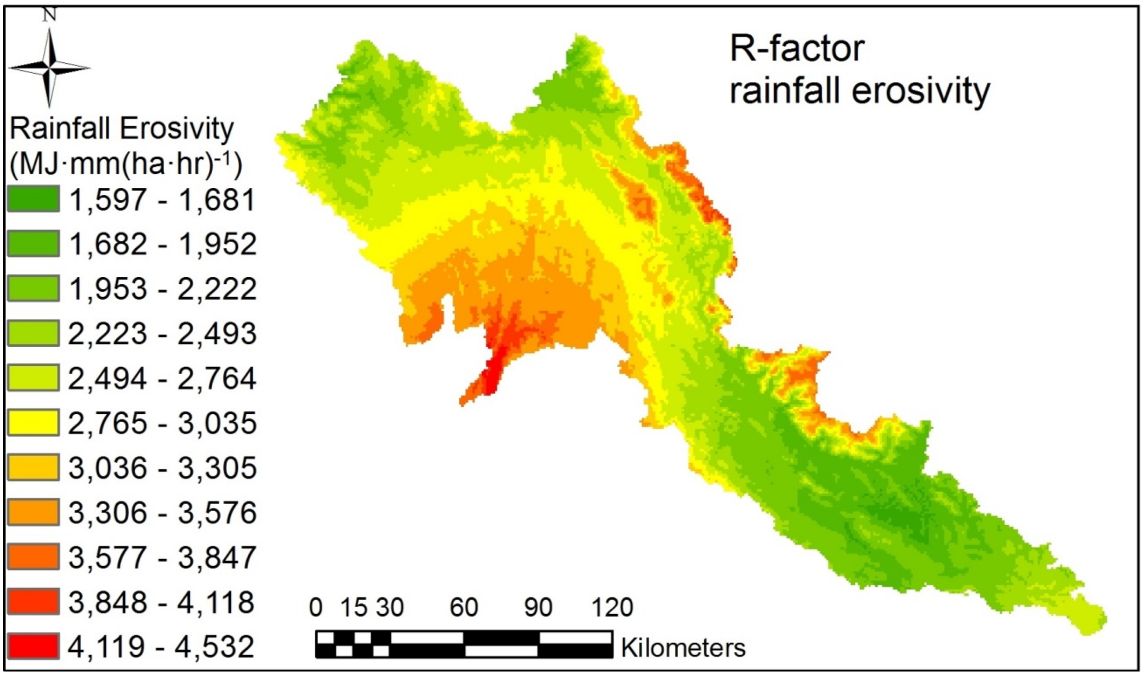
Data Set	Data Description	Data Source
1- For Developing FG scenarios		
Legal framework shaping forest governance arrangements	Forest property rights, benefit-sharing mechanisms, forest utilisation regulations, forest protection and development planning, the policy of payments for forest ecosystem services	The Law of Land, Law on Forest Protection and Development, and other government legal documents relating to forest land and forest allocation, benefit-sharing of forest resources and forest regulation
Forest governance practices	Allocation of forest and forest land, privatisation of forests and forest land, community-based forest management, payments for forest ecosystem services in practice	Government's and research institutions' reports on forest protection and development programs/projects (5MHRP, KWF7 project) and PFES policy implementation reports, and other publications relating to forest governance
Feasible forest governance arrangements	Possible future forest governance practices	Experts' opinions derived from unstructured expert interviews
2- For Mapping forest LULC under the generated FG scenarios		
The historical trend of forest cover changes	The changes of forest cover types (old growth forest, degraded forest, regrowth forest, planted forest) over time	VNForest, MARD
Priority of forest cover types under hypothetical FG scenarios	Which types of forest cover are more prioritised?	Experts' opinions summarised from structured expert interviews
Expected forest LULC transition likelihood under hypothetical FG scenarios	Likelihood of transition of each forest cover type	Experts' opinions summarised from structured expert interviews
Spatial-related factors determining land suitability for forest cover types	Proximity/distance to rivers, proximity/distance to roads, proximity/distance to villages, elevation, slope, current cover status	VN-Atlas 2006, ASTER GDEM2, LandSat 5TM images

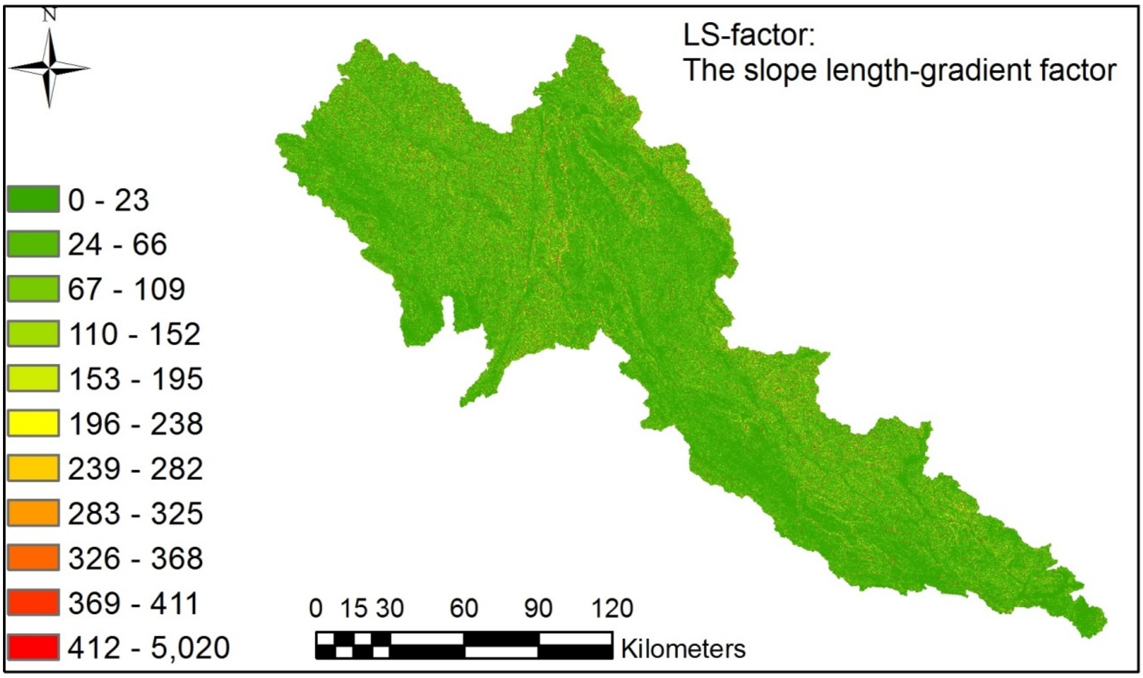
3- For Running the three InVEST models		
For all the three models		
Baseline LULC	Raster, WGS_1984_UTM_Zone_48N, cell size (30m x 30 m)	FIPI, Vietnam forest cover map in 2010
Future scenario of LULC (at 2020)	Raster, WGS_1984_UTM_Zone_48N, cell size (30m x 30 m)	Projected by the author
DEM	Raster, GCS_WGS_1984, 1-arc-sec	ASTER GDEM2
Watersheds	Vector, WGS_1984_UTM_Zone_48N	Delineated from DEM using ArcGIS 10.1's Hydrology Tools
Sub-watersheds	Vector, WGS_1984_UTM_Zone_48N	Delineated from DEM using ArcGIS 10.1's Hydrology Tools
For the Carbon Storage and Sequestration Model		
Carbon pools	Carbon stored in aboveground and underground biomass	Extracted from the work of Saatchi *et al.* (2011)
	Carbon stored in soil	Extracted from the work of Hiederer and Köchy (2012)
For the sediment delivery model		
Rainfall erosivity index (R-factor)	Raster, with an erosivity index value for each cell. Dependent on the intensity and duration of rainfall	Estimated through monthly rainfall using the formula developed by Wischmeier and Smith (1978), which is also suggested by Renard and Freimund (1994), Bagherzadeh (2014), Prasannakumar *et al.* (2012) and Fu *et al.* (2011), for tropical regions
Soil erodibility (K-factor)	Raster, with a soil erodibility value for each cell. It is a measure of the susceptibility of soil particles to detach and be transported by rainfall and runoff	Estimated from HWSD V1.2, with guidance from the OMAFRA factsheet Order No.12-051 (Hilborn 2012) and Renard *et al.* (1997) for the conversion from US customary units into the International System (IS)
C-factor	Cover-management factor	Estimated from NDVI by the formula proposed by Durigon *et al.* (2014) and Lin *et al.* (2002)
NDVI	Normalised difference vegetable index	Estimated from on LandSat 5TM images based on (Chander *et al.* 2009)

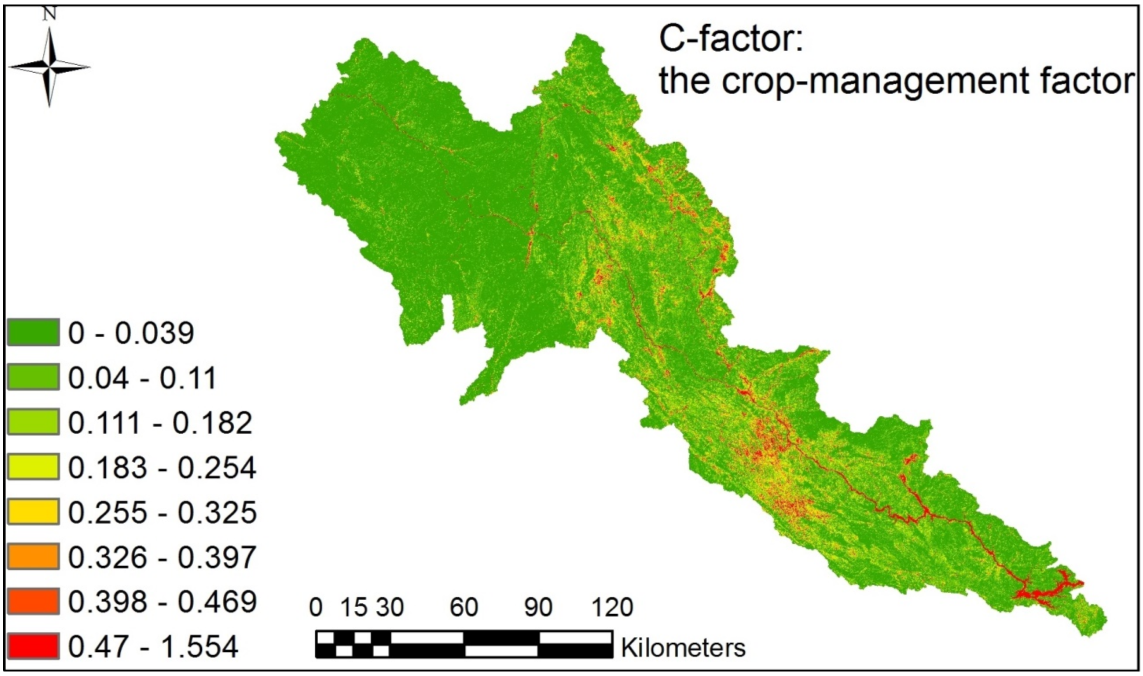
For the Water Yield Model		
Average annual precipitation	Raster, GCS_WGS_1984, 30 arc-second	WorldClim database: Averaged over the 1950-2000 period
Average annual reference evapotranspiration	Raster, GCS_WGS_1984, 30 arc-second	CGIAR-CSI, global-PET, averaged over the 1950-2000 period
Depth-to-root restricting layer	Raster, GCS_WGS_1984, 30 arc-second	Extracted from HWSD V1.2
Plant-available water fraction	Raster, GCS_WGS_1984, 30 arc-second	Extracted from HWSD V1.2
K_c	Plant evapotranspiration coefficient	Estimated from LAI using the equation proposed by Allen *et al.* (1998)
LAI	Leaf area index	Extracted from GLCF_ MODIS_LAI (MOD09A1)
Root depth	Root depth of particular forest cover	Calculated based on Canadell *et al.* (1996)
Z-parameter	Seasonality factor	Estimated through the formula suggested by Sharp *et al.* (2015)
AWC	Available water capacity	Extracted from HWSD V1.2
For Valuing the Forest Ecosystem services		
4- Valuing carbon storage and sequestration		
SSC	Social cost of carbon	The global SCC estimated in the published literature (Tol 2009; Nordhaus 2011)
Replacement cost of removing sediment deposited into reservoirs	Costs of removing sediment deposited into reservoirs in the north of Vietnam	The sediment removal cost calculated by Phuong (2009a)
Technical information of hydropower plants	Timespan, turbine type and efficiency of turbine, fraction of water used for hydropower generation, water height and installed capacity	EVN, Ministry of Industry and Trade
Economic information of the hydropower plants	Total investment, average annual energy production and electricity price	EVN, Ministry of Finance

Appendix 6. 3 The maps of the factors of the Sediment Delivery Ratio Model






N
LS-factor:
The slope length-gradient factor
0 - 23
24 - 66
67 - 109
110 - 152
153 - 195
196 - 238
239 - 282
283 - 325
326 - 368
369 - 411
412 - 5,020
0 15 30 60 90 120
Kilometers


N
C-factor:
the crop-management factor
0 - 0.039
0.04 - 0.11
0.111 - 0.182
0.183 - 0.254
0.255 - 0.325
0.326 - 0.397
0.398 - 0.469
0.47 - 1.554
0 15 30 60 90 120
Kilometers

Appendix 6. 4 Expert telephone interviews

Introduction

Hello, my name is Nguyen Minh Duc. I am conducting a study on Forest governance and economic values of forest ecosystem services in the Northwest region of Vietnam for my Ph.D. research project at The University of Sydney. I will be supervised by Associate Professor Tihomir Ancev while I conduct this study.

My research will greatly benefit from your participation. I will consult you about possible land use and land cover (LULC) changes in this region in the next five years. The results of my study may prove to be useful for decision makers in LULC planning in the future.

In the Northwest region, the LULC change pattern is examined under the current forest governance regime, and it is also considered for two hypothetical forest governance scenarios, namely a community-based forest governance regime and an individual household forest governance regime.

Please give me around 30 minutes of your time for a conversation over the telephone in which I will ask you to provide responses to the following questions:

I - THE CURRENT FOREST GOVERNANCE

Q1. In your opinion, what is the possibility that the following LULC transitions may happen in the next five years in the Northwest region? The possibility of transition is measured on a scale from 1 (not likely) to 10 (extremely likely).

Please choose the appropriate possibility of transition over the next five years.

From	**To**	**Possibility of Transition - Ordinal Scale from 1 (not likely) to 10 (extremely likely)**									
		1	2	3	4	5	6	7	8	9	10
Bare land/shrub land	Regrowth forest	○	○	○	○	○	○	○	○	○	○
Agriculture	Regrowth forest	○	○	○	○	○	○	○	○	○	○
Regrowth forest	Planted forest	○	○	○	○	○	○	○	○	○	○
Agriculture	Planted forest	○	○	○	○	○	○	○	○	○	○
Bare land/shrub land	Planted forest	○	○	○	○	○	○	○	○	○	○
Old growth forest (rich, medium)	Degraded forest	○	○	○	○	○	○	○	○	○	○
Old growth forest (rich, medium)	Bare land/shrub land	○	○	○	○	○	○	○	○	○	○
Degraded forest	Bare land/shrub land	○	○	○	○	○	○	○	○	○	○
Regrowth forest	Bare land/shrub land	○	○	○	○	○	○	○	○	○	○
Planted forest	Bare land/shrub land	○	○	○	○	○	○	○	○	○	○
Agriculture	Bare land/shrub land	○	○	○	○	○	○	○	○	○	○
Bare land/shrub land	Agriculture	○	○	○	○	○	○	○	○	○	○
Degraded forest	Agriculture	○	○	○	○	○	○	○	○	○	○
Regrowth forest	Agriculture	○	○	○	○	○	○	○	○	○	○
Planted forest	Agriculture	○	○	○	○	○	○	○	○	○	○

Q2. Do you think the following LULC will increase in the next five years?

LULC Type	Increase in the Next Five Years	
	Yes	No
Old growth (rich and medium) forest	○	○
Degraded forest	○	○
Regrowth forest	○	○
Planted forest	○	○
Agriculture	○	○
Bare land/shrub land	○	○

Q3. Among the LULC types that may increase in the next five years, what LULC is most prioritised in the current forest governance system? Please indicate the priority.

LULC Type	Priority Score: 0 (the least prioritised) to 10 (the most prioritised)										
	0	1	2	3	4	5	6	7	8	9	10
Old growth (rich and medium) forest	○	○	○	○	○	○	○	○	○	○	○
Degraded forest	○	○	○	○	○	○	○	○	○	○	○
Regrowth forest	○	○	○	○	○	○	○	○	○	○	○
Planted forest	○	○	○	○	○	○	○	○	○	○	○
Agriculture	○	○	○	○	○	○	○	○	○	○	○

II - COMMUNITY-BASED FOREST GOVERNANCE REGIME

The government has implemented forestry sector reform and strengthened the process of decentralisation of the forest governance regime. After reviewing the policy of forest land and forest allocation that has been implemented since the Doi Moi (renovation) was started in the forestry sector, I found that community-based forest management is becoming a promising alternative forest governance regime for the future.

For the purpose of this research, I want you to consider a hypothetical scenario where the forest governance regime is transformed from a state-controlled system to a community-based (kinship family, village) governance regime. If this scenario were true, what would your expectations be about LULC change under this hypothetical scenario?

The main characteristics of the new governance regime can be briefly described as follows:

- The state-managed forests and forest land (currently managed by the state organisations: state-owned enterprises, protection forest management boards, communal people committees) is now allocated to local community populations. The local communities have long-term (e.g., 50 years) land use tenure of forest land and property rights of forests, legally guaranteed by land titles (Red Book) that are granted when the allocation process is taking place.
- The communities self-organise land use planning based on their own demands and in accordance with the regional strategy of forest protection and development.
- They self-manage the use of forest land, the access to and utilisation of forests based on their customary laws, which are documented and approved by local state authorities for implementation and enforcement.
- Communities obtain and manage financial resources from the state funds for forest protection and development to implement their planning. They also receive benefits of payments for forest ecosystem services and for forest protection and development.
- The government takes a hands-off approach, no longer influencing matters directly. However, it retains a supervisory role to ensure that the communities can carry out their functions, especially regarding technical matters and the enforcement of legal claims, such as sanctions for violations that go beyond communal authorities.

Please do not take into account what you have answered in the previous questions. Think about the new forest governance regime seriously and give your responses to the following questions:

[The sets of questions: Q1, Q2 to Q3 are repeated for this scenario]

III - INDIVIDUAL HOUSEHOLD FOREST GOVERNANCE REGIME

Another alternative forest governance regime which enhances the decentralisation process in the forestry sector is the individual household forest governance regime. After the Doi Moi (renovation) that took place in the forestry sector, individual households have played an important role in forest protection and development, especially in remote mountainous areas.

In the Northwest region, allocation of forest land to individual households has been implemented simultaneously with structural reform of state forestry enterprises. However, there have been some obstacles that prevent the potential of individual households from having stronger participation in forest protection and development. For example, allocation of forests has not been attached to the allocation of forest land. Most of the forest land areas allocated to households are degraded forests or bareland and the benefits obtained from the allocated forests are small and local households have a lack of financial resources, which means that they are not interested in engaging in forest protection and development. Furthermore, forest land areas allocated to the state sector (state-owned enterprises, forest management boards) are extensive and go beyond the self-management capacity of these organisations. In order to fulfill the functions of forest protection and development, these organisations make contracts to local households. There is a significant amount of forest land that has not been allocated, which is currently managed by local communal people committees. These areas are often seen as ‘free access’ areas and managed inefficiently.

Therefore, I want you to consider another hypothetical scenario where the individual households take over the roles of the state’s organisations in the forestry sector, in terms of forest protection and development. In this governance regime, the obstacles preventing the individual households’ potential participation in forest protection and development are resolved.

The main characteristics of the new governance regime can be briefly described as follows:

- The state-managed forests and forest land are classified as protected, and production forests that have previously been managed by the state organisations (state-owned enterprises, forest management boards and forests managed by communal people committees), are now transferred to local individual households. The households have long-term (e.g., 50 years) land use tenure of forest land and the forest use rights are legally guaranteed by land titles (Red Book) that are granted when the allocation process is taking place.
- The individual households acknowledge their forest land and forests on the state cadastral map, as well as on the field. The boundary of individual household’s forest land is identified on the field.

- The households self-organise land use planning based on their own demands and in accordance with the state's land use planning and strategy of forest protection and development.
- The households self-manage their forestry production on their planted production forests, receive benefits from timber exploitation, can gain access to financial support from the state's fund for forest protection and development and obtain technical support provided by state/private organisations. The households can also practice agro-forestry production on their allocated planted production forest.
- For natural production forests, they can receive benefits from timber exploitation and other non-timber products allowed by the law. They also receive benefits from Payment for Forest Ecosystem Services for forest protection and development, and from the state budget for forest protection and development. Furthermore, forest land tenure and forest use rights of natural production forests are granted as are those of planted production forests.
- Regarding protected forests, households receive payment from Forest Ecosystem Services for forest protection and development. They can also receive benefits from the state's fund for forest protection and forest development, and other benefits from extracting forest resources (e.g., non-timber forest products) allowed by the law. Benefits obtained from the forest are at least equal to market labour costs and materials spent for forest protection and development.
- The government takes a hands-off approach, no longer influencing matters directly. However, it retains a supervisory role to ensure that the households follow the legal framework in the forestry sector, resolve possible conflicts in land tenure and provide financial and technical services.

If this scenario were true, what would your expectations be about LULC change under this hypothetical scenario?

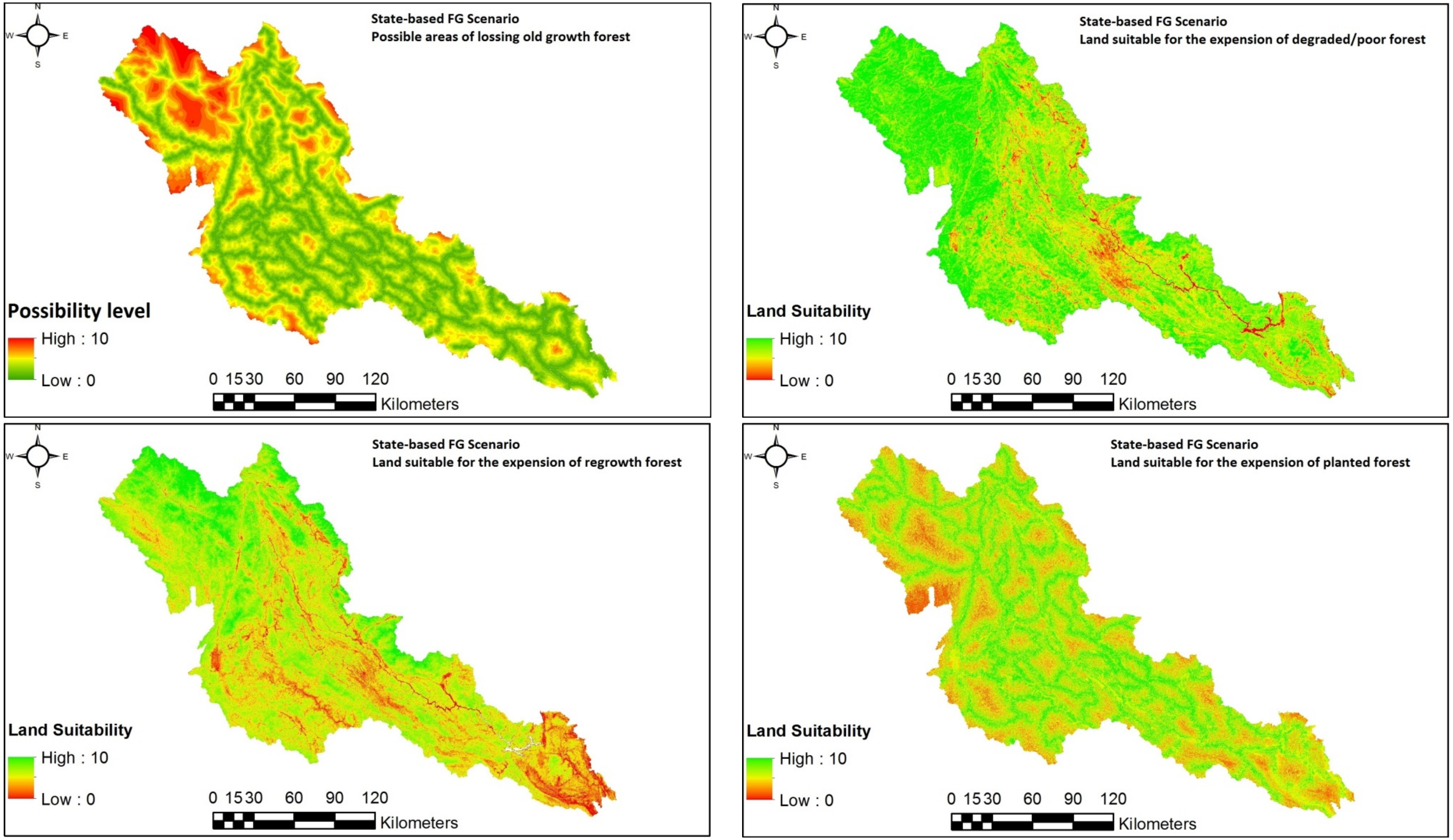
Please focus only on this forest governance regime, without taking into account your responses to previous questions. Please provide responses to the following:

[The sets of questions: Q1, Q2 to Q3 are repeated for this scenario]

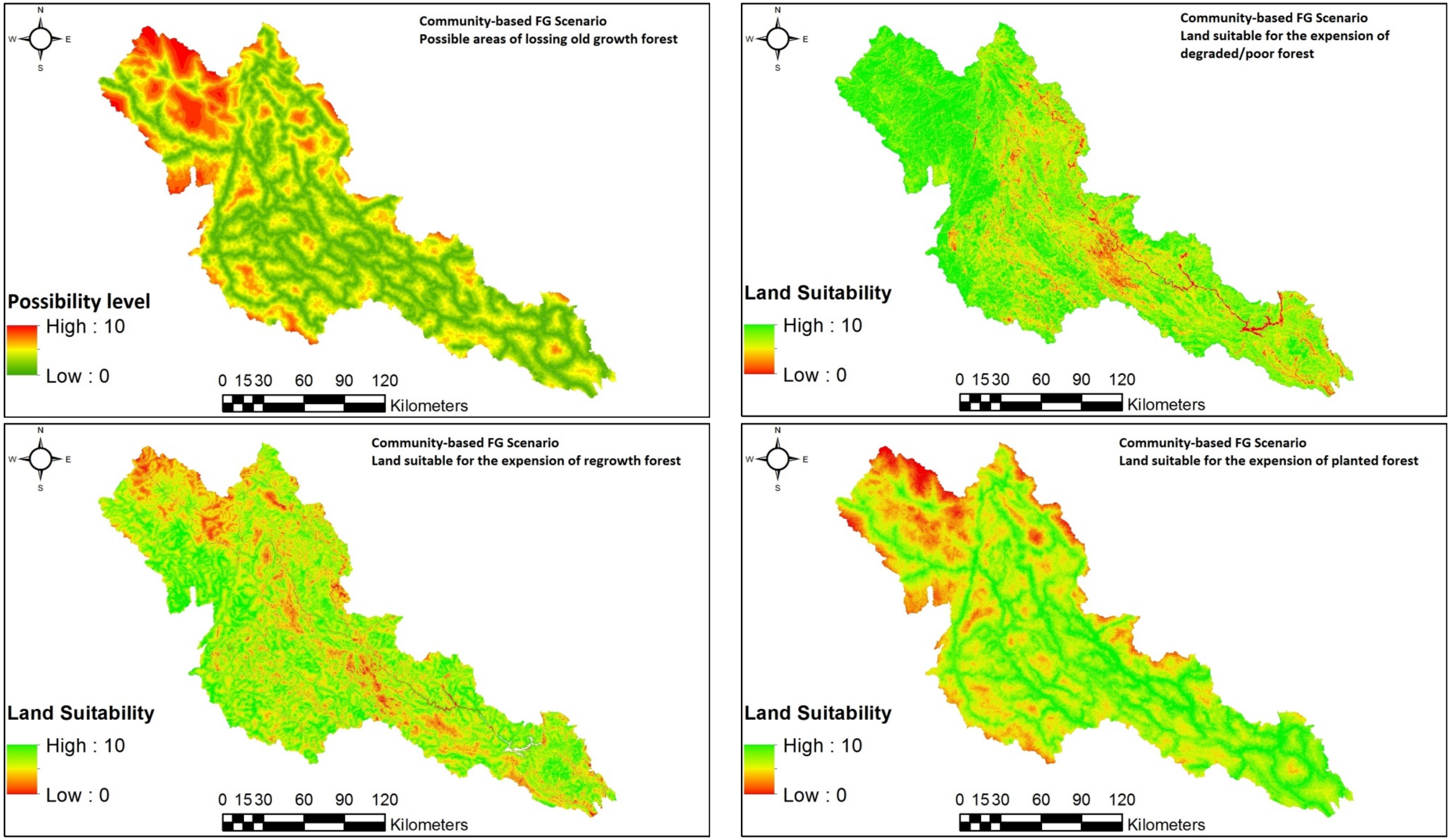
I truly appreciate your support, and

thank you very much for providing responses!

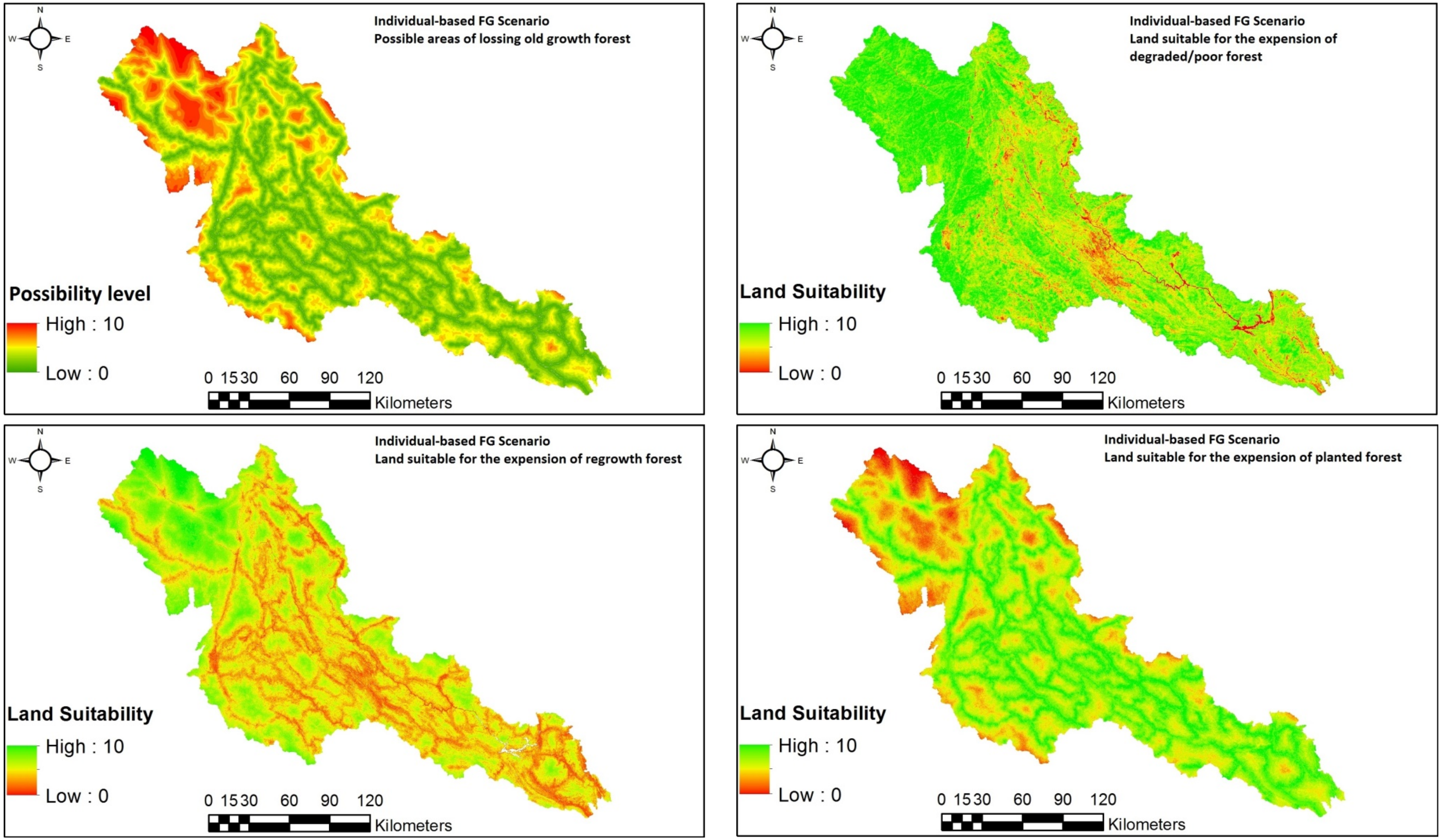
Appendix 6.5a Land suitability for different forest LULC types under the state-based scenario



Appendix 6.6b Land suitability for different forest LULC types under the community-based scenario



Appendix 6.7c Land suitability for different forest LULC types under the individual-based scenario



Appendix 6.5 Alternative forest governance regimes

Governance institutions	**Description**
Community-based FG	
Forest land tenure	The local communities have long-term (e.g. 50 years) land use tenure of forest land and legally guaranteed by land titles (Red Book).
Forest use rights/ownership	Forest use rights are granted with the land tenure.
Formal regulations of forest usage	The State Laws on Forest Protection and Development, the state policies of forest protection and development determine formal rules of forest usage.
Informal rules of forest utilisation	They self-manage the access and usage of forests utilisation based on their customary laws that do not conflict with the formal rules.
Benefit-sharing including PFES	Communities obtain and manage financial resources from the state funds for forest protection and development. They also get benefits from PFES for forest protection.
Roles of stakeholders	The communities self-organise their forests and forest land use planning based on their own demands and in accordance with the regional strategy of forest protection and development. The government takes a hands-off approach, no longer influencing matters directly. However, it retains a supervisory role to ensure that the communities can carry out their functions, especially regarding technical matters and the enforcement of legal claims such as sanctions for violations, which go beyond communal authorities.

Individual-based FG	
Forest land tenure	The households have long-term (e.g. 50 years) land tenure of forest land that is legally guaranteed by land titles (Red Book).
Forest use rights/ownership	Forest use rights are granted with the forest land tenure.
Formal regulations of forest usage	Regulated by the State Laws on Forest Protection and Development and the state policies of forest protection and development.
Benefit-sharing including PFES	For planted production forest areas, the households self-manage their forestry production on their planted production forest, get benefits from timber exploitation, can get access to financial support from the state's fund for forest protection and development and obtain technical support provided by state/private organisations. The households can also practice agri-forestry production on their allocated planted production forest. For natural production forests, they can obtain benefits from timber exploitation and other non-timber products allowed by the law. They also get benefits from PFES for forest protection and get financial support from the state budget for forest protection and development. Regarding protection forests, households mainly get benefits from payments for forest ecosystem services for forest protection. Besides these, they can also get benefits from the state's fund for forest protection and forest development, and other benefits from extracting forest resources (e.g., non-timber forest products) allowed by the law.
Roles of stakeholders	The households self-organise land use planning based on their own demands and in accordance with the state's land use planning and strategy of forest protection and development. The government takes a hands-off approach, no longer influencing matters directly. However, it retains a supervisory role to ensure the households follow the legal framework in the forestry sector, resolve possible conflicts in land tenure and provide financial and technical services.