Acta Universitatis Agriculturae Sueciae

Doctoral Thesis No. 2011:20
Faculty of Forest Sciences

Forest Restoration on Degraded Lands in Laos

Sovu

Forest Restoration on Degraded Lands in Laos

Sovu
Faculty of Forest Sciences
Southern Swedish Forest Research Centre
Alnarp

Doctoral Thesis
Swedish University of Agricultural Sciences
Alnarp 2011

Acta Universitatis agriculturae Sueciae
2011:20

Cover: Secondary forest recovered on swidden field abandoned 30 years ago in Phoukhaokhuay NBCA.
(Photo: Sovu)

ISSN 1652-6880
ISBN 978-91-576-7555-2

Print: SLU Service/Repro, Alnarp 2011

Forest Restoration on Degraded Lands in Laos

Abstract
Deforestation and land-use change have been, and still are, major threats to biodiversity and ecosystem services in the tropics, and forest areas are being lost at high rates in many countries in Southeast Asia, including Laos. Progress in the development and application of forest policies and programs, coupled with increasing trends to end damaging practices associated with swidden cultivation have created opportunities to restore forests. However, there is little scientific knowledge pertaining to forest restoration in Laos. Thus, the studies presented in this thesis examined factors influencing the recovery of secondary forests on abandoned fallows and tested site-specific approaches to restore forests on fallows, former grazing lands and a logged-over mixed deciduous forest. The results showed that 29% of the species richness, 39% of the stem density, 18% of the total basal area, as well as 41% and 7% of the density and basal area of commercial tree species of the surrounding natural forest fragments were recovered within a 20-year fallow period. The major factors influencing recovery are the distance from the forest edge, the fallow history and competition from increased occurrence of bamboo. A mixed-species planting trial established on fallows showed that the use of a mixture of pioneer and later-successional species results in good ecological compatibility, as evidenced by > 70% survival rates and subsequent growth of all planted species. Further, the addition of rice husk biochar to improve soil fertility resulted in 1.2-fold increase in seedling and sapling growth compared to the control and its effect was comparable with that of inorganic fertilizer application. An experiment involving direct seeding of former grazing lands using four native species showed that seedling establishment success was better for later-successional species (59-65%) than pioneer species (3-13%), and the establishment success varied with methods of sowing, the nature of the seeds, the seeding rate and site factors. In addition, an enrichment planting trial, involving five early- and late-successional species and two planting methods, showed that the shade-tolerant dipterocarps had better survival and growth rates than the light-demanding legumes in gaps than planting lines. Overall, the studies provide an important contribution, representing a major advance in the evidence base for selecting appropriate methods to accelerate forest restoration in Laos and other seasonal tropical environments in order to meet key economic and environmental objectives.

Keywords: biochar, direct-seeding, enrichment planting, Lao PDR, mixed planting, secondary forests

Author's address: Sovu, SLU, Southern Swedish Forest Research Centre, P.O. Box 49, 230 53 Alnarp, Sweden. *E-mail*: Sovu@slu.se, vuesovu@yahoo.com

Dedication

To my parents,
my wife Silavanh,
my daughter Naly Vue and
my son Anouvy Vue

Contents

List of Publications

This thesis is based on the work described in the following papers (I-IV), which are referred to by the corresponding Roman numerals in the text:

I Sovu, Mulualem Tigabu, Patrice Savadogo, Per Christer Odén, Lamphoune Xayvongsa. (2009). Recovery of secondary forests on swidden cultivation fallows in Laos. *Forest Ecology & Management,* 258, 2666–2675.

II Sovu, Mulualem Tigabu, Patrice Savadogo, Per Christer Odén. (2011). Facilitation of forest restoration on abandoned small-scale swidden fallows in Laos using mixed-species planting and biochar application. (Submitted manuscript).

III Sovu, Patrice Savadogo, Mulualem Tigabu, Per Christer Odén. (2010). Restoration of former grazing lands in the Highlands of Laos using direct seeding of four native tree species. *Mountain Research & Development,* 30(3), 232–243.

IV Sovu, Mulualem Tigabu, Patrice Savadogo, Per Christer Odén, Lamphoune Xayvongsa. (2010). Enrichment planting in a logged-over tropical mixed deciduous forest of Laos. *Journal of Forestry Research,* 21(3), 273–280.

Papers I, III and IV are reproduced with the permission of the publishers.

The contribution of Sovu to the papers appended to this thesis amounted to 80% of the total work load as outlined below:

I I designed the study together with my supervisors. I was solely responsible for data collection. The data analysis and writing of the paper were mainly my responsibility, with contribution from the co-authors.

II I also designed this study, together with my supervisors. I was fully responsible for data collection, and mainly responsible for data analysis and writing of the paper, with contribution from the co-authors.

III The direct seeding trial was established by the Namgum Watershed Cooperation Project, in which my university was a collaborator. Together with my supervisors, we developed revised objectives and scientific hypotheses to be tested I collected the data and was mainly responsible for the data analysis and writing of the paper, with contribution from the co-authors.

IV The enrichment planting trial was established in 2000 by PROFEP, implemented by FOF and GTZ, in which I am a project member. I was responsible for data collection, while the data were analysed and the paper was written with the support of my co-authors.

Abbreviations

FAO	Food and Agriculture Organization of the United Nations
FOF	Faculty of Forestry, National University of Laos
GIS	Geographical Information System
GOL	Government of the Lao PDR
GTZ	Deutsche Gesellschaft für Technische Zusammenarbeit
Lao PDR	Lao People's Democratic Republic
NAFRI	National Agriculture and Forestry Research Institute
NBCA	National Biodiversity Conservation Area
NFAP	National Forestry Action Plan
PROFEP	Promotion of Forestry Education Project
SER	Society for Ecological Restoration

1 Introduction

1.1 Background

Deforestation and land-use change are major threats to biodiversity and ecosystem services in the tropics, despite recent signs of abatement in several countries in the tropics. The FAO has estimated that tropical regions lost 15.2 million ha of forest per year during the 1990s and the net rates of forest loss were higher in Southeast Asia (2.4 million hectares per year) than in other regions (FAO, 2005). According to very recent estimates (FAO, 2010), net losses decreased to an estimated 0.7 million hectares/year in the period 2000–2010. However, many other countries in South and Southeast Asia continue to report high rates of net loss of forest area. Forest loss in the Lao Peoples Democratic Republic (Laos) is estimated to have amounted to 78 000 ha/yr in the period 2005-2010, which is fairly low compared to all countries in the lower Mekong basin (Cambodia, Thailand, Myanamar), except Viet Nam, where there was an estimated net 1.4 million ha/yr increment in forest cover during the same period (FAO, 2010). As shown in the land cover map presented in Figure 1, the current forest cover in Laos accounts for 41.5% of the country's total land area while unstocked forests, including bamboo and ray fields, cover 47.1%. The remaining land areas are other wooded/scrub lands (1.2%), permanent agricultural lands (5.1%), and non-forest areas (5.1%). The forest cover in Laos declined from 70% of the country's land area in the 1940s, to 64% in the 1960s and 41.5% in the 2000s (GOL, 2005).

According to official estimates, 70 000–220 000 ha of forest was lost annually in Laos in the latter part of the 20th century (GOL, 1999), primarily due to swidden cultivation (also known as slash and burn agriculture and shifting cultivation), exacerbated by a post-Vietnam War boom in the human population. Another potentially important factor was government policy encouraging agricultural expansion, which resulted in an estimated 0.5% per year decline in forest cover until the 1980s (Robichaud *et al.*, 2009). Logging (both legal and illegal) and permanent agricultural expansion have been major drivers of forest loss in the lowlands, while swidden cultivation has been the major cause of forest conversion and forest degradation in the highlands of the northern part of Laos (Giri *et al.*, 2003), as 70% of the Lao territory is mountainous and not suitable for establishing paddy fields – a primary farming practice for rice production. Alongside rice farming, forest resources underpin the majority of Laotians' livelihoods – about 70% of the population of 6.5 million people live in rural areas and depend heavily on forest products for their daily subsistence and income (WorldBank *et al.*, 2001; Yokoyama *et al.*, 2006). Thus, with increasing population, the pressure on the forests has also increased, leading eventually to forest degradation.

Deforestation in the tropics, and the associated loss of biodiversity and ecosystem services, has become a global concern, leading to a number of policy reforms intended to promote the sustainable management of forest resources. Accordingly, in recent years the Government of Laos has introduced a number of policy instruments and incentives to boost forest cover by promoting the development of forests throughout the country. These attempts to reverse deforestation include the creation of a National Forestry Action Plan (NFAP) in 1989, the demarcation of National Protected Areas and the implementation of a Forest Classification scheme in 1993 (GOL, 1999; Lestrelin, 2010). Consequently, production, protection and conservation forests covering 3.6, 6.6 and 4.7 million ha, respectively, have been established. The government also introduced a protected area system in 1993, which covered four sub-zones (the Anam Trung Son mountain chain, tropical lowland plain, hilly sub-tropical sector and mountain temperate zone) and 11 forest categories (Sawathvong, 2003). The Lao protected area system includes 20 National Biodiversity Conservation Areas (NBCAs) that cover about 13% of the country (Fig. 1). In addition, Forest, Land and Environmental protection laws were introduced during the period 1996-1999. For implementation of these laws, several sets of regulations were issued including the land and forest allocation program with the main objectives of promoting crop production and to replace swidden cultivation through allocation, protecting forests through forest classification, and utilizing the allotted forests on sustainable basis by villagers, households or communities. The land allocation program has continued with a view of achieving its target of effective reduction/stabilization of swidden cultivation to protect forest resources and the natural environment from further degradation. One of eight currently prioritized programs of the government is intended to protect natural resources whilst improving the living standards of rural people (GOL, 2005).

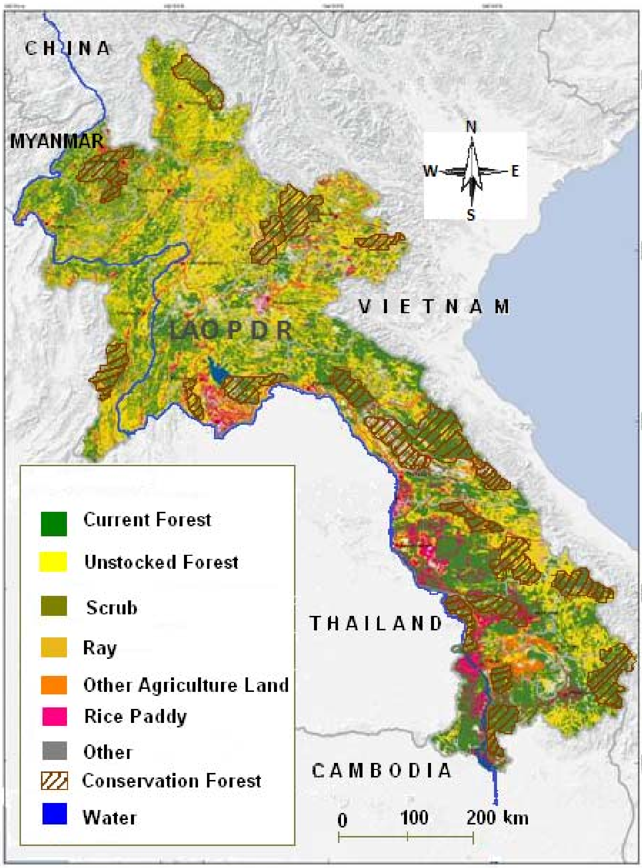

Figure 1. Land cover map of Lao PDR in 2002 (prepared by the Department of Forestry, Ministry of Agriculture and Forestry, Laos)

Moreover, reforestation is one of the main National Forestry Strategies, whereby the government intends to restore the forest cover to 70% by the year 2020 (GOL, 2005). Decree No. 186 and the forest law emphasized the promotion of tree planting, protection and restoration of natural forest. The reforestation program has started to take encouraging steps with increasing participation of individuals, communities and enterprises. As a result, the area of plantations, for instance rubber plantations, increased significantly from just under 1 000 ha in 1990 to over 200 000 ha in 2007; an expansion largely funded by foreign direct investment. Tree plantations are likely to make only a small contribution (covering perhaps 500 000 hectares) to the overall plan to restore 7 million hectares of forest (Phimmavong *et al.*, 2009). Conversion of degraded lands to secondary forests rather than to monoculture plantations of exotics is seen as the main restoration approach for meeting the diverse product needs of the local people, other stakeholders, and changing market demands, while simultaneously enhancing biodiversity and ecosystem services. The outcomes of pure plantation expansion are being criticized by conservation groups, non-government organizations and, most recently, members of the National Assembly of Laos. This stand point might be maintained by the observation of secondary forests that make up the largest share of forest cover in the lower Mekong basin, more than 80% of which is located in Laos and Cambodia (Heinimann *et al.*, 2007). Although there is scientific evidence that plantations can play a catalytic role in fostering the regeneration of native forest species in the understory (Parrotta *et al.*, 1997a; Carnus *et al.*, 2006), the re-colonization of the understory by native plants has limited value if they are cleared in preparation for the next cycle of planting.

Secondary forests have gained considerable attention and are believed to have enormous potential for restoring forest ecosystems in the Tropics, due to their socio-economic and environmental importance, their rapid growth and the current pressure on remaining old-growth forests (Finegan, 1996; Chokkalingam & Jong, 2001; Guariguata & Ostertag, 2001). Several studies have shown that tropical secondary forests are important sources of timber and non-timber forest products (Chazdon & Coe, 1999), and provide environmental services such as protection from soil erosion and sequestration of atmospheric carbon dioxide (Silver *et al.*, 2000). In addition, they are important templates for forest ecosystem restoration and refugia of biodiversity in fragmented landscapes (Lamb *et al.*, 1997; Lamb, 2011). In contrast, swidden agriculture has several major adverse effects, include reductions in soil nitrogen contents and stocks of soil organic carbon, coupled with increases in the pH and bulk density of top soil layers (Dalle and de Blois, 2006). Two types of swidden (shifting) cultivation systems are often distinguished in Laos: rotational and pioneering. In rotational shifting cultivation, the most common type in Laos, swiddeners keep their villages in the same place but shift their cultivated plots according to a crop/fallow cycle that depends upon several factors, including their ethnicity, culture and history. In the pioneering shifting cultivation system, swiddeners move their whole village settlements from one site to another after several years (Dalle and de Blois, 2006), mainly because the nearby forest has become exhausted.

Due to the advantages of secondary forests, and disadvantages of swidden cultivation, there are major reasons for converting swidden lands to secondary forests. In addition, the significant progress made towards developing and implementing forest policies, laws and national forest programs, with the accompanying abandonment of swidden fallows, has substantially improved prospects for restoring forests in Laos. However, sound forest restoration requires a good understanding of ecological processes, particularly succession (Hardwick *et al.*, 2004; Aronson & Van Andel, 2008), in order to assist natural recovery by maximizing the efficiency of applied measures and minimizing adverse human inputs and the cost of restoration work. A limitation in this respect is that most studies on secondary forest succession have focused on wet or humid tropical forests (Brown & Lugo, 1990; Finegan, 1996; Guariguata *et al.*, 1997; Guariguata & Ostertag, 2001; Peña-Claros, 2003), although some attempts have been made to study secondary succession in seasonally dry forests (Janzen, 1988a; Perkulis *et al.*, 1997; Kennard *et al.*, 2002; Kalacska *et al.*, 2004; Castro Marin *et al.*, 2009). Since scientific knowledge pertaining to forest restoration is not sufficiently well-developed in Laos, and there is a need for site-specific restoration approaches, further studies are required to improve understanding of factors influencing the recovery of secondary forests on abandoned lands (such as swidden fallows) and to develop optimal restoration techniques.

1.2 Restoration

1.2.1 Concepts and principles

Ecological restoration, as defined by the Society for Ecological Restoration (SER, 2004), is "the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed". It is an activity that is intended to initiate or accelerate the recovery of a given ecosystem in terms of its health, sustainability and integrity. Generally, ecosystems targeted for restoration have been damaged, degraded, transformed or destroyed as a consequence of human activities, either direct or indirect (SER, 2004; Lamb *et al.*, 2005; Falk, 2006; Rodrigues *et al.*, 2009). In some cases, damage to ecosystems caused by natural disturbances (e.g. wildfire, floods, storms, landslide or volcanic eruption) may be so severe that the ecosystems cannot recover their pre-disturbance or historic states (SER, 2004). However, in cases where restoration is possible, three basic restoration concepts (depending on the goals) can be distinguished: reclamation, rehabilitation and restoration (Mansourian *et al.*, 2005; Rodrigues *et al.*, 2009). Reclamation entails actions that usually involve site amelioration to permit vegetation to establish and colonize the site thereby increasing its utility or economic value. Indigenous ecosystems are rarely used as references in reclamation programs, and exotic species are generally used to overcome degradation. Rehabilitation entails actions that seek to repair damaged ecosystem functions, particularly productivity, quickly. Indigenous species and ecosystem structure and function are usually the rehabilitation targets, but exotic species might also be included. Restoration involves actions designed to restore degraded ecosystems to their presumed historic conditions. There are two main kinds of restorative actions (Lamb *et al.*, 1997): those that lead to full recovery of an ecosystem to its pre-disturbance state (restoration *sensu strict*o) and those intended to reverse degradation and re-direct the trajectory towards some aspect(s) of an ecosystem that previously existed at the site (restoration *sensu lato*). In this thesis, the term restoration is applied *sensu lato*, as there are no pristine natural forests to serve as reference ecosystems, or to harmonize long-term forest ecosystem restoration goals and short-term socio-economic development objectives.

Effective forest restoration should reestablish fully functioning ecosystems through direct or indirect actions based on three main objectives: (i) the reconstruction of species-rich functional communities that are capable of evolving; (ii) the stimulation of any potential for self-recovery that remains in the area (resilience) whenever possible; and (iii) the planning of restoration actions in a landscape perspective (Rodrigues *et al.*, 2009). In addition to these three main objectives, the cited authors note there are generally the following site-level goals: the removal or minimization of (adverse) human impact; the creation or protection of a forest structure that can provide permanent shade; the retention or increases in numbers of woody species, and promotion of invasion by other organisms; the provision of shelter and food sources in order to retain the local fauna permanently; and the effective management of invasive exotic species (Rodrigues *et al.*, 2009). In practice, it is not straightforward to identify the best restoration strategy for a given area because several strategies would often be suitable, and each area has a unique disturbance history, degree of resilience, reference information, surrounding landscape, and legal and socio-economic background (White & Walker, 1997; Holl *et al.*, 2000; Rodrigues *et al.*, 2009). Therefore, development of a suitable dichotomous key is important to facilitate identification of the optimal strategy in order to minimize costs and time requirements while maximizing the effectiveness of the restoration measures.

1.2.2 Approaches to forest restoration

Given the extent of land degradation, conservation efforts have increasingly focused on natural recovery and the active restoration of degraded ecosystems in order to restore both ecosystem services and biodiversity (Chazdon, 2008; Rey-Benayas *et al.*, 2010). Such restorative efforts range from minimizing anthropogenic disturbances (e.g. grazing, fire and removal of water from rivers), thereby allowing natural or unassisted recovery (also known as "passive restoration" *sensu* Dellasala *et al.*, 2003; Rey-Benayas *et al.*, 2010) to active human intervention in efforts to accelerate and/or influence the successional trajectory (also known as "active restoration"). With mounting evidence of ecosystems recovering over periods of decades without human intervention, there is a dilemma as to whether active restoration is always necessary. It is, therefore, important to analyze the likely outcome of a passive restoration approach based on the natural ecosystem's resilience, past land-use history, and the surrounding landscape matrix, to identify the specific goals of the project and to assess the available resources in order to select restoration approaches that are likely to use resources efficiently and maximize the success of restoration efforts (Holl & Aide, 2010).

As a general guiding principle, Chazdon (2008) proposed a restoration "staircase" that links the state of degradation of an initially forested ecosystem and a range of restoration approaches to at least partially restore levels of biodiversity and ecosystem services given adequate time and financial investment (Fig. 2). If previous disturbance has left some residuals (e.g. soil seed banks, remnant trees, sprouts) that can serve as "succession primers", and the goal of restoration is to enhance ecosystem services, natural regeneration is the cheapest approach to restore moderately degraded forest ecosystems (Chazdon, 2003). There is ample evidence that if left alone moderately degraded forests and/or the vegetation on abandoned lands will develop into secondary forest (Aide *et al.*, 2000; Finegan & Delgado, 2000; Castro Marin *et al.*, 2009; Sovu *et al.*, 2009). However, a closed canopy may take up to several decades to develop (Holl & Lulow, 1997), and such recovery could result in a community with a species composition that fails to meet management objectives (Brown & Lugo, 1990; Aide *et al.*, 2000; Hooper *et al.*, 2005). In addition, successful establishment of later successional forest species in deforested areas has proven difficult throughout the tropical world due to the short-lived nature of tropical forest seeds and their poor ability to form viable soil seed banks (e.g. Teketay & Granstrom, 1997; González-Rivas *et al.*, 2009). Thus, the scarcity of propagules, particularly those of late-successional species, is often a major limiting factor for forest recovery in abandoned areas (Guariguata, 1998; Holl *et al.*, 2000; Zimmerman *et al.*, 2000; González-Rivas *et al.*, 2009).

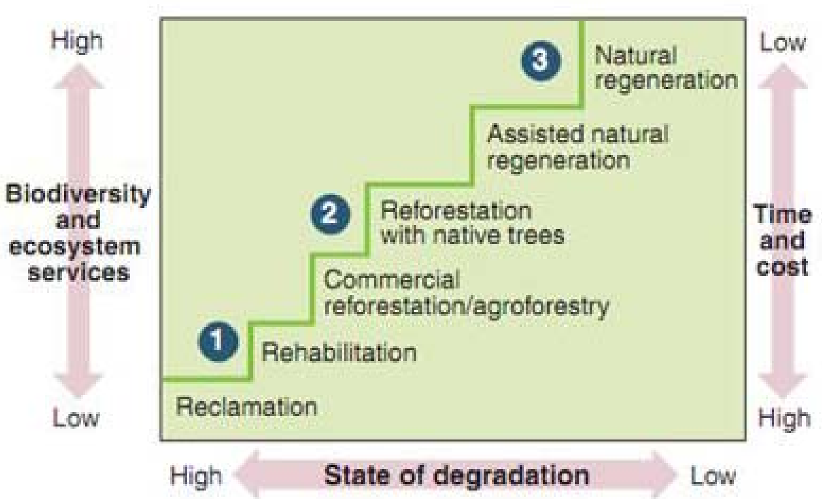

Figure 2. Chazdon's restoration staircase, linking the state of initial degradation of forest ecosystems and approaches to restore biodiversity and ecosystem services over time (Adapted from Chazdon, 2008).

The assisted natural regeneration (ANR) approach aims to balance the restoration of biodiversity in small areas through high-cost restoration planting and restoration of productivity through the establishment of commercial plantations in large areas. ANR accelerates natural succession by removing or reducing barriers to natural forest regeneration, such as paucity of propagules, soil degradation, competition with weedy species, and recurrent disturbances (e.g. fire, grazing, and wood harvesting). The techniques applied include precluding disturbance from regenerating sites, and mowing competing weeds around naturally established trees and shrubs (Dugan, 2000), using domestic animals for suppressing weeds and encouraging seed dispersal (Janzen, 1988b), erecting bird perches to facilitate seed dispersal (Holl, 1998; Scott *et al.*, 2000; Shiels & Walker, 2003), and direct seeding (Hardwick *et al.*, 1997; Engel & Parrotta, 2001; Cabin *et al.*, 2002; Camargo *et al.*, 2002; Woods & Elliott, 2004; Garcia-Orth & Martínez-Ramos, 2008), enrichment planting of missing primary species (Adjers *et al.*, 1995; Montagnini *et al.*, 1997; Ashton *et al.*, 2001; Paquette *et al.*, 2006), and/or soil amendment using commercial fertilizers (Ilstedt *et al.*, 2004) or readily available and affordable organic residues (Brunner *et al.*, 2004; Yamato *et al.*, 2006; Moyin-Jesu, 2007; Solla-Gullon *et al.*, 2008). Generally, it is a simple, cheap, and effective technique for converting degraded forests to more productive forests (Hardwick *et al.*, 1997; Shono *et al.*, 2007). Natural succession is often influenced by complex interactions of various factors including edaphic conditions, the amount and species composition of seed rain, and levels of seed and seedling predation. At the time of implementing ANR it is generally difficult to estimate the time required for restoration of the forest, but experience indicates that secondary forests of pioneer trees and shrubs regenerate in about three years following the implementation of successful ANR in former grazing lands (Dugan, 2000). In any case, if applied treatments are perceived to be failing to bring the desired changes in the vegetation sufficiently quickly, enrichment planting or a switch to conventional reforestation methods can be considered.

Restoration planting is widely recommended for heavily degraded sites, such as abandoned agricultural lands and excessively logged sites, or when the restoration goals include both conservation and production (Chazdon, 2008). The various restoration planting methods tested hitherto include dense planting of a large number of primary forest species (Miyawaki, 1999), staggered planting of primary forest species (Knowles & Parrotta, 1995), and the framework species method (Goosem & Tucker, 1995; Elliott *et al.*, 2003; Shono *et al.*, 2007). The major constraints of these methods are the high labor and financial inputs required, which limit their applicability to relatively small-scale projects (Lamb, 1998). Establishment of commercial tree plantations is another approach that can be used to overcome degradation, while ensuring financial returns (Lamb, 1998). Several studies have demonstrated that plantations can play a catalytic role in fostering the regeneration of native forest species in the understory (Lugo, 1997; Parrotta *et al.*, 1997b; Yirdaw, 2001; Senbeta *et al.*, 2002; Yirdaw & Luukkanen, 2003; Lemenih & Teketay, 2005; Carnus *et al.*, 2006). Integrating biodiversity conservation measures in commercial plantation regimes has, thus, becoming increasingly important, and planting high-value native trees and species mixtures is favoured in many countries (Lamb *et al.*, 2005), although monocultures of exotic timber species, such as *Acacia*, *Eucalyptus* spp., *Pinus* spp., *Tectona grandis* (teak) and *Gmelina arborea*, still dominate commercial plantations for their well-known silviculture and productivity.

1.3 Conceptual framework of the studies

The conceptual framework outlined below provides an overview of the functional linkages between deforestation and degradation of forests and the forest restoration techniques explored in this thesis in order to enhance the functionality of degraded and secondary forest in Laos (Fig. 3). As in many tropical areas, forest degradation in Laos starts with clearing forests for agricultural use or logging. While excessive logging leads to under-stocked open forest and bush/shrub land formations, clearing for permanent agriculture in the lowlands and swidden cultivation in the highlands of the northern part of Laos are the predominant causes of forest loss (Giri *et al.*, 2003). Some of the mountainous forests have also been heavily degraded and eventually converted to grazing lands. The other non-forest land, including pasture and grassland, may be converted to secondary forest or planted forest through active or passive restoration measures. Mismanaged secondary and planted forests may be converted to degraded land.

The conceptual framework indicates that in Laos primary forest is often converted to non-forest land use after land clearing for swidden cultivation. When the soil fertility status of swidden fileds diminishes and no more supports good crop growth and yield, crop cultivation is abandoned for a period of time to let the swidden field recover its fertility. During this fallow period secondary forests developed on abandoned swidden fields, which could in turn evolve into closed forest over a long time, if the abandonment is permanent. Generally, the diversity of woody species and structural attributes of secondary forests gradually increases with time since abandonment of fallow (Lawrence, 2004; Lebrija-Trejos *et al.*, 2008) and/ or fallow age (Derouw, 1995; Fujisaka *et al.*, 2000), crop-fallow rotation cycles, and distance to the remaining forest fragments for some species with dispersal ability that compensates for a negative response to repeated cultivation (Lawrence, 2005; Dalle and de Blois, 2006). As the first step in any forest restoration endeavor, abandoned fallows were assessed to quantify actual and potential levels of natural regeneration, to examine the barriers to natural regeneration and to determine whether interventions are needed to accelerate the natural regeneration process.

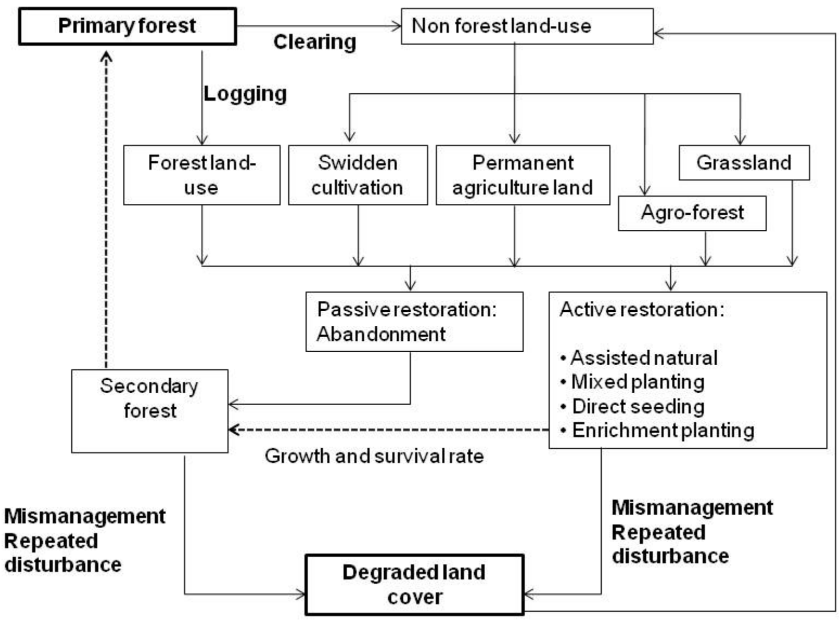

Figure 3. Overview of the functional linkages between forest and non-forest conditions and forest restoration techniques intended to enhance the functionality of degraded forest in Laos (- - -▶ Main goals of the restoration process in the thesis). (Adapted from ITTO, 2002).

The findings from this study revealed that the rate of recovery of secondary forests to mature forest attributes on fallows vary greatly and it appears that the successional stage is at establishment phase, dominated by bamboos (Sovu *et al.*, 2009). The challenges facing forest restoration on swidden fallow are, thus, to balance the species composition and to amend the soil fertility with economically feasible approaches. Normally, soil nutrients are depleted before swidden fields are left to fallow, hindering the subsequent recovery of forests (Johnson *et al.,* 2000, Moran *et al.* 2000). Changes in soil texture associated with swidden cultivation can also strongly affect forest recovery in terms of vegetation structure, species composition and rates of aboveground biomass accumulation during secondary succession (Johnson *et al.*, 2000; Moran *et al.*, 2000; Zarin *et al.*, 2001; Chinea, 2002). The low soil fertility in swidden fallows can be ameliorated by using readily available and affordable organic residues, such as rice husk, which is a readily available bioresource in Southeast Asia, where rice production is the dominant farming system. Carbonized rice husks (also called rice husk biochar, rice husk charcoal), produced by the partial combustion of rice husks, contain high proportions of plant nutrients — including silica (80.26%), phosphorus (0.38%), potassium (1.28%), magnesium (0.21%) and calcium (0.56%), according to Hashim *et al.* (1996) — and hence can be used for amending problematic soils, such as fallows. The application of biochar has been shown to improve soil chemical properties by neutralizing its pH and increasing its total nitrogen, available phosphorous and exchangeable cation contents, cation exchange capacity and base saturation, while inducing reductions in levels of exchangeable Al ions, which impair root growth (Ogawa and Okimori, 2010). Biochar also enhances the physical properties of soil, e.g. by raising its porosity and water holding capacity, as well as the inoculation efficacy of root nodule bacteria and mycorrhizae (Warnock *et al.,* 2007). Consequently, an active restoration measure was pursued to expedite forest recovery on small-scale swidden fallows to enable farmers meet their short-term needs for wood and wood products, using mixed-species planting and addition of rice husk biochar.

Financial and technical feasibilities are key considerations in any restoration endeavor. In mountainous areas like northern Laos, where establishing a nursery to produce and maintain plants can be costly, direct seeding can be utilized as a cost and time saving method to expedite the recovery process (Lamb, 2011). Direct seeding for reforestation has recently regained favor over the use of nursery-raised seedlings for planting due to the high cost of the latter (Hardwick *et al.,* 1997; Woods and Elliott, 2004). Despite its potential as a cheaper method of restoration, it is often considered to be less reliable (Brown and Lugo, 1990) and challenging (Willoughby *et al.*, 2004) due to seed and seedling mortality induced by biotic (pathogens, predation by granivores and herbivores, and competition) and abiotic (extreme temperatures, frost, drought, and sun scorch) factors. These drawbacks can be circumvented by choosing species suitable for direct seeding (Engel and Parrotta, 2001), seeding method and mechanical site preparation (Löf and Birkedal, 2009; Woods and Elliott, 2004), timing of seeding (Löf and Birkedal, 2009; Vieira *et al.*, 2008; Doust *et al.*, 2008), seed treatments that prevents seed predation (Birkedal 2010) and desiccation (Woods and Elliott, 2004) and combination of these approaches. The success of formerly established direct seeding trial established on former grazing land was evaluated in relation to the nature of the seeds, sowing methods and topographic locations.

Secondary succession on logged-over tropical deciduous forest is often characterized by a prevalence of low commercial value species. Enrichment planting is, therefore, the preferred method of assisting the natural regeneration of logged-over forests when desirable species are absent or are present at low densities (Appanah and Weinland, 1993). It involves planting of species of commercial or high local values using different approaches such as line-, under- or gap-planting. Line planting consists of cutting lines or transects of a given width through the existing vegetation and planting seedlings of the desired species at regularly-spaced distance along these lines. The light level created by the line depends on the direction and width of lines, and on the height of the surrounding vegetation, which are species-specific (Ådjers *et al.,* 1995; Montagnini *et al.,* 1997; Peña-Carlos *et al.*, 2002). Although line planting has been applied successfully in different places, it is not widely practiced due to the high cost of regular maintenance of the lines and rigid geometric patterns, making the stand look less natural. Contrary to planting in regularly spaced lines, gap planting has better resemblance to natural gap dynamics (Denslow, 1987), which could be considered advantageous in restoration programs. A study was, thus, conducted to identify the optimal enrichment planting method *vis-à-vis* planting gaps and planting lines, and to select species that are suitable for rehabilitating a heavily logged-over mixed deciduous forest.

1.4 Relevance of investigation for the restoration of degraded forests in Laos

Over-exploitation and fragmentation of natural forests have been typical problems in Southeast Asia, though its intensity varies from one country to another. In Lao PDR, as in all of the countries in the lower Mekong basin, there have been high rates of deforestation and forest degradation during the past half century. Forest cover declined from 70% of the land area in the 1940s to 47% or less by 1999. It has been estimated that about 129,000 ha per annum of forest was lost in the 1980s and 1990s (FAO, 1995). This represents a substantial loss in productivity, biodiversity and ecosystem function for the country. The causes of the forest losses include forest encroachment by permanent cultivation; swidden agriculture, forest fires, logging (legal and illegal), and infrastructure development. It has been estimated that 6.5 million ha of forests are affected by swidden agriculture (Messerli *et al.*, 2009), and industrial logging may become a more serious threat in the future as timber companies look for new sources of raw material (Thapa, 1998). As mentioned above, forest restoration on degraded and abandoned sites has gained increasing attention in Laos due to increasing abandonment of swidden fallows as a result of strict government policy to end the damaging practices associated with swidden cultivation since the 1980s (Kingsada, 1998). The Lao government plans to increase the national forest cover to 70% by the year 2020 through the establishment of plantations and natural regeneration of degraded forests, including forests on fallow land, as stipulated in the Forestry Strategy 2020 document. The core strategy of the national forest policy is to recover forest cover lost in recent decades while increasing forest biodiversity and improving economic benefits from forest resources.

Therefore, knowledge of factors that may impede forest regeneration and seedling establishment at deforested and abandoned sites would be highly valuable for guiding the national effort. For instance, distances to natural forest, disturbances and bamboo are important factors for the recovery of woody species in abandoned shifting cultivation lands in Laos as they affect seed dispersal and competition for nutrients at the stand level. Without management, rapid increases in the biomass of dwarf bamboo would prevent the regeneration of any tree species (Ito and Hino, 2007). Moreover, soil fertility is usually depleted before swidden fields are left to fallow, exacerbating the slow recovery of forests on swidden fallows (Lawrence, 2004; Lebrija-Trejos *et al.*, 2008). The challenge posed by low soil fertility in swidden fallows can be met using readily available and affordable organic residues such as sawdust, wood chips, tree bark, raw agricultural residues, and municipal biosolids that can improve the soil (Brunner *et al.*, 2004; Yamato *et al.*, 2006; Moyin-Jesu, 2007; Solla-Gullon *et al.*, 2008). However, there has been no prior investigation of the feasibility of using rice husk (biochar) for amending problematic soils before tree plantation. Converting some of the small-scale swidden fallows into mixed-species plantations of native species would be a viable option to achieve the desired economic and environmental goals such as diversifying rural livelihoods while maintaining ecosystem services. Therefore, efforts were made in the studies underlying this thesis to investigate factors influencing the recovery of secondary forests on abandoned fallows and to test site-specific approaches to restore forests on fallows, former grazing lands and a logged-over mixed deciduous forest in order to facilitate the design of appropriate restoration strategies in Laos.

2 Objectives

The general aim of the studies presented in this thesis was to generate knowledge that supports forest recovery on degraded lands to enhance ecosystem services and goods accrued from restored forests. The studies focused on forest recovery on abandoned swidden fallows, former grazing lands and logged-over forests and evaluated several possible restoration techniques to accelerate the recovery process. The specific research questions addressed were:

- What factors influence the recovery of secondary forests on abandoned swidden fallows? (Paper I)

- Are mixed-species planting and amendment of soil fertility using rice husk biochar effective methods for restoring forests on small-scale swidden fallows? (Paper II)

- How effective is direct seeding of native tree species to restore forests on former grazing lands? Do the selected seeding techniques, site conditions and the nature of the seeds influence the success of direct seeding? (Paper III)

- What enrichment planting methods (gap *vis-à-vis* line planting) are most suitable for restoring commercial tree species in logged-over forests? (Paper IV)

The overall hypothesis of the studies was that the optimal forest restoration depends on the state of site degradation and societal needs, and its success depends on various biotic and abiotic factors.

3 Materials and methods

3.1 Study sites

The studies were carried out in two provinces of Laos: Vientiane, representing the lowlands, and Xieng Khouang, representing the highlands (Fig. 4). The recovery of secondary forests on abandoned swidden fallows (Study I), the efficacy of mixed-species planting in combination with addition of rice husk biochar (Study II) and the utility of enrichment planting in a logged-over tropical mixed deciduous forest (Study IV) were investigated in Sang Thong District, located about 70 km north-west of Vientiane, the capital of Laos. Forests in this district were virtually undisturbed until the early 1970s, before being intensively logging, which was banned in the early 1990s. However, forest degradation continued as a result of ongoing illegal logging, the open access to forests due to the gaps and pathways created by loggers, the government policy of utilizing degraded forest for agriculture, and population pressure from immigration (Thapa, 1998; Ohlsson, 2009). Man-made fire (associated with swidden agriculture practices) is the main disturbance agent affecting the forest in the region; wind throw has been much less influential due to the low wind speeds and landlocked situation of the country. The sites are characterized by rugged topography with altitudes varying between 200 and 400 m above sea level. The region has a typical tropical monsoon climate, with distinct rainy (May to November) and dry seasons (November to April). Based on data collected by the Department of Meteorology in Vientiane from 1995 to 2005, the mean yearly rainfall amounts to 1647.1 ± 16.4 mm during the rainy season, the mean daily temperature during the year is 26.74 °C ± 0.66, while the relative air humidity varies between seasons from about 63-64% in

March and April to ca. 72±1.2% from June to September. The bedrock consists mainly of sandstone, and alisol is the dominant soil type (MAF, 1996). The direct seeding experiment (Study III) was established in Xieng Khouang Province, most commonly known for the intriguing 'Plain of Jars', located in north-central Laos (19 27'00 N, 103 11'00 E) about 173 km from Vientiane. It has mountainous topography, with altitudes varying between 1000 and 1100 m above sea level, mean (± SE) annual rainfall of 1467.96 ± 137.63 mm and mean daily temperature during the year of 20.40 °C ± 0.16. The mean monthly precipitation and temperature during the period 2002-2006 in the two regions (Xieng Khouang and Vientiane) are depicted in Figures 5 and 6. The mean annual wind speed at the site is 3.12 ± 0.16 m/s, and is the highest in the country. The soils are mainly yellow-red lateritic loamy soils derived from quartz, with pH varying between 3 and 5. The hills around the plain consist mainly of sandstone, granite and schist, with medium-rich loam soils.

The site selected for study I was located in an area of secondary forests developed on swidden fallows surrounded by mixed deciduous forests, comprised of clusters of bamboo and a tree layer with a low abundance of high-value commercial tree species, e.g. *Afzelia xylocarpa*, *Dalbergia cochinchinensis* and *Pterocarpus macrocarpus*. The site for Study II was a former swidden cultivation site dominated by bamboo and herbaceous species. Study III was conducted on former grazing lands surrounded by mixed coniferous-broadleaved forests. The main conifer species are *Pinus kesiya*, *Pinus merkusii* and *Keteleeria evelyniana*, while the main broadleaved species are *Schima wallichii*, *Alstonia rostrata* and *Quercus serrata*. The site selected for Study IV was a mixed deciduous forest that had been heavily degraded by logging.

Figure 4. Map of Laos, showing the locations of the study sites in the provinces.

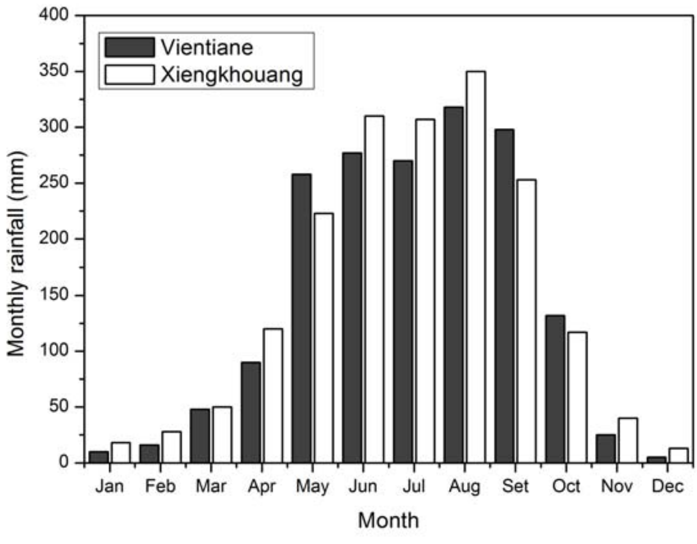

Figure 5. Average monthly precipitation during the period 2002-2006 at Vientiane and Xiengkhouang.

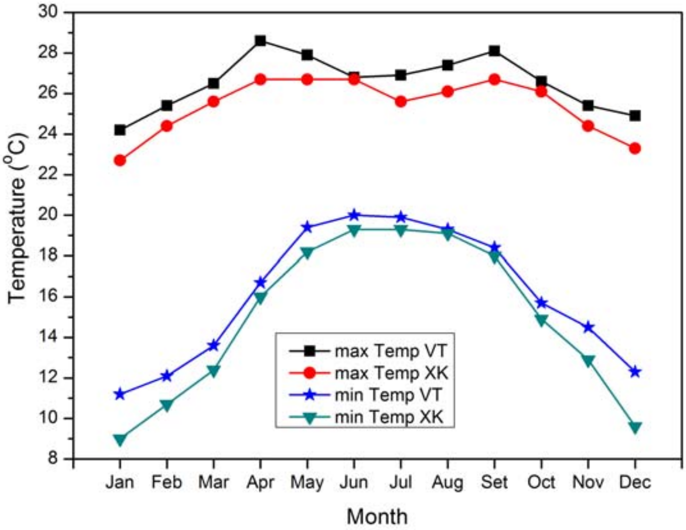

Figure 6. Average minimum and maximum temperatures (°C) at Vientiane (VT) and Xiengkhouang (XK)

3.2 Experimental design and data analysis

3.2.1 Study 1

This study examined the recovery of secondary forests on abandoned swidden fallows in relation to their distance from natural forest fragments, fallow age and crop-fallow rotation cycle. An inventory of the secondary forests was carried out in 2008 on 290 plots in 162 swidden fallows, laid along 15 parallel transect lines. Each transect line was 1000 m long, and was laid between two nearby natural forest fragments. The starting point of the first transect line was selected randomly, and other lines were spaced 200 m apart. Starting from the edge of the nearby natural forest fragment, sample plots (10 × 10 m) were laid out every 50 m along each transect line, and the last plots were also 1000 m away from the side of the other natural forest fragment. For sample plot which was positioned across the edge of a fallow patch the mirage method from Schmid-Hass (1969) was used. In each plot, all woody species with stem diameter >1 cm were identified, counted and their diameter at breast height (1.3 m, dbh) was measured. Bamboo species were also identified and the number of clumps was counted. Bamboo clumps encountered at the plot edge were counted if half of the clump's diameter was within the plot. Plants were identified *in situ* in the field by an experienced botanist from the Faculty of Forestry (FOF) and a villager with good knowledge of plant names in the local language. Specimens of the unidentified species were collected and later determined at the FOF, based on reference material.

Fallow age (the number of years since the swidden cultivation lands had been abandoned for more than six months) and crop-fallow rotation cycle were recorded through interviews with members of the local households responsible for preparing swidden fields, following the procedure described by Dalle and de Blois (2006). Farmers were asked to estimate the time since the forest was cleared and the number and length of previous crop-fallow rotation cycles. We subsequently corroborated the estimated years of the successive cycles by referring to key events (such as floods, the paving of access roads, and several government programs for agriculture) that most people in the area remember. In addition, we verified dates of specific events mentioned in the interviews, such as a marriage or land-use conflict, by consulting relevant certificates or other independent sources, which helped us to confirm estimated lengths of crop-fallow rotation cycles. Finally, each fallow was assigned to one of three crop-fallow rotation cycles (designated 1, 2 or 3, as illustrated in Figure 7).

Ecological indices were computed to characterize the floristic composition of secondary forest developed on the swidden fallows. The effect of distance from natural forest on the computed vegetation characteristics was then examined by curve-fitting to find the best function for describing the relationship. The effects of number of crop-fallow rotation cycles and fallow age on species richness, structural characteristics and diversity measures were analyzed using two-way ANOVA, and results of the statistical analyses were considered significant if $p < 0.05$ and to show tendencies if $0.05 < p < 0.1$. Moreover, Pearson correlation coefficients were computed to investigate the relationships between number of bamboo clumps and species richness, density, basal area, and diversity measures.

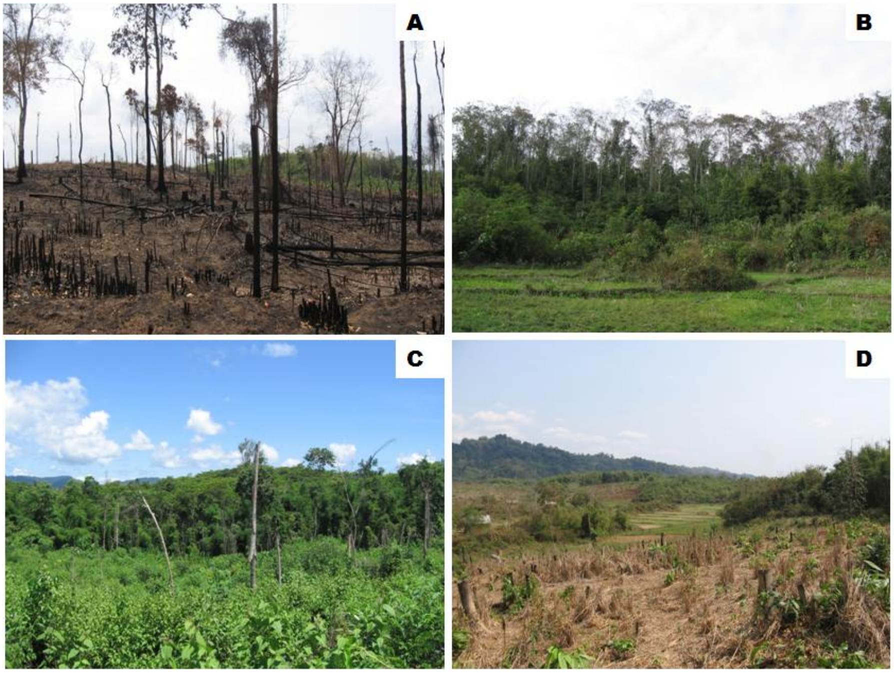

Figure 7. Fresh swidden field on degraded natural forest caused by logging (A); first cycle, 25-year-old fallow close to a permanent paddy field (B); second fallow cycle, 1-year-old fallow (C); third cycle, 3-month-old fallow after harvesting rice (D)

3.2.2 Study 2

In this study, the feasibility of amending soil fertility to expedite the establishment and growth of mixed-species planted on swidden fallows was evaluated. In 2006 a factorial experiment was established in Napo village, Sang Thong District, in which two soil amendment treatments (NPK fertilizer or rice husk biochar applications) were applied, with non-amendment controls, and eight native tree species were planted. The species planted in this study were *Afzelia xylocarpa* Craib, *Dalbergia chochinchinensis* Pierre, *Peltophorum dasyrachis* Kurz ex Baker, *Pterocarpus macrocarpus* Kurz, *Sindora siamensis* Teijsm. ex Miq. and *Xylia xylocarpa* (Roxb.) W.Theob, *Dipterocarpus alatus* Roxb and *Aquilaria crassna* Pierre ex Lecomte), which have contrasting ecological requirements and a variety of socio-economic uses (Table 1). The experimental site (1 ha) was divided into four 0.19-ha blocks and within each block three planting rows (160 m long) were established, with 4 m gaps both between rows and between blocks. In each block, the soil amendment measures were randomly assigned to the planting rows, while the species were randomly assigned to each planting row within each block. The inorganic fertilizer was applied at a rate of 100 g per seedling at the time of planting and 200 g per seedling three months after planting. The rice husk biochar was applied (mixed with soil excavated from the planting holes) two weeks before planting at a rate of 200 g per seedling, followed by addition of the same amount three months after planting, giving a total biochar application equivalent to 4 m ton/ha. As this study is the first attempt to apply rice husk biochar directly on planting sites, we considered an application rate lower than the rate recommended for crops (10-20 m ton/ha) by the Food and Fertilizer Technology Center (www.fftc.agnet.org) to avoid potential inhibition of seedling establishment and growth. The rice husk biochar was produced by incomplete combustion of the rice husks (60-70% by volume) for eight hours using the traditional charcoal-making method (Fig. 8). For each soil amendment measure, 10 nursery-grown seedlings per species and a total of 30 seedlings per block were planted along each planting row in May 2006 using the same planting hole-size (30 × 30 × 30 cm) for all treatments. Manual weeding was carried out twice per year in July and October during the rainy season. The survival rate, root collar diameter and height of the planted seedlings were assessed at the beginning of the rainy season in the two following years, while the diameter at 1.3 m and height of saplings that developed from them were measured four years after planting.

Mean survival rates and growth parameters of each species, in rows subjected to each soil amendment treatment, were computed per block (the blocks being considered as replicates during the statistical analyses). Prior to ANOVA, the survival rates were arcsin-transformed to meet homogeneity of variance and normality assumptions for analysis of variance (Zar, 1996), and three-way ANOVA was applied to compare differences in survival rate related to species, soil amendment measures and time since planting. In addition, two-way ANOVA was applied to examine differences in seedling and sapling growth related to soil amendment measures and species. Results of the statistical analyses were considered significant if Bonferonni-adjusted p-values < 0.016, and means that showed significant differences were compared using Tukey's test ($p < 0.05$).

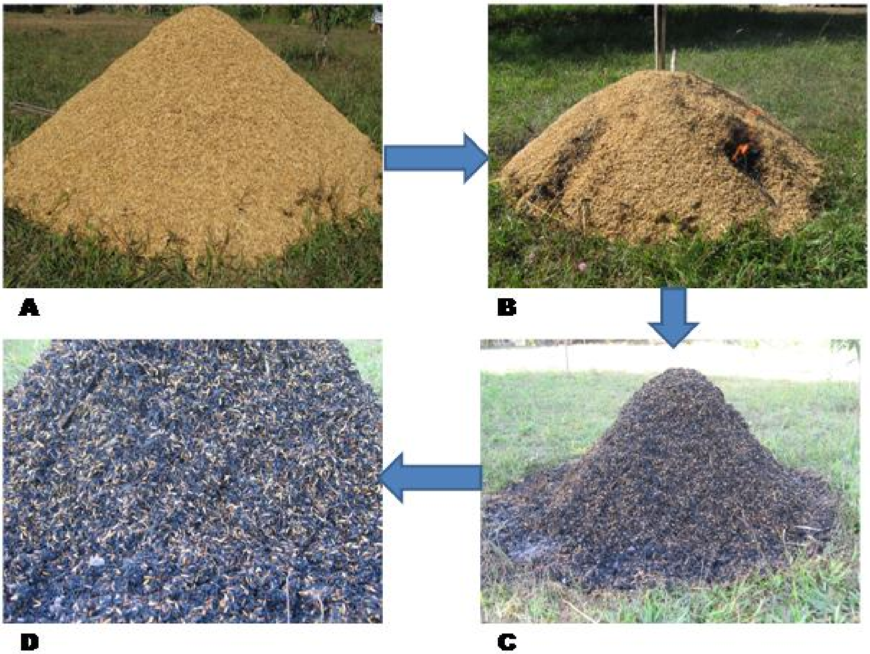

Figure 8. Production of biochar from rice husk (A) by igniting the heap at several points around their circumference with burning bamboo sticks (B), letting the rice husk burn slowly for 8 hours (C) then cooling the carbonized rice husks with water to prevent them turning into ashes (D)

3.2.3 Study 3

This study examined the suitability of direct seeding for forest restoration on former grazing lands. In May 2001, rehabilitation plots covering 29.8 ha were established using direct seeding of four species, *Schima wallichii* Choisy, *Quercus serrata* Thunb., *Pinus kesiya* Royle ex Gordon, and *Keteleeria evelyniana* Beissn (see Table 1 for species descriptions) by the Namgum Watershed Cooperation Project (NAWACOP) at 14 sites distributed within nine villages in Phookood, Pek and Phaxay districts, Xieng Khouang province. Seeds of *K. evelyniana* were sown at all of the sites, *P. kesiya* and *S. wallichii* seeds at ten sites and those of *Q. serrata* at five sites, owing to the limited availability of local seed sources. Although soil pH and NPK content do not differ significantly between the sites, the sites differ in terms of micro-climate, land use history and slope. Seeds for the rehabilitation trial were collected by hand from several mother trees in natural forests in close proximity to the experimental sites to ensure seed stocks were of local provenance. Before sowing, the open plots were ploughed to prepare a till suitable for sowing and to remove existing vegetation. Seeds were sown using two direct seeding techniques: even hand broadcasting (BC) for *P. kesiya, S. wallichii* and *K. evelyniana* seeds; and sowing with loose soil cover (seed burial) to a maximum depth of 0.5 cm (SB) for *K. evelyniana* and *Q. serrata* seeds. In SB methods, seeds were sown along the furrows at an interval of 30 cm. Seeds of the four species were sown mixed. Nine months after sowing (February 2002), four sample plots (4×4 m) were established for counting numbers of seedlings that had emerged. Seedling emergence rates were then calculated as the proportions of seedlings that emerged relative to the number of viable seeds sown per kg. Before sowing, the number of seeds per kg was estimated at 35,000, 250,000, 7,000 and 350 for *P. kesiya, S. wallichii, K. evelyniana* and *Q. serrata*, respectively. The growth performance of the species was subsequently assessed at three sites (Jar2, Nakhuan and Nahi), selected because they were well protected from disturbances by livestock grazing and well managed compared to the other sites. The height and root collar diameter of the planted seedlings were measured after three years while the height and dbh of saplings that developed from them were measured, using graduated poles and calipers respectively, after five years in 10 plots (5×5 m) established randomly at each site. Stem density was also counted one, three and five years after sowing on these sample plots.

Seedling establishment, stem density, height and diameter increments, and the annual rate of mortality were computed for each species and for each plot, then two-way ANOVA was applied to examine differences in these variables among species and sites. Before the analysis, the percentage of seedling establishment was transformed by an arcsine function to ensure normality. Since the species × site interaction was significant, one-way ANOVA was performed separately for each species. The results of the statistical analyses were considered significant if $P < 0.05$, and significant differences were further compared using Tukey's Honestly Significant Difference test. For *K. evelyniana*, which was sown using both broadcasting and seed burial methods, a pairwise t-test was performed to examine the effect of direct seeding methods on the studied parameters. The same test was used to compare establishment of *K. evelyniana* and *Q. serrata* seedlings, which were sown using seed burial as a direct seeding method.

3.2.4 Study 4

This study examined the feasibility of two enrichment planting methods (gap and line planting) to restore desirable species in a logged-over tropical mixed deciduous forest, established in 2000 by the Faculty of Forestry, National University of Laos, in an area of approximately 40 ha of logged-over forest. The forest was divided equally into two blocks (20 ha each), one of which was used for gap planting and the other for line planting. Before the rainy season, planting lines and gaps were created by selectively cutting down the aboveground layer (non-commercial trees and shrubs) and clearing the ground layer. A total of 16 gaps were created per ha; each gap covered an area of 8 × 8 m and the distance between two successive gaps was 20 m, measured from the gap center. For line planting, five lines 100 m long and 2 m wide were set per ha, spaced 20 m apart. The planting lines were oriented east to west to ensure the incidence of light. Within each block, seedlings of five indigenous tree species (*Afzelia xylocarpa* Craib, *Pterocarpus macrocarpus*, *Dalbergia cochinchinensis* Pierre, *Vatica cinerea* King and *Dipterocarpus alatus* Roxb) were planted (see Table 1 for species descriptions). Seedlings were raised at the experimental nursery of the Faculty of Forestry, National University of Laos, and in each gap 16 nursery-raised seedlings were planted at 2 × 2 m spacing, resulting in 256 seedlings per hectare. Each species was planted solely in a given gap. For line planting, 50 seedlings per line were planted at 2 m spacing, resulting in 250 seedlings per hectare. The lines and gaps were cleaned twice at the beginning and once per annum thereafter for the first five years. In 2007, seven gaps and lines each were randomly chosen for data collection for each species. During the inventory, all individuals in gaps and lines were counted, and their root collar diameter and height were measured.

Survival rate, mean height and root collar diameter were computed for each species and planting method and the resulting data were subjected to ANOVA. Prior to analysis of variance, the survival rate was arcsine-transformed to meet the normality and homogeneity of variance assumptions (Zar, 1996). Means that exhibited significant differences were compared using Tukey's test at the 5% level of significance. Diameter and height class distributions were constructed to examine the population structure of planted seedlings after seven years.

Table 1. *Distribution, habit, conservation status and economic importance of the species represented in the study*

Family	Species	Ecological distribution	Habit & status	Uses
Dipterocarpaceae	*Dipterocarpus alatus*★	Native to evergreen & dry deciduous forests of Cambodia, Laos, Myanmar, Philippines, Thailand, Vietnam, Bangladesh, Andaman islands; occurs gregariously along rivers at 0-500 m altitudes	Large tree up to 40 m tall and 150 cm in diameter; shade-tolerant and fast growing; endangered species	Timber, oleoresins, medicine, soil improvement
Dipterocarpaceae	*Vatica cinerea*★	Occurs in dry evergreen forests of Cambodia, Laos, Malaysia, Myanmar, Thailand, Vietnam	Large tree up to 30 m tall & 90-100 cm in diameter; evergreen tree; shade-tolerant; endangered species	Timber, medicine
Leguminoseae	*Afzelia xylocarpa*★	Native to Cambodia, Laos, Myanmar, Thailand, Vietnam; occurs in deciduous forests at 100-650 m altitudes	Tree up to 30 m tall & 150 c m in diameter; light-demanding; deciduous; endangered species	Timber, food, tanning, soil improvement
Leguminoseae	*Dalbergia cochinchinensis*★	Occurs in mixed deciduous forests of Cambodia, Laos, Thailand, Vietnam at altitudes of 400-500 m	Evergreen tree up to 30 m tall, 60-80 cm in diameter; drought-tolerant but slow growing; shade-tolerant when young; vulnerable species	Timber, soil improvement,
Leguminoseae	*Sindora siamensis*	Native to Cambodia, Laos, Vietnam, Thailand , Myanmar and Malaysia; occurs in dry dipterocarp, mixed deciduous and dry evergreen forest	Tree up to 30 m tall and 80 cm in diameter; light-demanding; deciduous; grows on poor and rocky sites; endangered species	Timber, medicine, soil improvement
Leguminoseae	*Peltophorum dasyrachis*	Native to Cambodia, Laos, Vietnam, Thailand and Myanmar; occurs up to 800 m altitudes; grows in secondary mixed deciduous and dipterocarp forests	Tree up to 30 m tall and 80 cm in diameter; light-demanding; deciduous, good colonizer as fast growing species; pioneer species with drought tolerance.	Timber, soil nutrient improvement, young leaves as fodder for cattle

Table 1 *(Continued)*

Family	Species	Ecological distribution	Habit & status	Uses
Leguminoseae	*Pterocarpus macrocarpus*	Native to Indochina, northern Thailand and Myanmar; introduced to India and the Caribbean	Medium-large tree (up to 25 m tall and 170 cm in diameter); light-demanding; deciduous, endangered species	Timber, medicine
Leguminoseae	*Xylia xylocarpa*	Native to Oceania, S.E Asia and South Asia; grows in mixed deciduous, dry dipterocarp and dry evergreen forests	Tree up to 30 m tall and 150 cm in diameter; light-demanding; deciduous, endangered species	Timber, medicine, soil improvement
Theaceae	*Schima wallichii*	Pioneer species that grows in wide ranges of climates, habitats and soils, up to 2400 m altitude.	Medium to large tree growing to 47 m in height; bole cylindrical, branchless, DBH up to 1 m	Heavy hard wood used for construction, pulp, firewood
Facaceae	*Quercus serrata*	Later-successional species, grows in deciduous forests below < 2000 m, in wet yellow-red ferralitic, rich loamy and calcareous soils in semi-shade (light woodland) or no shade.	Oak tree growing to 15 m in height and 1 m DBH	Wood - very hard, strong, red-brown in color, used for construction
Pinaceae	*Pinus kesiya*	Pioneer species, grows in areas with subtropical climates, with a distinct dry and rainy season, at 1000 to 1500 m altitudes.	Coniferous tree up to 40 m tall and 1 m DBH with straight, cylindrical bole, high light demanding and drought tolerant.	Soft & light timber used for paper pulp, construction,.
Pinaceae	*Keteleeria evelyniana*	Later successional species, grows in areas with wet, subtropical climates with a distinct dry and rainy season, at elevations > 600 m.	Coniferous tree up to 40 m tall and 1 m DBH; shade intolerant, prefers neutral or limestone soils.	Hard wood used in construction, mining and for timber
Thymelaea-Ceae	*Aquilaria crassna*★	Native to Tonkin and other countries of S. E. Asia; occurs in wide range of forest types, primary and secondary, and in the understory.	Tree up to 30 m tall and 80 cm in diameter; light-demanding; fast growing tree; endangered species	Aroma industry: perfume; medicine

★ IUCN Red list of Threatened Trees: cited as Critically Endangered, Endangered, and Vulnerable.
Sources: Bräutigam (1996); Ankarfjard and Kegl (1998); Lehmann *et al.* (2003); McCombe (1977).

4 Results

4.1 Secondary forests on abandoned swidden fallows

A total of 61 species, representing 56 genera and 27 families, were recorded in the secondary forests developed on abandoned swidden fallows compared to 204 species in a nearby natural forest fragment (Table 2). *Peltophorum dasyrachis* Kurz ex Baker and *Melanorrhoea laccifera* Pierre were the two most dominant species in the sampled fallows, accounting for 80.1% and 26.4% of the Importance Value Index (IVI), respectively. The rarest species were *Dialium cochinchinense* Pierre, *Ficus hispida* L.P, *Aglaia gagnepainiana* Pellegr. and *Chaetocarpus castanocarpus* Thwaites, which had similar IVI values of ca. 0.1%. With regard to commercial timber species, 19 were recorded in the secondary forests out of 84 timber species hosted in the natural forest fragment. The density of trees in the secondary forest was nearly half that recorded in the natural forest fragment while the basal area was nearly six times higher in the natural forest fragment than in the secondary forest. Commercial tree species accounted for 23% of stem density and 17% of the basal area of all trees recorded on fallows. In addition, two bamboo species, *Dendrocalamus lonoifimbriatus* and *Cephalostachyum virgatum*, were encountered, with 162 ± 5.9 clumps/ha and 3240 ± 159 culms/ha, respectively.

The floristic composition and structural attributes of secondary forests varied in relation to the distance from the forest edge, fallow age and number of crop-fallow rotation cycles. Distance from the forest edge influenced species richness, stem density, basal area and Simpson's index of dominance in a non-linear fashion (Table 3). Both species richness and stem density declined with increasing distance from the forest edge up to ca. 500 m, and increased thereafter. In contrast, the basal area was highest up to 300 m from the natural forest fragment, declining thereafter up to 800 m away and then started to increase again. Simpson's index showed a similar pattern to species richness and stem density. Distance from the natural forest edge did not significantly influence the number of bamboo clumps, or either Shannon-Wienner or Fisher's diversity indices ($p > 0.1$).

Table 2. *Vegetation characteristics of the nearby natural forest fragment and secondary forest developed on abandoned swidden cultivation fields, and the rate of recovery*

Vegetation characteristics	Secondary forest	Natural forest	Recovery rate (%)
Species richness	61	204	29
Stem density (no./ha)	1124	2917	39
Basal area (m^2/ha)	6.41	35.06	18
No. of commercial tree species	19	84	23
Density of commercial tree species (no./ha)	258	635	41
Basal area of commercial tree species (m^2/ha)	1.07	16.00	7

Table 3. *Relationships between diversity, structural characteristics and number of bamboo clumps on fallows with distances from natural forest edges and the number of crop-fallow rotation cycles (p-values within the bracket are for crop-fallow cycles) using non-linear regression.*

Vegetation characteristics	R^2	p-value
Species richness	0.46	0.08 ($p<0.01$)
Density	0.54	0.04 ($p<0.01$)
Basal area	0.63	0.06 ($p<0.01$)
No. of bamboo clumps	0.40	0.29 ($p<0.01$)
Fisher's diversity index	0.42	0.25 ($p>0.05$)
Shannon-Wienner index	0.36	0.16 ($p<0.01$)
Simpson's index	0.52	0.05 ($p>0.05$)

As the fallow age increased, the basal area of secondary forests increased significantly ($p < 0.01$) while stem density tended to decrease ($0.05 < p < 0.1$). The abundance of bamboo, as determined by the number of bamboo clumps, was not significantly affected by the fallow age. As the number of crop-fallow cycles increased from one to three, species richness, stem density, basal area and Shannon-Wiener index were reduced significantly ($p < 0.01$) by 28, 35, 72 and 23%, respectively. In contrast, the number of bamboo clumps increased significantly, by 45%. Species richness and diversity measures were not significantly affected by fallow age, while

Simpson's and Fisher's indices were not significantly affected by the number of crop-fallow rotation cycles ($p > 0.05$).

4.2 Mixed-species planting and biochar application

Neither rice husk biochar nor NPK fertilizer applications resulted in significant differences in survival rates of the planted seedlings during the two years following planting, relative to the control treatment ($p = 0.343$), but survival rates varied significantly among species ($p < 0.0001$). First and second order interactions between these variables were not significant ($p > 0.05$). Over all species and growth periods, the survival rates were 83%, 85% and 83% for the control, inorganic fertilizer and rice husk ash treatments, respectively. Among species examined in this study, the survival rates of *A. crassna* (72%) and *D. alatus* (73%) were significantly lower than those of *A. xylocarpa* (91%), *D. cochinchinensis* (88%), *X. xylocarpa* (88%), *P. macrocarpus* (87%), *P. dasyrachis* (86%) and *S. siamensis* (85%). The overall survival rates during the first two years following planting were 84% and 83%, respectively.

With regard to seedling growth, the root collar diameter of the seedlings one year after planting was significantly influenced by the soil amendment treatments ($p = 0.001$), species ($p < 0.0001$), and the interactions between them ($p = 0.002$). This variable was significantly enhanced by the application of NPK fertilizer, relative to the control treatment (increasing the mean diameter from to 20.8 ± 1.4 to 23.9 ± 1.5 mm), but not by rice husk biochar application (22.0 ± 1.2 mm) relative to either the control or NPK fertilizer treatments. During the subsequent year, the addition of both NPK. fertilizer and rice husk biochar resulted in greater root collar diameter than the control treatment (34.9 ± 2.5, 33.3 ± 2.2 and 30.5 ± 1.5 mm, respectively). The species with the smallest root collar diameter was *D. alatus* (13.0 ± 0.8 mm in the first year and 18.6 ± 1.0 mm in the second year) and the species with the largest diameter was *P. dasyrachis* (37.2 ± 1.5 mm in the first year and 62.9 ± 2.1 mm in the second year). The soil amendment treatments had significant effects on only two species: *A. xylocarpa* and *X. xylocarpa*. For *A. xylocarpa*, root collar diameter was significantly greater following NPK fertilizer (28.08 ± 1.0 mm) and rice husk biochar (24.03 ± 1.9 mm) applications than following the control treatment (17.35 ± 1.1 mm), while for *X. xylocarpa* the application of NPK fertilizer significantly increased the root collar diameter (30.31 ± 2.2 mm) compared to both the control treatment (21.46 ± 1.1 mm) and the addition of rice husk biochar (25.74 ± 1.4 mm). With regard to seedling height growth after one year, significant variation was observed among species ($p < 0.0001$), but it was not significantly influenced by either the soil amendment treatments ($p = 0.237$) or the interaction between these factors ($p = 0.291$). *P. dasyrachis* seedlings were the tallest (mean height, 172.4 ± 7.1 cm), and *D. alatus* the shortest (64.7 ± 4.4 cm). During the second year, seedling height was significantly ($P < 0.0001$) higher following NPK fertilizer application (245.0 ± 15.3 cm) than following rice husk biochar addition (225.4 ± 12.5 cm) and the control treatment (214.7 ± 15.3 cm). There were also significant between-species differences ($p < 0.0001$) in height in the second year; the three tallest species were *P. dasyrachis* (404.2 ± 12.1 cm), *X. xylocarpa* (263.8 ± 11.2 cm) and *A. xylocarpa* (234.5 ± 6.2 cm), while the shortest were *D. alatus* (127.1 ± 5.2 cm), *P. macrocarpus* (183.0 ± 7.2 cm) and *S. siamensis* (185.3 ± 8.8 cm).

Observations of the saplings that developed from the planted seedlings four years after planting showed that their diameter was significantly influenced by the soil amendment treatments ($p < 0.0001$) and species ($p < 0.0001$), but not the interactions between these factors ($p = 0.126$). The application of NPK fertilizer and rice husk biochar resulted in greater diameters than the control treatment (Table 4A). The species with the greatest mean diameter was *P. dasyrachis*, followed by *X. xylocarpa*, while the smallest diameters were recorded for *D. alatus* followed by *P. macrocarpus* and *D. cochichinensis* (Table 4A). The height of the saplings was also significantly affected by the soil amendment treatments ($p < 0.0001$) and species ($p < 0.0001$), but not by their interaction ($p = 0.341$). This variable was significantly enhanced by the application of rice husk biochar, relative to the control treatment, but not by NPK application relative to either the control or biochar treatments (Table 4B). Among species used in mixed planting, the mean sapling height was greatest for *P. dasyrachi*, followed by *X. xylocarpa*, and smallest for *D. alatus* followed by *P. macrocarpus* (Table 4B).

Table 4. *Diameter and height of saplings of the eight planted tree species following each of the soil amendment treatments (mean ± SE) after 4 years of planting. Overall means followed by different letter(s) are significantly different for the main effects of treatments (T) and species (S).*

A: Diameter (cm)

Species	Control	NPK fertilizer	Rice husk biochar	Main effect (S)
A. xylocarpa	3.63 ± 0.4	4.28 ± 0.5	3.50 ± 0.2	3.81 ± 0.2**c**
A. crassna	3.74 ± 0.4	3.58 ± 0.4	4.69 ± 0.2	4.01 ± 0.2**c**
D. chochichinensis	2.34 ± 0.2	3.51 ± 0.1	3.48 ± 0.6	3.11 ± 0.3**b**
D. alatus	1.61 ± 0.1	2.03 ± 0.1	2.63 ± 0.3	2.09 ± 0.2**a**
P. dasyrachis	9.34 ± 0.5	9.78 ± 0.4	9.73 ± 0.3	9.62 ± 0.2**e**
P. macrocarpus	2.23 ± 0.1	2.74 ± 0.2	3.46 ± 0.5	2.81 ± 0.2**b**
S. siamensis	3.12 ± 0.2	3.73 ± 0.5	3.28 ± 0.1	3.37 ± 0.2**bc**
X. xylocarpa	4.80 ± 0.6	5.89 ± 0.4	5.21 ± 0.4	5.30 ± 0.3**d**
Main effect (T)	3.85 ± 0.4**A**	4.44 ± 0.4**B**	4.50 ± 0.4**B**	

B. Height (m)

Species	Control	NPK fertilizer	Rice husk biochar	Main effect (S)
A. xylocarpa	4.38 ± 0.2	4.66 ± 0.5	4.04 ± 0.5	4.36 ± 0.3**c**
A. crassna	3.76 ± 0.3	3.97 ± 0.1	4.74 ± 0.2	4.15 ± 0.2**bc**
D. chochichinensis	3.12 ± 0.1	3.57 ± 0.2	4.49 ± 0.4	3.73 ± 0.2**bc**
D. alatus	2.43 ± 0.1	2.84 ± 0.2	3.36 ± 0.6	2.88 ± 0.2**a**
P. dasyrachis	9.32 ± 1.4	8.82 ± 0.8	10.79 ± 0.8	9.65 ± 0.6**e**
P. macrocarpus	2.94 ± 0.2	3.25 ± 0.1	4.12 ± 0.3	3.44 ± 0.2**b**
S. siamensis	3.77 ± 0.2	4.39 ± 0.2	4.20 ± 0.3	4.12 ± 0.1**bc**
X. xylocarpa	5.16 ± 0.6	5.84 ± 0.5	5.54 ± 0.5	5.52 ± 0.3**d**
Main effect (T)	4.36 ± 0.4**A**	4.67 ± 0.3**AB**	5.16 ± 0.4**B**	

4.3 Direct seeding

Proportions of seedlings that had established nine months after direct broadcasting of seeds differed significantly among species. They were higher for *P. kesiya* than *S. wallichii* and *K. evelyniana*, which had similar establishment success. For all species, there were conspicuous differences in seedling establishment between sites ($p < 0.01$), which was most successful at sites in School and Sui villages. The interaction between species and sites was also significant (p = 0.046), *P. kesiya* showing superior establishment at every site compared to the other species. Direct seeding with soil cover resulted in significantly higher seedling establishment success (41.63 ± 3.25%) than broadcasting (13.03 ± 1.30%) for *K. evelyniana*. For species planted using the seed burial method, seedling establishment was significantly higher for *Q. serrata* (59 ± 3%) than for *K. evelyniana* (42 ± 3%).

Table 5. *Seedling establishment success (%), height (cm) and diameter (cm) of four native species used in direct seeding trial. Values within brackets are root collar diameter after 3 years, and dbh after 5 years of sowing*

Species	Seeding method	Establishment success	Seedling height (root collar diameter)		
			9 months	3 years	5 years
P. kesiya	BC	38±3	11	156 (3.16)	275 (3.06)
S. wallichii	BC	14±2	17	165 (3.04)	267 (2.90)
K. evelyniana	BC; SB	13±1; 42±3	6	68 (0.99)	139 (1.34)
Q. serrata	SB	59±3	7	94 (1.29)	149 (1.55)

Analysis of subsequent seedling growth performance three years after sowing revealed significant differences among species, sites and interaction for both root collar diameter and seedling height. *K. evelyniana* seedlings had the smallest mean root collar diameter and shoot height, *P. kesiya* seedlings had the biggest mean root collar diameter and *S. wallichii* seedlings were the tallest (Table 5). Among restoration sites, seedling growth was superior in Nakhuan compared to Nahi and Jar2. Further assessment of growth at the age of five years showed that the diameter at breast height was greatest for *P. kesiya* saplings and at the Nakhuan and Nahi sites, while it was lowest for *K. evelyniana* saplings and at Jar2. Total height at the age of five years was also significantly higher for *P. kesiya* and *S. wallichii* than for the other species, but no significant difference in this respect was observed among rehabilitation sites.

The stocking density (number of stems/ha) of the four species used in the mixed direct seeding trial decreased with increasing age. The overall density per hectare five years after sowing was 5920, 5733, 2200, and 1573 for *S. wallichii*, *P. kesiya*, *Q. serrata* and *K. evelyniana*, respectively (Fig. 9). However, the rate of mortality during the first three years after sowing was highest for *S. wallichii* followed by *P. kesiya*, *Q. serrata* and *K. evelyniana*. Among rehabilitation sites, the annual rate of mortality during the same period was lower at the Jar2 site than at the Nakhuan and Nahi sites. During the subsequent assessment period (3-5 years after sowing), the annual rate of mortality was generally lower than during the first assessment period, and no significant difference was observed among species. Mortality was still lower at Jar2 than at Nahi during this period.

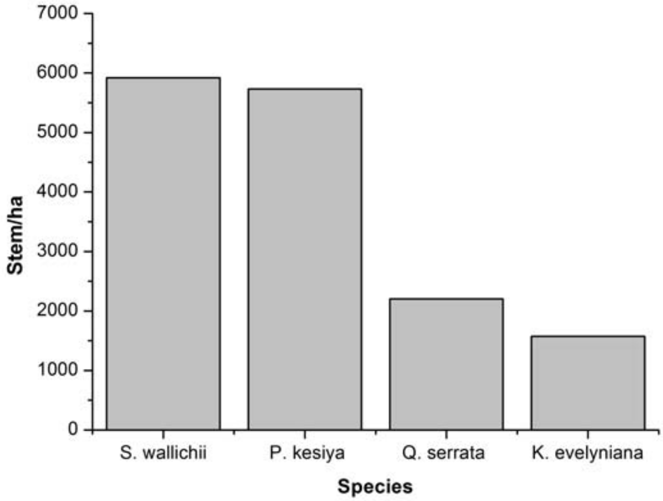

Figure 9. Mean stem density (number/ha), averaged over all sites, of the four native tree species five years after direct seeding

4.4 Enrichment planting

Seven years after planting, survival rates varied significantly ($p < 0.05$) among species, but not between enrichment planting methods. Root collar diameter and height varied significantly among species as well as between enrichment planting methods. Generally the survival rate was less than 55%, lowest for *P. macrocarpus* and *A. xylocarpa*, and highest for *D. alatus* and *V. cinerea*. Among species, *P. macrocarpus* and *A. xylocarpa* root collar diameters were significantly lower than those of *D. alatus* and *V. cinerea*, regardless of enrichment planting method. The root collar diameter of *D. cochinchinensis* did not differ significantly from that of *P. macrocarpus, A. xylocarpa* and *D. alatus,* while it was significantly low than that of *V cinerea*. After seven years *D. alatus* and *V. cinerea* plants were taller than those of the other species investigated in this study (Fig. 10). In addition, gap and line planting favored the height growth of *V. cinerea* and *D. alatus*, respectively, as evidenced by significant interaction effects of species by planting methods. Height growth did not differ significantly among the rest of the species.

The pattern of height class distribution also differed among species and between enrichment planting methods. A relatively large number of planted seedlings reached heights of 100-190 cm within seven years, for instance: *P. macrocarpus* in planting lines; *D. alatus*, *V. cinerea* and *D. cochinchinensis* in gaps; and *A. xylocarpa* in both gaps and planting lines. For all species, high proportions of planted seedlings grew to heights of at least 300 cm, and particularly high proportions of *D. alatus* in planting lines and *V. cinerea* in gaps were observed in the 300-390 cm and > 400 cm height classes, respectively.

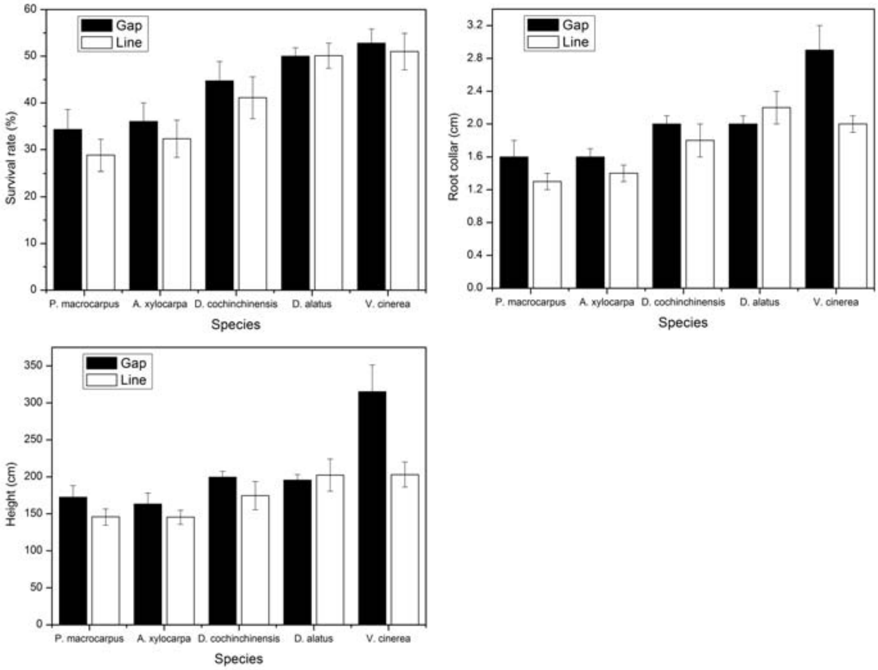

Figure 10. Survival rate, root collar diameter and height (Mean ± SE) of five tree species seven years after planting in gaps or lines in mixed deciduous forest.

5 Discussion

5.1 Natural regeneration

The results from the studies this thesis is based upon indicate that secondary succession on swidden fallows shows a similar pattern to those observed in many other tropical dry forests, with varying rates of recovery to forest with mature attributes (Kennard *et al.*, 2002; Molina Colon & Lugo, 2006; Lebrija-Trejos *et al.*, 2008). During 1-20 years of fallow, stem density increased to as high as 39% of the natural forest fragment value, but substantially less of the species richness and basal area recovered at our study sites during this period. The relatively slow recovery of species richness probably reflects the limited availability of propagules (seeds, live stumps or roots), which are crucial for initiating succession, since most species found in fallows are represented by few individuals and few species are generally present in them (Schmidt-Vogt, 2001). However, the stem density of commercial timber species recovered relatively rapidly, compared to their species richness and basal area, probably due to selective harvesting of timber species from the natural forests.

Contrary to previous studies which indicated a strong negative effect of distance from the forest edge on the number of species (Myster & Pickett, 1993; Cubina & Aide, 2001; Gunter *et al.*, 2007), the results from the present study showed only a declining tendency of species richness with increasing distance to the forest edge. This could be partly due to variations in the intensity of pre-fallow land use and the abundance of remnant trees in fallows, which may vary at different distances due to variations in farmers' swidden cultivation practices. The latter factor may substantially affect recovery rates, since remnant trees in fallows can serve as important nuclei for species establishment during secondary succession by providing both perches and food resources for seed dispersers, and hence promoting seed dispersal (Holl, 1999; Castro Marin *et al.*, 2009). In addition, as the intensity of land use increases, the potential of secondary forests to regenerate from soil-stored seeds or stump sprouts diminishes. In the study area, there are observable ethnic differences in the swidden farming practices of two groups of local people, Kammu and Lao, in terms (for instance) of numbers of crop-fallow rotation cycles and intensity of forest clearance. These differences could affect the fallow succession, as found in neighboring Thailand (Schmidt-Vogt, 1998). The Lao people fallow the land for longer periods than the Kammu people, who usually clear-cut all the vegetation without leaving some remnant trees on their fields (Delang, 2007). The lack of significant relationships between distance from the forest edge and diversity could be attributed to the dominance of a few species, especially *M. laccifera* and *P. dasyrachis*, across all distances from the natural forest, as also evidenced by the significant distance effect on Simpson's Index. These species are fast-growing pioneers, which establish quickly by exploiting opportunities created by disturbances.

Fallow history (age and number of cycles) appeared to be the major driver of secondary succession on swidden fallows. This finding is consistent with previous reports that the plant diversity, density, and composition of secondary forests that develop in abandoned swidden cultivation fields are all affected by the duration of fallow management and repetitions of swidden cultivation cycles (Staver, 1991; Derouw, 1995; Fujisaka *et al.*, 2000; Lawrence, 2004; Lawrence, 2005). The crop-fallow rotation cycle appears to have a strong negative effect on all vegetation attributes examined. The observed decline in species richness following repeated swidden cultivation could be attributable to systematic changes in the availability of "succession primers" as a result of repeated burning after clearing the fallows, which destroys most of the seeds stored in the soil, as well as root and stump sprouts and seedlings that appear after clearing (Whitmore, 1990; Nepstad *et al.*, 1991). Limits on the pool of species available to colonize fallows may cause long-term changes in tree diversity, which may be further exacerbated by inherent or induced soil impoverishment as land is subject to repeated cycles of swidden cultivation (Lawrence, 2005). The age of the fallows examined in the study presented in this thesis did not significantly affect species richness and diversity measures, mainly because 62% of all the species observed were encountered in fallows of all ages. In addition, the frequent occurrence of bamboo, irrespective of the age of the fallows, restricts the recruitment and establishment of woody flora due to its strong competitive ability (Arunachalam & Arunachalam, 2002). The tendency of stem density to decline with fallow age may be related to competition from bamboos, and the fairly high incidence of resprouting, which frequently leads to low individual densities and individual turnover rates (Kruger & Midgley, 2001).

5.2 Assisted restoration

Although succession proceeds relatively quickly during fallow periods, it is unlikely that species composition and re-establishment of primary forest species will reach pre-disturbance levels within a period of less than 200 years, as predicted in other parts of the world (Riswan *et al.*, 1985; Saldarriaga *et al.*, 1988). This challenge, coupled with low soil fertility in recent swidden fallows, should be overcome to enable small-scale fallow holders to obtain short-term economic benefits while maintaining ecosystem services. Planting a mixture of fast-growing pioneer and later successional native tree species in combination with soil amendment measures could be a viable option for restoring forests on swidden fallows to meet such short-term needs for wood and wood products, as evidenced by the good ecological combining ability of the species tested in our trial. The species used in the mixed planting test exhibited minor variations in survival rate, which generally ranged from 72% to 91%, depending on the species, irrespective of the soil amendment measures. This might be related to the nursing effect of the fast-growing pioneer species (e.g. *P. dasyrachis*) promoting the establishment of shade-loving species (e.g. *D. alatus*) by inducing increases in soil fertility or moisture, and providing protection from high irradiance, temperature, predation, or browsing (Aerts *et al.*, 2006; Gomes *et al.*, 2008). The soil amendment treatments did not result in significant improvements in survival rates in this trial, relative to the amendment-free control treatment. This may have been due to the use of container-grown seedlings, which have intact root systems with good soil contact, hence there is less resistance to water flow through the soil-plant-atmosphere continuum than when bare-rooted seedlings are used (Grossnickle, 2005).

The addition of rice husk biochar is expected to reduce bulk soil density, improve its aeration and water-holding capacity (Ogawa & Okimori, 2010). Hence, it should encourage lateral root formation and expansion of the rhizosphere zones that can be exploited by the roots, thereby promoting seedling growth. Contrary to this expectation, the addition of rice husk biochar did not improve the growth of seedlings of most species compared to the control treatment; even the inorganic fertilizer did not have significant effects. There are two possible explanations for this. First, the planting site was an 8-10 years old swidden fallow, and the soil fertility might have replenished during these years. Second, we slashed and burnt the existing vegetation (including the bamboos) while preparing the land for planting, thus ash from the burning might have temporarily replenished some soil nutrients (Wan *et al.*, 2001). However, the effect of rice husk biochar addition became more evident in the fourth year in between-treatment differences in the diameter and height of saplings of all planted species, especially the pioneer species (e.g. *P. dasyrachis* grew ca. 2.7 m/yr in height and 2.4 cm/yr in dbh while *D. alatus* grew 0.8 m/yr and 0.7 cm/yr in height and dbh, respectively, following the biochar treatment). This might be attributable to the slow release of nutrients (particularly phosphorus which is one of the growth limiting factors in the acidic soil of the study area) from the biochar over time, as rice husk biochar decompose slowly due to high silica content, which can have effects lasting up to ten maize rotations (Ogawa & Okimori, 2010). The marked increases observed in the diameter and height of the saplings are consistent with the "resource optimization hypothesis" (Agren & Franklin, 2003), which states that plants allocate relatively more to above-ground biomass than to their root systems when the availability of soil nutrients and water increases, and the changes in allocation pattern are relatively strong when the nutrient supply is varied (Poorter & Nagel, 2000).

The success of direct seeding as a restoration method depends on the method of sowing, the nature of the seeds, the seeding rate and the topography. Buried seeds of *Q. serrata* and *K. evelyniana* showed greater establishment success than broadcasted seeds of *P. kesiya*, *S. wallichii* and *K. evelyniana*. This is consistent with findings that exposed seeds are more susceptible to predation by granivores (Holl & Lulow, 1997; Woods & Elliott, 2004; García-Fernández *et al.*, 2008) and rapid seed desiccation during dry spells in the wet season (Vieira & Scariot, 2006; Yang *et al.*, 2006) than buried and excluded seeds. Seeds that are rich in tannins, such as *Q. serrata* acorns, have a better chance of escaping predation from rodents (Xiao *et al.*, 2006), as do seeds that germinate rapidly (Woods & Elliott, 2004). Early successional species are also more susceptible to high levels of seed predation than later-successional species (Garcia-Orth & Martínez-

Ramos, 2008), due to the high costs for predators of handling the large seeds of the latter species (Nepstad *et al.*, 1996). The high seeding rates of early successional species (c.f. 250, 000 *S. wallichii* seeds/kg versus 350 *Q. serrata* seeds/kg) also result in high rates of self-thinning due to high inter- and intra-species competition (Camargo *et al.*, 2002). This is further evidenced by the observed four-fold decline in density of *S. wallichii* seedlings versus a two-fold decline in *Q. serrata* seedlings three years after direct sowing; a difference that may not fully reflect self-thinning effects since some later-successional species are photo-inhibited under high light conditions (Loik & Holl, 2001). A further factor that should be considered is topography, since drainage, moisture, and nutrient levels vary from ridge tops to valley bottoms (Enoki & Abe, 2004), thus influencing seedling establishment. At higher positions on slopes, the ground water level is relatively low, hence the soil water content is often insufficiently high for successful establishment. In contrast, excessively high soil moisture levels at lower slope positions and in flat areas can lead to anoxia and poor seed germination, as can be inferred from the poor establishment of *K. evelyniana* and *S. wallichii* seedlings in areas that are flat and close to a river. Hence, seedlings appear to establish most readily at the foot of mountains and valley bottoms.

Between-site variations in seedling establishment may also be related to proximity to the surrounding natural forest fragments partly because the closer a planting site is to a forest fragment, the higher the probability of seed and seedling predation. Since nearby forest fragments provide shelter and concealment for granivores from their own predators (Nepstad *et al.*, 1996; Guariguata & Ostertag, 2001), they can readily cross or enter nearby open sites. This might explain the low seedling establishment success of *K. evelyniana* and *S. wallichii* we observed at one site (Nakhuan) located close to remaining forest fragments. Overall, the pioneer species showed significantly greater growth in both height and root collar diameter than the later-successional species. Rapid seedling growth is a desirable characteristic of plant species employed in restoring degraded areas.

Commercial tree species that are under-stocked or not present in logged-over forests can be restored by enrichment planting (Adjers *et al.*, 1995; Montagnini *et al.*, 1997; Ashton *et al.*, 2001; Paquette *et al.*, 2006). The establishment success of such species (or indeed any species) may be influenced by the method of planting and the ecology of the species considered. In our case, the survival rates of planted seedlings did not differ between gap and line plantings, and they were lower than rates reported in other studies of enrichment planting in the tropics (e.g. (Peña-Claros *et al.*, 2002; Marod *et al.*, 2004; Romell, 2008). This could be due to reductions in the level of irradiance, in both gaps and planting lines, arising from rapid closure of the canopy by bamboos and the intermediate canopy trees, starting in the second year. It has been well established that the quantity and quality of light are key factors influencing the survival rate and growth of under-planted seedlings (Peña-Claros *et al.*, 2002; Leakey *et al.*, 2003; Romell, 2008), and maintaining even light conditions in enrichment planting of a multistory mixed forest is challenging (Adjers *et al.*, 1995; Abebe, 2003). However, the low levels of irradiance seem to have favored the survival of shade-tolerant species, such as *D. alatus* and *V. cinerea*, while more seriously affecting the survival of light-demanding species such as *P. macrocarpus*. Generally, dipterocarps germinate and establish at sites with low irradiance under a forest canopy, albeit with considerable inter-species variations in growth responses to different light conditions (Tennakoon *et al.*, 2005; Romell, 2008). The dipterocarp species we examined also showed differences in diameter and height growth in relation to the level of light. *V. cinerea* appeared to be less shade-tolerant than *D. alatus*, as the former species performed better in gaps that were relatively open than in the heavily shaded planting lines. Seedling mortality may also occur as a direct result of non-drought stressors, such as herbivores, pathogens and competition (Gerhardt, 1996; Gerhardt, 1998; Engelbrecht *et al.*, 2005; Zida *et al.*, 2008).

Reliable seedling survival is a prerequisite for successful enrichment planting programs, and the development (and implementation) of efficient procedures for gap creation are important steps towards improving post-planting survival rates (Romell, 2008). Although the planting line method adopted in the enrichment planting study presented here had yielded good results elsewhere, the degree of canopy closure was high in our cases, which led to reduced survival and growth rates of light-demanding leguminous species. Similarly, the planting gaps favored shade-tolerant species over light-demanding species due to low levels of irradiance in the sub-canopy layer. Thus, the planting gaps need to be bigger than the 8 × 8 m gaps used in our study. Accordingly, Tuomela *et al.* (1996) considered 500 m^2 to be the optimal gap size for enrichment planting of dipterocarps in logged-over rainforests. In our study area, a gap size that balances the performance of both light-demanding and shade-tolerant species would be ideal, as both kinds of species yield valued products (see Table 1). Alternatively, the planting gaps should be tended not only by cleaning the sub-canopy vegetation but also by maintaining the canopy openness by thinning the canopy and intermediate (such as bamboo) layers periodically, since reducing stand density to an intermediate level has been shown to enhance the growth rate of under-planted seedlings (Paquette *et al.*, 2006). However, the latter option may be costly. With regard to line planting, increasing the planting width is generally believed to enhance seedling growth, but the difficulty in maintaining constant line width and the resulting unevenness of light conditions, coupled with the cost of annual tending operations, reduce its appeal.

6 Conclusions and recommendations

The development of secondary forests on fallows shows a similar pattern to those observed in many other tropical dry forests, i.e., with varying rates of recovery to forest with mature attributes. The distance from forest edges, fallow age, number of crop-fallow rotation cycles and abundance of bamboo are major factors influencing the recovery process. The success of secondary succession also depends on the frequency and intensity of disturbance, thus protecting secondary forests from further disturbances is vital. Given the important role bamboos play in the livelihood of rural people in the study area, one management option would be to transform parts of these secondary forests into bamboo forests. However, if management objectives include both production and biodiversity conservation, the bamboo population should be restricted to a level that is not detrimental to the establishment of other species. To meet short-term needs for wood and wood products as well as diversifying rural livelihood, some of the small-scale swidden fallows could be converted into mixed-species plantations using native species, as shown by the good ecological combining ability of the species tested in the mixed species study presented in this thesis. The subsequent growth of planted seedlings can be further increased by the addition of rice husk biochar, which is a readily available and affordable bioresource in Laos. However, further study is required to optimize the doses for fallows with different ages and rotations, for permutations of other site variables, and for biochar from other sources, in order to maximize its applicability for restoring forests. In addition, time and frequency of application as well as chemical analyses of the biochar itself and the amended soils need to be further studied to standardize the technique.

Direct seeding can be utilized as a cost and time saving method to expedite the recovery process, especially in inaccessible mountainous sites where establishing a nursery to produce and maintain plants can be difficult. It is more likely to be successful if: seeds are buried to avoid risks of seed desiccation and predation; the seeding rate of pioneer species is reduced to avoid high mortality; and species are well matched with sites to minimize the risk of establishment failure due to topography-related variations in microhabitat conditions. Commercial species missing in logged-over forests can be replaced by enrichment planting in gaps, which favors the growth of later successional species more than line planting since canopy closure is particularly rapid in the latter. The shade-tolerant diptrocarps, *D. alatus* and *V. cinerea*, performed best among the tested species in terms of mean survival and growth (diameter and height) rates, and hence are recommended for enrichment planting under mixed deciduous forests. For the legumes (*D. cochichinensis*, *P. macrocarpus* and *A. xylocarpa*) which had relatively low survival rates, increasing the gap size (to 400-500 m^2) or line width (to 4-6 m) may enhance light availability, thereby enhancing their establishment and growth. Alternatively, these leguminous species could be employed in plantations of mixed species in open sites or under canopies of young swidden forests. Overall, the findings provide insights that should be useful guides for future forest restoration programs in Laos and elsewhere in the region.

References

Abebe, T. (2003). The influence of selective logging on residual stand and regeneration in a tropical rain forest in southwestern Ethiopia. PhD dissertation, Swedish University of Agricultural Sciences, Umeå, Sweden, 68 p.

Adjers, G., Hadengganan, S., Kuusipalo, J., Nuryanto, K. & Vesa, L. (1995). Enrichment planting of dipterocarps in logged-over secondary forests - effect of width, direction and maintenance method of planting line on selected *Shorea* species. *Forest Ecology and Management* 73(1-3), 259-270.

Aerts, R., Maes, W., November, E., Negussie, A., Hermy, M. & Muys, B. (2006). Restoring dry Afromontane forest using bird and nurse plant effects: Direct sowing of *Olea europaea* ssp *cuspidata* seeds. *Forest Ecology and Management* 230(1-3), 23-31.

Agren, G.I. & Franklin, O. (2003). Root:shoot ratios, optimization and nitrogen productivity. *Annals of Botany* 92(6), 795-800.

Aide, T.M., Zimmerman, J.K., Pascarella, J.B., Rivera, L. & Marcano-Vega, H. (2000). Forest regeneration in a chronosequence of tropical abandoned pastures: Implications for restoration ecology. *Restoration Ecology* 8(4), 328-338.

Ankarfjard, R. & Kegl, M. (1998). Tapping oleoresin from *Dipterocarpus alatus* (Dipterocarpaceae) in a Lao Village. *Economic Botany* 52(1), 7-14.

Appanah, S., Weinland, G., 1993. Planting quality timber trees in Peninsular Malaysia : a review. Planting quality timber trees in Peninsular Malaysia : a review., xviii + 247 p.

Aronson, J. & Van Andel, J. (2008). Challenges for ecological theory. In: Van Andel J., Aronson J. (Eds.), Restoration ecology: The new frontier. Blackwell publishing, Washington, pp. 223-233.

Arunachalam, A. & Arunachalam, K. (2002). Evaluation of bamboos in eco-restoration of 'jhum' fallows in Arunachal Pradesh: ground vegetation, soil and microbial biomass. *Forest Ecology and Management* 159(3), PII S0378-1127(01)00435-2.

Ashton, M.S., Gunatilleke, C.V.S., Singhakumara, B.M.P. & Gunatilleke, I. (2001). Restoration pathways for rain forest in southwest Sri Lanka: A review of concepts and models. *Forest Ecology and Management* 154(3), 409-430.

Birkedal, M. 2010. Reforestation by direct seeding of Beech and Oak: influence of Granivorous Rodents and site preparation. PhD dissertation, Swedish University of Agricultural Sciences, Alnarp, Sweden, 41p.

Birkedal, M., Fischer, A., Karlsson, M., Lof, M., Madsen, P., 2009. Rodent impact on establishment of direct-seeded *Fagus sylvatica, Quercus robur* and *Quercus petraea* on forest land. *Scandinavian Journal of Forest Research* 24, 298-307.

Bräutigam, D. (1996). Some information and recommendations for the natural forest management and reforestation activities within the TMF of the VFC in Sangthong District. Report of the Lao-German Forestry Team at Ventiane Forestry College, 26 p.

Brown, S. & Lugo, A.E. (1990). Tropical secondary forests. *Journal of Tropical Ecology* 6, 1-32.

Brunner, I., Zimmermann, S., Zingg, A. & Blaser, P. (2004). Wood-ash recycling affects forest soil and tree fine-root chemistry and reverses soil acidification. *Plant and Soil* 267(1-2), 61-71.

Cabin, R.J., Weller, S.G., Lorence, D.H., Cordell, S., Hadway, L.J., Montgomery, R., Goo, D. & Urakami, A. (2002). Effects of light, alien grass, and native species additions on Hawaiian dry forest restoration. *Ecological Applications* 12(6), 1595-1610.

Camargo, J.L.C., Ferraz, I.D.K. & Imakawa, A.M. (2002). Rehabilitation of degraded areas of central Amazonia using direct sowing of forest tree seeds. *Restoration Ecology* 10(4), 636-644.

Carnus, J.M., Parrotta, J., Brockerhoff, E., Arbez, M., Jactel, H., Kremer, A., Lamb, D., O'Hara, K. & Walters, B. (2006). Planted forests and biodiversity. *Journal of Forestry* 104(2), 65-77.

Castro Marin, G., Tigabu, M., Gonzalez Rivas, B. & Oden, P.C. (2009). Natural regeneration dynamics of three dry deciduous forest species in Chacocente Wildlife Reserve, Nicaragua. *Journal of Forestry Research (Harbin)* 20(1), 1-6.

Chazdon, R.L. & Coe, F.G. (1999). Ethnobotany of woody species in second-growth, old-growth, and selectively logged forests of northeastern Costa Rica. *Conservation Biology* 13(6), 1312-1322.

Chazdon, R.L. (2003). Tropical forest recovery: Legacies of human impact and natural disturbances. *Perspectives in Plant Ecology Evolution and Systematics* 6(1-2), 51-71.

Chazdon, R.L. (2008). Beyond deforestation: Restoring forests and ecosystem services on degraded lands. *Science* 320(5882), 1458-1460.

Chinea JD (2002) Tropical forest succession on abandoned farms in the Humacao Municipality of eastern Puerto Rico. *Forest Ecology and Management* 167: 195–207.

Chokkalingam, U. & Jong, W.d. (2001). Secondary forest: A working definition and typology. *International Forestry Review* 3(1), 19-26.

Cubina, A. & Aide, T.M. (2001). The effect of distance from forest edge on seed rain and soil seed bank in a tropical pasture. *Biotropica* 33(2), 260-267.

Dalle, S.P. & de Blois, S. (2006). Shorter fallow cycles affect the availability of noncrop plant resources in a shifting cultivation system. *Ecology and Society* 11(2).

Delang, C.O. (2007). Ecological succession of usable plants in an eleven-year fallow cycle in Northern Lao P.D.R. *Ethnobotany Research and Applications* 5, 331-350.

Dellasala, D.A., Martin, A., Spivak, R., Schulke, T., Bird, B., Criley, M., Daalen, C.v., Kreilick, J., Brown, R. & Aplet, G. (2003). A citizen's call for ecological forest restoration: Forest restoration principles and criteria. *Ecological Restoration* 21(1), 14-23.

Denslow, J.S., 1987. Tropical rainforest gaps and species diversity. *Annual Review of Ecology and Systematics* 18, 431-451.

Derouw, A. (1995). The fallow period as a weed-break in shifting cultivation (tropical wet forests). *Agriculture Ecosystems & Environment* 54(1-2), 31-43.

Doust, S.J., Erskine, P.D., Lamb, D., 2008. Restoring rainforest species by direct seeding: Tree seedling establishment and growth performance on degraded land in the wet tropics of Australia. *Forest Ecology and Management* 256, 1178-1188.

Dugan, P. (2000). Assisted natural regeneration: methods, results and issues relevant to sustained participation by communities. *In: Forest restoration for wildlife conservation. International Tropical Timber Organization and the Forest Restoration Research Unit, Chiang Mai University, Chiang Mai,* 195–200

Elliott, S., Navakitbumrung, P., Kuarak, C., Zangkum, S., Anusarnsunthorn, V. & Blakesley, D. (2003). Selecting framework tree species for restoring seasonally dry tropical forests in northern Thailand based on field performance. *Forest Ecology and Management* 184(1-3), 177-191.

Engel, V.L. & Parrotta, J.A. (2001). An evaluation of direct seeding for reforestation of degraded lands in central Sao Paulo state, Brazil. *Forest Ecology and Management* 152(1-3), 169-181.

Engelbrecht, B.M.J., Kursar, T.A. & Tyree, M.T. (2005). Drought effects on seedling survival in a tropical moist forest. *Trees-Structure and Function* 19(3), 312-321.

Enoki, T. & Abe, A. (2004). Saplings distribution in relation to topography and canopy openness in an evergreen broad-leaved forest. *Plant Ecology* 173(2), 283-291.

Falk, D.A. (2006). Process-centred restoration in a fire-adapted ponderosa pine forest. *Journal for Nature Conservation (Jena)* 14(3-4), 140-151.

FAO, (1995). Forest resources assessment 1990 - Global synthesis. FAO forestry paper 124. FAO, Rome.

FAO (2005). State of the World's Forests. Food and Agriculture Organization of the United Nations, Rome.

FAO (2010). Global Forest Resources Assessment 2010, Country Report, Food and Agriculture Organization of the United Nations. Rome.

Finegan, B. & Delgado, D. (2000). Structural and floristic heterogeneity in a 30-year-old Costa Rican rain forest restored on pasture through natural secondary succession. *Restoration Ecology* 8(4), 380-393.

Finegan, B. (1996). Pattern and process in neotropical secondary rain forests: The first 100 years of succession. *Trends in Ecology & Evolution* 11(3), 119-124.

Fujisaka, S., Escobar, G. & Veneklaas, E.J. (2000). Weedy fields and forests: Interactions between land use and the composition of plant communities in the Peruvian Amazon. *Agriculture Ecosystems & Environment* 78(2), 175-186.

García-Fernández, C., Ruiz-Pérez, M. & Wunder, S. (2008). Is multiple-use forest management widely implementable in the tropics? *Forest Ecology and Management* 256(7), 1468-1476.

Garcia-Orth, X. & Martínez-Ramos, M. (2008). Seed dynamics of early and late successional tree species in tropical abandoned pastures: Seed burial as a way of evading predation. *Restoration Ecology* 16(3), 435-443.

Gerhardt, K. (1996). Effects of root competition and canopy openness on survival and growth of tree seedlings in a tropical seasonal dry forest. *Forest Ecology and Management* 82(1-3), 33-48.

Gerhardt, K. (1998). Leaf defoliation of tropical dry forest tree seedlings - implications for survival and growth. *Trees-Structure and Function* 13(2), 88-95.

Giri, C., Defourny, P. & Shrestha, S. (2003). Land cover characterization and mapping of continental Southeast Asia using multi-resolution satellite sensor data. *International Journal of Remote Sensing* 24(21), 4181-4196.

GOL (1999). The Government's strategic Vision for the Agricultural Sector. Ministry of Agriculture and Forestry, Vientiane.

GOL (2005). Forestry strategy to the year 2020. Government of the Lao PDR, Vientiane, Lao PDR. 89 pp.

Gomes, L.A.A., Rodrigues, A.C., Collier, L.S. & Feitosa, S.d.S. (2008). Production of lettuce seedlings using alternative substrates and fertilizer. *Horticultura Brasileira* 26(3), 359-363.

González-Rivas, B., Tigabu, M., Castro-Marín, G. & Odén, P.C. (2009). Soil seed bank assembly following secondary succession on abandoned agricultural fields in Nicaragua. *Journal of Forestry Research* 20(4), 349-354.

Goosem, S. & Tucker, N.I.J. (1995). Repairing the Rainforest: Theory and practice of rainforest re-establishment. Wet Tropics Management Authority, Cairns.

Grossnickle, S.C. (2005). Importance of root growth in overcoming planting stress. *New Forests* 30(2-3), 273-294.

Guariguata, M.R. & Ostertag, R. (2001). Neotropical secondary forest succession: Changes in structural and functional characteristics. *Forest Ecology and Management* 148(1/3), 185-206.

Guariguata, M.R. (1998). Seed ecology of dominant secondary forest tree species from northeastern Costa Rica. *Biotropica* 30(2 SUPPL.), 1-2.

Guariguata, M.R., Chazdon, R.L., Denslow, J.S., Dupuy, J.M. & Anderson, L. (1997). Structure and floristics of secondary and old-growth forest stands in lowland Costa Rica. *Plant Ecology* 132(1), 107-120.

Gunter, S., Weber, M., Erreis, R. & Aguirre, N. (2007). Influence of distance to forest edges on natural regeneration of abandoned pastures: A case study in the tropical mountain rain forest of Southern Ecuador. *European Journal of Forest Research* 126(1), 67-75.

Hardwick, K., Healey, J., Elliott, S., Garwood, N. & Anusarnsunthorn, V. (1997). Understanding and assisting natural regeneration processes in degraded seasonal evergreen forests in northern Thailand. *Forest Ecology and Management* 99(1-2), 203-214.

Hardwick, K., Healey, J.R., Elliott, S. & Blakesley, D. (2004). Research needs for restoring seasonal tropical forests in Thailand: Accelerated natural regeneration. *New Forests* 27(3), 285-302.

Heinimann, A., Messerli, P., Schmidt-Vogt, D. & Wiesmann, U. (2007). The dynamics of secondary forest landscapes in the lower Mekong basin: A regional-scale analysis. *Mountain Research and Development* 27(3), 232-241.

Holl, K.D. & Aide, T.M. (2010). When and where to actively restore ecosystems? *Forest Ecology and Management* (in press) doi:10.1016/j.foreco.2010.07.004.

Holl, K.D. & Lulow, M.E. (1997). Effects of species, habitat, and distance from edge on post-dispersal seed predation in a tropical rainforest. *Biotropica* 29(4), 459-468.

Holl, K.D. (1998). Do bird perching structures elevate seed rain and seedling establishment in abandoned tropical pasture? *Restoration Ecology* 6(3), 253-261.

Holl, K.D. (1999). Factors limiting tropical rain forest regeneration in abandoned pasture: Seed rain, seed germination, microclimate, and soil. *Biotropica* 31(2), 229-242.

Holl, K.D., Loik, M.E., Lin, E.H.V. & Samuels, I.A. (2000). Tropical montane forest restoration in Costa Rica: Overcoming barriers to dispersal and establishment. *Restoration Ecology* 8(4), 339-349.

Hooper, D.U., Chapin, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S., Schmid, B., Setala, H., Symstad, A.J., Vandermeer, J. & Wardle, D.A. (2005). Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. *Ecological Monographs* 75(1), 3-35.

Ilstedt, U., Malmer, A., Nordgren, A. & Liau, P. (2004). Soil rehabilitation following tractor logging: Early results on amendments and tilling in a second rotation *Acacia mangium* plantation in Sabah, Malaysia. *Forest Ecology and Management* 194(1-3), 215-222.

Ito, H., Hino, T., 2007. Dwarf bamboo as an ecological filter for forest regeneration. *Ecological Research* 22, 706-711.

ITTO (2002). ITTO guidelines for the restoration, management and rehabilitation of degraded and secondary tropical forests. International Tropical Timber Organization (ITTO) in collaboration with the Center for International Forestry Research. Policy Development Series No 13, 82 p.

Janzen, D.H. (1988a). Management of habitat fragments in a tropical dry forest - growth. *Annals of the Missouri Botanical Garden* 75(1), 105-116.

Janzen, D.H. (1988b). Tropical ecological and biocultural restoration. *Science* 239(4837), 243-244.

Johnson, C.M., Zarin, D.J. & Johnson, A.H. 2000. Post-disturbance aboveground biomass accumulation in global secondary forests. *Ecology* 81(5): 1395-1401.

Kalacska, M., Sanchez-Azofeifa, G.A., Calvo-Alvarado, J.C., Quesada, M., Rivard, B. & Janzen, D.H. (2004). Species composition, similarity and diversity in three successional stages of a seasonally dry tropical forest. *Forest Ecology and Management* 200(1-3), 227-247.

Kennard, D.K., Gould, K., Putz, F.E., Fredericksen, T.S. & Morales, F. (2002). Effect of disturbance intensity on regeneration mechanisms in a tropical dry forest. *Forest Ecology and Management* 162(2-3), 197-208.

Kingsada, K., 1998. Sustainable Forest Management and Conservation in Lao, Vision 2020. Paper presented in the 4th Donor Meeting 6-8 April 1998, Savannakhet, Lao PDR. Vientiane.

Knowles, O.H. & Parrotta, J.A. (1995). Amazonian forest restoration: An innovative system for native species selection based on phonological data and field performance indices. *Commonwealth Forestry Review* 74, 230-243.

Kruger, L.M. & Midgley, J.J. (2001). The influence of resprouting forest canopy species on richness in Southern Cape forests, South Africa. *Global Ecology and Biogeography* 10(5), 567-572.

Lamb, D. (1998). Large-scale ecological restoration of degraded tropical forest lands: The potential role of timber plantations. *Restoration Ecology* 6(3), 271-279.

Lamb, D., 2011. Regreening the Bare Hills-Tropical Forest Restoration in the Asia Pacific Region. Springer's World Forests VIII, Pages 547.

Lamb, D., Erskine, P.D. & Parrotta, J.A. (2005). Restoration of degraded tropical forest landscapes. *Science* 310(5754), 1628-1632.

Lamb, D., Parrotta, J., Keenan, R. & Tucker, N. (1997). Rejoining habitant remnants: Restoring degraded rainforest lands. *Tropical forest remnants: Ecology, management, and conservation of fragmented communities*, pp. 366-385.

Lawrence, D. (2004). Erosion of tree diversity during 200 years of shifting cultivation in Bornean Rain Forest. *Ecological Applications* 14(6), 1855-1869.

Lawrence, D. (2005). Biomass accumulation after 10-200 years of shifting cultivation in Bornean rain forest. *Ecology* 86(1), 26-33.

Leakey, A.D.B., Press, M.C. & Scholes, J.D. (2003). Patterns of dynamic irradiance affect the photosynthetic capacity and growth of dipterocarp tree seedlings. *Oecologia* 135(2), 184-193.

Lebrija-Trejos, E., Bongers, F., Garcia, E.A.P. & Meave, J.A. (2008). Successional change and resilience of a very dry tropical deciduous forest following shifting agriculture. *Biotropica* 40(4), 422-431.

Lehmann, L., Greijmans, M., Shenmen, D. (2003). Forests and Trees of the Central Highlands of Xieng Khouang, Lao P.D.R. A field guide, p 246.

Lemenih, M. & Teketay, D. (2005). Effect of prior land use on the recolonization of native woody species under plantation forests in the highlands of Ethiopia. *Forest Ecology and Management* 218(1-3), 60-73.

Lestrelin, G. (2010). Land degradation in the Lao PDR: Discourses and policy. *Land Use Policy* 27(2), 424-439.

Löf, M., Birkedal, M., 2009. Direct seeding of Quercus robur L. for reforestation: The influence of mechanical site preparation and sowing date on early growth of seedlings. *Forest Ecology and Management* 258, 704-711.

Loik, M.E. & Holl, K.D. (2001). Photosynthetic responses of tree seedlings in grass and under shrubs in early-successional tropical old fields, Costa Rica. *Oecologia* 127(1), 40-50.

Lugo, A.E. (1997). The apparent paradox of reestablishing species richness on degraded lands with tree monocultures. *Forest Ecology and Management* 99(1-2), 9-19.

MAF (1996). Report on soil of training & model forest of the Faculty of Forestry, National University of Laos (NUoL), Sangthong District (1520 ha), Vientiane Capital.

Mansourian, S., Lamb, D. & Gilmour, D. (2005). Overview of technical approaches to restoring tree cover at the site level. *Forest Restoration in Landscapes: Beyond Planting Trees*, 241-249.

Marod, D., Kutintara, U., Tanaka, H. & Nakashizuka, T. (2004). Effects of drought and fire on seedling survival and growth under contrasting light conditions in a seasonal tropical forest. *Journal of Vegetation Science* 15(5), 691-700.

McCombe, B.K.W. (1977). Properties of Laotian Timber Species. Manual for the woodworking industry development. UNDP/UNIDO. Bangkok, 72p.

Messerli, P., Heinimann, A., Epprecht, M., 2009. Finding Homogeneity in Heterogeneity-A New Approach to Quantifying Landscape Mosaics Developed for the Lao PDR. *Human Ecology* 37, 291-304.

Miyawaki, A. (1999). Creative ecology: Restoration of native forests by native trees. *Plant Biotechnology* 16, 15–25.

Molina Colon, S. & Lugo, A.E. (2006). Recovery of a subtropical dry forest after abandonment of different land uses. *Biotropica* 38(3), doi:10.1111/j.1744-7429.2006.00159.x.

Montagnini, F., Eibl, B., Grance, L., Maiocco, D. & Nozzi, D. (1997). Enrichment planting in overexploited subtropical forests of the Paranaense region of Misiones, Argentina. *Forest Ecology and Management* 99(1-2), 237-246.

Moran, E.F., Brondizio, E., Tucker, J.M., da Silva-Fosberg, M.C., McCracken, S. & Falesi, I. 2000. Effects of soil fertility and land-use on forest succession in Amazonia. *Forest Ecology and Management* 139: 93-108.

Moyin-Jesu, E.I. (2007). Use of plant residues for improving soil fertility, pod nutrients, root growth and pod weight of okra (*Abelmoschus esculentum* L). *Bioresource Technology* 98(11), 2057-2064.

Myster, R.W. & Pickett, S.T.A. (1993). Effects of litter, distance, density and vegetation patch type on postdispersal tree seed predation in old fields. *Oikos* 66(3), 381-388.

Nepstad, D.C., Uhl, C. & Serrao, E.A.S. (1991). Recuperation of a degraded Amazonian landscape - forest recovery and agricultural restoration. *Ambio* 20(6), 248-255.

Nepstad, D.C., Uhl, C., Pereira, C.A. & daSilva, J.M.C. (1996). A comparative study of tree establishment in abandoned pasture and mature forest of eastern Amazonia. *Oikos* 76(1), 25-39.

Ogawa, M. & Okimori, Y. (2010). Pioneering works in biochar research, Japan. *Australian Journal of Soil Research* 48(6-7), 489-500.

Ohlsson, B. (2009). Farmers and Forest Land Use in Lao PDR and Vietnam. PhD dissertation, Swedish University of Agricultural Sciences, Umeå, Sweden, 68 p.

Paquette, A., Bouchard, A. & Cogliastro, A. (2006). Survival and growth of under-planted trees: A meta-analysis across four biomes. *Ecological Applications* 16(4), 1575-1589.

Parrotta, J.A., Turnbull, J.W. & Jones, N. (1997a). Catalyzing native forest regeneration on degraded tropical lands. *Forest Ecology and Management* 99(1-2), 1-7.

Parrotta, J.A., Turnbull, J.W. & Jones, N. (1997b). Introduction - Catalyzing native forest regeneration on degraded tropical lands. *Forest Ecology and Management* 99(1-2), 1-7.

Peña-Claros, M. (2003). Changes in forest structure and species composition during secondary forest succession in the Bolivian Amazon. *Biotropica* 35(4), 450-461.

Peña-Claros, M., Boot, R.G.A., Dorado-Lora, J. & Zonta, A. (2002). Enrichment planting of *Bertholletia excelsa* in secondary forest in the Bolivian Amazon: effect of cutting line width on survival, growth and crown traits. *Forest Ecology and Management* 161(1-3), PII S0378-1127(01)00491-1.

Perkulis, A.M., Prado, J.M.R. & JimenezOsornio, J.J. (1997). Composition, structure and management potential of secondary dry tropical vegetation in two abandoned henequen plantations of Yucatan, Mexico. *Forest Ecology and Management* 94(1-3), 79-88.

Phimmavong, S., Ozarska, B., Midgley, S. & Keenan, R. (2009). Forest and plantation development in Laos: history, development and impact for rural communities. *International Forestry Review* 11(4), 501-513.

Poorter, H. & Nagel, O. (2000). The role of biomass allocation in the growth response of plants to different levels of light, CO_2, nutrients and water: A quantitative review (vol. 27, p. 595, 2000). *Australian Journal of Plant Physiology* 27(12), 1191-1191.

Rey-Benayas, J.M., Galvan, I. & Carrascal, L.M. (2010). Differential effects of vegetation restoration in Mediterranean abandoned cropland by secondary succession and pine plantations on bird assemblages. *Forest Ecology and Management* 260(1), 87-95.

Riswan, S., Kentworthy, J.B. & Kartawinata, K. (1985). The estimation of temporal processes in tropical rain forest: A study of primary mixed dipterocarp forest in Indonesia. *Journal of Tropical Ecology* 1, 171-182.

Robichaud, W.G., Sinclair, A.R.E., Odarkor-Lanquaye, N. & Klinkenberg, B. (2009). Stable forest cover under increasing populations of swidden cultivators in central Laos: the Roles of intrinsic culture and extrinsic wildlife trade. *Ecology and Society* 14(1). 33-59.

Rodrigues, R.R., Lima, R.A.F., Gandolfi, S. & Nave, A.G. (2009). On the restoration of high diversity forests: 30 years of experience in the Brazilian Atlantic Forest. *Biological Conservation* 142(6), 1242-1251.

Romell, E.H., G; Karlsson, A; Garcia Ch (2008). Artificial canopy gaps in a *Macaranga* spp. dominated secondary tropical rain forest-Effects on survival and above ground increment of four under-planted dipterocaps species. *Forest Ecology and Management* 255, 1452-1460.

Saldarriaga, J.C., West, D.C., Tharp, M.L. & Uhl, C. (1988). Long-term chronosequence of forest succession in the upper Rio Negro of Colombia and Venezuela. *Journal of Ecology* 76, 939-958.

Sawathvong, S. (2003). Participatory land management planning in biodiversity conservation areas of Lao PDR. PhD dissertation, Swedish University of Agricultural Sciences, Umeå, Sweden. 44 pp. +Papers I-V.

Schmid-Haas, P. 1969. Stichproben am waldrand. *Mitt. Schweiz Anst. Forstl. Versuchswes.* 45:234–303.

Schmidt-Vogt, D. (1998). Defining degradation: The impacts of swidden on forests in Northern Thailand. *Mountain Research and Development* 18(2), 135-149.

Schmidt-Vogt, D. (2001). Secondary forests in swidden agriculture in the highlands of Thailand. *Journal of Tropical Forest Science* 13(4), 748-767.

Scott, D., Welch, D., Thurlow, M. & Elston, D.A. (2000). Regeneration of *Pinus sylvestris* in a natural pinewood in NE Scotland following reduction in grazing by *Cervus elaphus*. *Forest Ecology and Management* 130(1-3), 199-211.

Senbeta, F., Teketay, D. & Näslund, B.Å. (2002). Native woody species regeneration in exotic tree plantations at Munessa-Shashemene Forest, southern Ethiopia. *New Forests* 24, 131-145.

SER (2004). Society for Ecological Restoration International Science & Policy Working Group. 2004. The International Primer on Ecological Restoration. www.ser.org & Tucson: Society for Ecological Restoration International.

Shiels, A.B. & Walker, L.R. (2003). Bird perches increase forest seeds on Puerto Rican landslides. *Restoration Ecology* 11(4), 457-465.

Shono, K., Cadaweng, E.A. & Durst, P.B. (2007). Application of assisted natural regeneration to restore degraded tropical forestlands. *Restoration Ecology* 15(4), 620-626.

Silver, W.L., Ostertag, R. & Lugo, A.E. (2000). The potential for carbon sequestration through reforestation of abandoned tropical agricultural and pasture lands. *Restoration Ecology* 8(4), 394-407.

Solla-Gullon, F., Santalla, M., Perez-Cruzado, C., Merino, A. & Rodriguez-Soalleiro, R. (2008). Response of *Pinus radiata* seedlings to application of mixed wood-bark ash at planting in a temperate region: Nutrition and growth. *Forest Ecology and Management* 255(11), 3873-3884.

Sovu, Tigabu, M., Savadogo, P., Oden, P.C. & Xayvongsa, L. (2009). Recovery of secondary forests on swidden cultivation fallows in Laos. *Forest Ecology and Management* 258(12), 2666-2675.

Staver, C. (1991). The role of weeds in the productivity of Amazonian bush fallow agriculture. *Experimental Agriculture* 27(3), 287-304.

Teketay, D. & Granstrom, A. (1997). Seed viability of Afromontane tree species in forest soils. *Journal of Tropical Ecology* 13, 81-95.

Tennakoon, M.M.D., Gunatilleke, I., Hafeel, K.M., Seneviratne, G., Gunatilleke, C.S. & Ashton, P.M.S. (2005). Ectomycorrhizal colonization and seedling growth of *Shorea* (Dipterocarpaceae) species in simulated shade environments of a Sri Lankan rain forest. *Forest Ecology and Management* 208(1-3), 399-405.

Thapa, G.B., 1998. Issues in the Conservation and Management of Forests in Laos: The Case of Sangthong District. *Singapore Journal of Tropical Geography* 19, 71-91.

Tuomela, K., Kuusipalo, J., Vesa, L., Nuryanto, K., Sagala, A.P.S. & Adjers, G. (1996). Growth of dipterocarp seedlings in artificial gaps: An experiment in a logged-over rainforest in South Kalimantan, Indonesia. *Forest Ecology and Management* 81(1-3), 95-100.

Vieira DLM, Lima VV, Sevilha AC, Scariot A. 2008. Consequences of dryseason seed dispersal on seedling establishment of dry forest trees: Should we store seeds until the rains? *Forest Ecology and Management* 256(3):471–481.

Vieira, D.L.M. & Scariot, A. (2006). Principles of natural regeneration of tropical dry forests for restoration. *Restoration Ecology* 14(1), 11-20.

Wan, S.Q., Hui, D.F. & Luo, Y.Q. (2001). Fire effects on nitrogen pools and dynamics in terrestrial ecosystems: A meta-analysis. *Ecological Applications* 11(5), 1349-1365.

Warnock, D.D., Lehmann, J., Kuyper, T.W., Rillig, M.C., 2007. Mycorrhizal responses to biochar in soil - concepts and mechanisms. *Plant and Soil* 300, 9-20.

White, P.S. & Walker, J.L. (1997). Approximating nature's variation: Selecting and using reference information in restoration ecology. *Restoration Ecology* 5(4), 338-349.

Whitmore, T.C. (1990). Tropical rain forest dynamics and its implications for management. *Gomez-Pompa, a., T. C. Whitmore and M. Hadley (Eds.). Man and the Biosphere Series, Vol. 6. Rain Forest Regeneration and Management. Xxiii+457p. Parthenon Publishing Group: Park Ridge, New Jersey, USA; Carnforth, England, UK; United Nations Educational, Scientific and Cultural Organization (UNESCO), Paris, France. Illus. Maps*, 67-90.

Willoughby, I., Jinks, R.L., Kerr, G., Gosling, P.G., 2004. Factors affecting the success of direct seeding for lowland afforestation in the UK. *Forestry* 77, 467-482.

Woods, K. & Elliott, S. (2004). Direct seeding for forest restoration on abandoned agricultural land in northern Thailand. *Journal of Tropical Forest Science* 16(2), 248-259.

WorldBank, Sida & Finland, G. (2001). Lao PDR Production Forestry Policy: Status and Issues for Dialogue (vol. 1). World Bank, Washington, DC. 65 p.

Xiao, Z.S., Wang, Y.S., Harris, M. & Zhang, Z.B. (2006). Spatial and temporal variation of seed predation and removal of sympatric large-seeded species in relation to innate seed traits in a subtropical forest, Southwest China. *Forest Ecology and Management* 222(1-3), 46-54.

Yamato, M., Okimori, Y., Wibowo, I.F., Anshori, S. & Ogawa, M. (2006). Effects of the application of charred bark of *Acacia mangium* on the yield of maize, cowpea and peanut, and soil chemical properties in South Sumatra, Indonesia. *Soil Science and Plant Nutrition* 52(4), 489-495.

Yang, J.-C., Lin, T.-P. & Kuo, S.-R. (2006). Seed storage behavior of Taiwan cow-tail fir (*Keteleeria davidiana* (Franchet) Beissner var. *formosana* Hayata). *Taiwan Journal of Forest Science* 21(2), 179-189.

Yirdaw, E. & Luukkanen, O. (2003). Indigenous woody species diversity in *Eucalyptus globulus* Labill. ssp *globulus* plantations in the Ethiopian highlands. *Biodiversity and Conservation* 12(3), 567-582.

Yirdaw, E. (2001). Diversity of naturally-regenerated native woody species in forest plantations in the Ethiopian highlands. *New Forests* 22(3), 159-177.

Yokoyama, S., Tanaka, K. & Phalakhone, K. (2006). Forest policy and swidden agriculture in Laos. *Paper presented at SEAGA Conference 28-30 November, 2006*, p 14.

Zar, J. (1996). Biostatistical Analysis. Prentice-Hall Inc., New Jersey.

Zarin DJ, Ducey MJ, Tucker JM & Salas WA (2001) Potential biomass accumulation in Amazonian regrowth forests. *Ecosystems* 4: 658–668.

Zida, D., Tigabu, M., Sawadogo, L. & Oden, P.C. (2008). Initial seedling morphological characteristics and field performance of two Sudanian savanna species in relation to nursery production period and watering regimes. *Forest Ecology and Management* 255(7), 2151-2162.

Zimmerman, J.K., Pascarella, J.B. & Aide, T.M. (2000). Barriers to forest regeneration in an abandoned pasture in Puerto Rico. *Restoration Ecology* 8(4), 350-360.

Acknowledgements

First of all, my gratitude is extended to his Excellency Phimasone Luangkhama, former Minister of Education, who authorized me to take this special opportunity. Special thanks are also due to Professor Dr. Soukkongseng Xaignaleuth, President of the National University of Laos, who facilitated and smoothed the internal process.

I am very grateful to my main supervisor, Professor Per Christer Odén, for assisting me to overcome the challenges encountered and for being very supportive throughout my PhD research studies. I highly appreciate his positive attitude, kindness and continuous support and encouragement in all matters. I could not have written this thesis without your patient assistance. Associate Prof. Dr Mulualem Tigabu and Dr Patrice Savadogo, my assistant supervisors, have aroused my interest in forest restoration since 2005 when I was starting my research project at SLU. Their assistance in planning experiments, reviewing several drafts of manuscripts, practical advice and crucial help during scientific writing, as well as openly sharing their wealth of knowledge regarding forest restoration, have all been instrumental for the successful completion of this thesis. My special thanks also go to Associate Prof. Dr Lamphoune Sayvongsa, my local assistant supervisor, for his assistance during the field work in Laos.

I would like to acknowledge Prof. Olavi Luukkkanen, Viikki Tropical Resources Institute, University of Helsinki, Finland, for giving me the opportunity to attend the course on Tropical Forest Landscape Restoration in Southeast Asia in Thailand. I am also very grateful to Mr. Khamsy, Head of Xieng Khouang Provincial Forestry Division, for providing secondary data and allowing me to do research at the project site (NAWACOP). I would like to take this opportunity to thank Dr. Mats Sandewall and Kajsa Sandewall not only for the nice trip to Skuleskogen National Park but also for supporting me in both professional and private matters during my stay in Umeå. Special thanks are due to Kajsa Sandewall for her delicious dishes. My thanks go also to Issa Ouedraogo for his kind support in mapping and GIS referencing of study site for Paper III. I am especially indebted to Dr. Matts Linbladh, head of our Department, Professor Magnus Löf, Margareta Kelemen, Magnus Mossberg for administrative and technical support.

Similarly, I am grateful to Professor Somsy Ngophanxay and Associate Prof. Houngphet Chanthavong for administrative and technical support during my field study in Laos as well as staff members of the Forestry Faculty, NAFRI, and National Tree Seed Centre. My thanks also go to Associate Prof. Latsamy Chanthalansy, Mr Bounthanome, Mr Phouthasone, and Mr Bounsong, staff members of the Training Model Forest (TMF) for their assistance during data collection and for providing accommodation during the fieldwork in Sangthong District. I express my gratitude to members of the Sida/NUoL coordination unit at the Faculty of Forestry, in particular, Associate Prof. Bounthene Phasiboliboune, Assoc. Professor Souksompong Prixar and Saykham Bouthavong for their strong support. I must also take this opportunity to thank staff members of Medical College 105 for providing my family with continuous hospital support. Thanks to the stakeholder villagers of Ban Napo and Ban Nongboua for valuable land use information during interviews. Especially thanks to Thongchan, Thitlo, and Thonglo for identifying forest species in local names during fieldwork and Thongmy for maintenance of experiment plots during the study period. Thanks to Dr Bounheuang Ninchaleun for sharing his experience, fruitful discussions about professional life and having fun together.

I am also very grateful to all colleagues in our research group: Indra Prasad Sapkota, Djibril Dayamba, Catherine Ky-Dembele, Issa Ouedraogo, Pascaline Lingani, Hari Priyadi and Abolfazel Daneshvar for their friendship and discussions (scientific and non-scientific). During my stay in Sweden, all my Lao friends in Umeå: Bounphan Tonpheng, Lemthong Lathdavong, Phimpho Luangrath, Thongloon Vilaythong, Chounlamany Kongsi, Keophouthong Bounyasone, Ngouay Keosada, Damdouane Khouangvichit, Phouxay Kapmanivanh, Phoutsavath Sounthala, Saythong Phommavong, and many others made my stay hospitable for which I convey my heartfelt gratitude. Ngouay Keosada and Vichith Lamxay are due special thanks for reviewing the Lao summary of the thesis.

I thank my parents, brothers, sisters, relatives, neighbors and friends who have been supported me in various ways and contributed to the completion of this work. Special thanks to family members Paagnia Vue, Tshou Vue, and Nengyong Vue for their support during my study. My thanks also go to my brother-in-law and sister, Yangthao Yangluexay, Maichee Vue for their moral support and encouragement. The National University of Laos (NUoL) and the Swedish International Development Cooperation Agency (Sida) are gratefully acknowledged for financial support, and particularly Dr Gity Behravan, Dr Aina Tollefsen and Associate Prof. Dr Khamphong Phomavongmixay for their great efforts to make the research cooperation fruitful. My special thanks go to Dr. John Blackwell, Sees-editing Ltd, for extensive and meticulous language-editing of the thesis summary and the individual papers.

Last, but far from least, my beloved wife, Silavanh, deserves my heartfelt gratitude for standing by my side all the time and taking care of all family matters during my absence. My beloved daughter, Naly Vue was a friend and has my thanks for visiting and comforting me while I was alone in Sweden. My lovely boy, Anouvy Vue, spices up our family life and I also thank him for understanding while I was away from home for long times.

Summary in LAO

ການຕັດໄມ້ທຳລາຍປ່າ ແລະ ການປ່ຽນແປງການນຳໃຊ້ທີ່ດິນຍັງຄົງເປັນໄພຮຸກຮານທີ່ສຳຄັນຕໍ່ກັບການຄູ້ມຄອງປ້ອງກັນປ່າ ແລະ ການພັດທະນາຄວາມລາກຫລາຍທາງຊີວະນາງພັນ ແລະ ລະບົບນິເວດເຂດຮ້ອນຢູ່ຫລາຍປະເທດໃນອາຊີໃຕ້ ແລະ ອາຊີຕາເວັນອອກສຽງໃຕ້ ລວມທັງ ສປປ ລາວ ທີ່ກຳລັງປະສົບກັບບັນຫາການສູນເສຍພື້ນທີ່ປ່າໄມ້ຢູ່ໃນລະດັບສູງ. ຄວາມກ້າວໜ້າໃນການດຳເນີນນະໂຍບາຍ ການພັດທະນາປ່າໄມ້ດ້ວຍກົດໝາຍ ແລະ ການນຳໃຊ້ປ່າຢ່າງມີແຜນການ ໄດ້ປະກອບສ່ວນເຮັດໃຫ້ການຖາງປ່າເຮັດໄຮ່ຫລຸດລົງ ທັງເປັນການສ້າງໂອກາດ ແລະ ປັດໃຈໃຫ້ແກ່ການຟື້ນຟູທີ່ດິນປ່າເຊື່ອມໂຊມໃຫ້ດີຂື້ນເທື່ອລະກ້າວ. ໃນປະເທດ ລາວ ວິທະຍາສາດກ່ຽວກັບການຟື້ນຟູປ່າຍັງບໍ່ທັນມີການພັດທະນາຢ່າງກວ້າງຂວາງເທື່ອ ດັ່ງນັ້ນ ການສຶກສາທີ່ນຳມາສະເໜີໃນບົດຄົ້ນຄວ້າສະບັບນີ້ ມີຈຸດປະສົງ ເພື່ອກວດສອບປັດໃຈຕ່າງໆ ທີ່ມີອິດທິພົນຕໍ່ກັບການຟື້ນຕົວຄືນຂອງໄມ້ໃນພື້ນທີ່ປ່າເລົ່າ ແລະ ທົດສອບວິທີທາງການຟື້ນຟູປ່າ ໂດຍສະເພາະໃນປ່າເລົ່າ, ທົ່ງລ້ຽງສັດ ແລະ ປ່າປະສົມທີ່ມີການຂຸດຄົ້ນຫລາຍເກີນໄປ ເພື່ອໃຫ້ປ່າດັ່ງກ່າວ ສາມາດສະໜອງຄວາມລາກຫລາຍທາງດ້ານເສດຖະກິດ ແລະ ສິ່ງແວດລ້ອມຄືນສູ່ສະພາບດີຕາມຈຸດປະສົງ. ເພື່ອສົມທຽບຊະນິດພັນໄມ້ຢືນຕົ້ນລະຫວ່າງປ່າເລົ່າກັບປ່າທຳມະຊາດຢູ່ໃນເຂດໃກ້ຄຽງ, ປ່າເລົ່າທີ່ມີອາຍຸຕ່າງກັນໃນລະຫວ່າງ 20 ປີລົງມາ ພົບວ່າ ການຟື້ນຕົວຄືນຂອງປ່າເລົ່າດ້ານຄວາມລາກຫລາຍຂອງຊະນິດພັນມີ 29ເປີເຊັນ, ຄວາມໜາແໜ້ນມີ 39 ເປີເຊັນ, ເນື້ອທີ່ໜ້າຕັດໄມ້ມີ 18 ເປີເຊັນ ແຕ່ ເມື່ອສົມທຽບສະເພາະສ່ວນຂອງຊະນິດພັນໄມ້ການຄ້າແລ້ວ ຄວາມໜາແໜ້ນ ແລະ ເນື້ອທີ່ໜ້າຕັດໄມ້ມີ 41 ແລະ 7 ເປີເຊັນຕາມລຳດັບ. ປັດໃຈຕົ້ນຕໍທີ່ມີອິດທິພົນຕໍ່ກັບ ຂະບວນການຟື້ນຕົວຄືນຂອງໄມ້ໃນປ່າເລົ່າມີຄື: ໄລຍະຫ່າງຂອງປ່າເລົ່າຈາກປ່າທຳມະຊາດ, ອາຍຸຂອງປ່າເລົ່າ, ຮອບວຽນຄວາມຖີ່ໃນການຖາງເພື່ອປູກພືດໝູນວຽນ ແລະ ການຊ່ວງຊີງຂອງໄມ້ຈິງກັບການເກີດຂື້ນໃໝ່ຂອງໄມ້ປ່ອງ. ການສູນເສຍຊະນິດພັນໄມ້ທີ່ມີຄວາມສຳຄັນທາງດ້ານການຄ້ານັ້ນສາມາດປູກເສີມຂື້ນໄດ້ ແຕ່ ບັນຫາຄວາມຈຳເປັນສຳລັບໄມ້ໃຊ້ ແລະ ຄວາມລາກຫລາຍທາງວິຖີຊີວິດ ກ່ຽວກັບຊີວິດການເປັນຢູ່ຂອງປະຊາຊົນໃນຊົນນະບົດແມ່ນຕ້ອງການວິທີທາງເລືອກທີ່ເໝາະສົມ ດັ່ງນັ້ນ ຈຶ່ງທົດລອງປູກໄມ້ປະສົມຫລາຍຊະນິດ ມີທັງໄມ້ທີ່ຕ້ອງການແສງນ້ອຍ ແລະ ໄມ້ຕ້ອງການແສງຫລາຍ, ພົບວ່າໄມ້ທຸກຊະນິດສາມາດເຕີບໂຕໄດ້ດີດ້ວຍກັນທັງໝົດ ເຊິ່ງສະແດງໃຫ້ເຫັນໃນລະດັບ

ອັດຕາການຫລອດຕາຍສະເລ່ຍຂອງເບັ້ຍໄມ້ມີສູງກ່ວາ 70 ເປີເຊັນຂຶ້ນໄປ, ການຈະເລີນເຕີບໂຕຂອງເບັ້ຍໄມ້ສ່ວນຫລາຍກໍ່ດີ ເຊິ່ງອາດເປັນຜົນດີໃຫ້ກັບລະບົບນິເວດຂອງປ່າໃນອານາຄົດ. ນອກຈາກນີ້ ການນຳໃຊ້ຝຸ່ນແກບເຜົາ ເປັນວິທີການນຶ່ງ ທີ່ສາມາດແກ້ໄຂປັບປຸງຄວາມອຸດົມສົມບູນຂອງດິນໃຫ້ດີຂຶ້ນ. ໜ້າຕ້າງຄໍຮາກຂອງເບັ້ຍໄມ້ເພີ້ມຂຶ້ນ 1.2 ເທົ່າ ຖ້າທຽບກັບດອນກວດກາ, ສ່ວນດອນທີ່ໃຊ້ຝຸ່ນເຄມີ ຜົນຂອງການເຕີບໂຕມີລັກສະນະຄ້າຍກັນ. ການທົດສອບທີ່ກ່ຽວຂ້ອງກັບການຫວ່ານແກ່ນໂດຍກົງໃສ່ພື້ນທີ່ທົ່ງຫຍ້າໂດຍນຳໃຊ້ໄມ້ສາຍພັນພື້ນເມືອງ 4 ສາຍພັນ ພົບວ່າ ເບັ້ຍໄມ້ສາຍພັນທີ່ຕ້ອງການແສງນ້ອຍມີສະຖານນະພາບການແຕກງອກ (59-65%) ສູງກວ່າສາຍພັນໄມ້ທີ່ຕ້ອງການແສງຫລາຍ (3-13%). ປັດໃຈເຮັດໃຫ້ອັດຕາການແຕກງອກສຳເລັດ ແມ່ນຂຶ້ນກັບວິທີການຫວ່ານແກ່ນ, ລັກສະນະທາງທຳມະຊາດຂອງແກ່ນ, ຈຳນວນແກ່ນທີ່ຫວ່ານ ແລະ ສະຖານະພາບທາງໂຄງສ້າງສະເພາະຂອງພື້ນທີ່. ການທົດລອງການປູກເສີມໂດຍໃຊ້ໄມ້ຫ້າຊະນິດ ແລະ ສອງວິທີທີ່ແຕກຕ່າງກັນ ພົບວ່າ ໄມ້ຕະກຸນຍາງ (dipterocarps) ໄດ້ຮັບຜົນດີກວ່າໄມ້ຕະກຸນຖົ່ວ (legumes) ໃນທາງດ້ານຂອງການຫລອດຕາຍ ແລະ ການເຕີບໂຕ. ພ້ອມດຽວກັນນັ້ນ ການປູກໄມ້ແບບເປັນດອນສຳລັບໄມ້ຕະກຸນຍາງກໍ່ໄດ້ຮັບຜົນດີກວ່າວິທີການປູກແບບເປັນແລວ ບໍ່ວ່າດ້ານຂອງການຫລອດຕາຍກໍ່ຄືການເຕີບໃຫຍ່. ໂດຍລວມແລ້ວ ການສຶກສາໄດ້ໃຫ້ການສະນັບສະໜູນທີ່ສຳຄັນ, ທັງເປັນການສະໜອງຂໍ້ມູນດ້ານຄວາມເຂົ້າໃຈທີ່ເປັນປະໂຫຍດ ແລະ ຫລັກຖານສຳຄັນ ສຳລັບການຄັດເລືອກ ວິທີການທີ່ເໝາະສົມ ເພື່ອເປັນທິດທາງໃນການສ້າງແຜນການຟື້ນຟູປ່າໄມ້ໃນອານາຄົດຂອງ ລາວ ແລະ ປະເທດອື່ນໆ ໃນເຂດຮ້ອນທີ່ມີສະພາບແວດລ້ອມຄ້າຍກັນ ເພື່ອໃຫ້ບັນລຸວັດຖຸປະສົງສຳຄັນທາງດ້ານເສດຖະກິດ ແລະ ສິ່ງແວດລ້ອມ.

Acta Universitatis Agriculturae Sueciae
Doctoral Thesis No. 2011:20

Deforestation and land-use change are still major threats to biodiversity and ecosystem services in Southeast Asia, including Laos. Progresses in developing forest policies and programs, coupled with increasing trend of stabilization of swidden farming, have created an opportunity to forest restoration on degraded lands. Thus, factors influencing the recovery of secondary forests on abandoned swidden fallows and a variety of site-specific restoration approaches were examined and documented.

Sovu received his graduate education at the Southern Swedish Forest Research Centre, SLU, Alnarp, and his MSc. in Forestry at the Technical University of Dresden, Germany.

Acta Universitatis Agriculturae Sueciae presents doctoral theses from the Swedish University of Agricultural Sciences (SLU).

SLU generates knowledge for the sustainable use of biological natural resources. Research, education, extension, as well as environmental monitoring and assessment are used to achieve this goal.

Online publication of thesis summary: http://epsilon.slu.se/eindex.html

ISSN 1652-6880

ISBN 978-576-7555-2