FACTORS AFFECTING THE RECRUITMENT OF TREE SPECIES IN RESTORED TROPICAL FOREST, CHIANG MAI PROVINCE, THAILAND

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DOCTOR OF PHILOSOPHY
IN BIOLOGY

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**A THESIS SUBMITTED TO CHIANG MAI UNIVERSITY IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY
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Yingluck Ratanapongsai

หัวข้อวิทยานิพนธ์	ปัจจัยที่มีผลต่อการกลับคืนของชนิดไม้ยืนต้นในพื้นที่ฟื้นฟูป่าเขตร้อน จังหวัดเชียงใหม่ประเทศไทย	
ผู้เขียน	นางสาวยิ่งลักษณ์ รัตนผ่องใส	
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บทคัดย่อ

หน่วยฟื้นฟูป่า (FORRU-CMU) ทำการศึกษาพัฒนาวิธีการฟื้นฟูป่าด้วยวิธีพรรณไม้โครงสร้าง (Framework Species (FWS)) เพื่อปรับให้มีความเหมาะสมกับพื้นที่ป่าทางภาคเหนือของประเทศไทย ตั้งแต่ปี พ.ศ. 2537 ซึ่งมีจุดมุ่งหมายช่วยเร่งกระบวนการการแทนที่ป่า และฟื้นฟูระบบนิเวศป่า โดยเพิ่มทั้งชีวมวลและความหลากหลายในป่าฟื้นฟู โดยกระบวนการนี้พึ่งพาการคัดเลือกพรรณไม้ที่มีความสามารถดึงดูดพาหะนำเมล็ดไม้เข้ามาในพื้นที่ และยังช่วยปรับปรุงระบบนิเวศให้เหมาะสมต่อการตั้งตัวได้ของ “ชนิดต้นไม้ที่กลับคืนมา (recruit tree species)” ดังนั้นการศึกษานี้มีจุดมุ่งหมายเพื่อเพิ่มความเข้าใจ และบ่งชี้ประสิทธิ์ภาพของวิธีพรรณไม้โครงสร้างต่อการกลับคืนมาของกล้ามธรรมชาติ ในแปลงเก็บข้อมูล ป่าฟื้นฟูต่างอายุ 3 ปี ได้แก่ อายุ 6 ปี (R6), อายุ 10 ปี (R10), และ 14ปี (R14) เพื่อเปรียบเทียบกับแปลงควบคุมที่เกิดการทดแทนตามธรรมชาติอายุ 14 ปี (Control Site) และป่าธรรมชาติดงเส้ง (DSF) ของหมู่บ้านแม่สาใหม่ โดยการสำรวจกล้าไม้ชนิดไม้ยืนต้นท้องถิ่นทั้งยังสำรวจต้นแม่ในป่าธรรมชาติ และปัจจัยโครงสร้างต้นไม้ใหญ่ ปัจจัยแสงและค่าความจุภาคสนาม (Soil Field Capacity - FC) เพื่ออธิบายอัตราการกลับคืนมา และอัตราการตายของกล้าไม้ ร่วมถึงความสัมพันธ์ของปัจจัยต่างๆ ที่มีผลกับการกลับคืนมาของกล้าไม้เหล่านี้

ผลการศึกษาพบชนิดพันธุ์กล้าไม้ที่เจริญในป่าฟื้นฟูทั้ง 3 ชั้นอายุ คิดเป็น 51% จากชนิดพันธุ์ทั้งหมดที่พบในการสำรวจชนิดพันธุ์ต้นแม่ ปัจจัยความหนาแน่นของต้นแม่ (Adult Tree Density) ส่งผลอย่างมีนัยยะสำคัญต่อความสำเร็จของชนิดต้นไม้เข้ามาตั้งต้นในป่าฟื้นฟู ในขณะที่ปัจจัยขนาดเมล็ด (Seed

Size) พาหะช่วยกระจายเมล็ด (Seed-dispersal Mode) และลักษณะ Successional Status ของชนิดพันธุ์ต้นไม้ นั้นไม่ใช่ปัจจัยที่ส่งผลอย่างมีนัยยะสำคัญ กล้าไม้ของชนิดที่คืนกลับมา (Recruit Species) มีค่าความหลากหลาย (Shannon Diversity Index) ใกล้เคียงแปลงป่าธรรมชาติ (DSF) กล้าไม้เด่นในแปลงปลูกคือชนิดที่มีเมล็ดเล็ก และมีสัตว์ช่วยกระจายเมล็ด (Animal-dispersed Species) ซึ่งมีมากในแหล่งชนิดพันธุ์ อัตราการกลับคืนมาของกล้าไม้ชนิด Non-pioneer ไม่แตกต่างอย่างมีนัยยะสำคัญระหว่างแปลงฟื้นฟูทั้ง 3 อายุ ส่วนกล้าไม้ชนิด pioneer นั้นมีอัตราการการตายในแปลงปลูกอายุน้อย (R6) น้อยกว่าแปลงฟื้นฟูอายุมาก (R10 และ R14) โครงสร้างต้นไม้ที่ปลูกในป่าฟื้นฟูฟื้นคืนสภาพใกล้เคียงป่าธรรมชาติภายใน 6 ปี อีกทั้งผลสอดคล้องกับปริมาณแสงที่กล้าไม้ได้รับในพื้นที่ที่ลดลง ค่าเฉลี่ย LAI และ GSF ของแปลงฟื้นฟูไม่แตกต่างอย่างมีนัยยะสำคัญกับแปลงป่าธรรมชาติ ส่วนค่าความจุความชื้นภาคสนาม (FC) ของป่าฟื้นฟู บ่งบอกถึงความสามารถเก็บกักความชื้นของดินต่ำ และมีสภาพดินที่แน่นอยู่ เมื่อเทียบกับดินในป่าธรรมชาติปัจจัยพื้นที่หน้าตัดต้นไม้ (BA) และสภาพแสงในแปลงฟื้นฟู เป็นสองปัจจัยสำคัญที่ส่งผลควบคุมการกลับคืนมาของกล้าไม้ในพื้นที่ ในแปลงที่มีค่า BA และ LAI ต่ำ ช่วยส่งเสริมอัตราการกลับคืนมาของกล้าไม้ชนิด Non-pioneer ในขณะที่แปลงที่มีค่า BA สูง แต่มีค่า GSF ต่ำ จะส่งผลให้อัตราการกลับคืนมาของกล้าไม้ชนิด Pioneer ลดลง ป่าฟื้นฟูอายุน้อย (R6) มีสภาพป่าที่ส่งเสริมกล้าไม้ ชนิด Pioneer ในพื้นที่ มากกว่าแปลงฟื้นฟูที่มีอายุมากกว่า 10 ปีขึ้นไป นอกจากนี้ผลที่ได้ชี้ชัดว่าแปลงฟื้นฟูเองตามธรรมชาติ (Control Site) นั้นพัฒนาช้ามาก กล้าไม้ของชนิดที่มีเมล็ดขนาดใหญ่ไม่สามารถคืนกลับมาเจริญในพื้นที่ได้

ผลที่ได้จากการศึกษา แสดงให้เห็นประสิทธิภาพของการฟื้นฟูป่าโดยวิธีพรรณไม้โครงสร้าง วิธีการนี้สามารถคืนสภาพป่า โครงสร้างต้นไม้ใหญ่ ปัจจัยแสงในป่า ให้กลับมาใกล้เคียงป่าธรรมชาติ และมีประสิทธิภาพกว่าการปล่อยให้ฟื้นฟูเองตามธรรมชาติ (Control Site) อีกทั้งแปลงป่าฟื้นฟูยังช่วยฟื้นคืนความหลากหลายของกล้าไม้คืนกลับมา โดยทั้งนี้ยังมีข้อเสนอแนะเพื่อแก้ไขปรับปรุงประสิทธิภาพในการฟื้นฟูป่าให้ดีมากขึ้น เช่นการศึกษาเพิ่มเติมในการปรับปรุงสภาพดิน ในพื้นที่มีข้อจำกัดพบความหนาแน่นของต้นไม้แม่พันธุ์ต่ำ การคัดเลือกชนิดพันธุ์ที่เป็นชนิดหายากเป็นพรรมไม้โครงสร้างจะช่วยเพิ่มความหลากหลายทางชีวภาพของป่าฟื้นฟู การใช้วิธีการฟื้นฟูป่าแบบผสม เพื่อแก้ไขข้อจำกัดของปัจจัยต่างๆในแต่ละพื้นที่ เป็นอีกวิธีที่สามารถเพิ่มประสิทธิภาพในการฟื้นฟูป่าให้มากขึ้นได้อีกเช่นกัน

คำสำคัญ**:** Framework species, Restoration forest, Seedling dynamic, Seedling recruitment, Tree density, Leaf area index (LAI)

Thesis Title Factors Affecting the Recruitment of Tree Species in Restored Tropical Forest in Chiang Mai Province, Thailand

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Degree Doctor of Philosophy (Biology)

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ABSTRACT

FORRU-CMU has been developing and adapting the framework species method of forest restoration (FWS) to restore tropical forest ecosystems to degraded forest land in northern Thailand, since 1994. The technique accelerates natural forest succession by selecting tree species that attract seed-dispersing animals and create conditions conducive for seed germination and early seedling establishment of incoming ‘recruit’ tree species. The research reported here, therefore, tested the effectiveness of the method in achieving seedling recruitment of native tree species, by field surveys of FWS plots at Baan Mae Sa Mai (BMSM). The study site consisted of FWS plots of 3 different ages: 6, 10 & 14 years since commencement of restoration activities (abbreviated to R6, R10 and R14), compared with 14-year-old natural regeneration site (control site) and old-growth, natural, remnant forest (DSF). Seedlings of native species and adult trees were systematically surveyed. Leaf area index (LAI), global site factor (GSF) and soil field capacity (FC) were also evaluated as factors likely to affect seedling recruitment and mortality rates. Fifty-one per cent of total adult species present in the vicinity of the plots (including FWS planted species) successfully recruited into the seedling communities, across all R plots. The most important limiting factor was species rarity in the surrounding area. Seed size, seed-dispersal modes and successional status of tree species did not significantly influence recruitment success or failure. After 6-14 years of restoration, the diversity of recruit species approached levels similar to those of the natural remnant forest (DSF). Seedlings of animal-dispersed species were more common and had higher recruitment rates in all R plots. The recruitment rate of seedlings of non-pioneer species was similar across all 3 ages of R plots, whilst the mortality rates of pioneer species was lower in the R6 plots than in the R10-14 plots. Density and basal area of the planted FWS trees exceeded those of naturally established trees in the DSF plots, after 6 years of restoration. Consequently, the dense forest canopy of the FWS trees reduced light availability at ground level in the R plots to levels similar to those in the DSF plots. However, soil moisture-holding capacity (per cent moisture at field capacity, FC) of all the R plots was lower than that of the DSF plots. Both tree BA and light conditions strongly influenced recruitment and mortality rates of both pioneer and non-pioneer tree seedling species. Low LAI and BA of understorey trees in the R forests supported recruitment of seedlings of non-pioneer species, whilst low GSF and high BA of canopy trees decreased seedling recruitment of pioneer species. Conditions in R6 were more favourable for pioneer species establishment than they were in older plots. Forest structure of in the natural regeneration control plots (after 14 years) was poorly developed. Large-seeded spices failed to recruit into such plots. Better conditions for seedling establishment occurred along the R14-control plot boundaries.

The FWS technique efficiently accelerated forest habitat recovery and native species recruitment, compared with DSF and the control, natural regeneration, sites. The FWS trees were well develop, providing shady habitat for tree seedling establishment and showing signs of catalyzing habitat recovery beyond the plot boundaries. However, further study of soil FC improvement is recommended, since its recovery after 14 years was unexpectedly slow. Indigenous tree species that are absent or rare in the surrounding landscape should be included in FWS plantings, to enhance diversity, since they are unlikely to recruit without intervention. A combination of several restoration methods might help researchers overcome site limitations and accelerate forest recovery for larger landscapes.

Key worlds: Framework species, Restoration forest, Seedling dynamic, Seedling recruitment, Tree density, Leaf area index (LAI)

ข้อความแห่งการริเริ่ม

ข้าพเจ้าขอรับรองว่างานวิทยานิพนธ์ในเล่มนี้ยังไม่เคยได้รับการตีพิมพ์หรือเผยแพร่ที่ใดมาก่อน และข้อความ บทความที่ปรากฏในวิทยานิพนธ์เล่มนี้ไม่มีเนื้อหา หรือข้อความที่เขียน หรือตีพิมพ์โดยผู้อื่นมาก่อน ยกเว้นในส่วนของเนื้อหาที่มีการอ้างอิงถึงเท่านั้น

โดยที่วิทยานิพนธ์นี้ ได้นำเสนอหลักฐาน และข้อมูลใหม่เกี่ยวกับการกลับคืนมาของชนิดพันธุ์ไม้ รวมถึงปัจจัยที่มีผลต่อกล้าไม้ ในป่าฟื้นฟูโดยวิธีพรรณไม้โครงสร้าง ความรู้ ความเข้าใจที่ได้จากวิทยานิพนธ์เล่มนี้สามารถนำไปปรับใช้ ปรับปรุง และพัฒนาวิธีการฟื้นฟูป่า โดยเฉพาะอย่างยิ่งป่าในจังหวัดเชียงใหม่ ประเทศไทยให้มีประสิทธิภาพดียิ่งขึ้น

STATEMENT OF ORIGINALITY

The work contained in this thesis has not been previously submitted for a degree or diploma at any other higher education institution. I certify that, to the best of my knowledge, the thesis contains no material previously published or written by another person except where due references are made.

This thesis provides evidences and new information of native recruit species and progressive forest recovery by the technique called Framework species (FWS). This knowledge can be efficiently adapted and used for the improvement of forest restoration, especially in Chiang Mai Province, Thailand.

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LIST OF ABBREVIATIONS

A	Animal-dispersed species
BA	Basal Area
BD	Total bulk density
BMSM	Baan Mae Sa Mai village (study site), Chiang Mai, Thailand
cm	Centimeter
CRD	Cord Distance index
DBH	Diameter at Breast Height
DSF	Dong Seng forest the natural remnant forest of BMSN
FC	Soil Field Capacity
FORRU-CMU	Forest Restoration Research Unit of Chiang Mai University
FWS	Framework Species
GSF	Global Site Factor
ha	Hectare
HemiView	Canopy analysis software for Hemispherical photographs
LAI	Leaf Area Index
M	Mortality Rate
m	Meter
mm.	Millimeter
N-A	Non-animal-dispersed species
na	Not available
N-P	Non-pioneer species
P	Pioneer species
PCA	Principle Component Analysis
Plot	Circular (5 m-radius) sample plots
R	Recruitment Rate
R forest	Restoration forest
RCD	Root Collar Diameter

Recruit seedling	Non-planted species that is new emerge seedling after first census
Recruit species	Native species that was not FWS planted species or mother tree of species that present in sample plots
RGR	Relative Growth Rate
RGR−H	Relative growth rates of height
RGR−R	Relative growth rates of root collar diameter
Seedling	The tree and treelet plants since have true leaves until their height not over 1.5 meter
T	Turnover Rate

CHAPTER 1

Introduction

1.1 Global Forest Losses and the Threat to Biodiversity

The problems of deforestation and rapid increase of atmospheric CO_2 in recent decades are well documented and public concern about the loss of forests and biodiversity, as well as carbon storage, is growing. Conversion of large areas of forest to various human activities continues. Although the rate of deforestation has recently declined, tropical forests are still threatened. In 2010, the FAO (FAO, 2010-a) reported that South America and Africa continue to have the highest rates of net forest loss. Between 2000 and 2010, South America lost about 4.0 million hectares of forest per year, whilst Africa was ranked second, losing 3.4 million hectares per year. During the 1990s, the net forest loss rate in Asia was 600,000 ha per year. Due to large-scale afforestation in China, the forest area there actually increased by about 2.2 million ha per year in 2000s. However, net forest loss by deforestation increased in most other countries in in South and Southeast Asia.

Plants uptake carbon dioxide (CO_2) from the atmosphere and convert it into carbohydrates, which are then stored, releasing oxygen during the process of photosynthesis. Forests, as the main terrestrial vegetation types, have huge potential as carbon stocks and for sequestering further CO_2 from the atmosphere. The world's forests store more than 650 billion tonnes of carbon (FAO, 2010-a). Righelato and Spracklen (2007), pointed to the conservation of existing forests and savannahs, and the restoration of natural forest habitat as short term solutions should be focused on in order to mitigate carbon dioxide–driven global warming. Furthermore, terrestrial forests, especially tropical forests, are an important repository of biodiversity on Earth (Whitmore and Sayer, 1992). Forest loss not only emits CO_2 into the atmosphere, but it also reduces future carbon sequestration potential and results in loss of wildlife habitat and consequently leads to biodiversity losses as well.

1.2 Thailand - Forest Degradation and Biodiversity Threat

Thailand's forest cover decreased by an average of 99,600 ha per year from 1991 to 1998 and 63,000 ha per year from 2000 to 2004. Forests in the north accounted for 52% of national forest cover in 2010 (FAO, 2010-b). The upland water catchment areas of Northern Thailand are important sources of water, which supply Thailand's main river systems. They are also home to many wildlife species, including many rare or threatened species. Doi Suthep-Pui National Park in Chiang Mai Province (where the study related here was carried out) was once home to highly diverse plant and animal communities including 5 hornbill species (*Dichoceros bicornis*, *Anthracoceros coronatus*, *Aceros undulatus*, *Aceros nipalensis*, and *Ptilolaemus tickelli*) as well as wild elephants (*Elephas maximus*), which have now become extirpated (Round, 1984 in Elliott, 1994; Brodie, 2007). Maxell and Elliott (2001) reported 1,285 species of vascular plants recorded in the park.

Deforestation and conversion of forest to agriculture is a major environmental problem in northern Thailand, particularly in the upland water catchment areas, causing droughts, floods, landslides and contributing to global climate change. Historically, natural forest succession has been adequate at repairing deforested areas. Unfortunately, intensive cultivation and human activities on degraded area are now major factors that delay or inhibit the natural processes of forest succession. For this reason, forest restoration by human intervention is needed. It is capable of not only increasing forest cover, but also with proper management, it can increase biodiversity and carbon sequestration (FORRU, 2005). During the 1970s - 1980s, after a period of a rapid reduction in forest cover in Thailand, reforestation was attempted on a country-wide scale by established mono-plantations of Teak (*Tectona grandis*), Pine (*Pinus kesiya*) or fast-growing exotic species, particularly *Eucalyptus sp.* (Oberhauser,1997).

1.3 Forest Restoration Development

The definition of plantations and restored forests are deviation from those of "natural secondary forest" (Lugo, 1992). However, planted forests enhance the rate of forest restoration and forest succession.

Forest restoration is …

"Directing and accelerating ecological succession towards an indigenous target forest ecosystem of the maximum biomass, structural complexity, biodiversity and ecological functioning that can be self-sustained, within prevailing climatic and soil limitations." (FORRU, 2005)

… where management aims include:

a) delivery of long-term watershed services, such a reliable supply of clean water and prevention or reduction in soil erosion, floods, landslides and droughts (since restoration aims for a persistent, self-sustaining ecosystem);
b) carbon sequestration (since restored forests trend towards maximum biomass);
c) biodiversity recovery and/or the conservation of rare or endangered species (since restored forests trend towards maximum (or equilibrium) species richness) and/or
d) delivery of a *diverse* range of forest products is beneficial to local communities.

Reforestation in the past often paid little attention to biodiversity recovery, often using exotic tree species in mono-species plantation, such as *Eucalyptus spp.*, *Acacia spp*, or high-value, economic, native species, such as pines (*Pinus spp.*) and teak (*Tectona grandis* L.). Since the 1990's, the total area of "restored" forest has increased, but this has mostly been due to the spread of economic plantations, at the expense of natural forest (Niskanen, 1998).

Tree species selected for tropical plantations have a large effect on biodiversity recovery. Kanowski *et al.* (2003) showed that the density of regrowth of woody plants is significantly higher in broadleaf tree plantations than in conifer plantations in the Australian tropics (Kanowski *et al.* 2003). Catterall, *et al.* (2008) summarized diversity of plant seedling found in many researches of plantation, restoration forest and natural secondary forest. The diversity in restoration forest by timber plantations were averagely lower (1-10 species) those of ecological restoration plantations (20-50 species). In regions of high biodiversity and ecological structural complexity, such as tropical forests, economic plantations are not the best way to accelerate and recover ecological functioning and forest biodiversity. Many researchers encouraged technique of **"Ecological Restoration"** – using mixtures of several native species plantation (Rey-

Benayas *et al.*, 2009; Tucker & Murphy, 1997; Elliott, 2000; Souza & Batista, 2004, Kaewkrom *et al.,* 2005). Hardwick *et al.* (1997) and Shono *et.al.* (2007) advocated Assisted Natural Regeneration (ANR) to encourage and accelerate secondary forest succession. A low-cost protocol was developed, with or without minimal tree-planting, to reduce barriers to natural forest regeneration, such as weed competition and soil degradation. The technique maximizes the chances of germination of incoming seeds advocated and early seedling establishment. It not only restores forest, but also brings about biodiversity recovery (Hardwick *et al.* 1997)). However, the method relies on an adequate density of seed trees surviving nearby the restoration site and adequate populations of seed-dispersing animals. On severely disturbed sites, far from seed sources, ANR alone is ineffective and forest regeneration is delayed. Therefore, additional tree planting and management become necessary to achieve self-sustainable recovery (Elliott, 2000).

Kageyama and Castro (1989) proposed planting a ratio of 1:1 pioneer:non-pioneer tree species on degraded sites, to reduce the costs and time needed for soil recovery. Fast-growing pioneer trees were used as foster trees, to provide suitable conditions for shade-tolerant, climax species to grow back (Kageyama and Gandara, 2000; De Souza & Batista 2004). Later, De Souza and Batista (2004) modified their method to include higher proportions of pioneer species. They reported a higher density of recruit woody plants in old age forest (9 years old).

One method of forest restoration that has proved very successful in Queensland, Australia, is the so called 'Framework Species Method' (FSM), which Goosem and Tucker proposed to restore tropical rainforest (Tucker & Murphy, 1997; Goosem & Tucker, 2013). The technique involves planting 20-30 tree species, with high growth and survival rates in deforested sites, dense crowns, to shade out weeds and re-capture the sites and which provide wildlife resources, early in life, to attract seed-dispersing animals. Tucker and Murphy (1997) reported the return of 72 recruit species of native forest tree seedlings in 7 year-old restored forest (Tucker & Murphy, 1997).

In Thailand, in the 1990's, monoculture policy was drastically revised after the King suggested plantations of mixed species the "Ploor Pah Chalerm Prakiet" project. Since 1994, Chiang Mai University's Forest Restoration Research Unit (FORRU-CMU) has been adapting and developing a similar framework species approach to restore various forest types in Thailand and neighbouring countries, based on concepts developed in Queensland, Australia (Tucker & Murphy, 1997). The unit focused first on identifying framework species, suitable for restoring evergreen, seasonal forest (EGF) in northern Thailand (FORRU, 1998, 2000; *sensu* Maxwell and Elliott, 2001). Framework Tree Species (FWS) are indigenous forest tree species, planted to complement and accelerate natural regeneration of forest ecosystems and encourage biodiversity recovery, on degraded sites (Elliott *et al.,* 2003). The five essential characteristics of framework tree species are 1) high survival and growth rates, 2) broad, dense crowns to shade out herbaceous weeds, 3) produce fruits, nectar-rich flowers or other resources at a young age that attract seed-dispersal wildlife, 4) fire resilient and 5) easy to propagate. A mix of 20-30 framework species, including both pioneer and climax species is planted, followed by weeding and fertilizer application for 2-3 rainy seasons thereafter (FORRU 2008). Processes of natural forest succession resume and the ecosystems become self-sustaining within 5 – 10 years, as demonstrated in experimental plots at Ban Mae Sa Mai (BMSM) the upper Mae Sa Watershed in Doi Suthep-Pui National Park.

From its conception in Australia and development in Thailand, the FWS concept has been adapted and used in many countries, such as South Africa, where Walker *et al*. (2015) developed a species selection index for seed-based ecological restoration on the Peninsula Shale Renosterveld, near Cape Town. In Indonesia, Harrison and Swinfeld (2015) assessed its value in restoring logged humid tropical forests at the Harapan Rainforest Project.

1.4 Seedling Recruitment and Forest Restoration

As mentioned earlier, forest restoration relies on natural forest succession to recover biodiversity and forest habitat. Natural forest succession is the change of plant species composition and structure of a forest plant community over time. In natural secondary forest, it is initiated by the return of herbaceous weeds and grasses. Subsequently, a few pioneer tree species establish and grow to dominate the area, shading out herbs and grasses. The dominance of the light-loving pioneer tree communities then gradually declines as shade-tolerant tree species become established beneath the developing forest canopy. The latter become more abundant in mid- and late-succession (Brown & Lugo, 1990; Bazzaz, 1996).

Natural regeneration can be inhibited or accelerated, depending on: the frequency and intensity of disturbances, the density of surviving natural regenerants (seedlings, saplings and live tree stumps growing among the weeds) and the distance, density and species of seed sources, as well as elevation, soil conditions, climate and previous forest type (FORRU, 2005). The formation of an understory in restored forest marks an important milestone in the recovery of forest ecosystem structure and biodiversity (Ashton *et al*., 2001)

In pine plantations in Northern Thailand, Oberhauser **(**1997) reported that forest structural complexity increased with plot age, especially in the mid- and low-level canopy layers. Underneath these pine plantation, density and species richness of trees (size ≥1 cm-dbh) was low (only 13 species) in young age stands (7-year-old plot) but the latter increased to 41–47 in 12-, 21- and the 28-year-old plots. At the Mae Yuak Plantation Station, Lampang province, Thailand, Koonkhunthod *et al.* (2007) found only 9.6 tree species (size ≥4.5 cm-dbh) per 400-m^2-plot in a 37-year-old teak plantation. In Brazil, 9-year-old restored forest had developed a complex structure with a high density of understory trees; enough to catalyze natural secondary succession (Souza and Batista, 2004).

In framework species trial plots in northern Thailand, the species richness of tree seedlings increased from 20-30 (in control non-planted plots) to 73 recruit species in the restoration plots after 8 years (Sinhseni, 2008). Wydhayagarn *et al*. (2009) reported that more animal-dispersed tree seedling species recruited under FWS trees than wind-dispersed species in 8-year-old forest, restored by the FWS method. This result agrees with those of previous studies of restored forest in Australia (Tucker & Murphy, 1997) and Costa Rica (Oberhauser, 1997; Cusack & Montagnini, 2004).

Consequently, biodiversity increases with restoration plot age (De Souza & Batista 2004). Canopy closure and the development of structural complexity of the sub-canopy and understory in restoration forests influence the light regime on the forest floor (Montgomery & Chazdon, 2001; Valladares & Guzman, 2006). Age since tree planting and the species planted both influence light levels – a critical factor in the establishment of tree seedlings. For example, the open canopies of pine plantations (age 31year-old) enhanced recruitment of native woody species (size ≥1cm dbh) more than the dense canopies of *Cupressus lusitanica* plantations (age 28-year-old) in southern Ethiopia (Lemenih *et al.,* 2004). In contrast, Beckage *et al*. (2000) found no influence of gap formation on seedling recruitment in a secondary deciduous forest in America.

Plant morphology also plays an important role in the response of plants to variations in microhabitat conditions. Some species that are able to grow under low light levels can survive beneath dense forest canopies (Wright, 2002; Rüger *et al*., 2009). Seedling recruitment is limited when species fail to reach suitable sites, due to lack of seed-dispersal, distance from nearest seed source and forest habitat etc. Both seed-dispersal and seedling establishment limit seedling recruitment in restoration sites (De Souza & Batista, 2004) and are important determinants of local abundance (Dalling *et al.,* 1998). A lot of research on forest restoration in tropical forest has concentrated on *planted* tree development at different ages, as an indicator of forest recovery. Knowledge of seedling recruitment and successional dynamics in forests undergoing restoration is still lacking and more information about it is needed to improve forest restoration methods.

1.5 Rational

In Thailand, reforestation has been practiced for decades, based mostly by replacing natural forest regeneration with mono-species economic plantations. Use of diverse mixtures of native tree species, to achieve results, similar to natural forest ecosystems, is still mostly practiced at small experimental scales. Further study of the FWS restoration technique, developed to restore the evergreen forest in northern Thailand, is needed for a better understanding of forest dynamics and successional processes and to increase the knowledge of those factors that might limit the progression of forest restoration. Therefore, the research reported here focussed on the tree seedling community and recruitment rates within framework species trial plots of different ages, compared with a remnant of disturbed primary forest nearby. Factors related to the seedling community composition and recruitment levels were also tested, namely regeneration traits and seed-dispersal mechanisms. Light availability and forest soil field-capacity were also studied, as the two physical factors most likely to affect seedling dynamics and limit seedling recruitment.

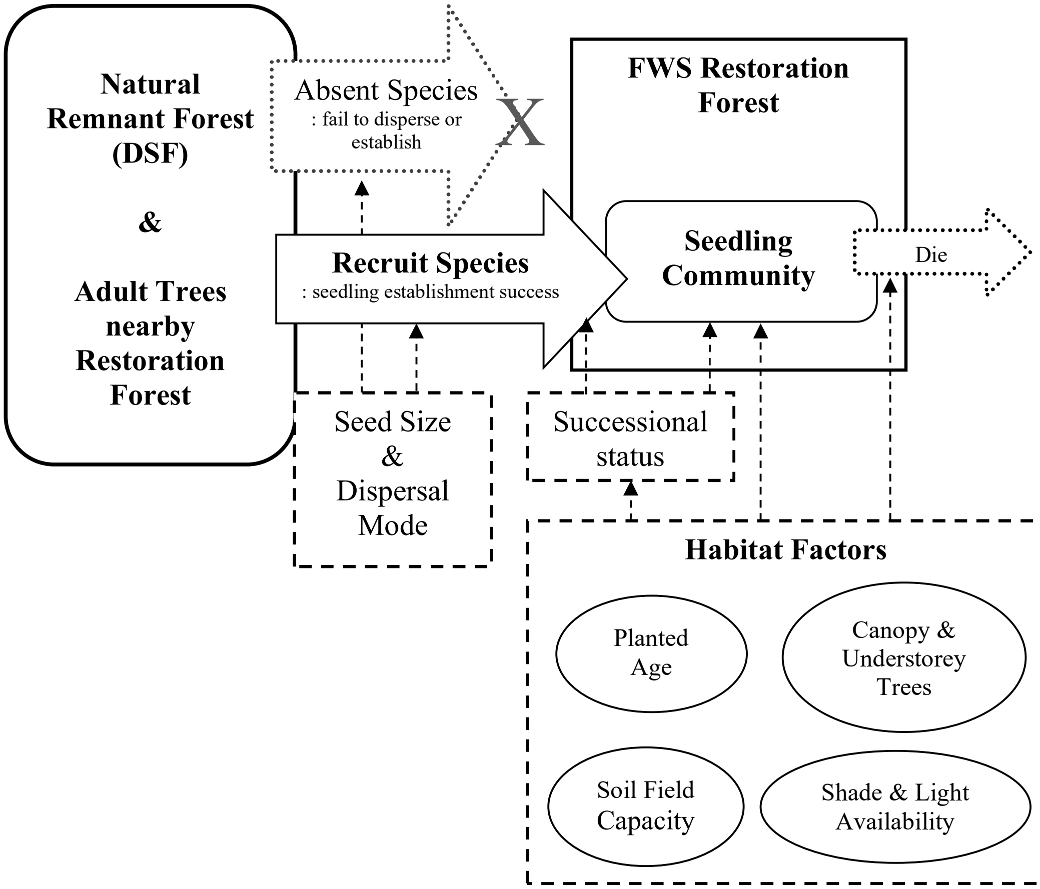


Figure 1.1: Schematic model of seedling recruitment and dynamics with possibility affecting factors (dashed line) in the FWS restoration forest.

Finally, this study monitored and increased understanding of succession dynamics and native seedling recruitment within forest restoration trials, to generate original knowledge of forest recovery trends, after planting and to facilitate improvements to restoration practices. The project addressed two key research questions about tree seedling diversity recovery, 1) how well did tree seedlings recruit within restored forest and 2) what are the major factors that limit recovery of tree seedling diversity in restored forest.

1.6 Research Objectives

To determine which tree species establish or fail to establish in restored plots, from the pool of seed trees present in nearby remnant forest and why? (see chapter 3)

1) To estimate the rate of species seedling dynamics and the diversity of recruit seedlings in restored forests, how they vary with plot age and with those in a natural remnant forest and non-planted control sites. (see chapter 3)
2) To identify relationships between species successional status and seed dispersal mechanisms of seedling species and establishment success in restoration plots. (see chapter 4)
3) To determine tree development and structure of canopy and understory trees in the restoration forest. (see chapter 5)
4) To evaluate the influence of FWS canopy and understory trees on seedling recruitment and dynamics in the restoration forests. (see chapter 5)
5) To evaluate the effects of light availability and soil field capacity on seedling dynamics and recruitment. (see chapter 6)

CHAPTER 2

Study Site, Ban Mae Sa Mai

Ban Mae Sa Mai (BMSM) is a Hmong hill tribe village community, located on the northern slopes of Doi Suthep-Pui National Park in Mae Rim district north of Chiang Mai province, northern Thailand. It comprises 239 households with a population of around 1,772 people and is the largest Hmong community in northern Thailand. It was founded in 1965, when villagers earned their living by growing opium poppies in slash and burn forest patches around the village.

After a visit by the King of Thailand in the 1970's, a station of the Royal Project was established near the village to support the development of alternative agricultural crops. The main crops were corn, cabbage, lychee and sweet peppers *etc.*, which need far more land than opium, to earn a living. Increases in the village population also caused large forest areas to be cleared for agriculture. Forest destruction led to decreased water supply, soil erosion, landslides and extirpation of many wildlife species (Elliott *et al.*, 2012).

Figure 2.1: Cleared forest and agriculture area in the past of BMSM

Figure 2.1: Cleared forest and agriculture area in the past of BMSM (cont.)

2.1 The Rise of Environmental Awareness

The villagers became aware that forest clearance was linked to diminished water supply, particularly during the dry season. Consequently in 1995, a few villagers formed the "BMSM Natural Resources and Environmental Conservation Committee" to encourage villagers to use the forest more sustainably and to ensure protection of water resources. Later in 1996, the BMSM villagers began to collaborate with the Forest Restoration Research Unit of Chiang Mai University (FORRU-CMU) and Doi Suthep-Pui Natural Park authority to establish the first FWS forest restoration trials on degraded former-agriculture land. Plantings continued annually, expanding the plot system each year until 2013.

Over the decades, around 89,000 trees were planted over 158 rai (about 25.3 hectares), forming a wildlife corridor along the upper watershed ridge. These plots have become a model FWS trial plot system for scientific research and for others to study the concept of forest restoration (Elliott *et al*. 2012)

2.2 Forest Habitat and Native Species

Researchers have been attracted to study the forest habitats and biodiversity of Doi Suthep-Pui since the beginning of the 20th century. Consequently, the area has become one of the best documented forests in Thailand (Elliott, 1994). Sawyer & Chermsirivathana (1969) mapped the vegetation of the south-eastern mountain slopes and recorded 679 vascular plants. Later, Cheke *et al.* (1979) studied the dynamics of secondary species in 5 forest types. They found high populations of animal seed-dispersal agents of both primary and secondary vegetation. The dominant tree species were *Dipterocarpus tuberculatus* Roxb. and *D. obtusifolius* Teysm.

Round (1984) reported the presence of 326 bird species, whilst 61 mammal species, 28 amphibian species, 50 reptile species, more than 500 butterfly species and more than 300 moth species are also known to inhabit the park (Elliott & Maxwell, 1995). The park also supports several forest habitats from lowland vegetation to hill evergreen forest. A previous studied by Maxwell in 2001, reported 2,247 vascular plant species, of which 21.6% are tree species (Maxwell & Elliott, 2001).

The Department of National Parks Wildlife and Conservation of Thailand (2009) recognizes 4 forest types in Doi Sutep-Pui National Park:

1) Deciduous Dipterocarp Forest; found at 330-900 m above sea level. The dominant tree species are *Shorea obtusa* Wall., *S. siamensis* Miq., *Dipterocarpus obtusifolius* Teijsm. Ex Miq, and *Schima wallichii* Korth (Wacharamusik, 1998; Khamyong, 2003).
2) Mixed Deciduous Forest; found at 330-600 m above sea level. The dominant tree species are *Dipterocarpus costatus* Gaertn. f., *Lithocarpus lindleyanus* A. Camus, *Helicia nilagirica* Bedd, *Terminalia mucronata* and *Pterocarpus macrocarpus* (Wacharamusik, 1998; Khamyong, 2003).
3) Dry Evergreen Forest; at 400–1,000 m. with dominant tree species: *Dipterocarpus turbinatus* Gaertn. f., *Aphanamixis polystachya* Parker, *Pinus kesiya* Royle ex Gard., *Castanopsis diversifolia* and *Castanopsis acuminatissima* (Cheke *et .al* 1979; Wacharamusik, 1998; Khamyong, 2003)
4) Lower Montane Forest; above 1,000 m.

2.3 Geography, Physical and Climatic Condition

The study site was above Ban Mae Sa Mai village (BMSM), located in tropical evergreen forest, 18°52’N and 98°51’E, and situated at 1,000–1,300 meters above sea level. The average annual rainfall was 703.5 mm. Low precipitation months are November to March, about 5 months of a year, and the average annual temperature was 26.8°C (Northern Meteorological Center (NMC), Chiang Mai province, 2012-2014). The data collection period was August 2012 to April 2014. Dominant indigenous tree species were mostly in the Fagaceae, e.g. *Castanopsis diversifolia* and *Castanopsis tribuloides*.

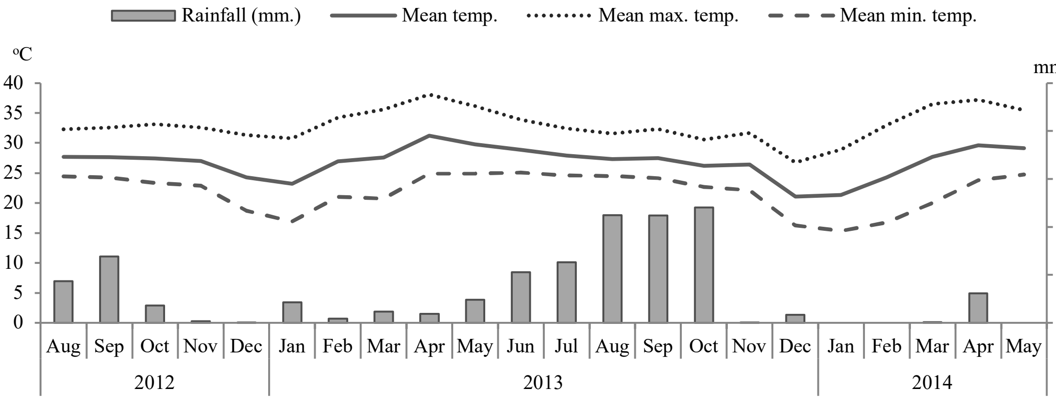


Figure 2.2: Precipitation (solid box) and average (solid line), maximum (dotted-line), and minimum (dashed-line) temperature of each month from year 2012 to 2014 (Northern Meteorological Center (NMC) Chiang Mai province, 2012-2014)

2.4 Study Plot Establishment

Sample plots were established in three major successional forest stages: i) restoration trial plots (R), planted with candidate framework tree species for testing at 3 different ages: R14, R10, and R6 (age 14, 10, and 6 year-old, respectively at the time of data collection), ii) 3 control sites were selected - former agricultural land adjacent to the R14 plots, which had undergone natural forest regeneration without any restoration planting or treatments for 14 years and iii) the nearest surviving remnant of natural forest, Dong-Seng forest (DSF), kept as a sacred forest by the villagers (disturbed primary forest, which had never been cleared for agriculture, but which had suffered minor disturbances being situated immediately adjacent to the village). All sites were triplicated.

Twelve circular plots of 5-m radius were permanently established (with iron poles to mark the centre) in each of the above habitats with 4 circles established in each of the triplicated sites. As much as possible, the circular plots were established at the same points where FORRU had originally established vegetation monitoring points for their pre-planting baseline surveys or as close as possible (in the control and the R sites). In natural remnant forest, 12 plots were set at 3 different altitudes at 1300, 1000, and 800 m.

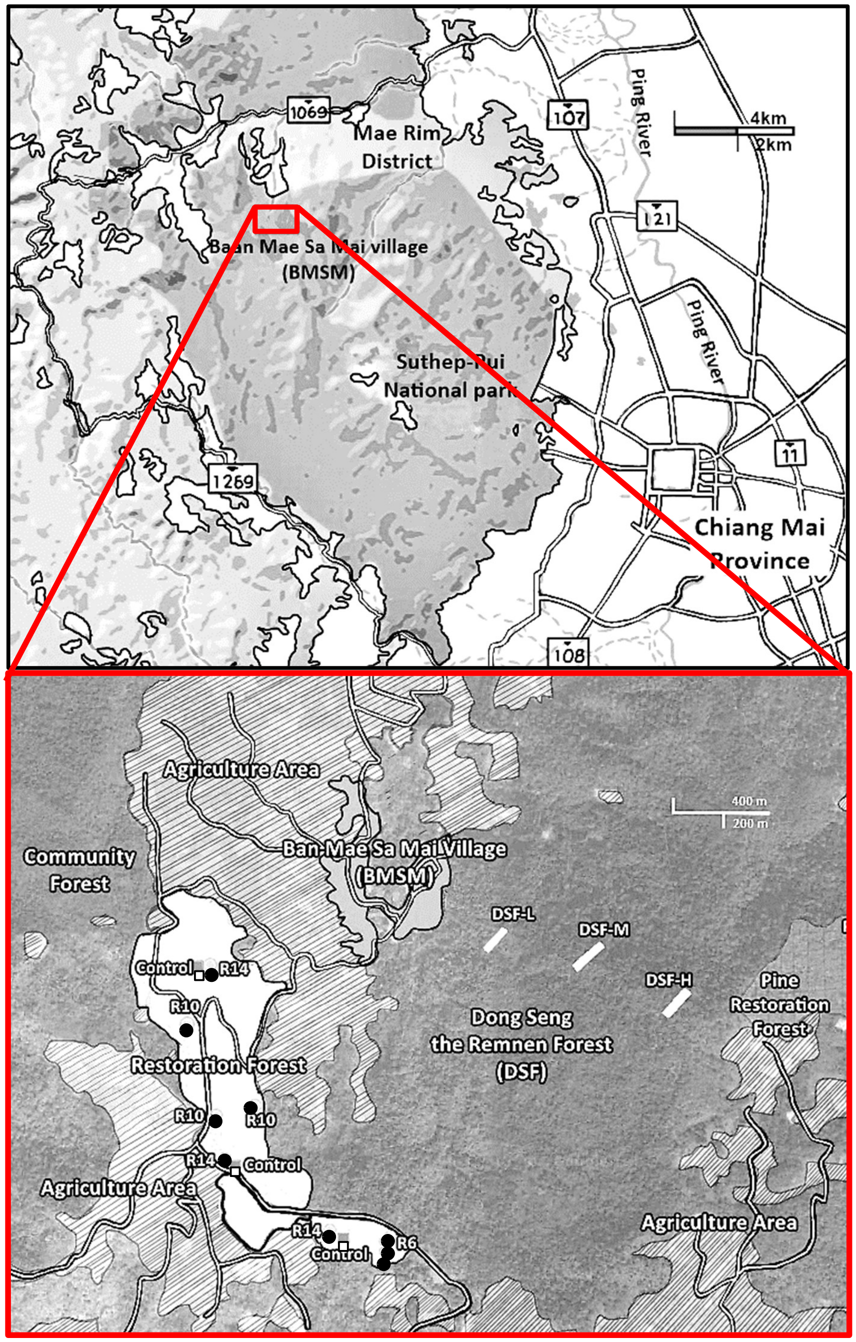


Figure 2.3: a) The study area of FWS restoration forest in Ban Mae Sa Mai village, Mae Rim district, Chiang Mai, Thailand, and b) Sample plots location in FWS restoration forest (●), control plot (□), and natural forest (DSF, ▭)
(Map: Google map)

Restoration forest age 14 years (R14)

Restoration forest age 10 years (R10)

Restoration forest age 6 years (R6)

Control site

Figure 2.4: The restoration forests and the control site during the study period

Figure 2.5: The natural remnant forest or Dong Seng forest (DSF) during the study period

CHAPTER 3

Tree Seedling Recruitment and Dynamics

3.1 Introduction

The framework species (FWS) method of forest restoration involves planting saplings of 20-30 framework tree species in deforested sites. Post-planting treatments, such as weeding and fertilizing, continues for about 2-3 years in order to reduce resource competition. Forest recovery then relies on the planted trees to attract seed-dispersing animals and create conditions, suitable for seed germination and early seedling establishment of incoming **'recruit'** tree species (i.e. a species that was not among those planted species) to accelerate forest succession process and recover biodiversity and ecological functioning (FORRU, 2008). The technique accelerates biodiversity recovery. For example, in North Queensland, 72 plant species, across all growth forms, had returned to framework species plots within 7 years (Tucker & Murphy, 1997), whilst in northern Thailand, species richness of forest tree seedlings taller than 50 cm increased from about 20-30 species to 84 species in FWS forests age 10 years (Sinhaseni, 2008).

In natural secondary forest, the tree seedling community, composition and dynamics define the progress of forest succession. In early succession, species diversity is low, but it increases with time (Denslow & Guzman, 2000; Capers *et al.,* 2005). Ribbens *et al.,* (1994) suggested that the seedling recruitment rate determines the species composition and dynamics of the tree seedling community, as succession progresses. Furthermore, studies of seedling diversity and species dominance could increase our understanding of the conditions that favour or limit seedling recruitment (Martinez-Ramos & Soto-Castro, 1993). Forest restoration seeks to enhance and direct forest succession to achieve a predetermined "target" forest ecosystem, defined by its biomass, structure biodiversity and ecological functioning (FORRU, 2005; Walker *et al.*, 2007; De Jong, 2010). Many studies of plantations and forest restoration approaches have focused on the re-establishment of forest structure and the diversity of trees and saplings, which vary according to the restoration method applied (Yirdaw, 2002**;** Lemenih *et al.,* 2004) and age (De Souza & Batista, 2004). Therefore, this chapter focuses on the seedling community of forests that are undergoing restoration by the framework species method and reports on the recruitment rate and dynamics of native seedlings therein. This information could help forest ecologists predict, understand and enhance successional processes within FWS forests.

3.2 Literature Review

3.2.1 Forest Regeneration and the Seedling Community

Natural forest succession is defined by recovery of forest structure and biodiversity over time (Lugo, 1992; Lamb *et al.*, 2005) and can happen rapidly in tropical climates. For example, in Borneo, sapling recruitment in secondary forest returns to levels similar to those of primary forest with 13 years (Bischoff *et al.*, 2005), whilst in Northern Thailand, plant diversity recovers rapidly during secondary succession on former slash and burn sites within 20 years although dominant species depended on land use history. In contrast forest recovery can be slow especially on isolated islands. For example, Chua *et al.* (2013) reported low natural forest recovery in a 56-year-old secondary tropical forest on Singapore by. The author suggested that limited seed dispersal and the presence of long-lived pioneer species were the reasons.

Most tropical forest tree species are dispersed by animals. Thus, dispersal limitation strongly influence which tree species do or do not arrive at any particular site and how quickly succession progresses (Corlett & Hau, 2000). The proportion of rare trees species that are large-seeded and animal-dispersed increases during late-succession (Caper *et al.* 2005). Asanok *et al.* (2013) reported that large seeded species are more limited in their recruitment in enrichment plantations and secondary forests, compared with primary forest of northern Thailand (seed dispersal will be addressed again in Chapter 4).

The seedling is a critical stage in a plant’s life cycle. They are small and are highly susceptible to adverse environmental factors and commonly have high mortality rates (Silvertown & Charlesworth, 2009; Eriksson & Ehrlén, 2008). Capers *et al.,* (2005) suggested that seedling recruitment and mortality rates are important indicators that determine the progress of natural succession in regenerating forests. Bischoff *et al.* (2005) investigated sapling composition and dynamics during forest succession in secondary dipterocarp rain forests, aged 8 and 13 years, and compared it with primary forest in Danum Valley, Malaysia. Sapling mortality and growth rates (>3.2 cm-DBH) were higher (*ca*. 60% and 37%, respectively) in secondary forest than in primary forest, although sapling recruitment rates in both forests were not significantly different. Moreover, increased species diversity is brought about through increased mortality of fast-growing pioneer species, in mid-successional tropical forest (Bischoff *et al.* 2005; Lasky *et al*., 2014) and their subsequent replacement by a diverse range of shade tolerant slower growing species.

3.2.2 Restored Forest and Seedling Community

Tree species recruitment is one of the most important indicators of restoration success (Tucker & Murphy, 1997; Capers *et al*., 2005; Singhaseni, 2008; McNamara, 2009; Asanok *et al*., 2013). Planted trees can grow and rapidly close canopy (recapture the site), creating a forest habitat in a similar way that naturally regenerating forest does (Koonkhunthod *et al.,* 2007; Asanok *et al*., 2013). In Australian rainforest, the FWS method of restoration has been extensively researched (Goosem &Tucker, 2003; Tucker & Murphy, 1997; Kanowski *et al*., 2003; Tucker & Simmons, 2009; Florentine *et al*., 2016). Tucker & Murphy (1997) reported increasing plant diversity (across all life form) with increasing restoration age. Younger forests support fewer plant species than older forests. White, *et al*. (2004) reported seed source distance to restoration sites as a factor which explained slower species recruitment at isolated sites, compared with adjacent sites. Recently, Florentine *et al*. (2016) studied natural recruitment of native plant species in a plantation and 3 FWS restoration forests of Queensland. They found greatest diversity and abundance of recruits in site with canopy cover was greatest and less grass cover.

In seasonally dry tropical forests of Lao and Vietnam, McNamara (2009) studied plant seedlings in restoration forest of 16 species enrichment plantings (4-9 years) compared to plantation of monocultures and mixture of native and exotic species (*Eucalyptus camaldulensis* and *Tectona grandis*) (7-10 years) and natural secondary forests. The researcher found survival rates of seedlings in restoration forest of enrichment was approximately 50%, and diameter increments of less than 0.2 cm per year after 6 to 9 years of growth. In addition, growth of pioneer species limited by light competition young age of secondary forest in Vietnam (Van Kuijk *et al.*, 2008).

In FWS restoration forest at BMSM (where this research was established) a lot of research about biodiversity recovery has been conducted in the young age restoration. Khopai *et al.* (2003) studied effects of forest restoration activities on recruitment of non-planted tree species. They found species richness and evenness of the ground flora increasing in the first year of planting before declined in the second year after closing of the forest canopy which reduced opportunities for establishment. Later study of seedlings in 2, 4 and 8 year-old FWS forests, Sinhaseni, (2008) reported the increasing tree seedling species recruitment with age. Seventy-two species were found in 8 year-old forest, and tree species diversity increased with plot age. Many species were attracted by planted FWS and recruited as seedlings beneath the restored forests. Later, a study of seedlings under 10 species of FWS tree in 8 year-old restored forest found a total of 26 recruited species (233 seedlings). Seedling density, species richness and growth rate were highest under *Prunus cerasoides* crowns (Wydhayagarn *et al.*, 2009).

Many studies in restoration forests have focused on tree species richness and planted tree structure. Information on tree seedling recovery is still lacking, especially seedling recruitment and dynamics over time, during the now commonly adopted FWS method of forest restoration. The crux of the FWS method is that it should promote seedling species recruitment, leading to increased species richness and forest structure, similar to that of old growth forest and subsequent biodiversity recovery. Therefore, studies of tree seedling dynamics, recruitment rates and community turnover are essential to determine the efficacy of the technique and how to modify it to achieve the best results.

3.3 Objectives, Questions and Hypotheses of the Chapter

3.3.1 Objectives

The study of this chapter aims to;

1) determine which tree species establish or fail to establish in restored plots, from the pool of seed trees present in nearby remnant forest and why.
2) compare the diversity and dynamics of the recruit tree seedling communities establishing in forest restoration plots of various ages and how they compare with those of natural remnant forest and non-planted control sites.

3.3.2 Questions

1) Which tree seedling species succeed/ fail to establish in restoration plots and why?
2) How fast and to what extent does species recovery of the tree seedling community in restoration forest plots occur, compared with remnant forest and the control sites?
3) How does age of restoration influence seedling recruitment and seedling dynamics?

3.3.3 Hypotheses

1) Seed tree species that are common in the landscape surrounding restoration sites become more highly represented among the recruit tree seedling community beneath the canopy of restoration forests than do those that are rarer.
2) The principles of FWS technique are that planted trees attract seed-dispersers into restoration plots and create the conditions amenable to natural tree seedling establishment. The development of such amenable conditions for germination takes time, so older plots should provide better conditions for tree seed germination and seedling establishment than younger plots do. Therefore:-

    a. Seedling recruitment is higher in older plots than it is in younger plots and the seedling population density of recruit species increase with the increasing planted age.
    b. As the number of species already present in the community approaches that of the species richness of natural remnant forest, the rate of species recruitment gradually decline with increasing plot age.
    c. Overall, seedling population density declines with increasing plot age, as the canopy of the restored forest becomes dense and shades out young seedlings.
    d. Species diversity of seedlings community increases with plot age, trending towards a natural remnant forest.
    e. The species composition of the tree seedling community becomes more similar to that of natural remnant forest and less similar to that of the control plots.
3) Age of restoration plots should also influence seedling dynamics mostly due to canopy closure. Therefore;
    a. Seedling growth rates increase with increasing plot age as closer canopy closure optimizes harsh environment for seedlings.
    b. Seedling mortality rates decline with increasing as habitat conditions for seedling survival improve.
    c. Consequently seedling turnover rates gradually decline with increasing plot age.

3.4 Methods

3.4.1 Data Collection

Tree seedling data were collected in circular (5 m-radius) sample plots, 12 plots in each of the following three restoration (R) forests (R14, R10, and R6), Control and DSF sites (details in Chapter 2). Totally 5 x 12 = 60 plots as close as possible to previously located plots (FORRU, 2005), but at least 5 m in from the edge of the plots. For the purposes of this study, a tree seedling was defined as a woody stem ≥ 5 cm-tall (or seedling with 2 true leaves) to ≤150 cm-tall (excluding lianas, herbs and shrubs). Every tree seedling was tagged with a metal label, identified to species level (following the species names of Maxwell & Elliott, 2001) and its height (cm) and root collar diameter (RCD, mm) measured.

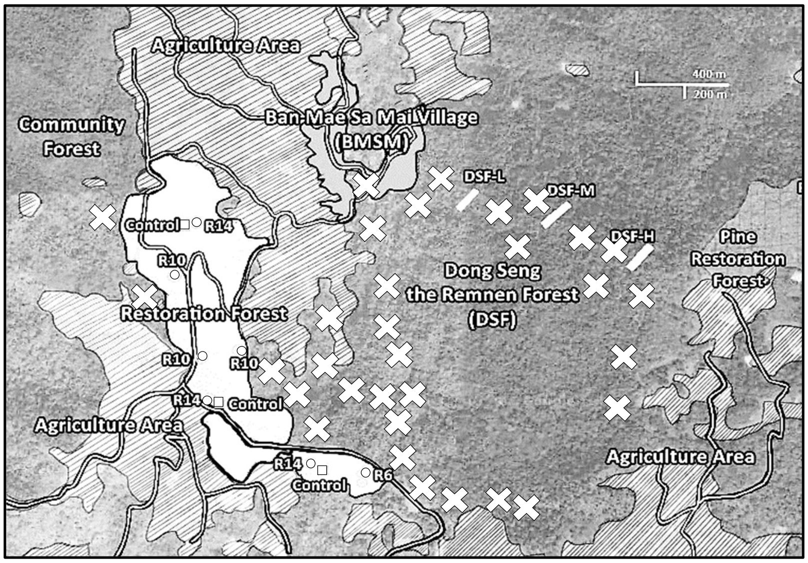


Figure 3.1: Belt-transects (100 × 10 m) location for adult trees (≥10 cm-DBH) surveyed outside study plots (✕)

Adult trees (≥10 cm-DBH) outside study plots were surveyed, counted and identified for species in 100 × 10 m belt-transects. Sixteen belt-transects were established in the DSF forest and 16 outside the R forests. Data collection was carried out at the end of the wet season in September to October and dry season in March to April, over 4 seasons (2 years) from September, 2012 to April, 2014. Please note that in the dry season of 2014 (the 4th census), 4 sample plots in the natural remnant forest were burnt by forest fire. Data from the burnt plots were excluded from the analysis of seedling dynamics.

3.4.2 Data Analysis

The species of tree seedlings establishing inside the restoration plots were compared with those represented by adult trees in the nearby old-growth forest and in the belt transects closer to the forest restoration plots. Species were then assigned to one of the following categories 1) recorded in the seed tree survey and present in the seedling community (i.e. both effective seed dispersal and good conditions for seedling establishment); 2) recorded present in the seed tree survey but not found in the seedling survey (i.e. seed dispersal ineffective and/or poor conditions for seedling establishment) 3) not recorded in the adult seed tree survey but present as seedling(s) in the restoration plots (i.e. either missed in the adult tree survey or seeds are capable of long-distance dispersal). Seedlings of planted species were excluded from this analysis. The population density of adult tree (stems per 100 m^2) was calculated for each species, to compare the mean population density of adult trees between the recruited species and the absent species.

In the seedling communities of the R forests and the control plots, species were classified as follows:- 1) those of planted framework species(i.e. not incoming as recruits) , 2) those of existing adult tree species that were mature and present before restoration was initiated (i.e. also by definition not incoming recruits) and 3) those of native tree species that grow only *outside* the restoration plots (i.e. true recruit species coming in as seeds from outside). The seedling species in DSF were obviously neither planted nor recruits (in the above sense) , so the DSF seedling community acted as the target seedling community (in terms of density, species richness and species composition) against which the seedlings communities of the R forests could be compared.

Stem density (stems per 100 m^2) and species richness were calculated, as well as mean seedling RCD (root collar diameter) for comparison among sites. Seedling height distributions were used, to compare seedling age-structure among the sites. Species richness, diversity index (Shannon diversity index) and plant community evenness (Buzzas and Gibson's evenness) were calculated for the all seedlings and for subset of recruited seedling species alone of each forest site**.** Rarefaction curves were used to examine the distribution of species and species richness in each forest. Cord Distance index (CRD) was used as an index of similarity among the communities of the various sites (Ludwig & Reynolds, 1988)

Annual relative growth rates (RGR, % per year) were calculated using root collar diameter data (RRGR), and seedling height data (RHGR) (equation (1) adapted from Hunt 1982; Welden *et al.*, 1991; Gomes *et al.*, 2003). Mean values were then compared among sites.

Equation (1): Relative Growth Rate (RGR) $RGR = \frac{\ln size_1 - \ln size_0}{(t_1 - t_0)} \times 360 \times 100$

Where ln is natural logarithm, size_0 and size_1 are size of seedling root collar diameter (mm) or height (cm) at time 0 census and at time 1, the next census, and Δt is time of census interval (in days).

Tree seedling species and individuals were listed as recruit species or dead seedlings in each census. Thus, in each forest sites, total dead seedlings were calculated for mean annual mortality rate (M) by equation 2 (Sheil *et al.*, 1995; Gomes *et al.*, 2003).

Equation 2: Mortality Rate (M) $M = \left\{1 - \left[\frac{N_0 - m}{N_0}\right]^{1/\Delta t}\right\} \times 100$

When; n_0 is the number of stems or species in the original population, "m" is the number of deaths among the initial population, Δt is time of the collection interval $(t_1 - t_0)$.

The mean annual rate of tree seedling recruitment (both individuals and species) (R) was calculated over 4 censuses by equation 3 (Sheil *et al.,* 1995; Sheil and May, 1996; Gomes *et al.,* 2003).

Equation 3: Recruitment Rate (R) $R = \left\{\left[\frac{N_0 + r}{N_0}\right]^{1/\Delta t} - 1\right\} \times 100$

When; n_0 is the number of stems or species in the original population, "r" is the number of recruited individuals or species, Δt is time of the collection interval $(t_1 - t_0)$.

The annual turnover rate of the seedling community was calculated from mean annual mortality and recruitment rates using equation 4 (Laurance *et al.,* 2009) to compare changes in the seedling community over the study period at each forest site.

Equation 4: Turnover Rate (T) $Turnover\ rate = \frac{[M+R]}{2}$

When; M is the annual mortality rate and R is the annual recruitment rate in the community**.**

3.4.3 Statistical Tests

Data sets of seedling composition and dynamics were compared among the R forest plots of different ages, the control plots and the DSF plots. Chi square (χ^2) compared among abundance and rare of adult trees at 95 % confident, with df =1, P-values ≤ 0.05 when $\chi^2 \geq 3.841$. Student's t-test was used if the data followed a normal distribution or Mann-Whitney test if the data followed a non-normal distribution. Similarly data sets were compared among seasons. Paired t-tests were applied to compare parametric data, whereas the Wilcoxon test was applied to analyze non-parametric data. Species rarefaction curves were constructed to compare seedling species richness and accumulation among the forest sites (Hurlbert, 1971; Krebs, 1989). A Principle Component Analysis (PCA) was performed to evaluate correlations between variables of R forests. The software "Paleontological Statistics, PAST, version 3.0" was used to perform the statistical tests (Hammer *et al.,* 2001; Hammer and Harper, 2006).

3.5 Results

3.5.1 Absent Tree Species

A total of 208 adult tree species (≥10 cm-DBH) were recorded in the belt transect surveys in DSF forest and nearby the restoration plots, representing 69 families. Of these, 101 species were not recorded in circular plot of the seedling community in the restoration (R) forest plots (49 families, or 49% of total adult species recorded (Fig.3.2). Conversely, seedlings of 107 species (or 51% of total adult species) successfully established, and were recorded in the R plot seedling community survey. More than half of them (59, across all sample plots and censuses, or 28% of total adult species) were genuine recruit native species (i.e. of non-planted species having been brought into the plots from outside), although 16 of them could not be identified (most of the latter were very rare or had no leaves for identification). Thirty-five seedling species established in the plots were of species that had been planted, whilst 13 (17% of total species) were derived from mature, non-planted, seed trees that had initially existed in the R forests since before restoration commenced (Fig.3.2).

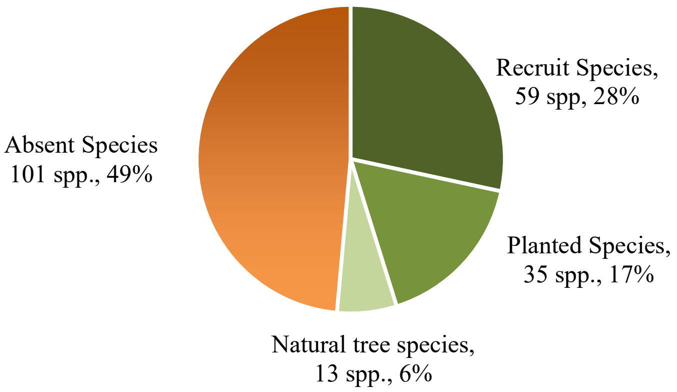


Figure 3.2: Amount and percentage of tree species recorded in the adult tree survey that were recorded/not recorded in the R forest seedling community survey (data combined across all plot ages and censuses)

One of the reasons why species failed to establish in the R plots may have been because they were rarer, as adult seed trees, in the surrounding landscape. Consider species that absent from R plots, percentage of rare adult trees species was significantly higher than those recruited in the R forests (χ^2 = 16.56 by Chi square in Fig. 3.3-a). The mean population density of adult trees of species that successfully recruited into the seedling communities of the R plots was 31.4 trees per 1,000 m^2, where as that of absent species was only 9.8 trees per 1,000 m^2 (Fig. 3.3-b, see hypothesis 1). In addition, 5 of all absent species (including *Pinus kesiya*) were found over 30 individuals (ca.1.88 trees per 1,000 m^2), whilst 57 species were found less than 10 individuals or >0.625 trees per 1,000 m^2.

Examples of species that were absent from the R forests seedling community surveys and rare in the surrounding landscape include: *Anogeissus acuminata*, *Broussonetia papyrifera*, and *Celtis tetrandra etc*. Furthermore, some of these rare species could not be found as seedlings in the DSF sample plots as well, indicating that both seed dispersal and seedling establishment are limited. Examples of recruit seedling species that were common as adult seed trees in the surrounding landscape included: *Litsea salicifolia*, *Engelhardia spicata* var. *integra* and *Turpinia pomifera* var. *pomifera*.

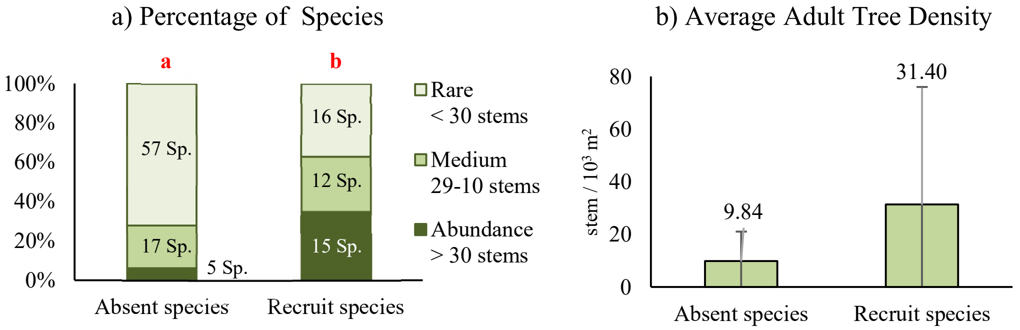


Figure 3.3: a) Percent of abundant, medium abundance and rare species and b) Mean adult tree density (stems per 1,000 m^2) between absent species and recruit species of seedling communities of the R forests. Different superscripts designate a significant difference between absent and recruit species by Chi square (χ^2) test.

3.5.2 Tree Seedling and Species Recruitment Rate in Restoration Forests

Seedling density and species recruitment rates did not decline with significantly with increasing plot age in the R forests (Fig. 3.4). In fact, the density of recruit seedlings in the older plots (R14 and R10) was significantly higher than in the youngest plots (R6) (P = 0.0004 and 0.0002, age respectively, by Mann-Whitney test in Fig. 3.4-a). Although, seedling recruitment rates did not differ significantly among the R plot ages (P > 0.05 by Mann-Whitney test in Fig. 3.4-b), this result shows slightly lower rate of those seedling in old age forests, R14 and R10, than site R6 (see hypothesis 2-a). Seedlings of control site had significantly higher recruitment rate than all R forests (P < 0.05 by Mann-Whitney test in Fig. 3.4-b) but amount of recruit seedlings was similar to the R14 and R10 forests (Fig. 3.4-a).

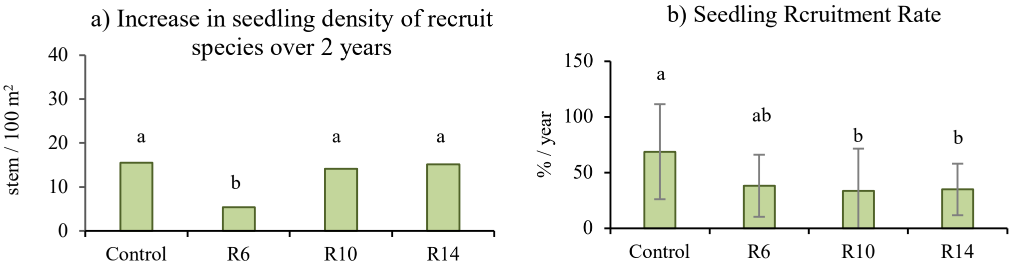


Figure 3.4: a) Seedling density of recruit species (stems per 100 m^2), b) Mean annual individual recruitment rate (% per year averaged over 2 years) in R forests and control sites. Columns that do not share the same superscripts are significantly different $P < 0.05$ (by Mann-Whitney test).

Throughout 4 censuses, the species recruitment rate was not related to plot age. The R10 plots accumulated more recruit species than both the R14 and R6 plots did (Fig 3.5, see hypothesis 2-b). There were only 3 and 5 new species recruited as seedlings in R6 and R14 sites respectively, within 1.5 years after first census (Fig 3.5-a). This result was leading to low species recruitment rates of those forests (9 and 13 % per year, R6 and R14 sites respectively in Fig 3.5-b). Whilst, site R10 attracted higher new species (15 species within 1.5 years) and had higher species recruitment rate, 18 % per year than other forests (Fig 3.5-a, and b).

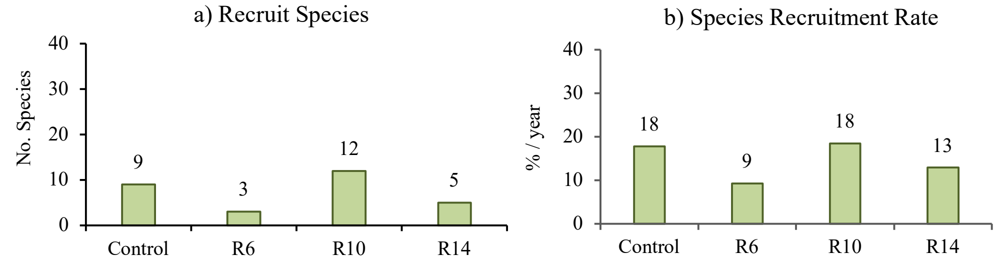


Figure 3.5: a) the numbers of recruit species (species per 12 plots) establishing as seedlings in the R plots and controls from 2012 to 2013 and b) species recruitment rate from initial census over the study period (% per year) in R forest and control site.

Some of the recruit species in the R plots were common species, found in the seedling community of DSF site, for example, *Lithocarpus polystachyus* in R14 site and *Actinoaphne henryi* in R10 and R6 sites (Table 3.1). Some of the recruit seedling species recorded in the R plots were absent from the DSF seedling community e.g. *Neolitsea zeylanica* and *Garuga pinnata* recorded in the R10 plots. Moreover, *Micromelum integerrimum var. intergerrimum* established abundantly in all the R forests and in the control site. In the R10 and R14 plots, four and five native species, sites respectively, were represented by more than 30 individuals (such as; *Micromelum integerrimum var. intergerrimum, Litsea salicifolia, Antidesma bunius, and Aporosa octandra var. octandra* etc.). Whilst in the R6 plots, most recruit species were rare. Only one species, *Artocarpus nitidus,* was represented by more than 30 individuals in this area (Table 3.1).

Table 3.1: The number of seedlings and percentage from total seedlings of ten most abundant recruit species in R forests and control site (data from 12 plots (total 942.3 m^3-area) at each forest site)

Rank	Control site			R6			R10			R14		
	Species	**No. stem**	**%**	**Species**	**No. stem**	**%**	**Species**	**No. stem**	**%**	**Species**	**No. stem**	**%**
1	*Micromelum integerrimum var. intergerrimum*	71	21.8	*Artocarpus nitidus*	45	0.8	*Micromelum integerrimum var. intergerrimum*	137	0.8	*Micromelum integerrimum var. intergerrimum*	103	1.0
2	*Archidendron clypearia ssp. clypearia var. clypearia*	35	8.7	*Litsea salicifolia*	19	0.7	*Antidesma bunius*	94	0.0	*Litsea salicifolia*	84	0.8
3	*Phyllanthus emblica.*	26	4.3	*Stereospermum colais*	10	0.6	*Litsea salicifolia*	86	0.0	*Aporosa octandra var. octandra*	61	0.6
4	*Aporosa octandra. var. octandra*	21	2.6	*Cinnamomum caudatum*	6	0.1	*Cinnamomum caudatum*	38	0.6	*Cinnamomum caudatum*	35	0.5
5	*Antidesma bunius*	16	2.0	*Engelhardia spicata var.integra*	6	0.1	*Wendlandia tinctoria ssp. tinctoria*	16	0.3	*Artocarpus nitidus*	30	0.3
6	*Olea salicifolia*	12	2.0	*Glochidion hongkongense*	6	0.1	*Stereospermum colais*	12	0.3	*Dalbergia cultrata*	16	0.3
7	*Litsea salicifolia*	7	1.7	*Micromelum integerrimum. var. intergerrimum*	5	0.1	*Sterculia lanceolata*	5	0.0	*Phyllanthus emblica.*	14	0.2
8	*Dalbergia cultrata*	6	1.5	*Alangium kurzii Craib var. kurzii*	4	0.1	*Actinodaphne henryi*	4	0.0	*Aporosa villosa*	8	0.2
9	*Sterculia lanceolata*	5	1.3	*Antidesma bunius*	4	0.0	*Olea salicifolia*	4	0.0	*Mallotus philippensis*	8	0.1
10	*Lithocarpus polystachyus*	4	0.9	*Aporosa octandra var. octandra*	3	0.0	*Alangium kurzii Craib var. kurzii*	3	0.0	*Litsea glutinosa var. glutinosa*	5	0.1

3.5.3 The Seedling Community in Restoration Forests

More than ten thousand tree seedlings were recorded in the sample plots across 5 sample sites over 4 censuses. Seedling population density was not consistently related to plot age (see hypothesis 2-c). The oldest (R14) and youngest (R6) age plots supported significantly higher total seedling densities (including both planted FWS and native recruit species) than the medium aged plots (R10). Seedling density of site R10 was comparable to that in the remnant forest, DSF (Fig. 3.6).

Most of seedlings in the R forests belonged to planted species and the super-abundant species, such as *Castanopsis calathiformis* in the R14 plots, a native planted tree, which produced dense seedling carpets beneath their crowns (*ca.* >1,000 stems per 100 m^2), and small seedlings of *Litsea monopetala* of site R6 (*ca.* 700 stems per 100 m^2 in Fig. 3.6). When the super-abundant species are removed from the data, seedling density was not much different among R forests with 274, 274, and 305 stems per 100 m^2, in R6, R10 and R14, respectively (Fig. 3.6). Moreover, these densities were similar to those of the DSF seedling community (263 stems per 100 m^2). As expected, the seedling communities of all the R forest plots were significantly denser than that of the control plots, undergoing natural regeneration (about 95 stems per 100 m^2, P <0.005 by Mann-Whitney test in Fig. 3.6).

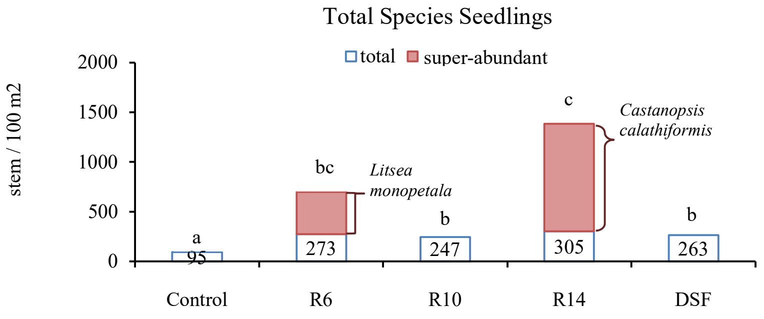


Figure 3.6: Seedling density (stems per 100 m^2) with super-abundant species in R forest and control site. Columns not sharing the same superscripts are significantly different P<0.05 (by Mann-Whitney test).

Over the study period, 24 species were found in all sample plots at all forest sites, 12 of which were found in all censuses (Table 3.2). Five species recruited abundantly in the R forests, across all 3 planted ages; 1) *Litsea salicifolia* (Roxburgh ex Nees) Hook. f., 2) *Artocarpus lacucha* Buch.-Ham. ex D.Don, 3) *Antidesma bunius* (L.) Spreng., 4) *Aporosa octandra* (B.-H. ex D. Don) Vick. var. *octandra* and 5) *Micromelum integerrimum* B.-H. ex DC.) Wight & Arn. ex M. Roem. var. *intergerrimum* (Tables 3.2 and 3.3, see pictures in **Appendix F**). Moreover, many of the common species were planted framework species such as *Eugenia fruticosum DC., Phoebe lanceolata (Wall. ex Nees) Nees,* and *Prunus cerasoides* B.-H.ex D.Don, etc., and natural mother trees in the study plots such as *Litsea monopetala Roxb.) Pers.,* and *Schima wallichii* etc.

Table 3.2: Common seedling species (with over 30 individuals) found in all forest sites, (in alphabetical order). Asterisks (*) indicate recruit species whist most species are planted framework tree species.

Found in all censuses	Found in only some censuses
*Antidesma bunius**	*Archidendron clypearia**
*Aporosa octandra**	*Artocarpus nitidus**
*Artocarpus lacucha**	*Bridelia glauca*
Castanopsis diversifolia	*Eugenia albiflora*
Cinnamomum caudatum *	*Heynea trijuga*
Eugenia fruticosum	*Magnolia baillonii*
Litsea monopetala	*Markhamia stipulata*
Litsea salicifolia *	*Micromelum integerrimum* *
Machilus gamblei	*Olea salicifolia**
Phoebe lanceolata	*Pterocarpus macrocarpus*
Prunus cerasoides	*Sterculia lanceolata**
Schima wallichii	*Stereospermum colais**

The most abundant seedling species in the R plots were of the planted species, *Castanopsis calathiformis* (Skan) Rehder & E.H.Wilson. It was by far the most abundant in the R14 plots (where it had been planted as a candidate framework species in 1998) with over 6,000 individuals (mostly in close proximity to mother trees) or *ca*. 80% of the total seedlings recorded over all censuses (Table 3.3). No seedlings of the species were recorded in the R6 plots, about 100 m away from the R14 plots. Five other species also established abundantly, with more than 100 individuals being recorded in the seedling survey. *Litsea monopetala*, was the most abundant species in the R6, R10 and Control plots, whilst *Helicia nilagirica* and *C. tribuloides* were the most abundant species in the DSF plots. However, the *H. nilagirica* population crashed, as a result of a fire shortly before the 4th census. (Table 3.3).

Table 3.3: The number of seedlings and percentage of total seedlings of ten most abundant species in R forests, control and DSF site (data from 12 plots (total 942.3 m^3-area) at each forest site)

Rank	Control site			R6			R10			R14			DSF		
	Species	**No. stem**	**%**	**Species**	**No. stem**	**%**	**Species**	**No. stem**	**%**	**Species**	**No. stem**	**%**	**Species**	**No. stem**	**%**
1	*Litsea monopetala*	284	35.9	*Litsea monopetala*	4027	75.4	*Litsea monopetala*	1073	53.2	*Castanopsis calathiformis*	6326	78.3	*Helicia nilagirica*	384	16.5
2	*Castanopsis calathiformis*	104	13.1	*Bauhinia variegata Linn.*	909	17.0	*Micromelum integerrimum*	138	6.9	*Heynea trijuga*	522	6.5	*Castanopsis calathiformis*	229	9.9
3	*Micromelum integerrimum*	71	8.9	*Phoebe lanceolata*	53	1.0	*Phoebe lanceolata*	123	6.1	*Litsea monopetala*	238	2.9	*Lithocarpus polystachyus*	194	8.4
4	*Phoebe lanceolata*	39	4.9	*Artocarpus lacucha*	49	0.9	*Antidesma bunius*	94	4.7	*Phoebe lanceolata*	226	2.8	*Castanopsis tribuloides*	188	8.1
5	*Archidendron clypearia*	35	4.4	*Artocarpus nitidus*	45	0.8	*Litsea salicifolia*	86	4.3	*Erythrina subumbrans*	106	1.3	*Castanopsis acuminatissima*	180	7.8
6	*Schima wallichii*	32	4.0	*Markhamia stipulata*	38	0.7	*Prunus cerasoides*	74	3.7	*Micromelum integerrimum*	103	1.3	*Phoebe lanceolata*	168	7.2
7	*Phyllanthus emblica*	26	3.3	*Prunus cerasoides*	37	0.7	*Machilus gamblei*	57	2.8	*Litsea salicifolia*	84	1.0	*Machilus gamblei*	123	5.3
8	*Aporosa octandra var. octandra*	21	2.6	*Castanopsis diversifolia*	28	0.5	*Bauhinia variegata*	53	2.6	*Aporosa octandra var. octandra*	60	0.7	*Eugenia albiflora*	121	5.2
9	*Antidesma bunius*	16	2.0	*Litsea salicifolia*	19	0.4	*Artocarpus lacucha*	48	2.4	*Prunus cerasoides*	43	0.5	*Styrax benzoies*	98	4.2
10	*Heynea trijuga*	16	2.0	*Heynea trijuga*	11	0.2	*Cinnamomum caudatum*	38	1.9	*Eugenia fruticosa*	42	0.5	*Aporosa octandra var. octandra*	68	2.9

3.5.4 Species Diversity of Seedlings

The species diversity of the seedling community under the R forests was considered as a) total species (planted and recruited species) and b) recruit species. The diversity of both total species and recruit species was not related to plot age (Shannon diversity index, Fig. 3.7-a). Diversity indices (including all categories of seedlings) in the R forest plots were generally low, compared with those of DSF site, especially in R6 and R14 (arrows in Fig. 3.7-a). Low species evenness of total seedlings in R6 and R14 forests was due to a few super-abundant species (Fig. 3.8-a).

Considering just the recruit native species, diversity of all R forests was higher than those diversity of overall species and approached that of DSF forest (in Fig.3.7-b, see hypothesis 2-d). Species evenness of these species was also higher than those of overall species in all 3 of the R forests (Fig. 3.8-a, and -b). In control site, the diversity of overall seedling community were high, compared with that in the R forests (Fig. 3.7-a). However, species diversity and evenness of recruit species in the control site were slightly lower those in the R forests (Fig. 3.8-b and 3.8-b).

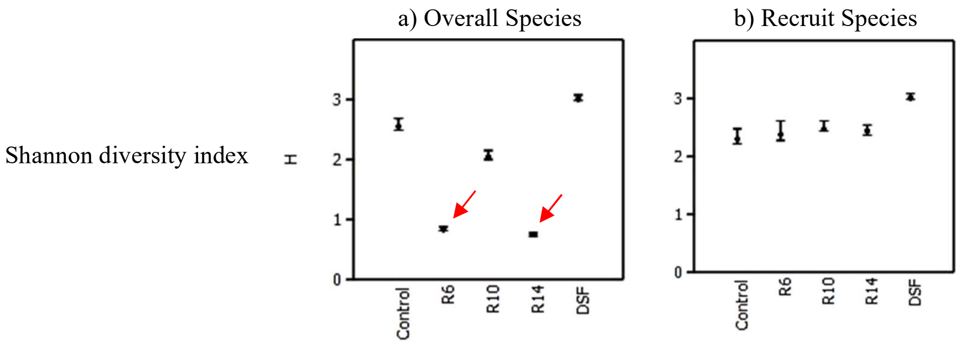
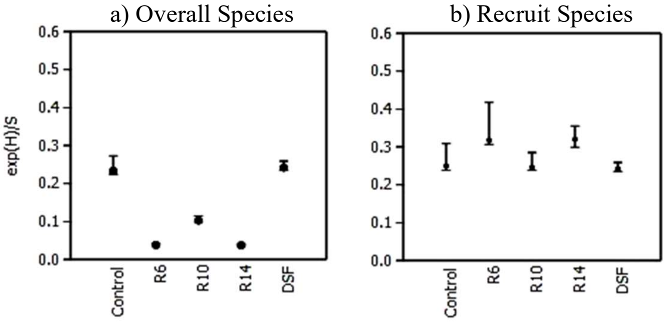


Figure 3.7: Shannon diversity index of a) overall species, and b) recruit species seedlings found in each forest site.

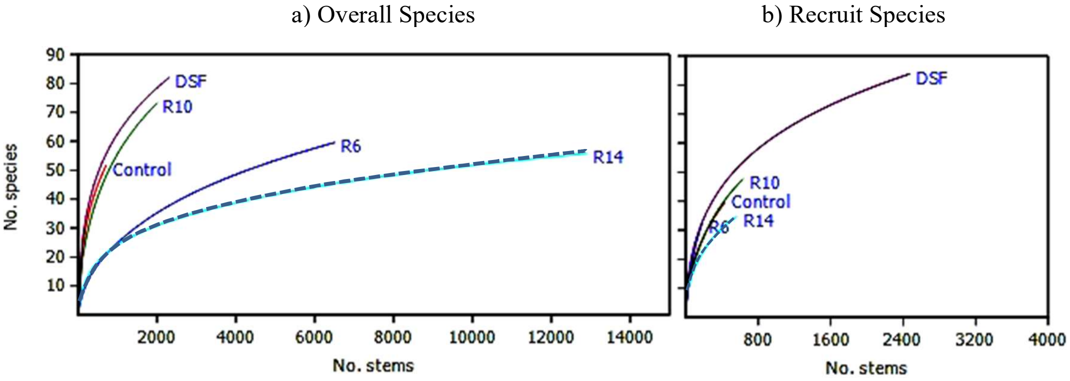
Evenness index


a) Overall Species

b) Recruit Species

Figure 3.8: Evenness index of a) total species, and b) recruit species seedlings found in each forest site.

Considering of both a) overall and b) recruit species, rarefaction curves showed lower species accumulation in seedling community of the R forests compared with the DSF site. Species richness did not appear to be related to plot age (Fig. 3.9 see hypothesis 2-d). Among the R forests, seedling community of R10 forest contained the highest species richness of overall species (Fig. 3.9-a) and recruit species (Fig. 3.9-b). Unexpectedly, the control site collected native species as many as those R14 plots (Fig 3.9). According to the low seedling density, the control plots had high species diversity similar to the R plots.


a) Overall Species

b) Recruit Species

Figure 3.9: Rarefaction curve of a) total species, and b) recruit species in R forests and control site compared to total species of site DSF.

3.7.5 Seedling Community Similarity

The species composition of the overall seedling communities in the R forests differed substantially from that of the DSF site, with Cord Distance Index (CRD) (Table 3.4-a). The seedling community of DSF plots was slightly more similar to older age (R14) plots, than younger age (R10 and R6) forests (CRD = 1.114, 1.337, and 1.372 unit, age respectively in Table 3.4-a). Species composition of R14 sites further away from those of other R forests with CRD > 1.38 unit (Table 3.4-a). Whist, species composition of young age forest, R6 and R10, and control site shown closer CRD (less than 0.5 unit) among them (* in Table 3.4-a).

Considering just the communities of the recruit species, the CRD indicate closer distance of the composition similarity among all 3 R forests, with CRD between 0.67 to 1.17 units, than the distance from the DSF site to those R forests, with CRD ≥1.25 units (Table 3.4-b, see hypothesis 2-e). Furthermore, the communities of R14 site were more similar to that of the R10 site (with CRD = 0.674 units) than other sites (** in Table 3.4-a).

Noticeably, the species composition of the overall seedling communities of the R14 plots was also similar to that of the control plots (index of 1.123 units, in Table 3.4-b). Since the control sites were positioned immediately adjacent to the R14 sites they shared common boundaries. Consequently, a “spillover effect” could be expected, with seedlings establishing in the control plots being derived from seeds dispersed from the adjacent R14 plots (* in Table 3.4).

Table 3.4: Cord Distance Index (CRD) of seedlings among the R forests, control site and DSF site for a) overall species and b) recruit species. Values indicate distance (*d*) of community similarity apart between sites ($0< d >\sqrt{2}$), ** indicates close distance (<0.7) index of similarity.

a) Overall Species

Forest site	R14	R10	R6	Control site
DSF	1.114	1.337	1.372	1.236
R14	-	1.381	1.387	1.123
R10	-	-	0.226 **	0.414 **
R6	-	-	-	0.484 **

b) Recruit Species

Forest site	R14	R10	R6	Control site
DSF	1.253	1.357	1.344	1.245
R14	-	0.674 **	1.009	0.763 *
R10	-	-	1.166	0.772
R6	-	-	-	1.294

3.5.6 Tree Seedling Relative Growth Rate (RGR)

Considering only the recruit native species, mean RGR of root collar diameter (RRGR) and height (RHGR) of seedlings (averaged across all recruit species) were not statistically related to restoration age ($P > 0.1$ by Mann-Whitney test in Fig. 3.10, see hypothesis 3-a). Recruit species seedlings, averaged across all species, in the R14 forests and control site grew taller more rapidly than those in the DSF site ($P = 0.026$ and 0.035, site respectively, in Fig. 3.10-b).

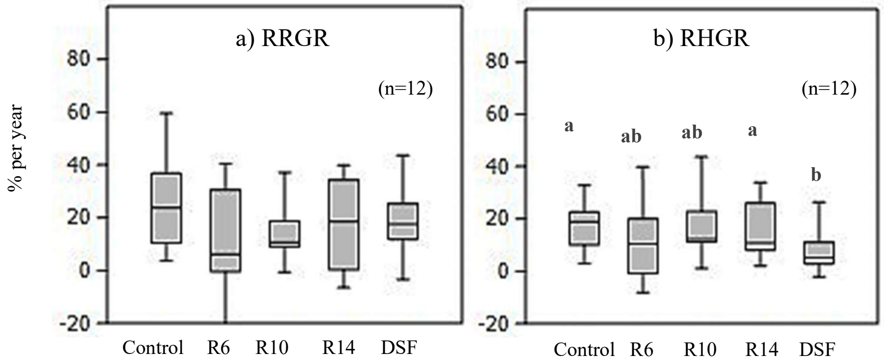


Figure 3.10: Average relative growth rate of a) RCD (RRGR), and b) height (RHGR) of recruit species in R forests and control site compare to DSF site. The same superscripts are significantly different $P < 0.05$ by Mann-Whitney test.

Similarly, mean RGR of overall species, both RRGR and RHGR, appeared unaffected by plot age (Fig. 3.11), but the RHGR of seedlings in the R6 forest plots was significantly lower than that of seedlings in the R10 and R14 plots ($P = 0.0042$ and 0.0086, site respectively, by Mann-Whitney test in Fig. 3.11-b). On average, seedlings in the R forests grew by less than 20% per year (both RCD and height), but the mean rates did not differ significantly from those recorded in the DSF plots (Fig. 3.11). Both height and RCD growth rates of seedlings in the control plots were more rapid than those of seedlings in the R6 plots ($P = 0.025$ and 0.0006, RCD and height respectively, in Fig. 3.11)

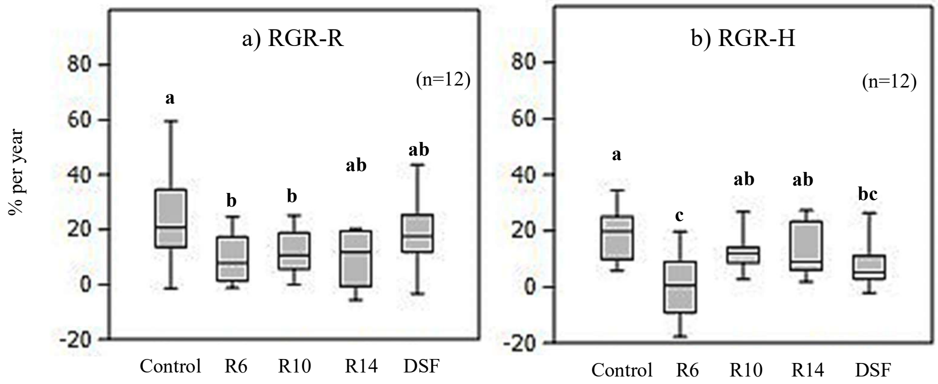


Figure 3.11: Average relative growth rate of a) RCD (RRGR), and b) height (RHGR) of **overall species** in each forest. The same superscripts are significantly different *P*< 0.05 by Mann-Whitney test.

3.5.7 Seedling Mortality Rate

The mean seedling mortality rate of **recruit species** increased with restoration plot age (Fig. 3.12-b). In the R6 plots, the rate was significantly lower than that in the R14 forest (*P* = 0.04 by Mann-Whitney test) with the rate in R10 plots ranked between (Fig. 3.12-b, see hypothesis 3-b). The number of seedlings that died from the 1st to the 4th censuses in R6 forest was distinctively lower than in R10 and R14 forests (*P* = 0.003, and 0.0001, respectively, in Fig. 3.12-a).

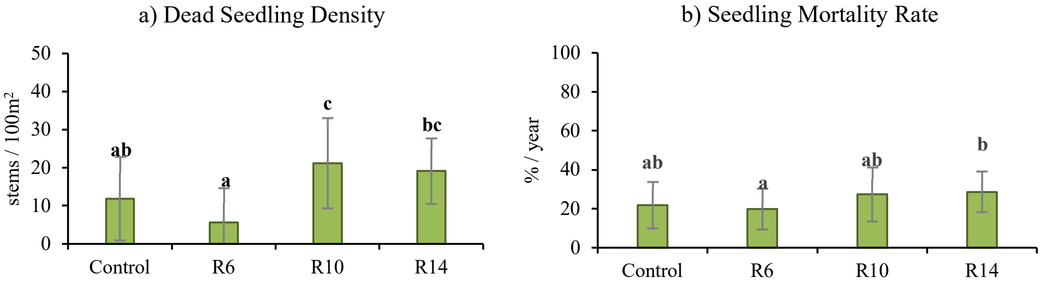


Figure 3.12: Recruit species: a) dead seedling density per year (stems per 100 m^2), and b) annual mortality rate (% per year) of recruit species seedling in each forest site. Different alphabets represent the significant difference among forest sites by Mann-Whitney test.

For **overall species,** mean annual seedling mortality rates decreased with increasing plot age (20±5, 18±7, 16±11 % per year in site R6, R10 and R14, respectively, in

Fig. 3.13-b, see hypothesis 3-b). Seedling mortality rates in all R forests were not significantly different from those in the DSF and control plots (p >0.05 by Mann-Whitney test in Fig. 3.13-b). Although the number of seedlings in site R14 that died from 1st to 4th censuses was very high, compared with the DSF and control plots (Fig. 3.13-a), the mortality rate was low and not significantly from those sites (Fig. 3.13-b).

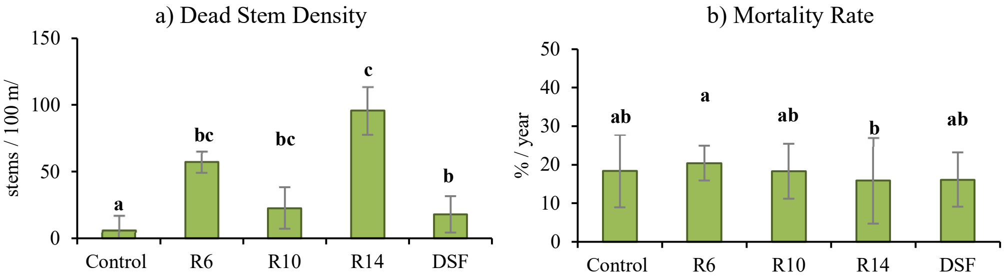
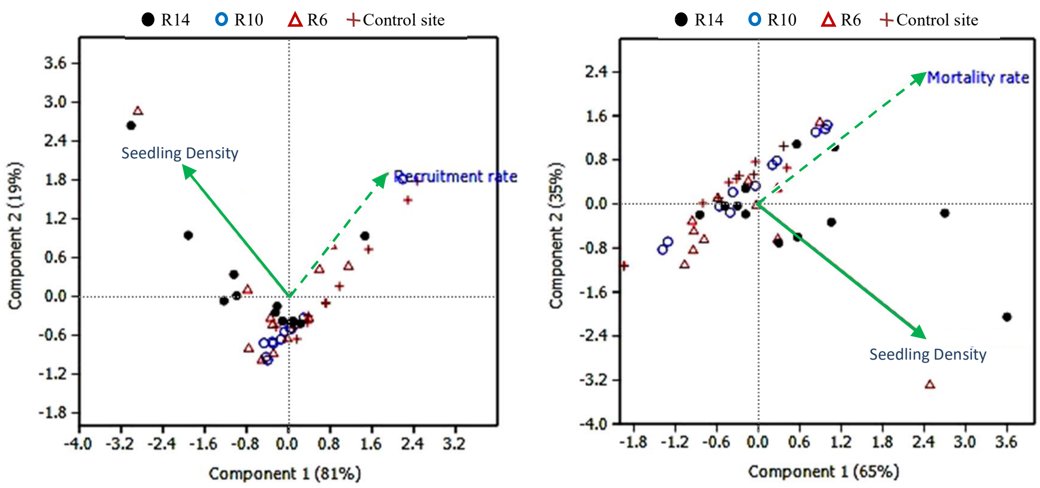


Figure 3.13: a) dead seedling density (stems per 100 m^2), and b) annual mortality rate (% per year) of total seedling in each forest. Different alphabets represent the significant difference among forest sites by Mann-Whitney test.

Crowding of seedlings (overall species) in the R forests did not limit recruitment rate or enhance mortality of recruit species (Fig. 3.14). Principle Component Analysis (PCA) shows no or weak correlation between these 2 factors (Fig. 3.14-b).



a) Recruitment Rate

b) Mortality Rate

Figure 3.14: Principle Component Analysis (PCA) between variables of overall seedling density (arrow) vs. a) recruitment rate and b) mortality rate of recruit species (dashed-arrows) in the R forests and control site. (Each spot illustrates a sample plot in each forest, n=12, and arrows indicate low to high strength of variables)

3.5.8 Seedling Community Turnover

The **seedling turnover rates** of recruit species did not differ significantly among the R forest plots of different ages ($P \geq 0.079$ by Mann-Whitney test in Fig 3.15-a). The numbers of seedlings of recruit species arriving and dying were similar over 1 year among all 3 R forests. However, the same variable for overall seedlings significantly declined with increasing R forest plot ages (Fig. 3.15-b). Apart from R6 and control sites, the rates of seedling turnover in the R10 and R14 forests were not significantly different from those at the DSF site ($P \geq 0.061$ in Fig. 3.15-b). The high turnover rate of R6 sites, 54.6 %, implies dramatically changed of seedling in the community. About half of those seedlings died and were replaced by new arrival seedlings over a year (Fig. 3.15-b, see hypothesis 3-c).

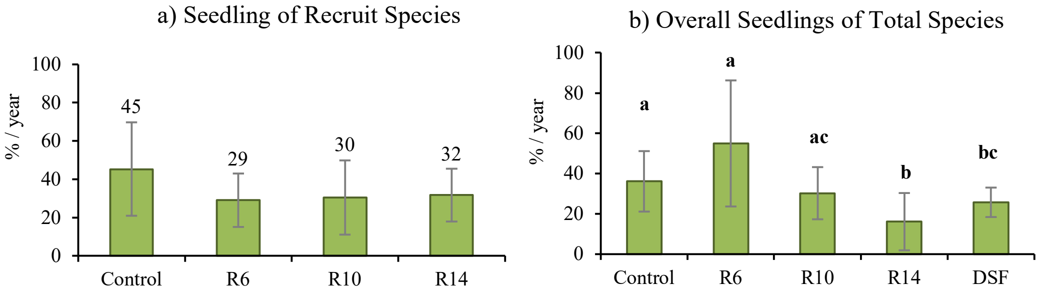


Figure 3.15: Turnover rate of a) individuals of recruit seedling species (non-significant) and b) individuals of *all* species, including planted species (% per year). Bars in b), not sharing the same superscripts, are significantly different (Mann-Whitney test for stem turnover rate)

Turnover of **recruit species** in the R forest was rapid (*ca.* 40 % of recruit species disappeared and were replaced by other species per year) under the R forest older than 10 years (Fig. 3.16-a) with no significant differences among plots of different ages (Fig. 3.16-a). Over all the R plots about 30-40 % of recruit native species were replaced by new species per year (see hypothesis 3-c).

Considering, **overall species,** (Fig. 3.16-b) species turnover rate in the R10 plots was significantly higher than in the R6 plots. Apart from that, other differences were not significant. In particular species turnover rate in all the R plots not differ significantly from that in the control plots and from that in the DSF plots (Fig. 3.16-b).

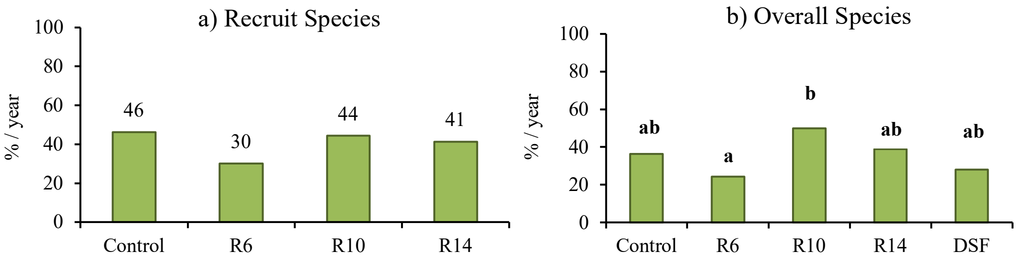


Figure 3.16: Species turnover rate of a) recruit species and b) overall species (including planted species) in each forest site (% per year). The different alphabets designate significant differences among forest sites by Chi-square test.

3.6 Discussion

3.6.1 The Absent Species

The FWS restoration forests collectively accumulated over half (51%) of total trees species recorded in the surrounding landscape within 6-14 years (including planted species) – a remarkable achievement, considering the small sizes of the plots. Conversely, 49% of local tree species failed to establish as seedlings in the R forests, most of which were rare in the surrounding landscape, suggesting that the absence of such species may have been due to limited seed availability. Several other authors support this view. In Panama, Hooper *et al.* (2004) reported that seed dispersal limits seedling density and species richness in naturally regenerating native tropical forest. In Costa Rica, Capers *et al*. (2005) reported that it takes at least 30 years after pasture abandonment for seedlings of rare tree species to become well established, since most of them have large, animal-dispersed seeds. Therefore, although the FWS method of forest restoration obviously catalyzes rapid recruitment of native forest tree species from nearby forest and isolated remnant trees, the effectiveness of the technique, in this regard, is limited by the availability of mature seed tree species, within reasonable dispersal distances of the restoration plots.

3.6.2 Recruit Species Seedlings

Although both seedling density and species richness increased by 35-38% and 9-18% per year in the R plots over 2 years, both these rates appeared to be unrelated to plot age and the latter was also unrelated to the number of tree species already present (hypothesis-2-b). Seedlings of native species recruited at similar rates in all 3 ages (6, 10, and 14 years) of R forests, which is contrary to chapter's hypothesis. Tucker & Murphy (1997) found higher species richness of recruit woody plant in 7-year-old than 5-year-old restoration forest of Australia. However, species richness included all plant habits and they found variation in species richness among restoration plots of similar ages. After 10-14 years of restoration, the FWS forests accumulated diverse native species communities. Several recruit species were common in the natural remnant forest such as *Lithocarpus polystachyus*. Results reflected the attractiveness of FWS restoration plots which confirmed the effectiveness of the techniques area (FORRU, 2005).

Hypothesis 2b stated that "*as the number of species already present in the community approaches that of the species richness of natural remnant forest, the rate of species recruitment gradually decline with increasing plot age*". The results presented above did not support this hypothesis. The species richness of adult trees in surrounding area revealed plenty of unrepresented tree species were left to recruit. However, species recruitment rate was low across all plots (Fig.3.4-b). Some of tree species are also absent from seedling community of the natural remnant forest (DSF). The result might project similar cohort of native species that able to disperse from seed source to neighbourhood area.

Although, seedlings of recruit species was more diverse than those of overall species and approached that of the seedling community of the remnant forests (DSF), restoration age was not related to species diversity and seedling density in these area. These results contrast to Chapter's hypothesis 2-b and several studies in secondary forests of Panama (Denslow & Guzman 2000) and Costa Rica (Caper *et al.* 2005). Thus, age of restoration is not a factor of species diversity recovery in this FWS forest. Khurama and Singh (2001) suggested that differences in forest structure of different restoration stand ages provide different habitats and/or create microhabitat heterogeneity, which impact the dynamics and composition of seedling communities and consequently affect forest succession patterns. Different succession stages (pioneer and non-pioneer) and dispersal mechanisms (non-animal and animal-disperse) and seed dispersal barrier are important factors of recruit species. The proportion of animal-dispersed tree seedlings increased with plantation age (Viani, *et al.*, 2015), Size of seed (White, *et al.*, 2004). The factors, mentioned above, were investigated and discussed again in the following Chapter 5-6.

3.8.3 Seedling Community

Several patches of seedling community, under the R forests, had a high seedling density, due to the accumulation of seedlings of planted tree directly beneath or close to the parent trees. The most dramatic example was *Castanopsis calathiformis* – a superabundant species in the R14 plots and of *Litsea monopetala* in 6-year-old plots (see pictures in Appendix F). This peculiarity resulted in low species diversity and evenness in the oldest R plots (14 years) and in the youngest (6 years) ones. If such species are excluded, native species diversity in the R forest approaches that of natural remnant forest after 6 years. Although the phenomenon of *C. calathiformis* could increase the chances of a deflected succession as suggested by Hopkins (1990), seedling dynamics of native species were not related to seedling density of planted species in community. Besides, area that effected by seedling carpet of *C. calathiformis* is limited under mother trees. Moreover, most of seedlings of *L. monopetala* often die after a few seasons.

The dense accumulation seedling carpets beneath C. *calathiformis* (supper-abundance species) is contrary to the Janzen–Connell hypothesis of density-dependence, which proposes that high densities of seedlings, beneath parent trees attract large numbers of seed/seedling predators and pathogens (Janzen, 1970; Connell, 1970). The accumulation of such high densities of *C. calathiformis* seedlings, therefore may have been due to the absence of such species-specific predators or pathogens. Lan *et al*. (2012) reported that the Asian house rat (*Rattus tanezumi*) is a major predator and disperser of *Castanopsis* species nuts in China's Xishuangbanna National Natural Reserve. Sharp (1995) and Thaiying (2003) found rats of many kinds are very common in these R forests area. Thus, lacking of pathogens possibility is a factor of this phenomenon, however, data and research about this issue are not yet carried out in this area.

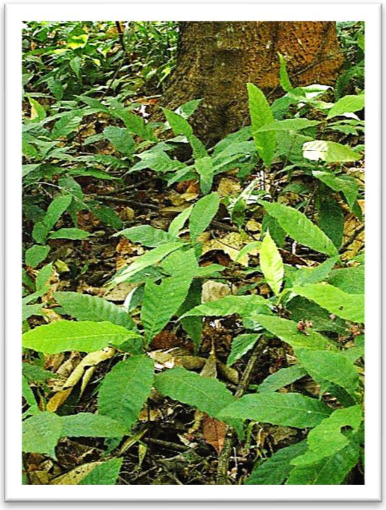


Figure 3.17: Seedling carpets of *Castanopsis calathiformis*

Forests 14-year-old, show sign of similar recruit species derived from natural remnant forest than younger age forests. Most other authors have noted increasing similarity of seedling communities with natural forest as restoration proceeds. For example, Bischoff *et al.* (2005) reported high similarity of recruit sapling species composition among 8-13-year-old secondary forests and primary dipterocarp rain forests in Malaysia.

Nicotra *et al*. (1999) suggested that low light levels lowers seedling density in secondary forests, although seedling density rises later in succession due to the development of greater structural complexity and consequent "light heterogeneity" (light gaps interspersed with deep shade) (this will be addressed in more detail in Chap. 5). Thus, high seedling density in the 14 year-old forest might also be explained by a similar mechanism.

In the control sites (natural regeneration), although seedling density was low, the species diversity index was actually higher than that of the R forest plots. The similarity of seedling community between the control plots and the R14 suggests that the R14 sites were potentially seed source of the control seedling community. Control plots adjacent to the R14 plots supported more trees than grassy plots further away did. This suggests that the R14 sites were actually expanding into the adjacent control sites. Several papers have reported high and diverse seedling recruitment in plantations situated close to natural forest (Zanne & Chapman, 2001; Lee *et al*., 2005). These results are consistent with the forest restoration strategy known as "applied nucleation" which involves re-establishing small forest patches and letting them expand and eventually coalesce (Corbin & Holl, 2012). Therefore, the FWS forest can act as a catalyst for wider-scale landscape restoration. However, the expansion was short-distance ca.10 m in 14 years. At this rate, forest restoration by applied-nucleation method alone might take decades to expand and recover small area, especially in a harsh condition of large-grass-dominated sites. Several methods of forest restoration might need to be combined, applied, and adapted for the most effective strategy, practical and economical purpose in large-scale area. For example, the combination of applied-nucleation and ANR techniques that the ANR technique can prove the area more suitable for seedling establishment near the island of applied-nucleation sites to accelerate forest expansion.

3.6.4 Seedling Relative Growth Rate

Relative growth rate, of both RCD and height of recruit seedlings, was not significant difference among age of restoration. However, seedling of overall species in the 10- and 14-year-old forests grew taller than those in the younger forest plots. Selaya *et al.*, (2007) found that plants typically respond to surrounding neighbor plants through increased stem elongation (grow tall), reduction in stem diameter (such as RCD). The dense seedling, especially with supper-abundance species, of the 14-year-old forest might be factors as neighbor plants influence other seedling to grow tall faster than other forests.

The chapter's hypothesis proposed that the closure of canopy trees promotes seedling growth rate by creating a more optimized environment. However, average RGR across all species in the R forests reflects heterogeneity among species present in the area. Many researchers have suggested that plant responses to habitat in different degrees of growth and survival according to plants' succession stages (shade-tolerant species and light-demanding or shade-intolerant species in a study in tropical forest of Mexico by Popma and Bongers, (1988), in restoration forest of Ethiopia by Yirdaw (2002) and in secondary forest of Costa Rica by Chazdon *et al*. (2010). In early secondary forest succession in Vietnam, light competition is importance limitation for growth of pioneer species (Van Kuijk *et al.*, 2008). Therefore, the habitat factor of 3 different age of the R forest might create similar environment for seedlings underneath. The study of plants' succession stages and habitat factor will be proposed again in Chapter 4 and 5.

3.6.5 Seedling Mortality and Community Turnover Rate

Seedling mortality rates of recruit species, increased with plot age (Fig.3.12-b). Lu, *et al*. (2016) proposed that higher basal area of adult trees in older regeneration forests in secondary forest of China increases seedling mortality due to competition for abiotic resources. A similar result was obtained in Malaysia (Bischoff *et al.,* 2005). Shade brought about by canopy closure was suggested as a strong factor that decreases seedling density in secondary forests in Costa Rica (Capers *et al*., 2005). Above-ground plants generally struggle for light resource (Gorchov & Trisel, 2003). Furthermore, Chazdon & Pearcy (1991) and Montgomery (2004) reported that radiant light partitioning by different structure in forests strongly influences the seedling community composition. Larger FWS trees of older R forest might be a factor limiting light resource for seedlings of recruit species (see Chapter 5).

In contrast, the mortality rate of overall seedlings (including planted species) declined with older age of R forest, although, those rates were not significantly different from those in natural remnant forest. Bischoff *et al.* (2005) reported that higher abundance of pioneer saplings causes higher mortality rates of overall saplings in the secondary forest (8- and 13-year-old) than in primary forest of Malaysia. Therefore, seedlings of pioneer species composition of overall species in the R forest might cause the different mortality rate of those recruit species. However, the factor of species succession stage (non-pioneer and pioneer species) will be addressed in following Chapter4.

The high turnover rate in this study reflects unstable seedling population of native recruit species in the R forests with 1/3 of seedlings were replaced by new native species every year (Fig. 3.14-a). Whilst, seedling community (all seedlings including planted species) of the older age R forest was more stable than the younger age plots (Fig. 3.14-a). This result consistent to 13–16-year-old secondary forests in Costa Rica (Caper *et al*. 2005). Forest environment of older age forest may more suitable for seedling establishment and persistence.

These results demonstrate that the framework species method of forest restoration facilitates recovery of seedling species diversity and restores seedling dynamics to levels similar to those of natural forest within about 14 years. Age of restoration did not seem to affect seedling recruitment of recruit species. Various conditions of regeneration such as dispersal limitation, lack of seed dispersed (Hardwick *et al*., 1997; Farwig *et al*., 2009) or light condition and forest structure (Denslow, 1987; Tabarelli & Peres, 2002; Markesteijn & Poorter, 2009; Viani *et al*., 2015), are suggested as important factors influencing seedlings recruitment of the R forest (the factor of dispersal limitation, forest structure and light condition will be addressed in following Chapter of 4-6).

3.7 Conclusions

1) Of the 208 mature native forest tree species, recorded in or near the restored forest plots, 59 successfully recruited into the R 6-14 plots (28%), whilst seedlings of 35 planted species (17%) also established and 13 species (6%) were present as mature established trees (not planted). Slightly less than half (101 species (49%)) failed to recruit, mainly due to low density of parent trees.
2) Although the number of seedling individuals, establishing in the plots, increased by 34-38% per year, *species* recruitment (over 2 years) increased by only 9-18% per year and was not related with restoration age.
3) Seedling recruitment (individuals) and mortality were not affected by existing seedling density.
4) Five species recruited abundantly in the R forest, across all 3 ages and study periods: 1) *Litsea salicifolia*, 2) *Artocarpus lacucha,* 3) *Antidesma bunius,* 4) *Aporosa octandra*, and 5) *Micromelum integerrimum* (see pictures in Appendix F)
5) Diversity indices of *recruit* seedling species in all R plots were similar to that of remnant forest (DSF), but when seedlings of planted species were included, diversity was lower in the R6 and R14 plots, due super-abundance of a few planted species skewing the result (Fig.3.6).
6) After 6-14 years of restoration, native species diversity approached levels similar to those of the natural remnant forest (DSF), once the effects of the unnatural dominance of one of the super-abundant species (*C. calathiformis* in R14) were removed. Super-abundant species caused low species diversity of seedling community. This was probably due to the loss of seed predators or/and dispersers from the local fauna.
7) Seedling growth rates did not vary with plot age, except for those in the younger R6 plots which had unusually low growth rates. This may be explained by more unfavorable conditions for growth in the younger R6 plots (see Chapter 6).
8) Mortality rates of recruit native species increased with increasing plot age.

9) The seedling community of the 6-year-old forest changed rapidly and the seedling dynamics was similar to that of the control site (natural regeneration site). After 10 years of restoration, however, the seedling community appeared to become more stable.

10) The framework species (FWS) plots were progressing moderately and steadily, with the accumulation of native species seedlings resulting in the seedling community tending towards that of the nearby natural remnant forest (DSF).

11) The natural regeneration sites (control plots) accumulated high species diversity, but many of these seedlings originated from the adjacent FWS plots. Excluding this spill-over effect, forest regeneration on the control sites was slow.

12) Further research of forest regeneration, involving the combination of several techniques, is recommended to improve site limitations and determine the economics of restoration management.

Table 3.5: Summary of tree seedling communities and dynamics in each forest.

	Forest sites				
	Control	R6	R10	R14	DSF
Seedling composition					
Dominant species					
- Overall species	*Litsea monopetala*	*Litsea monopetala*	*Litsea monopetala*	*Castanopsis calathiformis*	*Helicia nilagirica*
- Recruit native species	*Micromelum integerrimum*	*Artocarpus nitidus*	*Micromelum integerrimum*	*Micromelum integerrimum*	-
Density(stems per 100m^2)					
- Overall species	95	700	247	1385	263
- Recruit native species	16	5	15	15	-
Species richness (overall)	51	57	73	49	84
Family (overall)	27	29	34	25	40
Unidentified species	8	5	8	4	7
Diversity (Shannon index)					
- Overall species	2.6	0.8	2.0	0.7	3.0
- Recruit native species	2.3	2.4	2.5	2.4	-
Seedling dynamics (recruit native species)					
Recruitment rate (% per year)					
-Individuals	69	38	34	35	-
-Species	18	9	18	13	-
Growth rate (% per year)					
-RCD	24.6	5.9	14.0	17.0	-
-Height	16.1	13.0	17.2	15.6	-
Dead seedling					
-Density (stems/100 m^2)	12	6	21	19	-
-Mortality rate (% per year)	22	20	27	29	-
Turnover rate (% per year)					
-Individuals	45	29	30	32	-
-Species	46	30	44	41	-

CHAPTER 4

Successional and Seed-Dispersal Status of Native Species

4.1 Introduction

Plant species can be classified according to several life history traits that play an important role in recruitment, survivorship and growth of seedlings during forest succession (Bazzaz, 1991). Such traits include the reproductive strategy, such as seed production and dispersal, and the regeneration strategy such as evolution towards a light-demanding (pioneer) strategy or a shade-tolerant species (Bazzaz, 1991; Barbour *et al.,* 1999). Forest restoration by the framework species method involves planting approximately 10% of the native tree flora and it relies on seed dispersal to increase tree species richness to former levels. Therefore, lack of seed dispersal is one of the most critical factors that could limit seedling recruitment in restored forest (Elliott, 2000; Souza and Batista, 2004). Dispersal distance is often closely related to seed size, since large seeds can be dispersed by few large animal species, whilst small seeds can exploit a much wider range of seed-dispersing animal species and therefore may be transported over long distances (Wunderle, 1997).

After seeds are deposited into restoration sites, plant succession status determine survival and growth of the young seedlings and whether they survive to maturity. As restoration proceeds, the forest floor becomes more and more shaded (Kanowski, *et al.,* 2003). Consequently, seedlings of light-demanding pioneer species struggle for light with shade-tolerant species. The relative abundance of seedling species with combinations succession status can therefore indicate forest habitat condition, forest structure and the extent to which restoration has progressed. Therefore, this chapter investigates how seed-dispersal modes and succession status affect the species composition of seedling communities beneath FWS forests of three restoration ages.

4.2 Literature Review

4.2.1 Seed Size and Dispersal Mechanism

Seed size can affect dispersal, germination and seedling survival (Milberg and Lamont, 1997). Small seeds are easily dispersed over long distances by the wind, birds and mammals of a wide range of different body sizes. In contrast, large seeds are rarely dispersed over long distances, since large animals, capable of ingesting them, are much rarer than smaller animals and they are too heavy to be dispersed by the wind (Wunderle, 1997). Vertebrates are major dispersers in wet tropical forests and their presence or absence is a major factor influencing forest dynamics (Wunderle, 1997; Corlett, 1998a).

In an Amazonian forest, Parrotta *et al.* (1997) recorded a higher proportion of small-seeded climax species, dispersed by mammals and birds, than large-seeded species in a 10-year-old mixed native tree species plantation, purportedly due to a high abundance of small seed-dispersing animal species (e.g. bats and birds) and low abundance of large-bodied animal dispersers (e.g. toucans, tapirs, deer and primates). Corlett and Hau, (2000) reviewed seed dispersal in the tropics and suggested that large mammals are key dispersers in intact forest. Thus, declining animal populations, as a result of deforestation, can easily affect the population dynamics of animal-dispersed plant species. A study of seed dispersal in Doi Suthep-Pui National Park supported this view. Brodie (2007) found 80-95% of *Choerospondias axillaris* fruits was not dispersed, possibly due to hunting having reduced wildlife populations.

Seed dispersal is important for forest regeneration, especially where the seed bank is sparse or where disturbance is intense. Limited seed dispersed limits forest recovery (Hardwick *et al*. 1997). In Kakamega forest, Western Kenya, Farwig *et al*. (2009) revealed that seedling recruitment in plantations strongly depends on seeds from neighbouring sites. Furthermore, natural succession can be greatly accelerated by planting trees that attract seed-dispersed birds and mammals (Tucker and Murphy, 1997; FORRU, 2005). Frugivorous birds are effective dispersal modes, bringing native species into planted sites (Tucker and Murphy, 1997; Wydhayagarn *et al.,* 2009). A study in 2-8-year-old FWS restoration forests at BMSM, Thailand, found about 75% of the recruit species that established in the restoration plots were animal-dispersed (Sinhaseni, 2008). A similar result was obtained by Asanok *et al*., (2013), who compared secondary forests with enrichment plantations in Suthep-Pui National Park, Chiang Mai, Northern Thailand. In the highlands of Ethiopia, large-seeded non-pioneer species are absent or rare in the understorey of restoration plots, whilst many small-seeded pioneer species are abundant (Yirdaw, 2002). Attraction of seed-dispersing animals increases with plantation age. Tabarelli & Peres (2002) reported that plant species, dispersed by vertebrates, gradually increase with plot age. In South-Eastern Brazil, Gorne Viani *et al*. (2015) reported that the proportion of animal-dispersed seedlings increases with plantation age as well.

4.2.2 Species Successional Status

Plant species can be classified basically into two species types, based on their successional status; 1) pioneer (light-demanding) species and 2) non-pioneer (climax or shade-tolerant) species. The pioneer species colonize open areas during the early stages of forest succession. Their seeds require light exposure for germination. They are fast growing, especially in height, and their seedlings and saplings are light–demanding. In contrast, seeds of non-pioneer (shade-tolerant) species can germinate under shaded-habitat. Their seedlings and saplings are shade-tolerant (but not necessarily shade-dependent) and when shaded, they grow slowly (Barbour et al., 1980; Richards, 1996; FORRU, 2005). However, some species are able to grow well under dense canopy or low light habitat (Wright, 2002; Rüger *et al*., 2009). Some species can grow well in both shady and sunny conditions (Popma & Bongers, 1988). Breugel, *et al*. (2007) proposed the hypothesis of the Initial Floristic Composition (IFC) which suggests the recruitment of pioneers sharply declines from early to old aged forest succession.

Hutchings *et al*. (2003) suggested that habitat heterogeneity of physical resources and conditions regulates the responses of different plant species. Differences in successional status can contribute and sustain diversity in a plant community, especially in tropical forests (Kraft *et al*., 2008). In tropical forest, forest gaps accumulate many small-seeded pioneer tree species, whilst medium-to-large-seeded shade-tolerant species recruit abundantly beneath dense forest canopies (Dupuy & Chazdon, 2008).

In FWS restoration forest, Sinhaseni (2008) found many species of pioneer species recruited in all framework species demo plots (2-8-year-old) at BMSM, Chiang Mai, Northern Thailand. Moreover, the proportion of climax species tended to increase with age of restoration. Growth and survival of plant species with different species succession status are also influenced by forest habitat. Pioneer tree species grow faster than shade-tolerant ones, which tend to grow better in diameter and crown under forest canopies (King, 1990 in Montgomery & Chazdon, 2001). Plant succession status are strongly related to seedling composition and regeneration in forest succession (Chazdon *et al.,* 2010). However, Baraloto *et al*. (2005) proposed that dispersal limitation may be more important than the responses of species in different regeneration traits in determining forest structure.

Both seed-dispersal mode and species succession status contribute to variation in seedling recruitment. Therefore, this chapter investigates how seed-dispersal modes and species succession status affect seedling recruitment and how they might explain changes in the seedling community of restoration forest plots with increasing age.

4.3 Objectives, Questions and Hypotheses of the Chapter

4.3.1 Objectives

The objective of this chapter ~~was~~ was to identify relationships between the successional status (pioneer and non-pioneer species) and seed-dispersal modes of tree seedling species and their establishment success in restoration plots.

4.3.2 Questions

1) Do seed-dispersal mode, seed-size, and species succession stages influence the success/failure of tree species in restoration forest plots?
2) Does age of restoration influence the relationships determined in 1) above and if so, how?
3) Do seedling recruitment, growth, and mortality rate differ between pioneer and non-pioneer species?

4.3.3 Hypotheses

1) Tyree s~~S~~pecies ~~of~~ with small seeds are easily dispersed over longer distances than large-seeded species are. Therefore, species absent from the R forest plots are more likely to have large, animal-dispersed seeds, whilst recruit species are more likely to ~~be~~ have small ~~-~~seeds. ~~ed species.~~

2) One of the critical characteristics of framework tree species is that they attract seed dispersing animals. Old restoration plots provide habitat that should ~~attracts more~~be more conducive for the establishment of ~~-~~seedlings of large-seeded and animal-dispersed species than younger forests do~~es~~. Therefore;

    a. Older ~~age~~ plots support relatively more large-seeded, animal-dispersed species in the seedling community, compared with younger plots, whilst small-seeded seedling species are more equally distributed across all ages of restoration plots.

    b. In older plots, seedling recruitment rate of animal-dispersed species is higher, compared with younger plots. Non-animal dispersed tree species distribute equally across all plots of all ages.

3) ~~Succession status of tree species have evolved under various habitat conditions.~~ Restoration plots of different ages provide different micro-habitats that are likely to affect the establishment, growth and mortality of seedlings with different successional status differently. ~~Therefore;~~

    a. Non-pioneer (shade-tolerant) species ~~will~~ recruit more frequently within older plots compared with younger plots. Conversely, pioneer species seedlings recruit in the younger plots more frequently than they do in older plots.

    b. The growth rates of pioneer species decline with increasing plot age. Inversely, seedlings of non-pioneer species grow better in older plots than pioneer species do.

    c. Seedling mortality of non-pioneer species declines and that of pioneer species increases with increasing plot age.

4.4 Methods

4.4.1 Data Collection

Seedlings data form the surveys described in Chapter 3 were classified according to seed-dispersal modes and species succession status, derived from literature: Maxwell and Elliott (2000), Ashton *et al.* (2001), the FORRU database and online research (particularly The Flora of China (eFloras, 2008)) (A table of seed-dispersal mode and the succession status of each species along with referenced sources of such secondary data is presented in **Appendix B**) as followed;

1) Seed-dispersal mode classification;
    a. Animal-dispersed species (A) refers to those that rely mostly on animals as seed-dispersal agents.
    b. Non-animal-dispersed species (N-A) refers to those mostly dispersed by non-animal agents, such as by wind, water, or ballistics etc.
2) Species succession stage classification;
    a. Pioneer species (P) refers to shade-intolerant and light-demanding species, including early- and medium-successional species.
    b. Non-pioneer species (N-P) refers to shade-tolerant and climax species, including late-successional species.

Data on seed size were collected by directly measuring seeds of the species listed in Chapter 3, if seeds were available. If not, seed size data were obtained from FORRU's seed collection and database, and from literature research especially from the flora of China (eFloras, 2008). In this case, seed mass could not be measured for all species. Thus, seed volume was used for a comparison. The length, width and thickness of seeds were used to calculate seed volume (cm^3) (equation 4.1) to compare size between tree species.

Equation 4.1: $\text{Seed volume}\ (cm^3) = Width\ (cm) \times Length\ (cm) \times Thickness\ (cm)$

Then. The data sets were classified into for 3 seed-size classes as 1) small (<2.2 cm^3), 2) medium (2.2-20.2 cm^3) and 3) big (>20.2 cm^3) (modified from Tabarelli & Peres, (2002) integrating s micro- and tiny-seed size in to one small size).

4.4.2 Data Analysis

The tree seedling species were classified into 3 classes of seed size and a pie chart constructed to represent the proportions species in each seed-size. Segments were then then further subdivided into absent, recruit, and planted species in order to compare absent species percentage among all seed-sizes. Data for species of both seed-dispersal mode (N-A and A) and species succession stage (P and N-P) were similarly analyzed.

For the recruit species of the R forest, number species and seedlings were also classified according to 3 seed-sizes in each forest plots to investigate seed-size composition of all 3 ages of R forest. The proportion of recruit seedlings and recruit species of each forest was compared and r recruitment rates calculated between the 2 seed-dispersal modes (N-A and A) and between two successional status (P and N-P). For species of 2 succession stages, relative growth rates of root collar diameter (RGR−R) and height (RGR–H) of recruit species were calculated and compared among the forests of different ages (equation 3.2 in Chapter 3). Likewise, annual mortality rate (% per year) was calculated by the equation 3.3 (in Ch. 3).

Seedling recruitment, growth, and mortality rate (data from 12 replicated plots) were compared among the forest sites by Mann-Whitney test (non-parametric data). Between categories of seed size and seedling dynamics, Wilcoxon test was used to compare within site. Chi-square (χ^2 in equation 4.2) test was used to compare the different proportion of seed-sizes, seed-dispersal modes and species succession stages among forest sites.

Equation 4.2: Chi-square (χ^2) $$\chi^2 = \sum \frac{(O-E)^2}{E}$$

When O is the observed values and E is the expected values, the degree of freedom is 1.

Principle Component Analysis (PCA) was used to evaluate correlation between seedling density and the rate of recruitment and mortality seedlings of the R forests.

The software “Paleontological Statistics, PAST, version 3.0” was used to perform the statistical tests (Hammer *et al.,* 2001; Hammer and Harper, 2006).

4.5 Results

4.5.1 Seed Size of Native Species

Most of tree species, in the vicinity of the plots, were small-seeded (125 spp or 60% of total adult species). Fifty-three (26%) were medium-seeded and 23 (11%) were large-seeded species (Fig.4.1-a). Seven species were unclassifiable due to lack of data (Appendix B). Within all seed size categories, about half of the species (41-52%) failed to recruit in the restoration plots. Although the proportion of large-seeded species that were absent from the R forests was slightly larger than that of small-seeded species, the differences among all 3 seed-size class were not significant (χ^2 = 1.4817, *P* = 0.4767). This result suggests that seeds of all sizes have the same opportunity to recruit into the R forests (see hypothesis 1).

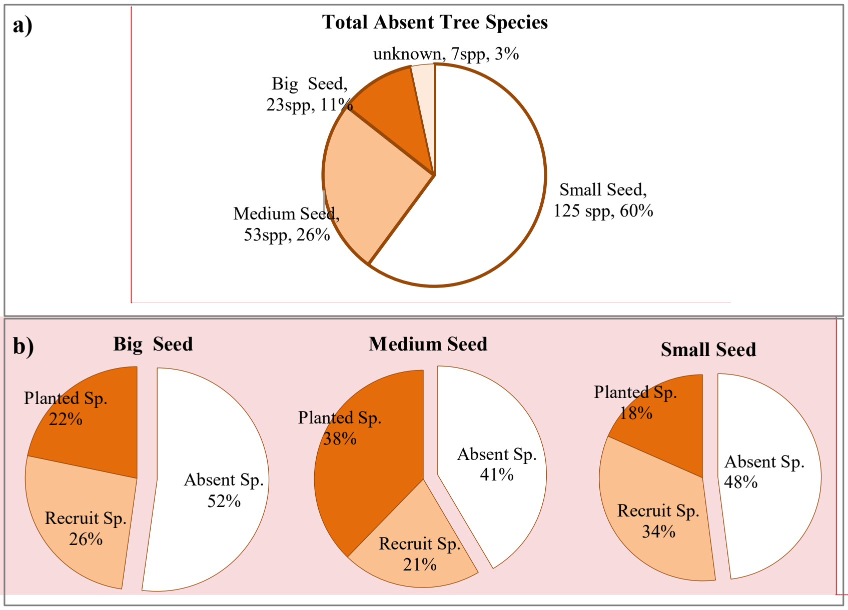


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Figure 4.1: a) Number of absent species and b) % absent species of each seed size category found in adult tree species survey around DSF and R forests.

The percent species of big-seeded species establishing slightly increased with plot age, but not significantly ($\chi^2 \leq 0.809$ in Fig.4.2, see hypothesis 1 and 2-a). Most of the recruit species were small-seeded species (Fig.4.2 and 4.3).

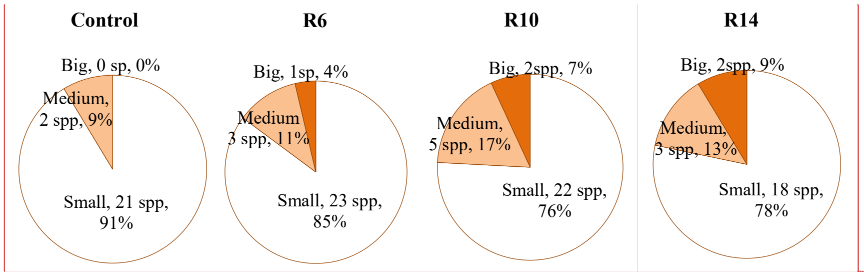

Figure 4.2: Number and percent species of 3 seed-sizes that recruited in the R forests and control site.

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About 14-18% of small seeded adult trees in the vicinity of the plots species established as recruit species in the R plots (Fig.4.3), whilst only 6-9% of large-seeded adult trees did so. Examples of big-seeded species that established in the R14 and R10 forests included *Horsfieldia amygdalina,* and *Mangifera caloneura.* However, big-seeded species were completely absent from the control sites. Species of small and medium-sizes adult species established in similar proportion in all the R forests ($\chi^2 \leq 0.995$).

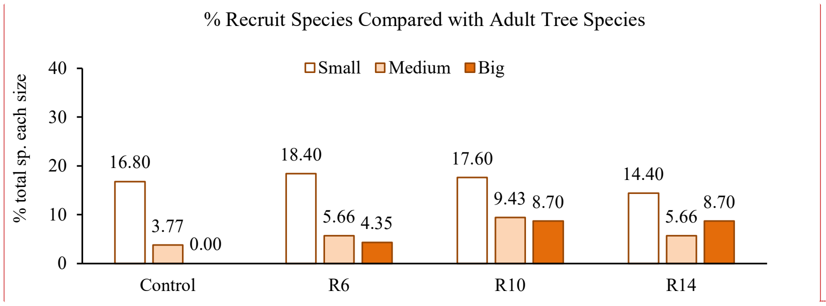

Figure 4.3: Percent recruit species compared with the seed trees of each category in the R forests and control site.

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Considering numbers of individuals, most of the seedlings of recruit species in the R forests and control site were small-seeded species. The percent of total seedling individuals recorded of medium-seeded species in the R6 plots was significantly higher than in the other forest plots ($\chi^2 \geq 19.753$ in Fig.4.4). However, forty-five out of forty-seven seedlings belonged to a single species: *Artocarpus nitidus* (seed-size = 3.73 mm^3), due to the close proximity of a fruiting tree of this species to the R6 plots.

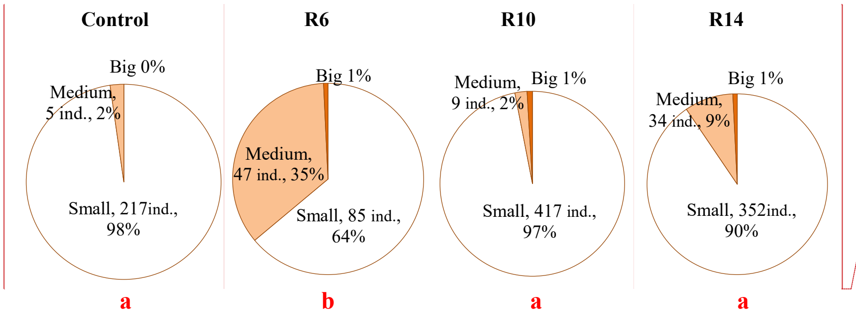

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Figure 4.4: Number seedlings and percentage of 3 seed-sizes that recruited in the R forests and control site. Alphabets, below each pie chart, indicate significant different proportion of all seed-size among forests by Chi-square (χ^2).

4.5.2 Recruit Species Seedlings: Seed-dispersal Modes

Considering seed-dispersal mechanisms, most of the adult tree species in the vicinity of the plots were animal-dispersed (154 sp.) whilst, 48 were non-animal-dispersed and 6 were of unknown dispersal mode (Fig.4.5). Almost half of the animal-dispersed species failed to recruit in the R forests (Fig.4.5-b and c, and see hypothesis 1). Absent percentage within each mechanism reveals slightly higher percent of animal-dispersed species (49%) than non-animal-dispersed species (40%). Both non-animal-dispersed and animal-dispersed species had similar percent species (31% and 29 %, respectively) recruited into the R forests (Fig.4.5-b and -c).

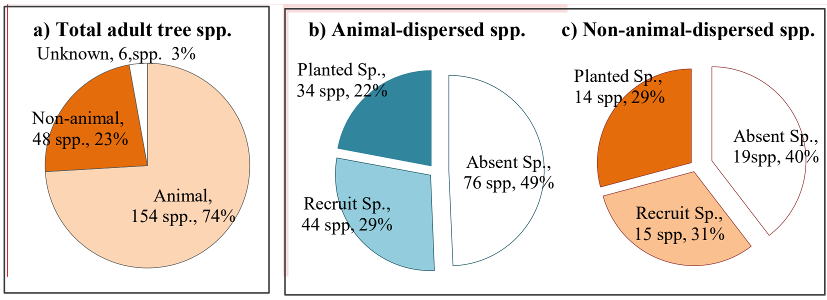

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Figure 4.5: Number species of seed-dispersal modes a) total adult species and number species that absent, recruit, and planted found of each b) animal-dispersed and c) non-animal-dispersed species

Almost all recruit seedlings in the R forests were animal-dispersed species which reflected the dominance of such seedlings in natural forest, the DSF site (Fig. 4.6-a). The percentage of seedlings of the two dispersal modes did not differ significantly with plot age ($\chi^2 \leq 3.724$ in Fig. 4.6-a, see hypothesis 2-b). However, the percentage of animal-dispersed seedlings in all R plots was significantly lower than that in the DSF site ($\chi^2 \geq 4.067$). The percentage of recruit species of animal-dispersed species declined with increasing restoration age. The percentage of animal-dispersed recruit species in the R6 plots was significantly higher than that in the R14 plots ($\chi^2 = 10.241$ in Fig. 4.6-b). However, the ratio between 2 dispersal modes of species in all 3 R forests was not significantly different from DSF and control site.

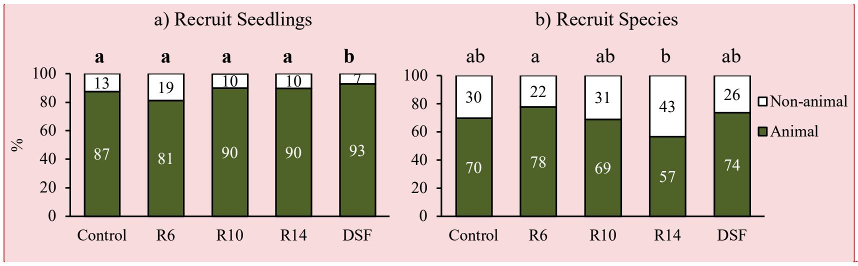

Figure 4.6: Percentage of recruit seedlings between seed-dispersal modes by a) seedling density, and b) number of species in the R forests. Different letters indicate significant differences in Chi-square test ($\chi^2 > 3.84$), among forests

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4.5.3 Recruit Species Seedlings: Non-pioneer and pioneer species

The successional status did not influence seedling species establishment in the R forests (across all sites in Fig. 4.7). Forty-four non-pioneer and 43 pioneer species established (Fig. 4.7-a). Probability of establishment or non-establishment was unaffected by successional status (Fig. 4.7-b).

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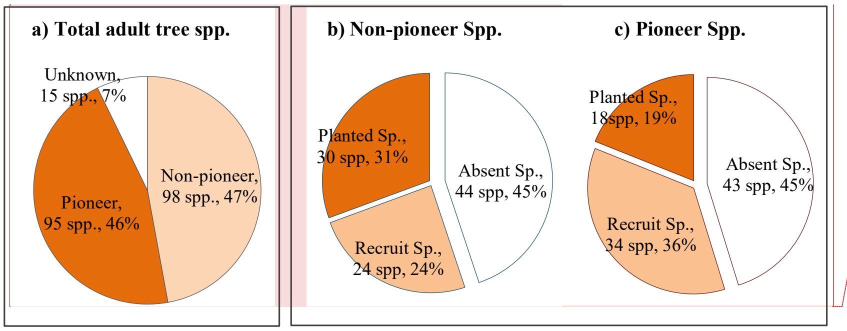

Figure 4.7: Number species of succession stage a) total adult species and number species that absent, recruit, and planted found of each b) non-pioneer and c) pioneer species

The seedling proportion of non-pioneer species establishing did not gradually increase with plot age, as hypothesized (Fig. 4.8). Each age of R forest, DSF and control site had different seedling ratios of non-pioneer: pioneer species ($\chi^2 > 8.28$ in Fig. 4.8-a). Nonetheless, pioneer species represented a higher proportion of the recruit seedlings than non-pioneer species did in the youngest plots (R6), whereas the reverse was true in the older plots (R10 & R14) (Fig. 4.8-a, see hypothesis 3-a). In addition, the proportion of non-pioneer seedlings in site R10 was significantly higher than in site R14 ($\chi^2 = 29.55$ in Fig. 4.8-a). For amount of recruit species, species proportion between pioneer: non-pioneer species was not significantly different among all R forests and DSF and the control sites ($\chi^2 < 2.74$ in Fig. 4.8-b).

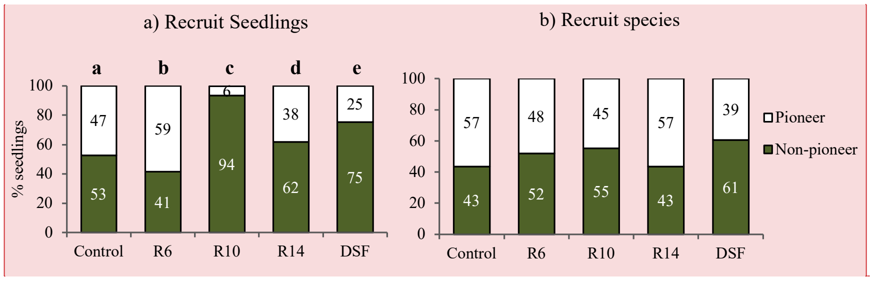

Figure 4.8: The percentage between pioneer and non-pioneer species of recruited seedlings by a) seedling density, and b) recruit species in the restored forests. Different letters indicate significant differences, with Chi-square test ($\chi^2 > 3.84$), among forest sites.

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4.5.4 Seedling Recruitment Rate

For seed-dispersal mechanisms, seedling recruitment rate of both non-animal and animal-dispersal species were not related to R plot age (Fig. 4.9-a). The recruitment rates of both species were not significantly different among three planted ages (*P* >0.05 in Fig. 4.9-a, see hypothesis 2-b and 3-a). Between two seed-dispersal modes, seedling recruitment rates of animal-dispersed species were significantly higher than those of non-animal-dispersed species in all 3 of the R forests (*P* = 0.008, 0.003 and 0.002, in R6, R10 and R14 respectively, by Wilcoxon test in Fig. 4.9-a). This result explains the larger percent of seedlings of animal-dispersed species than of non-animal-dispersed species found in those R forests (Fig. 4.6-a).

For species successional stage, seedling recruitment rate of both non-pioneer and pioneer species were not related to R plot age (Fig. 4.9-b). The recruitment rates of non-pioneer species were not significantly different among three planted ages (*P* ≥0.238 in Fig. 4.9-b, see hypothesis 2-b and 3-a). Seedlings of pioneer species in site R10 recruited significantly lower than those seedlings in site R14, R6, and control (*P* = 0.036 and 0.0009, 0.017, respectively, by Mann-Whitney test in Fig.4.9-b). Moreover, R10 forest was the only site that the recruitment rate of pioneer species was significantly lower than non-pioneer species (*P* = 0.005 by Wilcoxon test in Fig. 4.9-b). This event led to a very low ratio of pioneer seedlings of this R10 plot (Fig. 4.8-a). In the control plots, the recruitment rates of all seed-dispersal modes and both regeneration traits, were not significantly different from the R forest, except those animal, pioneer species in plots R10 (Fig. 4.9-b).

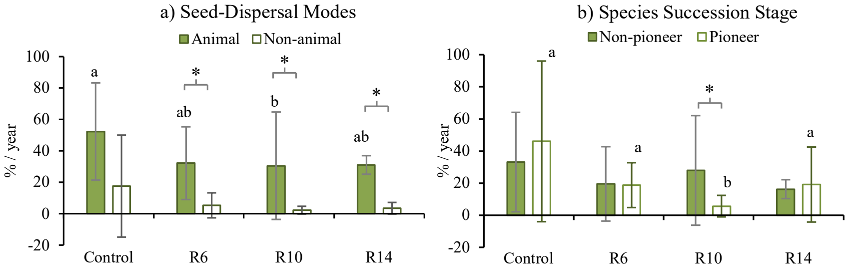

Figure 4.9: The recruitment rate of seedlings in each a) seed-dispersal mode and b) species succession stage, with standard deviation. Columns not sharing the same superscripts are significantly different by Mann-Whitney test. Asterisks (*) designate significant difference between dispersal modes and succession stages by Wilcoxon test.

4.5.5 Relative Growth Rate (RGR) of Species Successional Status

Relative growth rate of both non-pioneer and pioneer species were not related to plot age. Furthermore, growth rates of seedlings in all R forests were not significantly different from seedlings in the control and the DSF plots (*P* > 0.06), except RGR-H of seedlings in the R10 and DSF plots (*P* = 0.015 in Fig. 4.10, see hypothesis 3-b).

Comparing mean seedling RGR between non-pioneer and pioneer species, the growth rates were not significantly different between in all the forest plots, except in the control plots. In the control plots, seedlings of non-pioneer species grew significantly bigger RCD than pioneer seedlings (*P* = 0.047 in Fig. 4.10-b, see hypothesis 3-b).

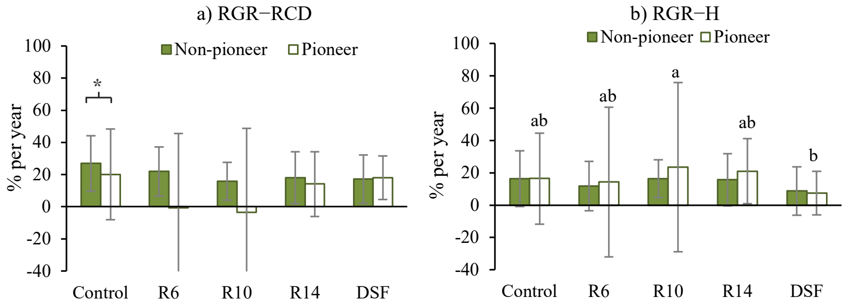

Figure 4.10: Mean growth rate of seedlings a) RGR−RCD and b) RGR−H of recruit species between pioneer and non-pioneer species in R forests and control site compare to those of total seedlings of site DSF. Asterisks (*) designate significant difference between pioneer and non-pioneer species by Wilcoxon test.

4.5.6 Seedling Mortality Rate of Species Successional Status

Seedling mortality did not ~~decrease~~ increase with increasing plot age for both pioneer and non-pioneer species (Fig. 4.11). Differences between ~~The~~the mortality rates ~~-~~of ~~both~~ non-pioneer and pioneer species were not significant~~ly different among~~ across all R forest ~~3~~ ages, ~~of the R forests,~~ although the mean mortality rate of pioneer seedlings ~~under youngest age,~~in R6 ~~,~~was significantly lower than ~~those~~ it was in the ~~under~~ R10 and R14 plots ($P \geq 0.57$). Moreover, the mean mortality rate of pioneer species in R6 plots ~~was~~ was significantly lower than it was in the ~~those of~~ DSF plots ~~sites~~ ($P = 0.0386$ by Mann-Whitney test). This result explains the overall ~~can describe the~~ low mean mortality rate of all recruit species ~~found~~ in the R6 plots because the R6 plots were ~~tha~~t dominated by pioneer species (Fig.3.12 in Chapter 3). Apart of that, none of all 3 R forests had significantly different mortality rates of non-pioneer and pioneer species among the R plots, DSF and control sites ($P > 0.6$ Fig. 4.11, see hypothesis 3-c).

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Comparing between seedlings of non-pioneer and pioneer species in each site, none of the R plots, DSF, and control sites had significant different mortality rates among the two groups of species ($P > 0.07$ by Wilcoxon test).

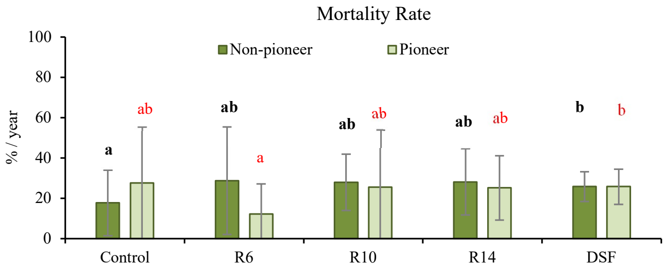
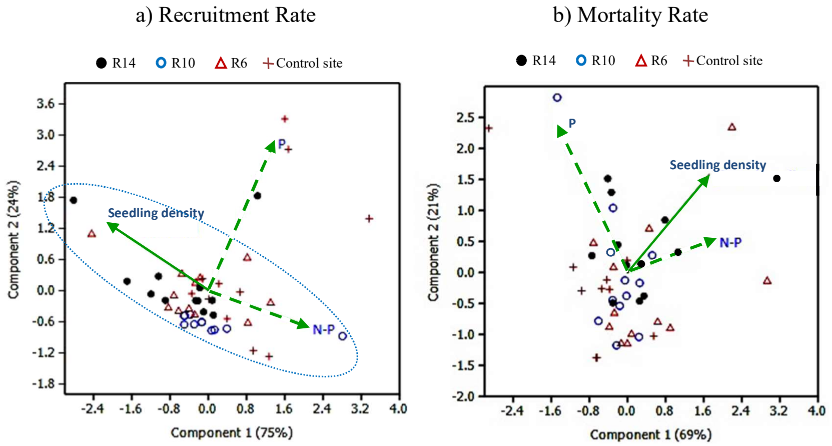


Figure 4.11: Relative mortality rate of recruit species in each regeneration trait in all forest sites. The different alphabets designate for the significant difference between sites by Mann-Whitney test.

Although in chapter 3, I reported no relationship between planted tree density and the dynamics of overall recruit species, the results of the Principle Component Analysis (PCA) here shows a negative correlation between planted tree density and recruitment rates of non-pioneer species (Fig. 4.12-a). Seedling density (lined-arrow) had an opposite trend to non-pioneer species recruitment rate (dashed-arrow) (dotted-eclipse in Fig. 4.12-a). This indicates a decline in the recruitment rate of non-pioneer species where seedlings grow densely. Moreover, the mortality rate of non-pioneer seedlings was highest in most dense plots (Fig. 4.12-b). This result explains the low recruitment rate of non-pioneer seedlings in the R14 and R6 forests (Fig. 4.9-b). At the same time, pioneer seedlings were not affected by the numbers of planted trees.


a) Recruitment Rate

b) Mortality Rate

Figure 4.12: PCA for the relationship between variables; seedling density of planted species and recruitment rate (dotted-lines), or b) mortality rate (dashed-lines) of pioneer and non-pioneer species in R forest and control site (each spot represents a sample plot, n= 12 of each forest site)

4.6 Discussion

4.6 1 Seed Size

Unexpectedly, seed size did not appear to limit species recruitment in the R forests (Fig. 4.1). Recruitment of tree species for all seed sizes were similar. This result contradicts previous studies in moist evergreen forest in Thailand (e.g. Kitamura, *et al.*, 2004) and in Panama (Hooper, *et al.*, 2005) where large-seeded size limited dispersal. Although dispersal of large seeds is a rare event, the R forests were quite close to least disturbed forest and seed-dispersal agents remained common in the landscape (as evidenced by camera trapping, carried out by FORRU-CMU, unpublished observations). Thus, lack of seed availability is more likely to limit recruitment than dispersal limitations (as found in Chapter 3). In addition, Sansupan *et al.* (2018), who studied in the same study sites of BMSM, reported a similar evident that the lack of seed availability or inadequate seed dispersal limit tree species to colonize in the FWS forests.

The comparison among planted age revealed that species of big-seeded occurred to a slightly extend in 10-14-year-old forests than in 6-year-old forest, although differences among sites were insignificant. Seedlings of small-seeded species were equally distributed across all R forests (Fig. 4.2). These results might be an initial sign of higher attractive potential of big-seeded species in >10 year-old R forests than young-age forests. Several studies reached similar conclusions, such as a study of 10-year-old native plantations in Amazonia (Parrotta *et al*., 1997), in the highlands of Ethiopia (Yirdaw, 2002) and in plantations in Chiang Mai, Northern Thailand (Asanok *et al*., 2013).

4.6 2 Seed-Dispersal Modes and Species Succession Status of Recruit Seedlings

Seed-dispersal mode and successional stage did not limit native species recruited in the R forest (across all ages) (Fig.4.5 and 4.7). In contrary, seed-dispersal agents are limited factors for species recovery in many studies of secondary forest (Wunderle, 1997; Corlett & Hau (2000); Terborgh *et al.*, 2008; Markesteijn & Poorter, 2009). The result of this chapter imply stronger effect of the rarity of seed trees available than dispersal agent limitation to the recruitment in the FWS forests. Therefore, rare species and/or species that less common in seed-source forest should be considered as one of the FWS for forest restoration in order to enhance species diversity in this area.

However, seedlings of non-pioneer and animal-dispersed species established abundantly in the seedling communities of all 3 age R forests but their number did not increase significantly with plot age, as hypothesized (Fig. 4.6 and 4.8) (see hypothesis no. 2 and 3). The result conforms to a study by Capers *et al.* (2005) who provided an evident of higher abundance of animal-dispersed species (including palms) than those with ballistically dispersed seeds. Birds are key dispersers of seed from forest into many forest restoration sites (Parrotta *et al*. 1997; Lozada *et al*. (2007). A previous study at this study site by Wydhayagarn *et al*. (2009) reported many bird species found in FWS restoration plots.

Babaasa *et al.,* (2004) proposed that density of late-successional (non-pioneer) species indicates the potential of different forest management sites to develop into more natural forests. Therefore, high density of non-pioneer species in age >10 years of the FWS restoration forests could indicate high potential of the FWS method to direct and accelerate natural regeneration towards seedling communities similar to those of natural remnant forest. However, the results did not follow the hypothesis of "Initial Floristic Composition (IFC)" by Breugel, *et al*. (2007) that applied to secondary tropical rain forest succession in abandoned agricultural fields. The hypothesis expects the recruitment of pioneers would sharply decline from early to old age of forest succession. In these R forests, the ratio of recruit pioneer seedlings sharply decreased from young age (6 years) forest to older age (10 years) forest but it rises again in the oldest age (14 years) forest. This unexpected result was affected by superabundance species that influenced only on the recruitment rate of non-pioneer species in R forests age 6 and 14 years.

In control site (natural succession site), seedling recruitment rate of animal-dispersed species were higher than those of R14 forest (Fig. 4.9). This result contrary to a study of planted plots and simulated natural succession in tropical forest of southern Mexico, by De La Peña-Domene *et al.* (2013). They found the recruitment of animal-dispersed trees accumulated in planted plots 10 times faster than in control plots. The shared boundaries between R14 and the control plots allowed seedlings of animal-dispersed species recruited due to the split out phenomenal (earlier mention in Chp.3) and seed dispersers attracted to the R14 plots.

4.6 3 Relative Growth Rate of Species Successional Status

Contrary to chapter's hypothesis, seedling growth rates of pioneer and non-pioneer species were not affected by age of restoration. The rates of them under R forests were also not different from those of natural remnant forest, except RGR-H of pioneer species in the R10 forest which was higher than in natural forest (Fig. 4.10). In the early stages of secondary forest succession, such as in the 6-year-old of the R forest, light levels are expected to be high, which should stimulate seedling growth (King, 1990; Van Kuijk *et.al.*, 2014). The indifferent growth rates, both RCD and height, among the R forests perhaps reflects similar light environments.

In addition, the low seedling growth rate (less than 27% cm per year) of all forests found in this study is consistent with a study by McNamara (2009) that found diameter increments of seedlings less than 0.2 cm/year after 6 to 9 years of growth in enrichment secondary forest in Vietnam and Lao.

~~Furthermore, the result of seedlings in the control site indicate that this area supported seedlings of non-pioneer species than pioneer species. The variations of control site habitat might benefit and enhance RCD growth rate of non-pioneer species seedlings. Dupuy and Chazdon (2008) suggested that high dense grasses could enhance establishment of non-pioneer species through increased humidity.~~

Furthermore, the result of seedlings in the control site indicate that seedlings of non-pioneer species grow bigger RCD than grow tall. The situation of dense grasses could enhance light resource competition. Baraloto *et al.* (2005) found an evidence of seedling height-growth and survival trade-offs among shade-tolerant (non-pioneer) and gap-establishment (pioneer) species. Therefore, the low mortality of non-pioneer species in the control site may explain the trade-off situation of seedlings to survive, wait, and grow vertically than tall, for the opportunity to grow tall when a light gap opens in the future.

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4.6 4 Seedling Mortality Rate of Species Succession Status

Although the mortality rate of both non-pioneer and pioneer recruit species were non-statistically different among all 3 age of the R forests, the low mean mortality rate of pioneer species under 6-year-old R forest might provide an explanation of high ratio of pioneer seedling in R6 plots. Similarly, to plant growth, forest habitat is an important factor controlling survive of seedlings in many studies (Kobe, 1999; Markesteijn & Poorter, 2009). Kobe (1999) found the mortality of several rain forest tree seedlings decreased with increase in light intensity (to 20% full sun) in Costa Rica. Therefore, the results of both growth rate and mortality rate reflected similar average shade and light environment found among the 3 ages of the R forest (see result in Chapter 6).

High seedling density can inhibit establishment of non-pioneer species. Negative density-dependent is strong for young seedling survival during the transition from seed-to-seedling stages (Harms *et al*. 2000, Metz 2007). Resource-limitation by neighbouring plants such as soil moisture in dry season or spatial light might increase mortality rate of seedling of non-pioneer species. Forget (1991) suggested that non-pioneer tree species less survived under a dense understorey plant than they did spared neighbouring plants. In addition, the result might be involve to dense adult trees that found negatively affect seedling of non-pioneer in the area species (see chapter 5).

Ultimately, seed trees availability was major limitation of species recruitment in all 3 R forests. However, there are factors such as tree structure and physical suggested as important factors controlling seedling recruitment of native species. Therefore, these factors were investigated and shown in the next chapters.

4.7 Conclusions

1) Seed Size dispersal mechanisms mode and successional status did not appear to limit species establishment in FWS restoration forest plots. Species of any seed size,~~ of seed,~~ dispersal mode and successional status had similar chances (c. 50%) to recruit into the R forests.

2) Among 3 ages of the R forests, seedlings of small-seeded species established equally frequently regardless of restoration plot age. Seedlings of animal-dispersed species were more common and had high recruitment rates in all ages of FWS restoration forests.

3) After seed arrival, the ratio of ~~s seedling between~~ pioneer to ~~and~~ non-pioneer seedling species did not differ significantly among the ~~of each~~three ~~-~~R forests ~~were significantly difference.~~ Seedlings of pioneer species are more common in 6-year-old than in 10- and 14-year-old forest plots.

4) Seedlings of non-pioneer species dominated the ~~in~~ 10-year-old plots. The very low seedling recruitment rates of pioneer species forest cause very low seedling of pioneer species in the 10-year-old plots

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5) Seedling growth rates of both pioneer and non-pioneer species were not significantly different among plot ages, control and DSF sites except that RGR-H of seedlings in the R10 plots was significantly higher than it was in the DSF plots.

6) The mortality rate of both non-pioneer and pioneer species was not related to plot age~~d~~. However, 6-year-old forest ~~is~~ was the only site ~~that~~ where seedling~~s~~ mortality rates of pioneer species was significantly lower ~~than~~ compared with the DSF site.

7) The FWS forest plots accumulated native trees species across all seed size classes, dispersal modes and successional statuses tending towards a similar seedling community to that of ~~the~~ natural remnant forest (DSF).

8) ~~The factor of s~~Seed availability was the ~~a~~ strongest factor, determining ~~limitation for~~ native tree species recruitment. Rare species and/or less common species in the vicinity of restoration sites should be selected for inclusion amongst those planted~~seed-source forest are considered as one of the FWS for forest restoration in order~~ to enhance species diversity.~~ in those areas.~~

9) N~~The n~~atural regeneration sites (control site) failed to recruit large-seeded ~~spicess~~species in the seedling community.

CHAPTER 5

Influence of Understorey and Canopy Trees on Seedling Recruitment

5.1 Introduction

Canopy closure and understorey formation are the first two major indicators of recovery of forest structure, biodiversity and ultimately forest restoration success (FORRU, 2005; Catterall *et al.,* 2008). Many studies of forest plantations and restoration have demonstrated that tree density and stand structure became more similar to those of secondary forests (Saha, 2001; Farwig *et al.* 2009) and ultimately to old-growth forests over time (Kanowski *et al.* 2003; Kaewkrom *et al.* 2005; Sansevero *et al.,* 2011). However, diversification of forest structure and increasing complexity vary due to the forest restoration techniques applied (Carnevale & Montagnini, 2002; Kanowski *et al*., 2003; Lemenih *et al*., 2004; McNamara, 2009) and restoration plot age (Khurama & Singh, 2001; Farwig *et al.* 2009) and etc.

Consequently, variations in the understorey and forest canopy trees influence the seedling communities beneath (Denslow *et al.*, 1991; Khurama & Singh, 2001; Jones *et al*., 2004). A closed canopy and high density of planted trees create shade and habitats that are more suitable for seedlings of shade-tolerant (non-pioneer species) trees to recruit (FORRU, 2005). In contrast, incomplete canopy closure, during early forest regeneration, has repeatedly been reported as creating habitat for shade-intolerant (pioneer species) (Finegan, 1996; Van Kuijk *et al.*, 2008). The FWS method aims to accelerate recovery of forest structure that supports species recruitment and enhances natural succession. Therefore, this chapter analyses the understorey and canopy trees that determine the recovery of forest structure, among restoration plots of 3 ages and compares them to those of nearby remnant old-growth forest. The second objective was to determine relationships between forest structure and native tree seedling recruitment and dynamics. These factors might help to indicate natural succession progress and efficiency of FWS for forest restoration.

5.2 Literature Review

5.2.1 Canopy and Understorey Trees Recovery

A closed canopy and high tree density are characteristic of secondary forests in Malaysia (Bischoff *et al.*, 2005), Khao Yai NP., Thailand (Tangprasertsri, 2005), Northern Thailand (Fukushima *et al.*, 2008), Vietnam (Van Kuijk *et al.*, 2014) and China (Lu, *et al.*, 2016). Increasing canopy cover and shade, cast by the canopy trees, indicate progression of forest succession (Corlett, 1995; Brown & Lugo, 1990). Royo & Carson (2006) pointed out that tree distribution patterns can determine rate and direction of successional pathways. In addition, land-use history also influences which tree species dominate secondary forest in Northern Thailand (Fukushima *et al.,* 2008) and Costa Rica (Guariguata & Ostertag, 2000).

Many plantations and restoration forests are established to reconstruct tree cover and canopy closure on deforested lands (Kamo *et al.*, 2002; FORRU, 2005; Kaewkrom *et al.*, 2005). In restoration forest (aged 5-22 yrs.) in Australia Kanowski *et al.*, (2003) reported canopy closure is denser in older restoration plots, although it is generally not as dense as that of intact rainforest. High tree density in restoration and plantation forests is usually attained by initially planting high numbers of tree seedlings or saplings (FORRU, 2005). Therefore, complexity of dense understorey and canopy trees are frequently reported as natural succession after tree planting. Oberhauser (1997), who studied pine (*Pinus kesiya*) plantations (age 7-28 yrs.) in the Northern Thailand, reported that structural complexity increases with plot age. Later studies by Kaewkrom *et al.* (2005) and Koonkhunthod *et al.* (2007) also demonstrated well-developed structure at the mid-storey level in a pine plantation in Northern Thailand. As natural forest succession progresses, the habitat characteristics of restoration forests approach those of old-growth forest. Many studies of restored forests provide evidence of forest structure becoming more similar to that of neighbouring secondary forests (Saha, 2001; Farwig *et al.*, 2009) and old-growth forests (Kanowski *et al.*, 2003; Kaewkrom *et al.*, 2005) over time. For example, Sansevero *et al.* (2011) reported that the density of understorey trees (size ≥1-5 cm-DBH) in plantations (age 9–11 yrs.) was comparable to that of old-growth forest in Brazil. Kamo *et al.* (2002) compared 12-year-old monoculture plantations of *Acacia sp*. and *Eucalyptus sp.*, mixed species plantations of indigenous species (*Dalbergia cochinchinensis, Xylia kerrii* and *Pterocarpus macrocarpus*) and natural forest in Sakaerat, Thailand. They reported 35 - 59 species of understorey plants in plantation plots (although plantation plots supported fewer understorey plant species natural forest did), with 31-51% of understorey species coming in from natural forest. Total biomass of understorey plants was highest in the *P. macrocarpus* plantation.

Factors that influence the structural characteristics of restoration forest include tree density, species planted, variation in growth rates and tree crown architecture (Lamb, 1998; Kanowski *et al.* 2003; Lemeniha *et al.* 2004; Jones *et al.*, 2004; Catterall *et al.*, 2008). However, some studies have argued that restoration design does *not* influence forest structure and dynamics, for example in Brazil (De Souza & Batista, 2004) and in Colombia (Rulz *et al,* 2005). Therefore, this chapter determined forest structure (density and basal area of trees) in the R forests, DSF, and control sites and identified differences among the sites.

5.2.2 Influence of Trees on Seedling Community

After canopy closure, various species are attracted to restoration forest plots. For example, in framework restoration forests in Queensland, Australia, Tucker and Murphy (1997) reported higher recruitment of plant species, across all growth forms, in 7-year-old restoration forest than in 5-year-old, forest. Keenan, *et al*. (1997) found 176 understorey plant species beneath both monoculture and native tree plantations (age 5-63 yrs.) in Northern Australia. In Northern Thailand, Asanok *et al.* (2013), who studied enrichment plantations and secondary forests, found that seedling/sapling densities of some species, such as *Castanopsis acuminatissima, Syzygium claviflorum* and *Sarcosperma arboretum,* were affected by forest structure (BA, tree density and forest types), and physical environment. The development of understorey and canopy trees, during natural forest succession, greatly affect seedling communities because factors such as canopy height, density and basal area of trees, in combination, create heterogeneous environmental conditions (Denslow *et al.* 1991; Kenoyer, 1992; Valladares & Guzman, 2006). Furthermore, in a *Picea glehnii* plantation (age 21-26 yrs.) in Japan, Nagai and

Yoshida (2006) reported a positive influence of high tree density on seedling density of both shade-tolerant and shade-intolerant species. However, most seedlings died before reaching the sapling stage in high density plantations. The phenomenon may be due to low understorey light availability, associated with canopy closure. Several studies suggest that forest gaps enhance recruitment of seedlings and saplings more than dense canopy forest without open gaps (Forget, 1991; Oberhauser, 1997; Lemeniha *et al.*, 2004; Yirdaw & Luukkanen, 2004; Dupuy & Chazdon, 2006). However, Beckage *et al.* (2000) suggest that rather than canopy gaps, it is the density of the forest understorey that strongly influences tree seedling recruitment. In their study, canopy gap formation did not affect tree seedling/sapling species richness or diversity if understorey density was high. High understorey density decreases light availability even where forest canopy cover is low.

Previous studies of the FWS restoration plot system at BMSM, Chiang Mai, Thailand, showed high rapid biodiversity recovery as FWP plots matured. Many frugivorous bird species visited the plots resulting in high species richness of seedlings of animal-dispersed tree species establishing beneath the restored forest canopy. (Wydtayakarn *et al.*, 2009). Singhaseni (2008) reported seedlings of 73 recruit (non-planted) tree species recolonized plots planted with 20-30 framework tree species. Furthermore, Jinto (2010) reported dense canopy closure of FWS trees in 9-year-old restoration forest, whilst the canopy of a nearby natural forest remnant was discontinuous with canopy gaps and lower stem density. Therefore, observations of understorey and canopy development presented in this chapter were used to determine correlations of density and basal area (BA) of understorey and canopy trees with seedling recruitment and the dynamics of native species found in this FWS restoration forests.

5.3 Objectives, Questions and Hypotheses of the Chapter

5.3.1 Objectives

This chapter:

1) compares tree development and structure of understorey and canopy trees in restoration forest, with the same in control and remnant forest.

2) evaluates the influence of the understorey and FWS canopy trees on seedling recruitment and dynamics of native recruit tree species in restoration forests.

5.3.2 Questions

1) Do density and BA of understorey and canopy trees increase with increasing plot age?

2) How do the density and BA of understorey and canopy trees affect tree seedling recruitment and seedling mortality?

5.3.3 Hypotheses

1) After tree planting, the density and BA of understorey and canopy trees increase, leading to increased complexity of forest structure which encourages further forest habitat recovery. Hypotheses tested were therefore;

   a. The density of understorey and canopy trees increase with increasing restoration age and is higher than it is in DSF forest, due to the high density of trees initially planted (1.8 m apart).

   b. BA of older restoration forest is higher than that of younger restoration forest.

   c. The density and BA of trees in older restoration plots starts to approach that of DSF and becomes more dissimilar to that of younger forests.

   d. Tree density and BA in the control plots remains significantly lower than in the other forest plots.

   e. Crown cover of understorey and canopy trees in old age forest is higher than it is in younger age forests.

2) The FWS technique aims to create forest structure suitable for seedling recruitment. The dense understorey and canopy trees create microhabitats that support seedling establishment of non-pioneer and

animal-dispersed species more than of pioneer and non-animal-dispersed species. The hypotheses tested were, therefore;

a. Recruitment of seedlings of animal-dispersed trees increase with tree density. Seedlings of non-animal-dispersed species are not affected by tree density.

b. Increasing tree density and basal area reduces seedling establishment of pioneer (shade-intolerant) species.

c. Seedling mortality rate of pioneer species increases with increasing density and basal area of understorey and canopy trees, whilst the rate of non-pioneer species is more independent of such conditions.

5.4 Methods

5.4.1 Data Collection

In all 60 sample plots in all sites (see Chapter 2), every tree (including planted trees) taller than 1.5 m (from sapling to canopy trees) was tagged, counted and measured for diameter at breast high (cm-DBH) and height (using a digital clinometer from Haglof HCC Digital Compass/ Clinometer) between August-September in 2012 (1st census) and 2013 (2nd census).

5.4.2 Data Analysis

Trees were classified as i) understorey trees (1.5-14.9 m tall) or ii) canopy trees (taller than 15 m). Stem density (stems per 100 m^2 area) and basal area (BA) (m^2 per 100 m^2 area) (equation 5.1) were calculated for both tree classes and the mean values used to compare among forests. Noted that the unit area "per 100 m^2" is used as same as the analysis of seedlings in chapter 3 in case of the comparison among them. Between 1st and 2nd censuses, the difference in total was calculated as per cent a per year to compare changes in BA among the forest plots

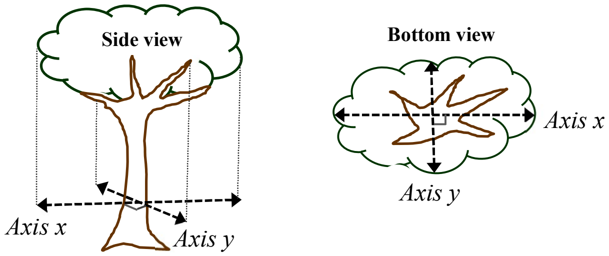
Equation 5.1: Basal Area (BA) $BA = \pi \left(\frac{DBH}{2}\right)^2$

… where DBH is diameter at breast high (cm) and π is 3.14159.

Tree density distribution and BA accumulation were used to indicate and compare tree structure among forests. Tree density and BA of each forest were classified into 5 height-categories as follows: 1) <5 m-tall, 2) 5-9.9 m-tall, 3) 10-14.9 m-tall, 4) 15-19.9 m-tall, and 5) >20 m-tall. The BA accumulation was calculated increasing of BA from top canopy categories (>20 m-tall.) to lowest categories (<5 m-tall).

Crown spread area (CSA) was used to compare crown cover between understorey and canopy trees among restoration plots of different ages, DSF and control site (equation 5.2).

Equation 5.2: Crown Spread Area $CSA = \left(\frac{Crown\ width\ x + Crown\ width\ y}{2}\right)^2$



…where, crown width X and Y = the distance of canopy width from one side of tree to another side with perpendicular axis X and Y through the centre of tree trunk (as shown in the figure above).

5.4.4 Statistical Tests

Average stem density and BA of understorey trees, canopy trees and total trees were compared among the sites and also compared between 1st and 2nd censuses. The data sets from 12 replicated plots, in each forest type, were used for statistical comparisons among the sites. To compare tree density and BA among the forest sites, Student's t-test was used in the case of normally distributed data and the Mann-Whitney test was used for non-normal data. Between the 1st and 2nd censuses, tree density and BA in each forest site were compared by paired t-tests (for normally distributed data) or by the Wilcoxon test (for non-normal data). Principle Component Analysis (PCA) was used to determine if tree density, total BA and seedling recruitment and seedling mortality rate were correlated in each forest types. The software "Paleontological Statistics, PAST, version 3.0" was used to perform the statistical tests (Hammer *et al.,* 2001; Hammer and Harper, 2006).

5.5 Results

5.5.1 Tree Density in Restoration Forests

Mean tree density in the R plots, (>1.5 m tall) was significantly higher than had been achieved by natural regeneration in the control plots (*P* <0.04 by Student's t-tests in Fig. 5.1, see hypothesis 1-d). Furthermore, it had reached and was maintained at the same level as in the remnant forest, even in the youngest restoration plots (R6) (*P* > 0.05 in Fig. 5.1, see hypothesis 1-a). Over a year, from the first to the second census, total tree density did not change significantly (*P* > 0.05) at all sites.

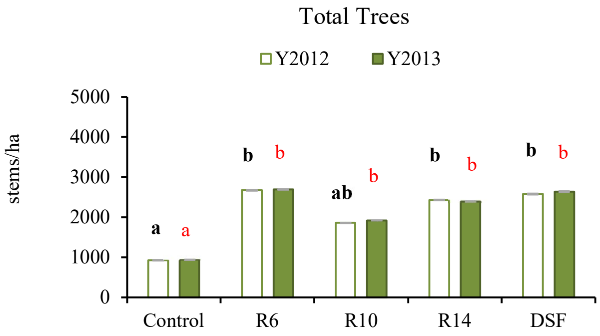


Figure 5.1: Mean density (stems per ha) of total trees in study plots (n=12) recorded in the first (Y2012) and second (Y2013) censuses. Different letters indicate significant differences (Student's t-tests among forests).

As expected understorey trees (less than 15 m-tall) vastly outnumbered canopy trees. Their density in the R plots generally exceeded that in the control plot (*P* <0.03), although not significantly so in R10. Understorey tree density was similar to that of remnant forest (except for the first survey of the R10 plots) (*P* <0.05).

Focusing on just canopy trees, density gradually increased from control site to old planted age forests (Fig. 5.2-b, see hypothesis 1-a). Mean canopy tree density in the R14 plots actually exceeded that of remnant forest (DSF) site due to planted density significantly (*P* = 0.0001 and 0.045 in 1st and 2nd census respectively (Fig. 5.2-b see hypothesis 1-a and -c).

As expected, both understorey and canopy trees in control site were obviously lower density than other forests (Fig. 5.1-2, see hypothesis 1-d). Between two censuses, the density of understorey trees in R14 forest significantly increased over a year (*P* = 0.0098), whilst canopy tree density significantly decreased (*P* = 0.011 by Paired t-test in Fig 5.2). Density of trees in R10 forests was stable. In site R6, the density of canopy tree significantly increased over one year (*P* = 0.041 in Fig 5.2-b).

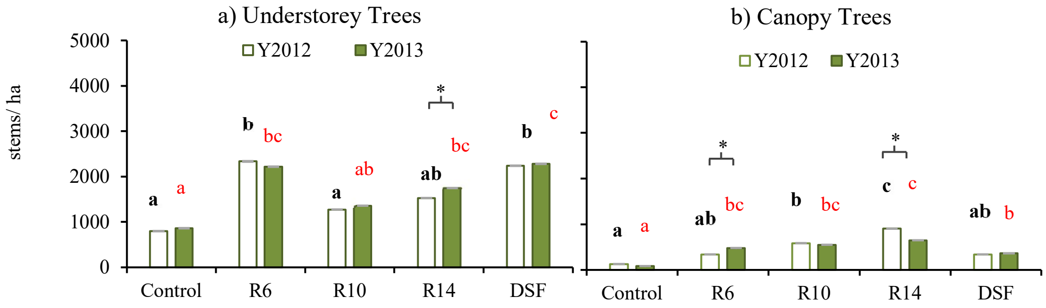


Figure 5.2: Mean density (stems per ha) of tree by height a) understory level (height between 1.5-15 m) and b) canopy level (height >15 m) of the first (Y2012) and second (Y2013) census. Different letters indicate significant differences among forests by Student's t-tests and asterisks (*) designate significant difference between censuses by Paired-t test.

Recruit trees included saplings that had grown above the 1.5 m threshold into the understorey level (recruited-understorey trees) and understorey trees that grown taller than 15 m into the canopy tree class (recruited-canopy trees), between the first and second surveys. Understorey recruits of pioneer species were found only in R6 and surprisingly in the DSF plots († in Fig. 5.3-a). Amount of non-pioneer seedling that reach understorey level are slightly increasing from control, R forests to DSF except R14 forest (Fig. 5.3-a). Recruited-canopy trees of both pioneer and non-pioneer (late successional) species decrease from young to old age of R forest because fewer FWS trees that no yet reach to canopy level. The recruited-canopy trees belong to both non-pioneer and pioneer planted FWS (Fig. 5.3-b).

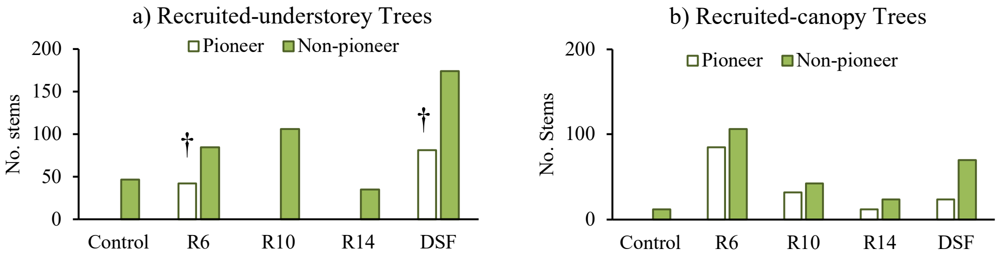


Figure 5.3: Total recruited stems (per ha) of a) understorey and b) canopy trees of pioneer and non-pioneer species in all forests.

In the R forests, 6-11 trees died (in a total 12 plots per forest) died in 1 year (Fig. 5.4-a). Almost all dead trees were understorey trees (Fig. 5.4-b). The most prevalent cause of death was cutting by villagers. Tree deaths from natural causes in the R forests were actually lower than those in the DSF and control sites. Cutting trees for timber frequently occurred over the study period. In one year, twelve trees were cut from about 3,000 m^2 (0.3 ha) area of all sample plots in the R forests and several trees were cut outside sample plots (Fig. 5.4-a). Moreover, some trees were cut at a certain height (about 1-2 m above ground) and stayed alive with some branches and leaves. These trees were not recorded as dead trees but were demoted according to their height reduction from canopy to understorey tree level. Such cutting reduced the opportunity for understorey trees to grow up and become canopy trees and opened up gaps in the R forests, which rapidly became invaded by grasses and herbs, increasing the risk of fire damage in the dry season.

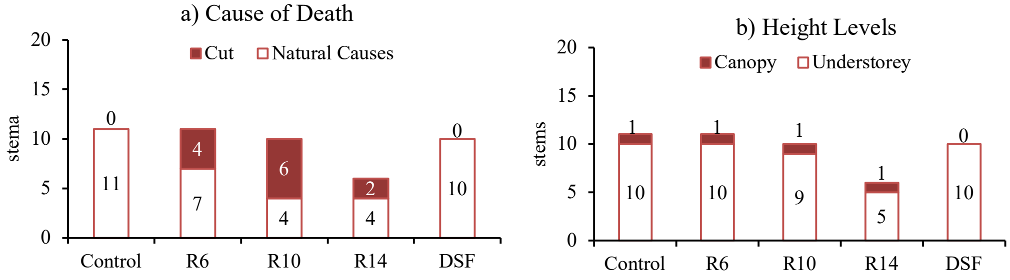


Figure 5.4: Total dead trees (per 12 plots) classified by a) cause of death and b) understorey and canopy trees in each forests.

5.5.2 Tree Basal Area (BA)

Mean basal area (BA) increased significantly with plot age, becoming the same as in natural forest by year 14 (*P* = 0.07). Trees in the oldest, R14, forest had distinctively higher total BA than trees of younger age (R10 and R6) forests (*P* = 0.01, and 0.04, site respectively in Fig. 5.5, see hypothesis 1-b). From control site to R14 forests, mean BA (of total trees) increased and approached that of DSF with no significant difference between R forests and DSF (*P* > 0.1 by Mann-Whitney test in Fig. 5.5, see hypothesis 1-c). Due to very low tree density in the control site, total BA in this site was significantly lower than in all other forest sites (see hypothesis 1 -d).

Furthermore, BA increased significantly from the first to the second survey at all the R sites (*P* = 0.005, 0.002, and 0.003, from R6, R10, to R14 respectively, in Fig. 5.5), due to tree growth, since the tree density did not increase. In contrast, the BA of trees in control and DSF sites barely changed over the same period (1 year).

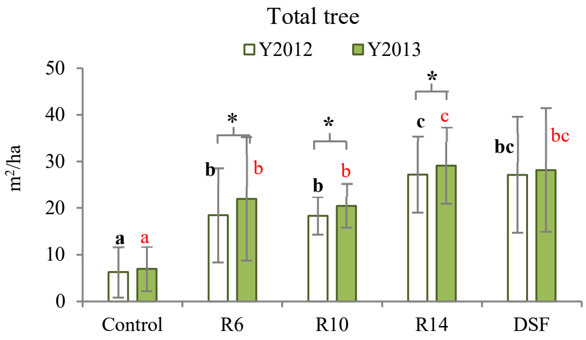


Figure 5.5: Mean basal area (BA-m^2 per ha) of trees in each forest type of the first (Y2012) and second (Y2013) census. Different letters indicate significant differences among forests in the Mann-Whitney tests. Asterisks (*) designate significant difference between censuses by Wilcoxon test.

Canopy trees contributed the most to mean BA in the R sites. Mean BA of understorey trees was significantly lower in the R forests than in the DSF plots only for the first census ($P \leq 0.037$ in Fig. 5.6-a see hypothesis 1-b). Mean BA of the canopy trees in the oldest age plots, R14, was significantly higher than in the younger plots, R10 and R6 ($P < 0.04$ in Fig. 5.6-a). However, BA of canopy trees in the R forests of all ages was not significantly different to that of the DSF site

(Fig. 5.6-b). Differences in mean BA of understorey trees among the R plots and control site (P ≥0.08 in Fig. 5.6-a), were not significant, whilst the BA of canopy trees in the control site was significantly lower than the same in the R forests (P < 0.01 in Fig. 5.6-b, see hypothesis 1 -d).

Mean BA of understorey and canopy trees did not change significantly between the two censuses at almost all sites, except for canopy trees in R14 and understorey trees in R6, where BA increased significantly over the year (*P* = 0.041 and 0.004, respectively, * in Fig. 5.6-a and –b).

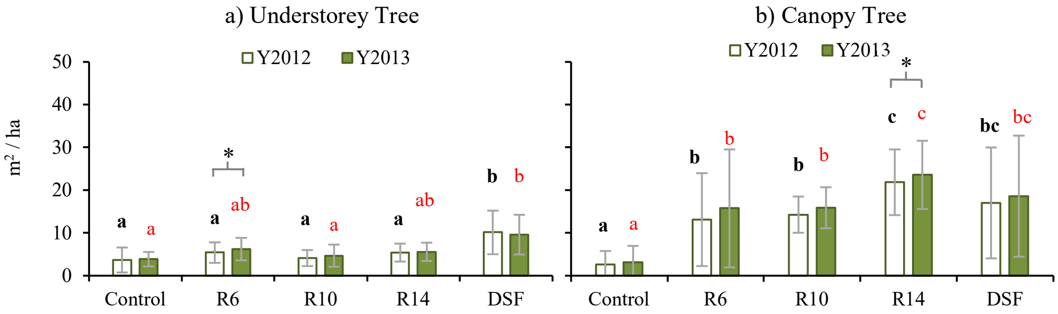


Figure 5.6: Mean basal area (m^2 per ha) of trees in each forest type, of the first (Y2012) and second (Y2013) census. Different letters indicate significant differences among forests by the Mann-Whitney tests. Asterisks (*) designate significant difference between year by Wilcoxon test with P-Value> 0.05.

Percent increase of BA over one year was significantly higher in the R6 than in R14 plots (*P* = 0.0141 by the Mann-Whitney tests). The per cent increase in BA per year was gradually decline from 19.82%, 11.49%, to 8.04% per year, from R6, R10 to R14 plots respectively ($P \leq 0.01$), however, none of 3 ages are significantly different from the rate of increase in BA per year (6.59%) in the remnant forest (DSF) ($P \geq 0.06$ in Fig. 5.7). BA in the control plots increased by about 31.18% from the 1st to 2nd census a significantly greater increase higher than in the other forest sites ($P \geq 0.01$), except for the R6 plots (*P* = 0.0926, in Fig. 5.7).

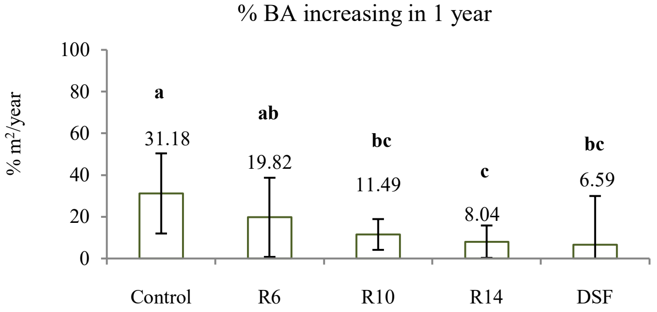


Figure 5.7: Percent increase in mean BA (% m^2/year) from the 1st to the 2nd censuses in each forest. Different letters indicate significant differences among the forests (Mann-Whitney tests $P<0.05$)

After 6 years of forest restoration, the density distribution and BA pattern of both canopy and understorey trees became more similar to those of DSF (Fig. 5.8-a and b, see hypothesis 1-c). There was a negative exponential distribution from low density of tall-canopy trees to high density of short-understorey trees in the R6 plots and DSF site (+ in Fig. 5.8-a). Furthermore, the number of trees in the R10, and R14 plots were similarly distributed from top canopy to understorey trees (* in Fig. 5.8-a). As expected, the density distribution and BA pattern in the control plots were different compared with the other forests (Fig. 5.8).

The BA accumulation pattern in the older R14 plots was similar to that of the DSF plots (Fig. 5.8-b). High BA accumulated at the lowest understorey level (<4.9 m-high), implying a similar total BA of trees for seedling community of the R14, and DSF plots. However canopy trees dominated a larger area in the R14 plots compared with the DSF plots (Fig. 5.8-b). These results suggest good development of FWS trees and rapid restoration of adequate forest structure in all the R forests, especially the oldest stand.

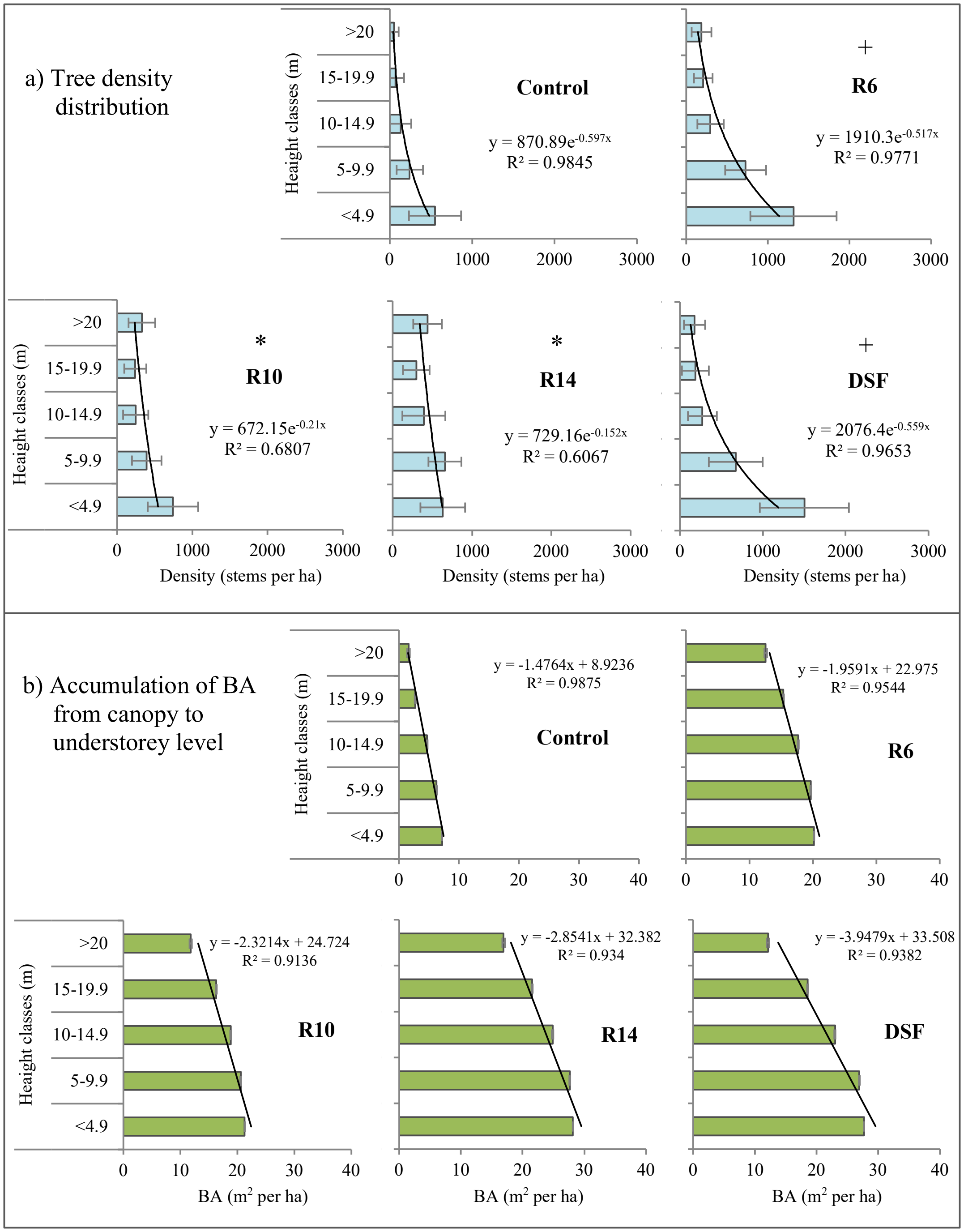


Figure 5.8: a) Tree density distribution (stems per ha) and b) tree BA accumulation from canopy to understorey levels (m^2 per ha) of each forest, average from 2 censuses, categorized by tree height with exponential (a) and linear regression (b) trend line and equation. Symbols (* and +) represent similar trend among sites.

5.5.3 Tree Crown Cover (CC) of the R Forests

Total crown cover (*ca.* 50 % per plot) and relative contributions towards crown cover from the understorey and canopy trees did not differ significantly among the differently aged R forests ($\chi^2 \geq 3.52$ in Fig. 5.9). Differences in the CC between the R10 and R14 plots were not significant ($\chi^2 = 0.06$) with canopy and understorey trees contributing towards overall CC more or less equally (Fig. 5.9). In the R6, control, and DSF plots, understorey trees contributed significantly more to total CC ($\chi^2 \leq 0.45$) than canopy trees did. The CC contribution of understorey and canopy trees in DSF plots were significantly different from the R10 and R14 plots ($\chi^2 =$ 3.95 and 5.19, respectively, in Fig. 5.9, see hypothesis 1-e).

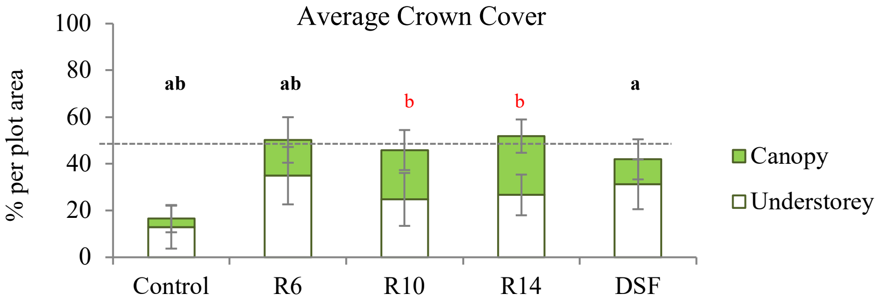


Figure 5.9: Average crown cover (CC) of understorey and canopy trees in each forest. Different letters indicate significantly different ratio of crown spread between understorey and canopy trees among forests by Chi-square tests ($\chi^2 >$ 3.84).

5.5.4 Influence of Forest Structure on Seedlings

Principle Component Analysis (PCA) revealed a negative correlation between BA and seedling recruitment rate, for tree species of both seed-dispersal modes and successional statuses (Fig. 5.10 and 5.11, respectively, see hypothesis 2-a). High density, CC and BA of understorey trees limited seedling recruitment rate of animal-dispersed species (A), whilst, high canopy trees limited those of non-animal dispersed species (Fig. 5.9~~-b~~). Therefore, the low non-animal dispersed species recruitment, found in every age of R forest, especially R10 and R14, may have been caused by very high density, CC and BA of canopy trees (doted-ellipse in Fig. 5.10, see hypothesis 2-a).

Tree density had stronger effect on animal dispersed species than it did on non-animal dispersed species. (Fig. 5.10, see hypothesis 2-a).

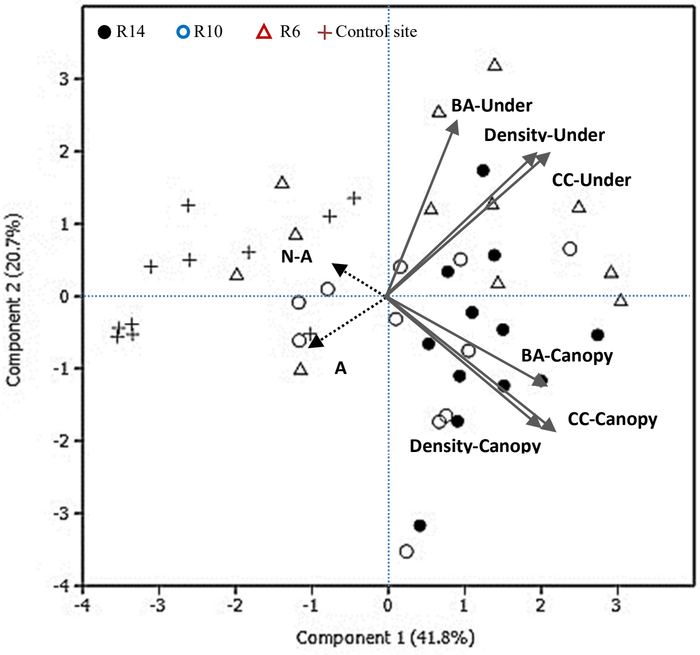


Figure 5.10: PCA between variables of tree factors (density, BA and CC) on seedling recruitment rate in R forest and control site. Dashed-arrow lines represent trend of recruitment rate of non-animal (N-A) and animal-dispersed species (A), when solid lines represent factors of understorey and canopy trees. (Each spot illustrates a sample plot in each forest, n=12, and arrow lines point out of centre indicate low to high strength of variables)

Considering species' successional status, seedling recruitment rate of non-pioneer species was limited by high BA, CC and density of understorey trees, whilst, those of pioneer species was limited by high BA , CC and density of canopy trees in the R plots. However, the recruitment rates of pioneer species were more strongly limited by high BA of canopy trees than non-pioneer species were by understorey trees (Fig. 5.11-a, see hypothesis 2-b). Accordingly, the low recruitment rate of pioneer species in many sample-plots was may have been due to the high BA of FWS canopy trees in the R forests (doted-ellipse in Fig. 5.11-a). In the control sites, low BA of canopy trees created more opportunities for the seedling establishment of pioneer species (solid-ellipse in Fig. 5.11-a).

Considering the relationship between tree structure and seedling mortality rate, the PCA showed that both tree density and basal area of understorey trees influenced the mortality rate of both pioneer and non-pioneer seedlings in the R forests (Fig. 5.11-b). The mortality rate of non-pioneer species increased with increasing BA, CC and density of understorey trees, In contrast, high BA, CC and density of understorey trees decreased mortality rate of pioneer species (Fig. 5.11-b, see hypothesis 2-c). Moreover, many sample-plots in the R6 forest had high density and BA of understorey trees, which may have lowered the mortality rate of pioneer species (dashed-ellipse in Fig. 5.11-b see hypothesis 2-c).

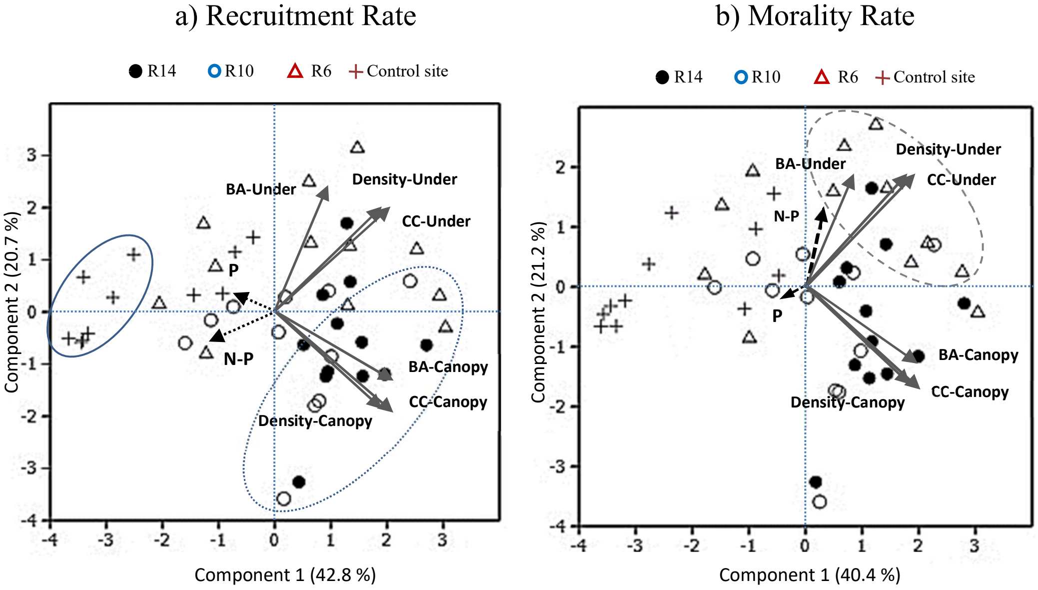


Figure 5.11: PCA between variables of tree factors (density, BA and CC) on a) seedling recruitment (dotted-arrow lines) and b) mortality rate (dashed-arrow lines) of pioneer (P) and non-pioneer species (N-P) in R forest and control site. Solid lines represent factors of understorey and canopy trees. (Each spot illustrates a sample plot in each forest, n=12, and arrow lines point out of centre indicate low to high strength of variables)

5.6 Discussion

5.6.1 Understorey and Canopy Trees in the FWS Restoration Forests

After 6 years of forest restoration by FWS technique, tree density exceeded that in natural remnant forest (DSF). The FWS method rapidly re-establishes tree cover due to dense initial planting of tree saplings (500/rai = 3,125/ha) (FORRU, 2005). After 14 years, mean total BA (per 100 m^2 area) surpassed that of the DSF and, statistically, both tree density and BA of all R plots (aged 6-14 years) did not differ from those of remnant forest (Fig.5.1 and 5.5). This result was consistent with a previous published study of secondary forest (age 8-13 yrs.) after logging in Malaysia, where Bischoff *et al.* (2005) found no significant difference in mean BA after 13 years of natural regeneration compared with primary forest. A high density of large trees in secondary forest, compared with primary forest has been reported by several researchers (Montgomery & Chazdon, 2001; Bischoff *et al.* 2005; Breugel *et al.*, 2006; Feldpausch *et al.*, 2007), plantation forests (De Souza & Batista, 2004) and restoration forests (Saha, 2001).

Tree density increased with increasing restoration plot age. Trees were larger in the older plots since they had had more time to grow, contributing most to increasing BA with age. This result also mirrored other published studies from regenerating secondary forest in Malaysia (Bischoff *et al.*, 2005), Bolivian Amazon forests (Pena-Claros, 2003), in dry tropical forest of Colombia (Rulz *et al*, 2005) and in Costa Rica (Brown & Lugo, 1990). Consequently, most understorey trees in the R6 plots were small saplings, mostly of the planted tree species.

In addition, the rapid increment in BA in the younger forest plots (R6) indicates rapid growth of the young trees there. This finding also agrees with studies in secondary forests under similar seasonal conditions, e.g. in Mexico, by Read and Lawrence (2003) and Amazonian secondary forest by Feldpausch *et al*. (2007). The slow increment of trees in oldest age forest plots (R14) similar to a study in the secondary forests of Malaysia that found mean BA increased by 7% per year (Bischoff *et al.*, 2005).

The tree size distribution and development of an understorey indicated rapid recovery of forest structure and the likely re-creation of habitats and niches for wildlife from R6 onwards tending towards the structural complexity of natural forest, although clear differences in structure could be discerned between the R6 and R10-14 plots. The younger plots consisted of dense small understorey trees with spared canopy trees, whilst the R10-14 plots had similar density of understorey and canopy trees.

Natural regeneration in the control plots, over 14 years since the trial plots had been established, failed to achieve the structural development attained in the R plots, which agrees with restoration studies in Queensland, Australia, by Tucker & Murphy (1997) that the natural regeneration sites were dominated by grass and herb with low density of canopy trees. The results of this study provide further evidence for the effectiveness of the FWS restoration technique in accelerating natural succession. Furthermore, increases in tree size and BA demonstrate the potential of FWS restoration forests to sequester large amounts of carbon at levels similar or higher than those of mono-species plantations (Petsri *et al.*, 2007; Jundang, *et al.*, 2010; Mukul, *et al.*, 2016). Carbon sequestration in the FWS restoration plots was reported about 105.8 tC/ha after 14 years that similar to carbon storage levels of those trees in nearby natural forest. (Jantawong *et al.*, 2017). Kavinchan, *et al.* (2014) report below-ground carbon in term of litterfall contained 6.82 t/ha in the 11-year-old FWS plots.

5.6.2 Influence of Trees on Seedlings of Recruit Species

Contrary to chapter hypothesis 2-a, high BA of canopy trees limited seedling establishment of non-animal-dispersed tree species, whilst, the BA, CC and density of understorey trees limited seedling establishment of animal-dispersed tree species. A dense forest canopy might act as a physical barrier that prevents wind-dispersed seeds from reaching the ground, whilst also providing habitat for perching/nesting etc. for bird/mammal seed dispersers. In the same study site, Wydhayagarn *et al.* (2009) reported many of frugivorous birds visiting in the R forests (12-year-old), which resulted in a high density of recruit seedling species establishing under individual FWS trees. Frugivorous birds are important agents of seedling establishment in pine plantations in Thailand (Oberhauser, 1997), in young forest restoration plots in Panama (Jones *et al.*, 2004), in Costa Rica (Lindell *et al.*, 2012) and in Brazil (Viani *et al.*, 2015) etc.

Consistent with this chapter's hypothesis 2-b, high BA, CC and density of canopy trees limited seedling recruitment of pioneer species. Moreover, high BA, CC and density of understorey trees both limited recruitment and enhanced mortality of non-pioneer species, whilst the condition decreased the mortality rate of pioneer species in the R forests (Fig. 5.11 and 5.12). Comita & Hubbell (2009) reported the negative effect of basal area (plant size >1cm-DBH) on seedling survival of shade tolerance (non-pioneer) species in tropical moist forest of Panama. The lack of canopy gaps might be a factor that influenced seedling recruitment. Forget (1991) reported that a dense understorey without gaps enhances seedling mortality of non-pioneer species than those under gap area. Dupuy and Chazdon, (2006) and Puerta-Pineroa, *et al.* (2007) supported that canopy gaps and a "heterogeneous environment" in regenerating forest provides opportunities for seedlings of shade-tolerant (non-pioneer) species to survive in a generally low-irradiance habitat. A previous study of trees in the same FWS forests as used in the study reported here (8 years old) indicated complete canopy closure in FWS forests (lack of gaps and more homogenous canopy structure), whilst in DSF the canopy was discontinuous (more gaps) (Jinto, 2010). Kanowski *et al.* (2003) also reported that in Australian forest restoration plots (mixture of planted trees and shrubs) also develop closed, dense canopy coverage that lacks gaps. Therefore, the continuous canopy of FWS of the R forests results in a lack of canopy gaps and a less "heterogeneous environment" that might indirectly limit seedling of recruit native species.

Many of the trees that died in the R forests had been chopped by the villagers. The local villagers regarded the restoration forests as community forest that they can access, cut and use for their own and the village's purposes. Cutting trees and fire destruction from nearby agriculture field are difficult to control. Consequently, these events resulted in a lower density and BA of trees in the R forest than they should be. Disturbance by human activities could delay forest succession as suggested by Chazdon (2003). The activities have a particularly negative effect on the slower growing non-pioneer species (Chapman *et al.,* 1999; Benıtez-Malvido & Martınez-Ramos, 2003; Farwig *et al.*, 2008). However, the cutting trees by the villager created heterogeneous environment for the R plots. The controlled tree cutting and gap formation can also increase biodiversity (Chapman & Chapman, 1999; Oberhauser, 1997; Lemeniha *et al.*, 2004; Yirdaw & Luukkanen, 2004; Dupuy & Chazdon, 2008).

In this study, the impact of human disturbances on seedling community could not be determined. Further research of the impact is needed, as well as community forest relations in aspect of social services and welfare (Sayer *et al*., 2004; Poffenberger, 2006; Jacobs, *et al*., 2015).

5.7 Conclusions

1) Forest restoration by the FWS technique returned forest structure to that typical of disturbed primary forest by 6-14 years. Within 6 years, framework trees created forest structure and complexity similar to that of natural forest. High density of understorey and canopy trees reflected the potential for forest succession in the future.

2) Density of canopy trees increased with increasing plot age. Differences in forest structure (distribution, density, or crown spread) indicated differences between the younger and older plots in niches/habitats created for wildlife. This indicated that a diverse plot system, containing plots of many different ages would be best for biodiversity conservation, rather than planting the whole area in a single year.

3) Development of forest structure in the non-planted control plots was very slow, indicating a failure of natural regeneration to achieve restoration goals.

4) Understorey trees influenced seedling dynamics more than canopy trees did. High BA, CC and density of understorey trees inhibited animal-dispersed, non-pioneer species and reduced the mortality rate of pioneer species. High BA, CC and density of canopy trees reduced seedling recruitment of non-animal-dispersed, pioneer species

5) Young restoration plots were characterized by low density of canopy trees and high density of understorey trees, which reduced seedling mortality rate of pioneer species.

6) The older plots (10- and 14-year-old) supported a high density of larger trees. High BA and density of canopy trees limited pioneer species recruitment, but low density of understorey trees supported seedlings of non-pioneer species more than those of pioneer species.

7) Human-caused disturbance of the FWS restoration forests is concerning, because it interferes with seedling dynamics, which could in turn delay succession. On the other hand, some tree cutting might create a more heterogeneous environment that encourages seedling establishment. Consequently, further research is needed to develop an appropriate community forestry management plan for the study site.

CHAPTER 6

Physical Conditions Related Seedling Dynamics

6.1 Introduction

Deforestation greatly changes light and soil conditions at ground level, exposing the soil surface to full sunlight, high temperatures, direct impact of rainfall and surface runoff, especially after intensive land use, such as an agriculture. Such conditions adversely impact soil quality of soil and water holding capacity (Islam & Weil 2000; Fukushima, *et al.* 2008; Lu *et al.*, 2016**).** Light availability and soil water content are essential resources for plant growth and survival in moist and wet tropical forests (Denslow, 1987; Mokhtaruddin, *et al.* 2001) and dry seasonal evergreen forests (Khurana & Singh, 2001; Van Kuijk, *et al.* 2014; Liu, *et al*. 2014; Lu, *et al.* 2016). Research has shown that the availability of soil water and nutrients can determine survival of recruit tree seedlings in monsoon Asia (Mokhtaruddin *et al*., 2001; Huete *et al.,* 2008) and in Neotropical forests (Chazdon, 2014). Chazdon *et al.* (2010) proposed that the response of seedlings to light is a major factor affecting regeneration dynamics, especially of understorey plants. Furthermore, several studies suggest such abiotic factors have an important impact on seedling recruitment (Valladares, 2004; Gomez Aparicio *et al.*, 2004).

The BMSM restoration area was once covered by evergreen forest that was cleared decades ago for agriculture. Two important purposes of FWS restoration are to achieve forest canopy closure and improve soil conditions through inputs of organic matter via litterfall, thus creating ideal seed bed conditions for tree seed germination and the establishment of recruit seedlings (FORRU 2005; Elliott *et al.* 2003). Therefore, this chapter aims to investigate the influence of light and soil field capacity on FWS restoration forest dynamics in order to a) determine differences in levels of abiotic factors among the 3 restoration ages, DSF and control site, and b) determine the influence of such conditions on seedling recruitment and dynamics.

6.2 Literature Review

6.2.1 Light Availability in the Forest

In secondary forest, light availability on the forest floor can be very low due canopy closure and lack of light gaps (Brown and Lugo, 1990; Bazzaz 1991; Denslow *et al.* 1991; Barbour *et. al.*, 1999; Van Kuijk *et al.*, 2014). Complexity and variation of light gaps strongly influence light distribution in secondary tropical forest of Costa Rica (Montgomery & Chazdon 2001). Species diversity of planted trees can also affect the variation in light availability and creat heterogeneity of niches for forest regeneration of tropical lowland rain forest (Rüger *et al.* 2009).

To study light availability in forest habitats, indirect methods, such as the measurement Leaf area index (LAI) and Global Site Factor (GSF) have been widely used due to limitation of cost and access **(**Clark *et al.* 1993; Valladares & Guzmán, 2006; Tang *et al.*, 2012). Clark *et al.* (1993) found that understorey light environments of almost all tropical rain forests fall within GSF = 2-11 %, whilst, pioneer species require maximum values of light about 25%. Several studies reveal variation of LAI among ecosystems. Comparing plantations of native and non-native tree species, aged 26 to 32 years in Hawaii, Harrington and Ewel (1997) reported insignificant differences in LAI among three plantations of *Eucalyptus saligna, Flindersia brayleyana,* and *Fraxinus uhdei*, although variation of native plant density was high. In a plantation (age 31 yrs.) of exotic conifers (*Cupressus lusitanica* and *Pinus patula)* and a broadleaved species plantation (aged 28-31 yrs.) of *Cordia africana* and *Eucalyptus saligna*, in southern Ethiopia, Lemenih *et al.* (2004) reported that the broadleaved species plantations had significantly lower LAI than the conifer plantation did.

Tang *et al.* (2012) published the mean values of total LAI in many kinds of forests of Costa Rica, such as, 1.74 m^2m^{-2} in open pasture, 5.2 m^2m^{-2} in secondary forests, 5.41 m^2m^{-2} in regeneration forests (after selective-logging), with the highest value being in old-growth forest, 5.62 m^2m^{-2}. In addition, light availability in deciduous forest and dry seasonal tropical is influenced by season. Light transmission increases during the dry season (Murphy & Lugo, 1986; Vieira & Scariot, 2006; McNamara, 2009). Behera *et al.* (2015) reported LAI values at ground level ranging from 1.2 to 7.7 m^2m^{-2}, during the post-monsoon season, and from 0.3 to 4.5 m^2m^{-2} during summer in dry deciduous forest of India. In the tropical seasonal forests of northern Thailand, Yoshifuji, *et al.* (2006) found that LAI increases at the end of dry season or beginning of the rainy season and decreases in the dry season.

6.2.2 Light Availability and the Seedling Community

Different light conditions in a forest habitat influence the plant community, especially seedlings. Low light levels increase seedling mortality (Chen *et al.* 1992) and decrease seedling density (Nicotra *et al.* 1999). However, in case of Pantai Aceh Forest Reserve of Malaysia, Turner (1990) found that potential direct radiation did not affect seedling mortality (varies from 6 to 30%). They also found that seedlings grew taller in gaps over the 16 months of their study period. In tropical lowland forests of Panama, Rüger *et al.* (2009) report recruitment of almost of woody plant species (from about 250 spp.) increased with increasing light. Nine species (3%) had recruitment declining with light.

Under plantations and restoration forest, Kamo *et al*. (2002), showed that canopy closure shaded out grasses and vines, enough for tree seedlings to establish in young age (12 years) monoculture plantations of exotics species (*Acacia sp.* and *Eucalyptu sp.*) and indigenous species (*Dalbergia cochinchinensis, Xylia kerrii* and *Pterocarpus macrocarpus*) at Sakaerat, Thailand. Similar evidence was found in the FWS restoration forests at BMSM (Khopai & Elliott, 2003; FOURRU, 2005). However, in the later stage of forest succession, the development of structural complexity and light heterogeneity increases seedling density in heterogeneous habitats (Nicotra *et al.* 1999; Capers *et al.* 2005). Asanok *et al.* (2013) found that soil and light conditions were strongly effected on seedling regeneration of 8-10 spp. out of 30 tree species, whilst 12-15 spp. were affected by forest type and structure in enrichment and secondary forest of northern Thailand.

6.2.3 Forest Soil Field Capacity and Seedling Community

Deforested sites are exposed to direct impact of rainfall, surface runoff and soil erosion. These conditions greatly affect soil properties (Ewel *et al*. 1991; Nepstad *et al.*, 1996; Holl, 1999) and reduce both macroporosity and saturated hydraulic conductivity (Islam & Weil, 2000; Giambelluca, *et al*. 2000; Giertz *et al*. 2005).

Islam & Weil (2000) demonstrated reduced water holding capacity in a cultivated area (22.9%) compared with forest (27.9%) in natural forest. Moreover cultivated soil was more compact and had higher bulk density than forest soil did. Low water holding capacity was caused by low soil organic matter

Planted trees add organic matter to the soil through litter fall (Bonet, 2004), which increases soil moisture capacity (Aguiar & Sala 1994). Hudson (1994) found this phenomenal happened in all soil texture groups. Bentos *et al*. (2013) point at water availability as a limiting factor for the recruitment success of Amazon secondary forest. They found higher seedling emergence and growth in high-moisture zones than in low-moisture zones. Madsen (1994) also revealed that, at the highest level of soil water content, seedling growth linearly increases with increases light.

Water deficit is a major cause of seedling death in gaps by Brown and Whitmore (1992) and Tuner (1990). An explanation was provided by Nepstad *et al.* (1996), who reported that soil moisture below the permanent wilting point inhibits seedling survival during the dry season in Brazilian pastures. Slot and Poorter (2007) showed that periodic drought strongly inhibited establishment of seedlings of early successional tree species. In a tropical seasonal forest in northern Thailand, Yoshifuji, *et al.* (2006) reported that declines in tree transpiration were directly controlled by soil moisture conditions at the beginning of the dry season. The dry period increases seedling death in open gaps of 37-year-old teak (*Tectona grandis*) plantation. Ray and Brown (1995) suggested that drought, together with the light requirements of seedling species are the main factors that determine seedling growth and survival.

Seasonal drought, along with light availability are major limiting factors for plants in the forests (Lugo *et al.*, 1978; Gerhardt, 1993; Hardwick *et al.*, 2004). Therefore, this chapter investigated two important abiotic factors, 1) light availability and 2) soil field capacity (FC), in the FWS restoration forests of BMSM. Soil field capacity (FC) was used as an indicator moisture availability during the dry season, since soils achieve field capacity during rainy season and gradually decline in soil moisture availability as the dry season progresses. The higher the FC, the longer it will take before the wilting point is reached during the dry season. Little is known of the relationship between soil field capacity (FC) and the seedling recruitment and mortality rates in restoration forests.

6.3 Objectives, Questions and Hypotheses of the Chapter

6.3.1 Objectives

This chapter evaluates the effects of light availability and soil field capacity on seedling dynamics and recruitment.

6.3.2 Questions

1) How do light availability and soil field capacity in restoration forest vary with increasing age and compared to the natural remnant forest?
2) How do light and soil field capacity relate to tree seedling recruitment and mortality rates in restoration plots of different ages?

6.3.3 Hypothesis

1) Canopy closure progressively reduces light reaching the forest floor. Hypotheses tested were, therefore:
    a. Light availability at forest floor declines with increasing plot age and approaches that of natural remnant forest.
    b. Light availability in control sites is higher than that in the forest sites, since canopy coverage is lower.
2) Light influences seedling recruitment and mortality rates in restoration forest plots. Moreover, seedlings of each successional status (pioneer and non-pioneer species) respond differently to light conditions. Therefore, the hypotheses tested were:
    a. Where light availability is low, seedling recruitment rate of non-pioneer (shade-tolerant) species is higher than that of pioneer species.
    b. Low light levels increase seedling mortality, particularly of pioneer (light-demanding) species. The mortality rate of non-pioneer species is lower than that of pioneer species.
3) Forest restoration increases water absorption due to accumulation of organic matter in the top soil. Soil field capacity (FC) indicates soil

recovery affects seedling dynamics. Therefore, the hypotheses tested were:

a. Soil field capacity (FC) increases with increasing restoration plot age and approaches that of the natural remnant forest.
b. High soil FC enhances seedling recruitment rate.
c. Increasing soil FC reduces seedling mortality.

6.4 Methods

6.4.1 Light availability

Light availability was measured by hemispherical photography (Pic. 6.1) a convenient and rapid technique. Leaf Area Index (LAI) and Global Site Factor (GSF) were derived from the photographs by "HemiView Canopy Analysis Software". Leaf Area Index (LAI) is the cumulative square metres of leaf area above 1 square meter of ground. Global Site Factor (GSF) is the proportion of global radiation calculated from both direct and diffuse radiation, penetrating forest canopy trees ("total site factor" of Anderson, 1964).

Hemispherical photographs were taken using a cannon digital camera (model f10) with a fisheye lens (Nikon 180 degree model). Photos were taken at the center of seedling sample plots 50 cm above ground at f5 with fast shutter speed and using the self-timer to prevent blurring. All shots were adjusted for horizontal level and magnetic north orientation. Leaf or branch cover 1 m above the lens and direct sun were avoided. The photographs were taken at the same time as tree seedling data collection in the wet and dry seasons (see Chapter 2). The "HemiView Software" (HemiView Canopy Analysis Software v. 2.1 1999, Delta-T Devices Ltd., Cambridge, UK) was used to analyze all hemispherical photographs for Leaf Area Indices (LAI) following Beer's Law and Global Site Factor (GSF) in equation 6.1.

Equation 6.1: GSF $G (\theta) = e – K (\theta).L$

Where: G is gap fraction, K (θ) is the extinction coefficient at angleθ, and L is LAI, θ is zenith angle (User Manual for *HemiView,* HemiView Manual Revision Number: 2.1 HemiView Manual 1996-1999 Helios Environmental Modelling Institute, LLC, Copyright 1998-1999 Delta-T Devices, Ltd.)

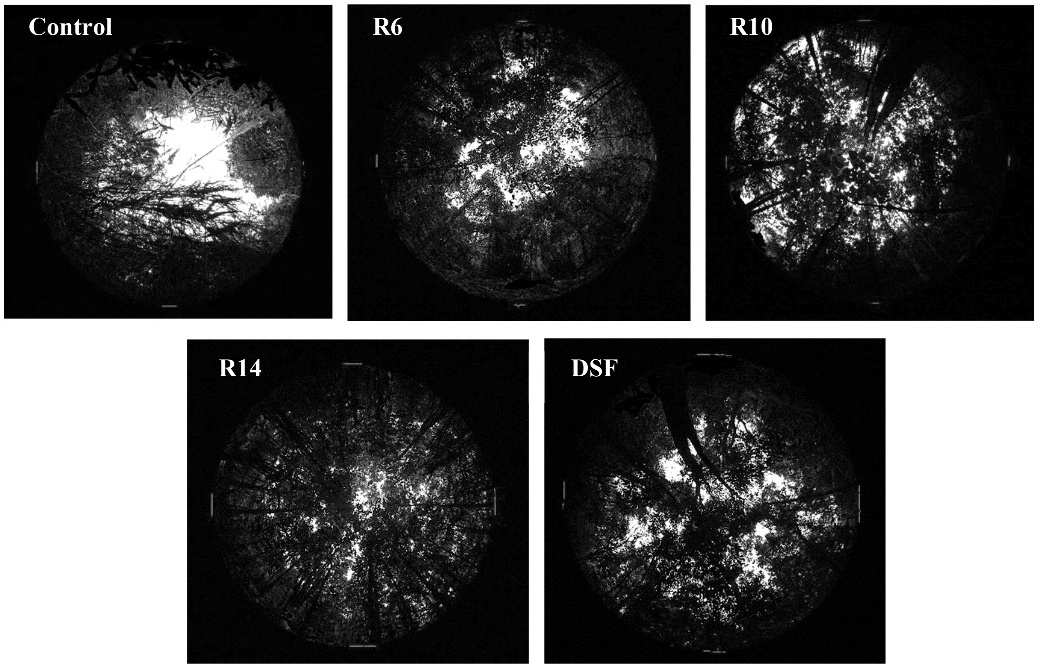


Figure 6.1: Examples of hemispherical photograph collected in each forest for LAI and GSF analysis by program "HemiView".

6.4.2 Soil Field Capacity

Field Capacity (FC) is maximum weight of water (in gm) that can be held against gravity by 1 gram of dry soil (Brady & Weil 2010 in Khamyong, *et al.* 2014). It is the water available to plants from the start of the dry season until the first rains, because by the end of the rainy season, most soils are saturated. Soil samples were collected from 12 plots at two depths 1) 0-10 cm, and 2) 10-20 cm. A plastic tube (volume 98.15 cm^3) was used for soil collection. Field capacity (FC) was measured in laboratory according to the method of Brady and Weil (2010). In laboratory, water was added into soil collections until completely saturated and allowed water to drain against gravity. Then the moisture content of the saturated soil sample (FC) was measured. Samples were oven-dried at 105°c until maintain the constant weights, and the moisture contents by volume were later measured. Moisture content (MC) was calculated using the equation 6.2 for the percentage of water by "dry soil weight (g)" and equation 6.3 for the percentage of water by "soil volume (cm^3)". Total bulk density (BD) was calculated by equation 6.4 in order to indicate the compaction of soil.

Equation 6.2: Moisture content by soil weight (g) $MC = \frac{M_w}{M_s}$

Equation 6.3: Moisture content by soil volume (cm^3) $MC = \frac{V_w}{V_t}$

Equation 6.4: Total bulk density $BD = \frac{M_w}{V_t}$

When, V_w is the water volume (1 cm^3), V_t is the total soil volume (98.15 $cm.^3$), M_w is the water weight (1 g), and M_s is dry soil mass (100 cm^3) (Brady and Weil, 2010).

6.4.3 Data Analysis

The average value of LAI and GSF, percentage of water by soil weight (g) and the percentage of water by soil volume (cm^3) at field capacity and total bulk density were calculated for each forest. Student's t-test was used to compare between sites. Paired t-tests were used for comparison between seasons within each forest site. If the data set was non-parametric, Mann-Whitney test was used to compare data between sites. The Wilcoxon test was used to compare between seasons within forest site. Principle Component Analysis (PCA) was used to find the relationship between factors (light and soil) and seedling dynamic (data from chapter 3). The software "Paleontological Statistics, PAST, version 3.0" was used to perform the statistical tests (Hammer *et al.,* 2001; Hammer and Harper, 2006).

6.5 Results

6.5.1 Light Availability under FWS Restoration Forests

LAI did not increase significantly with restoration plot age (3.1− 3.9 m^2m^{-2}, $P < 0.06$, Mann-Whitney test), although mean LAI was highest in the R14 plots. (Fig. 6.2-a) and significantly higher than in DSF ($P = 0.01$ in Fig. 6.2-a).

Mean GSF in the R6 plots was significantly higher than it was in the R10 and R14 plots ($P < 0.03$ in Fig. 6.2-b, see hypothesis 1-a). After 10 years of restoration, the mean GSF was about 8.8% (Fig. 6.2-b), but not significantly lower than in the DSF ($P > 0.19$).

In the control sites, mean LAI was significantly lower than those of the R14 plots, but invert for mean GSF ($P = 0.021$, and 0.00012, respectively, see hypothesis 1-b).

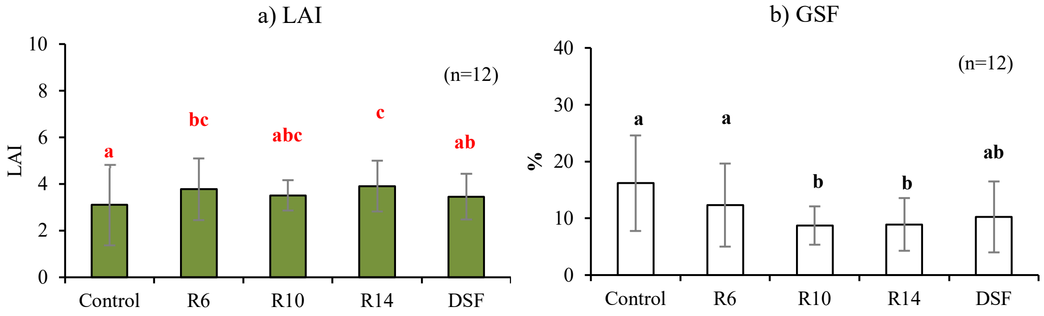


Figure 6.2: Average LAI and GSF over 4 censuses in R forest compared with DSF and control sites. Bars not sharing the same superscripts are significantly different (Mann-Whitney test $P < 0.05$).

Seasonal variations were evident. In the dry season, mean LAI and GSF were influenced by age of restoration. Mean LAI increased with plot age causing GSF to decline (Fig. 6.3). In wet season, LAI in the R6 plots was significantly higher than it was in the R10 plots ($P = 0.01$) but not in R14 forests ($P = 0.08$ in Fig. 6.3-a). However, differences in GSF among the R forest plots were insignificant (Fig. 6.3-b).

In the R6 plots and control plots, LAI and GSF changed between seasons. LAI was very low with high GSF in the dry season with the reverse being true in the wet season ($P < 0.003$, by Wilcoxon test in Fig. 6.3). However in the R10-14 plots LAI did not differ significantly between the dry and wet seasons ($P = 0.2$ and 0.75, sites respectively by Wilcoxon test in Fig 6.3-a), although, site R14 was the only site where mean GSF did not differ significantly between wet and dry seasons ($P = 0.139$). GSF in the R10 plots was not significantly different from that in R14 plots ($P= 0.175$ in Fig. 6.3). These results imply a more stable light environment throughout in R forests older than 10 years.

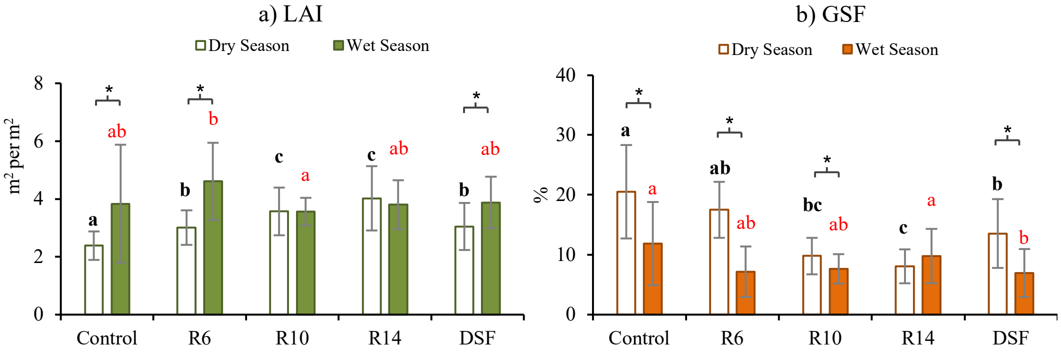


Figure 6.3: Average LAI and GSF (n=12) with standard deviation in wet and dry season of each forest. Bars not sharing the same superscript are significantly different (Mann-Whitney test, $P < 0.05$) Asterisks (*) represent significant differences between season (Wilcoxon test, $P < 0.05$).

6.5.2 Influence of Light Availability on Seedling of Native Recruit Species

The PCA showed a positive correlation between mean GSF on the seedling recruitment rate of non-pioneer species (N-P) and pioneer species (P), whilst, mean LAI negatively affected seedling recruitment rate of both non-pioneer species (N-P) and pioneer species (P) (Fig. 6.4-a, see hypothesis 2). Recruitment rates of non-pioneer species increased with increasing GSF but decreasing LAI, whilst those of pioneer species were slightly limited under high LAI (Fig. 6.4-a, see hypothesis 2-a).

Mortality rates of seedlings were also affected. The mortality rate of non-pioneer species was greatly enhanced by low GSF (Fig. 6.4-b). On the other hand, the mortality rate of pioneer species declined under high LAI (Fig. 6.4-b, see hypothesis 2-b). Many of the R6 plots had high LAI (eclipse)-resulting in low mortality rates of pioneer species (Fig. 6.4). In addition, factor lines of LAI and GSF (solid lines) were angled close to 90° to the recruitment rate of pioneer species implying a weak influence of light conditions.

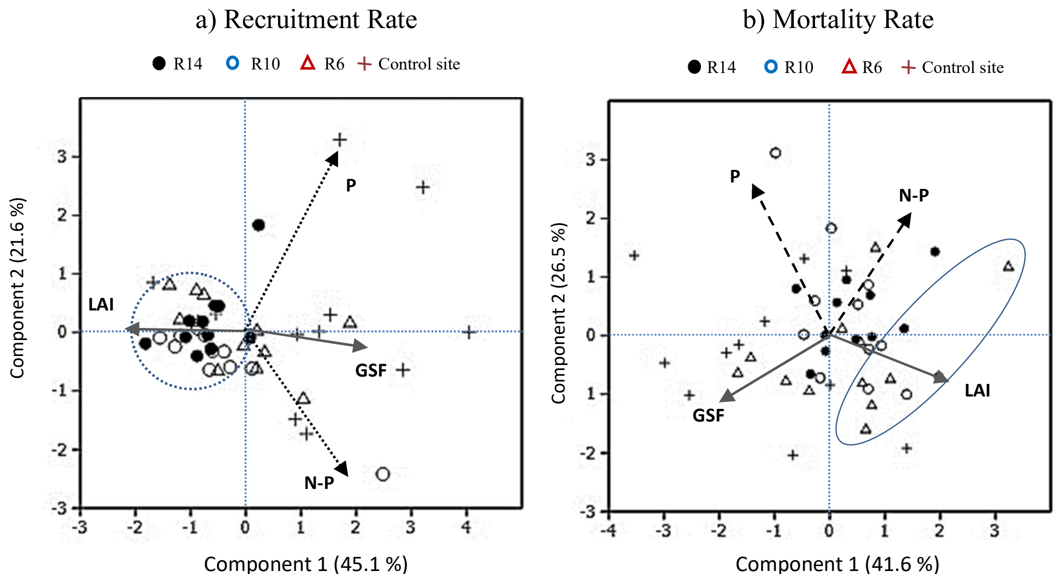


Figure 6.4: PCA between variables of light factors (LAI and GSF) on seedling a) recruitment (dotted-arrow lines) and b) mortality rate (dashed-arrow lines) of pioneer (P) and non-pioneer species (N-P) in R forest and control site. Solid lines represent factors of LAI and GSF. (Each spot illustrates a sample plot in each forest, n=12, and arrow lines point out of centre indicate low to high strength of variables)

6.5.3 Soil Field Capacity (FC) of FWS Restoration Forests

Percent moisture at field capacity (FC) in all the R forest plots was significantly lower than in the DSF plots ($P < 0.03$ by Student t-tests in Fig. 6.5) and it did not increase with plot age (hypothesis no. 3-a). Soil of site R10 had significantly lower FC, of both soil weight and volume, than other sites ($P < 0.04$ by Student t-tests in Fig. 6.5). Whilst, mean Soil FC was not significantly different among R14, R6, and control site ($P > 0.05$ in Fig. 6.5, see hypothesis 3-a). Due to the same pre-restoration land-use history and connected boundary of R14 and control site, mean soil FC was not significantly different among these two sites (Fig. 6.5).

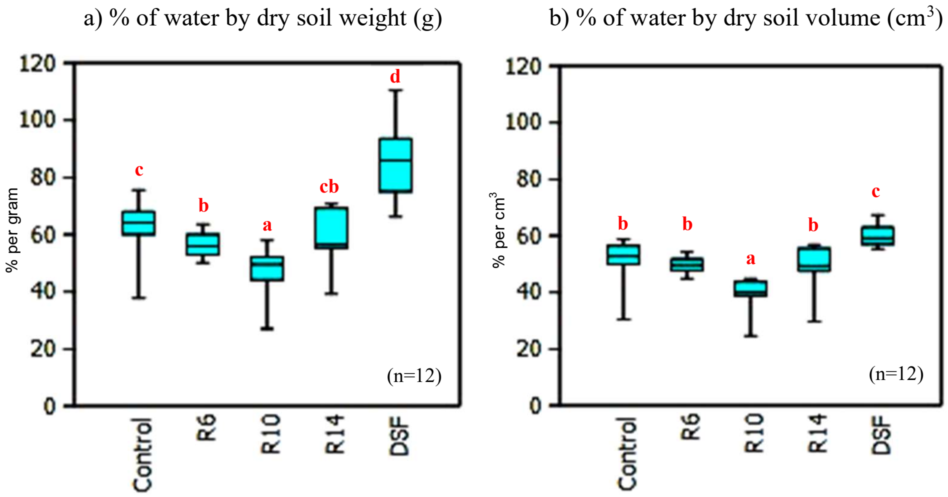


Figure 6.5**:** Soil Field Capacity (FC): the average percent of water capacity by a) dry soil weight (% per gram), and b) dry soil volume (% per cm^3) of soil at 0-20 cm depth from ground surface, in each forest. Bars not sharing the same superscripts are significantly different ($P < 0.05$).

The bulk density data indicated high soil compaction in all R forest and control plots. Bulk density in the R forest plots was high (0.84-0.9 g/cm^3, both top and deeper soil levels). Differences among plot ages insignificant ($P > 0.05$ by Student-t test in Fig. 6.6). In contrast DSF top soil significantly less dense than all other plots ($P < 0.0001$ in Fig. 6.6), although compaction increased in the deeper soil levels ($P < 0.0005$ by paired-t test, in Fig. 6.6).

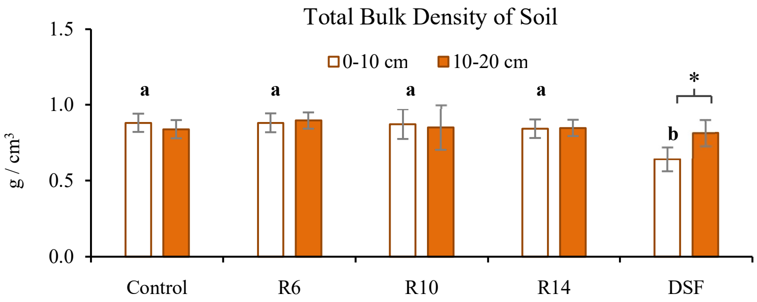
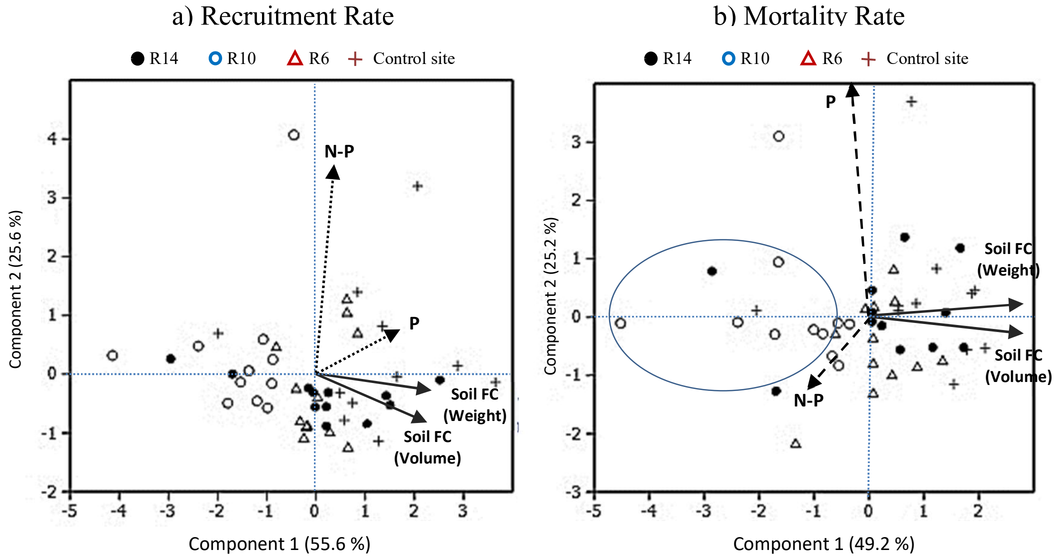


Figure 6.6**:** Average total bulk density (g/cm^3) of topsoil (0-10 cm-depth) and deeper soil (10-20 cm-depth) from ground surface in each forest with standard deviation. Bars not sharing the same superscripts are significantly different among forests (Student t-tests $P < 0.05$) Asterisk (*) designates a significant difference between the 2 soil depths (paired-t test, $P < 0.05$).

6.5.4 Influence of Soil Field Capacity on Native Recruit Species

The PCA indicated a weak influence of soil FC on seedling dynamics of native recruit species in the R forests. Increasing FC did not influence seedling recruitment rate of non-pioneer species (N-P) but slightly increased those of pioneer species (P) (Fig 6.7, see hypothesis 3-b and c). Mortality rates of N-P seedlings were increased by lower FC, whilst those of pioneer species was barely affected by them (Fig 6.7, see hypothesis 3-c). Since many of R10 plots had distinctively low soil FC, the high mortality rate of non-pioneer species found in the results of Chapter 4 might be caused by this condition together with tree structure (circles in Fig. 6.7).


a) Recruitment Rate

b) Mortality Rate

Figure 6.7: PCA between variables of soil FC factors on seedling a) recruitment (dotted-arrow lines) and b) mortality rate (dashed-arrow lines) of pioneer (P) and non-pioneer species (N-P) in R forest and control site. Solid lines represent factors of soil FC by dry soil weight and volume. (Each spot illustrates a sample plot in each forest, n=12, and arrow lines point out of centre indicate low to high strength of variables)

6.6 Discussion

6.6.1 Light Availability of Restoration Forest

The FWS restoration forests had already attained shady conditions, similar to those of natural remnant forest after 6 years, due to the rapid recovery of forest structure. Consequently plot age did not affect LAI. However, mean GSF measurements clearly indicated higher light available levels for seedlings in young (6 years) than in old age (>10 years) of the R forests. This result is consistent with the results on forest structure (Chapter 5). Moreover, Clark *et al.* (1993) found that GSF is weakly but significantly correlated with sapling height and crown position in secondary forest in Costa Rica. Therefore, the different structure of understorey and canopy trees in the area are an important factor controlling light environment in these R forests such as found in other forests (Bazzaz, 1991; Denslow, *et al.,* 1991; Montgomery & Chazdon, 2001).

Seedlings in the R6 plots were more exposed to light those in the older R plots, although tree density and LAI did not differ significantly among sites. However, LAI and GSF fluctuation by season were more similar to those of natural remnant forest than to old age R forests. Differences in structure of the understorey and forest canopy could cause light availability in the R plots as Nicotra *et al.* (1999) and Capers *et al.* (2005) suggested that variations in forest structure and the complexity lead to light heterogeneity. Furthermore, the forest structure of the R6 plots was more sensitive to seasonal change, which led to fluctuating conditions between the dry and wet seasons. This result was consistent with Yoshifuji, *et al.'s* (2006) study of tropical seasonal forest in northern Thailand mentioned earlier**.**

In contrast the R10-14 plots supported more constant low light conditions at ground level throughout a year, which could be explained by the continuous canopy with fewer, smaller gaps in such forest reported by Jinto (2010). Whitmore, (1978) and Lieberman *et al.*, (1989) (in McManara, 2009) highlighted the importance of the presence, size, and abundance of canopy gaps as affecting light heterogeneity and microclimate in primary tropical forests.

The relatively low LAI of FWS restoration forests aged 6-14 years (3.1-3.9 m^2m^{-2}) was lower than that of 5.20 m^2m^{-2} in secondary forests, 5.41 m^2m^{-2} in regeneration forests after selective-logging, and 5.62 m^2m^{-2} in old-growth forests at La Selva, Costa Rica (Tang *et al.*, 2012). However, Kalacska *et al.* (2004) ranked LAI from early to late succession in tropical secondary moist forest of Costa Rica. Using their scale, the FWS forest at age 6-14 years could be ranked as "intermediate stage of regeneration" with LAI= 2.4-4.0 m^2m^{-2}. Further study between FWS trees and light condition in a longer period might help researchers to determine effect of planted FWS tree structure on the variation of light condition such as the fluctuation of light in young age plots.

6.6.2 Influence of Light Availability on Native Recruit Species

High LAI inhibited recruitment of non-pioneer species which disproved hypothesis 2-a. In Panama, Rüger *et al.,* (2009) reported that the recruitment of almost all woody plant species (97% from total 250 spp.) increased with increasing light. As mentioned in Ch. 5, juveniles of non-pioneer tree species survived better in gaps than they did under a dense understorey without gaps (Forget, 1991). Thus, non-pioneer seedlings were also inhibited by dense LAI in the R plots. As expected, high GSF level enhanced, but weakly, recruitment of pioneer species. Whilst, the mortality rate of pioneer species declined with high LAI. Veenendaal, *et al.* (1996) reported that seedlings of pioneers are not very light sensitive because they have an intermediate response to differences in light conditions. Van Kuijk, *et al.* (2008) suggested that pioneer species have similar light-capturing efficiencies and photosynthetic rates per unit mass to non-pioneer species. Therefore, pioneer species can establish under low light condition as well as high light condition (Kitajima 1994; Poorter 1999). The correlation between of light availability and seedling species recruitment supports the previous results about forest structure in Chapter 5.

6.6.3 Soil Field Capacity

Soil of FWS restoration forest at age 6-14 years was dense and had low ability to absorb water content compared to soil in DSF, particularly the top soil (0-10 cm-depth from surface). Mean soil FC was independent of restoration age due to the overriding influence of previous land use history: an effect shown in several other studies (Giambelluca, *et al*. 1999; Giertz *et al*. 2005; Kalacska *et al*., 2004). Nevertheless, there were signs that FC was increasing over time. Initial moisture at field capacity of R14 forest was 34.8% in 1998 when the plots were first established (Elliott *et al.* (2000) rising to 49.5 ±7.8 % by 2013 after 14 years. Furthermore, earlier studies of litter in this FWS forests provided evidence of increasing organic matter in the R10-14 plots over time. Organic matter in the R14 plots had increased from 5.37 % in 1998 just before restoration activities commenced (Elliott *et al.*, 2000) to about 7 % after 11 years of restoration in 2009 (Kavinchan, 2014). Kavinchan (2014) also reported high litterfall by 7 and 11 years after restoration commenced of 0.11-0.64 t/ha/month and 0.17 – 0.65 t/ha/month (in the R10 and R14 plots used in my study).

The distinctly lower FC of R10 soil compared with the other R plots, DSF and control sites may be explained by former cultivation of the sites affecting hydraulic conductivity (Islam & Weil, 2000; Giertz *et al.* 2005). Unfortunately, land-use history, intensity, and activities of the FWS restoration sites could not be determined. There were some planted FWS, such as *Hovenia dulcis* Thunb., that plated in order to recover soil nitrogen and nutrient condition. However, the result suggested that plated FWS alone might not sufficient for soil FC improvement. Plant species that have complex root system can loosen compact soil and improve soil FC (Greacen, & Sands, 1980). Soil microbial agents are important factor for the decomposition system (Islam & Weil, 2000) that helps to add organic matter into soil (Greacen, & Sands, 1980). Therefore, further research of soil FC condition and the factor that helps to accelerate soil improve are needed.

6.6.4 Soil Field Capacity and Native Recruit Seedlings

FC did not appear to strongly influence seedling species recruitment. However, in seasonally dry zones, several studies suggest that drought is a main cause of seedling mortality, e.g. in Ghana (Lieberman & Li, 1992), in Mediterranean forest (Hartman & McCarthy, 2004), in tropical moist forest, Panama (Engelbrecht *et al.*, 2005). Rainfall records show that BMSM has a 5-6 month dry period each year. The low soil FC of many R10 plots (Fig.6.6) might correspond with the high mortality rate of non-pioneer species of R10 forests found in chapter 4. The lower BA of canopy trees in R10 than in the R14 plots (Fig.5.6-b) should lower seedling mortality rate of non-pioneer species, however, the mortality rate of both forests were similar (Fig.4.11). Since soil FC was low in all R forests, soil improvement might positively influence seedling recruitment. Therefore, further research of soil FC improvement effected on seedling recruitment is suggested.

6.7 Conclusions

1) Light availability in the R plots had returned to levels similar to those of natural forest from 6 years onwards. Differences in mean light availability among different ages of R plots and the DSF site were not significant.
2) Seasonal fluctuations in light availability occurred in the R6, control and the DSF plots, whilst, light availability in the R10-R14 plots was more stable throughout a year.
3) Mean light availability in the control plots was similar to that in the R6 plots, although dense grasses and shrubs were the main shade source in the control plots (compared with tree crowns in the R plots).
4) High LAI and low GSF decreased seedling recruitment rate of non-pioneer and pioneer species. Whilst, mortality rate of non-pioneer species was enhanced by high GSF, and those of pioneer species by high LAI.
5) After 14 years, soil FC of the R plots was significantly lower than soil of natural remnant forest (DSF). Further research for soil FC improvement is recommended.
6) FC was not related with R forest age. R10 had lowest soil FC compared with other sites.

CHAPTER 7

Summary

7.1 Overview

The results presented in this thesis demonstrate native species recruitment and progressive forest recovery by the FWS method. In the following table (7.1), the results of each analysis are summarized in relation to the hypotheses that they were designed to test. Finally, relationships among factors that affect recruitment of native species seedlings are summarized in the model presented in Fig. 7.1.

Table 7.1: Study's hypothesis summary

Hypothesis	Supported	Evidence	Explanation
3.1. Seed tree species that are common in the landscape surrounding restoration sites become more highly represented amongst the recruit tree seedling community beneath the canopy of restoration forests than do those that are rarer.	Yes	Fig. 3.2	− Adult tree density of recruited species was 3 times higher, on average, than those of absent species. − Seed availability is a limiting factor for species recruitment in the FWS plots.
3.2-a. Seedling recruitment is higher in older plots than it is in younger plots and the seedling population density of recruit species increases with increasing plot age.	No	Fig. 3.3	Seedling recruitment of native species was similar across all R plots regardless of age. (see 4.2-b)
3.2-b. As the number of species already present in the community approaches that of the species richness of natural remnant forest, the rate of species recruitment gradually declines with increasing plot age.	No	Fig. 3.4	− Species recruitment was not affected by R plot age. − About half of the native tree species in the surrounding landscape had yet to recruit into the R plots.

Table 7.1: (cont.)

Hypothesis	Supported	Evidence	Explanation
3.2-c. Overall, seedling population density declines with increasing plot age, as the canopy of the restored forest becomes dense and shades out young seedlings.	No	Fig. 3.5	− Seedling density measurements were skewed upwards by dominance of a few super-abundant species in the R6 and R14 plots. However, such species affected only a very limited area within each plot.
3.2-d. Species diversity of the seedling community increases with increasing plot age, trending towards that of remnant forest (DSF).	No/Yes	Fig. 3.6-b, Fig. 3.8	− Super-abundant species resulted in low species diversity of the seedling communities of the R6 and R14 plots. − When super-abundant species were excluded from the diversity calculations, species diversity of seedlings increased with increasing plot age towards that of DSF
3.2-e. The species composition of the tree seedling community becomes more similar to that of (1) natural remnant forest and less similar to that of (2) the control plots.	(1) Yes (2) No	Table 3.4-b	(1) The R14 Seedling communities were of more similar species composition to the DSF communities than they were to the R6-10 plots (2) The "spillover effect" caused similarity of the seedling community between the control and R14 plots.
3.3-a. Seedling growth rates increase with increasing plot age, as canopy closure optimizes environmental conditions to favour seedling establishment.	No	Fig. 3.9	Mean growth rates of native species did not differ significantly among R plot ages (see 4.3-b).
3.3-b. Seedling mortality rates decline with increasing plot age as habitat conditions for seedling survival improve.	No	Fig. 3.12-b	Mortality rates of recruit native species increased with increasing plot age. (see 5.2-c)
3.3-c. Seedling turnover rates gradually decline with increasing plot age.	(1) No (2) Yes	Fig. 3.14, Fig. 3.15	(1) The turnover rates of *recruit species* did not differ significantly among plots of different ages. (2) The seedling community of *all species* are more stable in the older plots.

Table 7.1: (cont.)

Hypothesis	Supported	Evidence	Explanation
4.1. Species absent from the R forest plots are more likely to have large, animal-dispersed seeds, whilst recruit species are more likely to be small-seeded species.	No	Fig. 4.1, Fig. 4.2, Fig. 4.5-b and c	− Seed size and dispersal modes did not significantly affect species recruitment in the R plots. − Consequently, seed availability was a stronger factor that influenced species recruitment in the R plots than seed size and dispersal mode were.
4.2-a. Older plots support relatively more large-seeded, animal-dispersed species in the seedling community compared with younger plots, whilst small-seeded seedling species are more equally distributed across all ages of restoration plots.	No	Fig. 4.2, Fig. 4.3	− Small-seeded animal-dispersed seedlings were common in all R plots because of abundant seed sources in the surrounding landscape. − The control site failed to recruit seedlings of large-seeded species.
4.2-b (1) In older plots, seedling recruitment rate of animal-dispersed species is higher compared with younger plots. (2) Non-animal dispersed tree species distribute equally across all plots of all ages.	(1) No (2) Yes	Fig. 4.6-a, Fig. 4.9,	(1) Seedling recruitment rates of animal-dispersed species were similar among R plots of all ages. (2) Seedlings of non-animal-dispersed species were equally distributed across R plots of all ages. (see 5.2-b)
4.3-a. Non-pioneer (shade-tolerant) species recruit more frequently in the older plots than in the younger plots. Conversely, pioneer species seedlings recruit in the younger plots more frequently than they do in the older plots.	No	Fig. 4.9 Fig. 4.8-a	− Seedling recruitment rates of both non-pioneer and pioneer species were not related to R plot age. − However, pioneer species represented a higher seedling proportion than non-pioneer species in the youngest plots (R6), than in the older plots (R10 & R14)
4.3-b. The growth rates of pioneer species decline with increasing plot age. Conversely, seedlings of non-pioneer species grow better in older plots than pioneer species do.	No	Fig. 4.10	− Relative growth rates of both non-pioneer and pioneer species were not influenced by plot age.
4.3-c. Seedling mortality of non-pioneer species declines and that of pioneer species increases with increasing plot age.	No	Fig. 4.11	− The mortality rates of both non-pioneer and pioneer species did not differ significantly among R plots of all ages. − However, the mortality rate of pioneer seedlings in the R6 plots was lower than it was in the R10-14 plots (see 5.2-c).

Table 7.1: (cont.)

Hypothesis	Supported	Evidence	Explanation
5.1-a. The density of (1) understorey and (2) canopy trees increase with increasing restoration age and is higher than that of DSF forest, due to the high density of trees initially planted (1.8 m apart).	(1) No (2) Yes	Fig 5.1 Fig 5.2	(1) Understorey trees grew more densely in the R6, plots than they did in the R10-14 plots. Many of planted trees had not grown up into the main canopy level in R6 plots. (2) The FWS canopy trees were larger in the older R plots due to the longer growth period. Tree density in all the R plots exceeded that of DSF.
5.1-b. Basal area (BA) of older restoration forest is higher than that of younger restoration forest.	Yes	Fig. 5.5 Fig. 5.6	− BA was becoming the same as in natural forest by year 14. − Canopy trees contributed the most to mean BA in the R sites. − Mean BA of understorey trees was significantly lower in the R forests than in the DSF plots
5.1-c. The density and BA of trees in the older restoration plots approach those of DSF and become more dissimilar to those of younger forests.	No	Fig. 5.5 Fig. 5.8	From 6 years onward, the density and BA of trees had reached and were maintained at the same levels as in the DSF plots.
5.1-d. Tree density and BA in the control plots should remain significantly lower than those in the other forest plots.	Yes	Fig. 5.1 Fig. 5.2 Fig. 5.5 Fig. 5.6	− Both density and BA of trees in control site were poorly developed compared with the R plots. − Many control plots were dominated by tall grasses and shrubs.
5.1-e. Crown cover of understorey and canopy trees in older forest is higher than it is in younger forests.	No	Fig. 5.9	Total crown cover and relative contributions towards crown cover from the understorey and canopy trees did not differ significantly among the planted ages from 6 to 14 years old (see 6.1)
5.2-a. Recruitment of seedlings of animal-dispersed trees increases with tree density. Recruitment of non-animal-dispersed seedling species are not affected by tree density.	No	Fig. 5.10 Fig. 5.11	− High BA, CC and density of understorey trees caused low seedling recruitment rate of seed-dispersal species − High BA , CC and density of canopy trees caused low non-animal dispersed species recruitment

Table 7.1: (cont.)

Hypothesis	Supported	Evidence	Explanation
5.2-b. Increasing tree density and basal area reduces seedling establishment of pioneer (shade-intolerant) species.	Yes	Fig. 5.11	− High BA of canopy trees strongly limited the recruitment rates of pioneer species. − However, high BA, CC and density of understorey trees also limited non-pioneer species recruitment.
5.2-c. Seedling mortality rate of (1) pioneer species increases with the increasing density and basal area of understorey and canopy trees, whilst (2) the rate of non-pioneer species is more independent of such conditions	(1) No (2) No	Fig. 5.12	(1) High BA and density of understorey trees decreased mortality rate of pioneer species (2) The mortality rate of non-pioneer species increased with increasing BA of understorey trees.
6.1-a. (1) Light availability at the forest floor declines with increasing plot age and (2) approaches that of natural remnant forest.	(1) No (2)Yes	Fig. 6.2	− Mean LAI did not differ among R plots of all ages. However, mean GSF in the R6 plots was significantly higher than it was in the R10 and R14 plots − Light availability in the young plots fluctuated seasonally, whilst it was more stable in older plots.
6.1-b. Light availability in the control plots is higher than that in the forest sites, since canopy coverage is lower.	Yes	Fig. 6.2	− Lower tree density caused higher light availability in the control plots than in the R plots. − Large-grasses were main shade source in most of the control plots.
6.2-a. Where light availability is low, seedling recruitment rate of non-pioneer (shade-tolerant) species is higher than that of pioneer species.	No	Fig. 6.4	− The recruitment of non-pioneer species was limited under high LAI and low GSF. − However, recruitment of pioneer species slightly increased with increasing GSF and decreasing LAI.
6.2-b. Low light levels increase seedling mortality especially of (1) pioneer (light-demanding) species. (2) The mortality rate of non-pioneer species is lower than that of pioneer species.	(1) No (2) No	Fig. 6.4	(1) The mortality rate of pioneer species was increased by low LAI. (2) The mortality rate of non-pioneer species was enhanced by low GSF.

Table 7.1: (cont.)

Hypothesis	Supported	Evidence	Explanation
6.3-a. Soil field capacity (FC) increases with increasing R plot age and approaches that of the natural remnant forest.	No	Fig. 6.5	− After 14 years, soil FC of all the R forests was still lower than that of the DSF plots, probably reflecting previous land use (for agriculture). − Further soil improvement may be needed.
6.3-b. High soil FC enhances seedling recruitment rate.	Yes	Fig. 6.7	− Soil FC positively influenced recruitment of pioneer seedling species − However, soil FC was less important as a factor that influenced seedling recruitment than light conditions and tree structure were.
6.3-c. Increasing soil FC reduces seedling mortality.	Yes	Fig. 6.7	High soil FC reduces the mortality rate of non-pioneer species.

The FWS technique recovered forest habitat, brought about high seedling recruitment of native species and accelerated forest succession. Within 6 years, FWS forest restoration achieved two major milestones of structural recovery, high tree basal area (BA) and canopy closure, with BA exceeding that of natural remnant forest. Consequently, the development of FWS trees reduced light availability at ground level in R forests from at least 6 years old onwards, resulting in light conditions similar to those of natural remnant forest. The model shown in Fig 7.1 highlights the importance of forest structure, light conditions and seed available as the major factors that affect seedling species recruitment.

Whether or not a species recolonized the restoration plots depended more on its abundance in the surrounding landscape, rather than on seed size and seed-dispersal mode. This study suggested that tree species that are absent or rare in the surrounding landscape should be planted intermixed with accepted FWS, to enhance diversity, since they are unlikely to recruit without such intervention. Knowledge of seed-tree abundance in areas around restoration sites can therefore be extremely useful in the planning of successful forest restoration projects.

After 10 years of restoration, canopy cover became continuous canopy (Jinto, 2009), resulting in a stable low-light environment that limited seedling recruitment. Seedlings of pioneer species in particular were negatively influenced by habitat conditions in the older forest. In the long term it is likely that seedlings pioneer species will continue to decline in older restoration plots. Nonetheless, long-term monitoring of seedling community in the R forest would help researchers to understand and predict the forest succession of the FWS forest.

After 14 years, soil compaction and low water holding capacity persisted in all the R forest plots, perhaps reflecting a prolonged legacy from previous cultivation. Although such conditions did not appear to affect forest development, research on soil improvement might yield methods to further accelerate forest recovery. Trees with complex root system (Greacen, & Sands, 1980; McNabb & Froehlich, 1983) and/or soil invertebrates such as earth worms or soil microbial agents (Islam & Weil, 2000) might help to loosen compact soil of post-agriculture history area. Further study of trees' root systems and soil microbial agents are therefore recommended.

FWS forests appear to catalyze wider-scale landscape restoration and as such could be usefully incorporated into the "applied nucleation" technique proposed by Corbin & Holl (2012). This method involves planting small islands or nuclei of forest from which tree spread their seeds outwards, eventually leading to coalescence of the nuclei. However, the rate at which seedlings of FWS trees invaded adjacent control plots was quite slow. Nevertheless, trials of combinations of forest several restoration are recommended. For example, combining nuclei of FWS forest with ANR in the surrounding landscape should accelerate spread and eventual coalescence of the nuclei and might achieve restoration goals more cost-effectively than planting whole areas with FWS (see more researches of restoration and plantation forests in Appendix D).

7.2 Future Research Suggestion

1) Since super-abundant species affect species diversity and the recruitment rate of non-pioneer species in a community, further research is needed to investigate the potential positive or negative effects of controlling such species.
2) To improve soil conditions, investigations of tree root-systems, soil invertebrates and soil microbes are needed, particularly on former agricultural sites, which retain a legacy of previous long-term soil degradation.
3) Disturbance, such as fire and small scale tree cutting, found during the study period, could have both immediately negatively effects (i.e. destroying seed sources and seedlings) and longer term positive effects (i.e. creating heterogeneity in the light/shade conditions). Therefore, further research is needed on the effects of disturbance and recommendation for prevention or control as needed.
4) In order to balance exploitation and conservation in framework forests, studies of restoration resilience are needed. This would enable managers to understand and predict "when, what and how" to harvest timbers and forest products in restored forest, in ways that support both ecological functioning as well as forest utility. Such studies would raise public awareness of the value of forest restoration, among local communities.

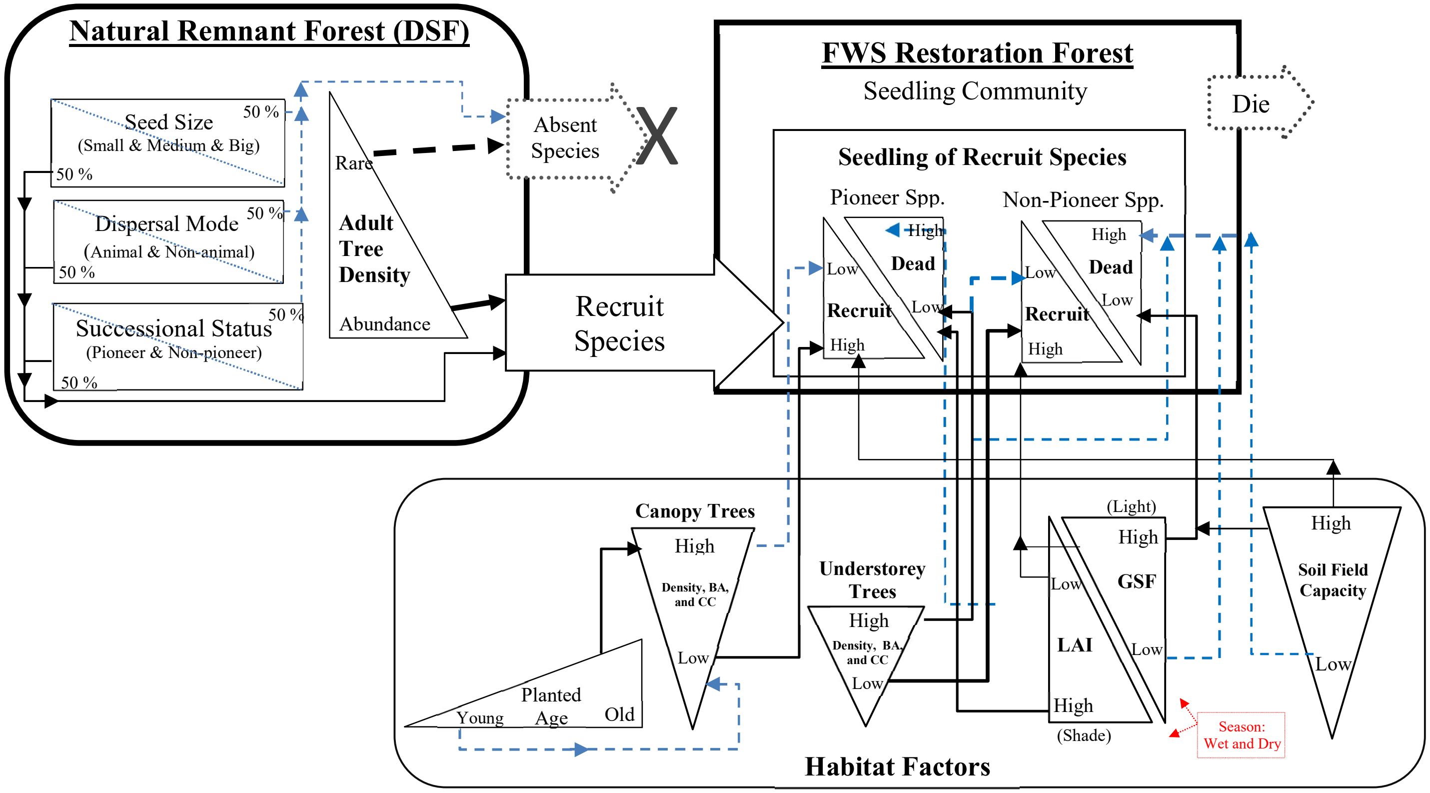


Figure 7.1: Relationship model of factors that affect dynamics and recruitment of native species seedlings in the FWS restoration forest of BMSM. Solid- line arrows present positive (+) effect, and dashed-line arrows present negative (-) effect between factors. Thickness of line represents higher degree of influence

REFERENCES

Alvarez-Aquino, C., Guadalupe Williams-Linera, and Adrian C. Newton. (2004). *Experimental Native Tree Seedling Establishment for the Restoration of a Mexican Cloud Forest Restoration Ecology*. Vol. 12, No. 3, pp. 412-418.

Anderson, M. C. (1964). Studies of the woodland light climate: I. The photographic computation of light conditions. The Journal of Ecology, 27-41.

Asanok, L., Marod, D., Duengkae, P., Pranmongkol, U., Kurokawa, H., Aiba, M., ... & Nakashizuka, T. (2013). Relationships between functional traits and the ability of forest tree species to reestablish in secondary forest and enrichment plantations in the uplands of northern Thailand. *Forest ecology and management*, 296, 9-23.

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

Babaasa, D., Eilu, G., Kasangaki, A., Bitariho, R., & McNeilage, A. (2004). Gap characteristics and regeneration in Bwindi Impenetrable National Park, Uganda. *African Journal of Ecology*, 42(3), 217-224.

Baraloto, C., Goldberg, D. E., & Bonal, D. (2005). Performance trade-offs among tropical tree seedlings in contrasting microhabitats. *Ecology*, 86(9), 2461-2472.

Barbour, M. G., Burk, J. H., & Pitts, W. D. (1980). *Terrestrial plant ecology.* Benjamin/Cummings.

Barbour, M. G., Burk, J. H., Pitts, W. D., Gilliam, F. S., & Schwartz, M. W. (1999). Methods of sampling the plant community. *Terrestrial Plant Ecology, 3rd Edition. Wesley Longman, Inc., Don Mills, ON*. p649.

Bazzaz, F. A. (1991). Regeneration of tropical forests: physiological responses of pioneer and secondary species. *Rain forest regeneration and management*, 91-118.

Bazzaz, F. A. (1996). *Plants in changing environments: linking physiological, population, and community ecology*. Cambridge University Press.

Beckage, B., Clark, J. S., Clinton, B. D., & Haines, B. L. (2000). A long-term study of tree seedling recruitment in southern Appalachian forests: the effects of canopy gaps and shrub understories. *Canadian Journal of Forest Research,* 30(10), 1617-1631.

Behera, S. K., Behera, M. D., & Tuli, R. (2015). An indirect method of estimating leaf area index in a tropical deciduous forest of India. *Ecological indicators*, 58, 356-364.

Benayas, J. M. R., Newton, A. C., Diaz, A., & Bullock, J. M. (2009). Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. *Science*, 325(5944), 1121-1124.

Bentos, T. V., Nascimento, H. E., & Williamson, G. B. (2013). Tree seedling recruitment in Amazon secondary forest: Importance of topography and gap micro-site conditions. *Forest ecology and management*, 287, 140-146.

Bischoff, W., Newbery, D. M., Lingenfelder, M., Schnaeckel, R., Petol, G. H., Madani, L., & Ridsdale, C. E. (2005). Secondary succession and dipterocarp recruitment in Bornean rain forest after logging. *Forest Ecology and Management,* 218(1-3), 174-192.

Brady, N. C., & Weil, R. R. (2010). Elements of the nature and properties of soils (No. 631.4 B733E.). *Upper Saddle River, NJ: Pearson Educational International*.

Brodie, J. F. (2007). *Effects of seed dispersal by gibbons, sambar, and muntjac on Choerospondias axillaris demography, and the disruption of this mutualism by wildlife poaching* (Doctoral dissertation, University of Montana).

Brown, N. D., & Whitmore, T. C. (1992). Do dipterocarp seedlings really partition tropical rain forest gaps?. *Phil. Trans. R. Soc. Lond.* B, 335(1275), 369-378.

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

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

Capers, R. S., Chazdon, R. L., Brenes, A. R., & Alvarado, B. V. (2005). Successional dynamics of woody seedling communities in wet tropical secondary forests. *Journal of Ecology*, 93(6), 1071-1084.

Carnevale, N. J., & Montagnini, F. (2002). Facilitating regeneration of secondary forests with the use of mixed and pure plantations of indigenous tree species. *Forest ecology and management,* 163(1-3), 217-227.

Catterall, C. P., Kanowski, J., & Wardell-Johnson, G. W. (2008). Biodiversity and new forests: interacting processes, prospects and pitfalls of rainforest restoration. *Living in a dynamic tropical forest landscape*, 510-525.

Chazdon, R. L. (2014). *Second growth: the promise of tropical forest regeneration in an age of deforestation*. University of Chicago Press.

Chazdon, R. L., & Pearcy, R. W. (1991). The importance of sunflecks for forest understory plants. *BioScience*, 760-766.

Chazdon, R. L., Finegan, B., Capers, R. S., Salgado-Negret, B., Casanoves, F., Boukili, V., & Norden, N. (2010). Composition and dynamics of functional groups of trees during tropical forest succession in northeastern Costa Rica. *Biotropica*, 42(1), 31-40.

Chazdon, R. L., Letcher, S. G., Van Breugel, M., Martínez-Ramos, M., Bongers, F., & Finegan, B. (2007). Rates of change in tree communities of secondary Neotropical forests following major disturbances. Philosophical Transactions of the Royal Society B: *Biological Sciences*, 362(1478), 273-289.

Cheke, A. S., Nanakorn, W., & Yankoses, C. (1979). Dormancy and dispersal of seeds of secondary forest species under the canopy of a primary tropical rain forest in northern Thailand. *Biotropica*, 88-95.

Chua, S. C., Ramage, B. S., Ngo, K. M., Potts, M. D., & Lum, S. K. (2013). Slow recovery of a secondary tropical forest in Southeast Asia. *Forest ecology and management*, 308, 153-160.

Clark, D. B., Clark, D. A., & Rich, P. M. (1993). Comparative analysis of microhabitat utilization by saplings of nine tree species in neotropical rain forest. *Biotropica*, 397-407.

Coll, L., Potvin, C., Messier, C., & Delagrange, S. (2008). Root architecture and allocation patterns of eight native tropical species with different successional status used in open-grown mixed plantations in Panama. *Trees,* 22(4), 585.

Comita, L. S., & Hubbell, S. P. (2009). Local neighborhood and species' shade tolerance influence survival in a diverse seedling bank. *Ecology*, 90(2), 328-334.

Connell, J. H. (1970). A predator-prey system in the marine intertidal region. I. Balanus glandula and several predatory species of Thais. *Ecological Monographs*, 40(1), 49-78.

Corbin, J. D., & Holl, K. D. (2012). Applied nucleation as a forest restoration strategy. *Forest Ecology and Management*, 265, 37-46.

Corlett, R. T. (1995). Tropical secondary forests. *Progress in Physical Geography*, 19(2), 159-172.

Corlett, R. T. (1998). Frugivory and seed dispersal by vertebrates in the Oriental (Indomalayan) Region. *Biological reviews*, 73(4), 413-448.

Corlett, R. T., & Hau, B. C. (2000). Seed dispersal and forest restoration. *Forest restoration for wildlife conservation*, 317-325.

Cosset, C. C., & Edwards, D. P. (2017). The Effects of Restoring Logged Tropical Forests on Avian Phylogenetic and Functional Diversity. *Ecological Applications.*

Cusack, D., & Montagnini, F. (2004). The role of native species plantations in recovery of understory woody diversity in degraded pasturelands of Costa Rica. *Forest Ecology and Management*, 188(1), 1-15.

Dalling, J. W., Hubbell, S. P., & Silvera, K. (1998). Seed dispersal, seedling establishment and gap partitioning among tropical pioneer trees. *Journal of Ecology*, 86(4), 674-689.

De Jong, W. (2010). Forest rehabilitation and its implication for forest transition theory. *Biotropica*, 42(1), 3-9.

De La Peña-Domene, M., Martínez-Garza, C., & Howe, H. F. (2013). Early recruitment dynamics in tropical restoration. *Ecological Applications*, 23(5), 1124-1134.

De Souza, F. M., & Batista, J. L. F. (2004). Restoration of seasonal semideciduous forests in Brazil: influence of age and restoration design on forest structure. *Forest ecology and Management*, 191(1), 185-200.

Denslow, J. S. (1987). Tropical rainforest gaps and tree species diversity. *Annual review of ecology and systematics*, 18(1), 431-451.

Denslow, J. S., & Guzman, G. (2000). Variation in stand structure, light and seedling abundance across a tropical moist forest chronosequence, Panama. *Journal of Vegetation Science,* 11(2), 201-212.

Duncan, R. S., & Chapman, C. A. (2003). Consequences of plantation harvest during tropical forest restoration in Uganda. *Forest Ecology and Management*, 173(1-3), 235-250.

Dupuy, J. M., & Chazdon, R. L. (2008). Interacting effects of canopy gap, understory vegetation and leaf litter on tree seedling recruitment and composition in tropical secondary forests. *Forest Ecology and Management*, 255(11), 3716-3725.

eFloras (2008). Published on the Internet http://www.efloras.org [accessed March 2012]*' Missouri Botanical Garden, St. Louis, MO & Harvard University Herbaria, Cambridge, MA.

Elliott, S. (1994). The Effects of Urbanization on Soi Suthep-Pui National Park. *Urbanization and Forests.*

Elliott, S. (2000). Defining forest restoration for wildlife conservation. *Forest restoration for wildlife conservation. Chiang Mai University, Thailand*, 13-17.

Elliott, S., Kuaraksa, C., Tunjai, P., Toktang, T., Boonsai, K., Sangkum, S., ... & Blakesley, D. (2012). Integrating scientific research with community needs to restore a forest landscape in northern Thailand: a case study of Ban Mae Sa Mai. In *A goal-oriented approach to forest landscape restoration* (pp. 149-161). Springer Netherlands.

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), 177-191.

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

Eriksson, O., & Ehrlén, J. (2008). *Seedling recruitment and population ecology. Seedling ecology and evolution*. Cambridge University Press, Cambridge, 239-254.

FAO. (2010-a). *Global forest resources assessment 2010*, FAO Forestry Department, Food and Agriculture Organization of the United Nations. Available: http://www.fao.org/forestry/fra/fra2010/en/.

FAO. (2010-b). *Global forest resources assessment 2010: country report, Thailand.* FAO Forestry Department, Food and Agriculture Organization of the United Nations. Available: http://www.fao.org/forestry/fra/67090/en/tha/

Farwig, N., Sajita, N., & Böhning-Gaese, K. (2009). High seedling recruitment of indigenous tree species in forest plantations in Kakamega Forest, western Kenya. *Forest ecology and management,* 257(1), 143-150.

Florentine, S. K., Pohlman, C. L., & Westbrooke, M. E. (2016). The effectiveness of different planting frameworks for recruitment of tropical rainforest species on ex-rainforest land. *Restoration Ecology*, 24(3), 364-372.

Forget, P. M. (1997). Effect of microhabitat on seed fate and seedling performance in two rodent-dispersed tree species in rain forest in French Guiana. *Journal of Ecology*, 693-703.

FORRU. (2005). *How to Plant a Forest: The Principles and. Practice of Restoring Tropical Forests*. Biology Department, Science Faculty, Chiang Mai University, Thailand.

FORRU. (2008). *Research for Restoring Tropical Forest Ecosystems: A Practical Guide*. Biology Department, Science Faculty, Chiang Mai University, Thailand.

Fukushima, M., Kanzaki, M., Hara, M., Ohkubo, T., Preechapanya, P., & Choocharoen, C. (2008). Secondary forest succession after the cessation of swidden cultivation in the montane forest area in Northern Thailand. *Forest Ecology and Management*, 255(5-6), 1994-2006.

Geldenhuys, C. J. (1997). Native forest regeneration in pine and eucalypt plantations in Northern Province, South Africa. *Forest Ecology and Management,* 99(1), 101-115.

Gerhardt, K. (1993). Tree seedling development in tropical dry abandoned pasture and secondary forest in Costa Rica. *Journal of vegetation science*, 4(1), 95-102.

Giambelluca, T. W., Fox, J., Yarnasarn, S., Onibutr, P., & Nullet, M. A. (1999). Dry-season radiation balance of land covers replacing forest in northern Thailand. *Agricultural and Forest Meteorology*, 95(1), 53-65.

Giambelluca, T. W., Nullet, M. A., Ziegler, A. D., & Tran, L. (2000). Latent and sensible energy flux over deforested land surfaces in the eastern Amazon and northern Thailand. *Singapore Journal of Tropical Geography,* 21(2), 107-130.

Giertz, S., Junge, B., & Diekkrüger, B. (2005). Assessing the effects of land use change on soil physical properties and hydrological processes in the sub-humid tropical environment of West Africa. *Physics and Chemistry of the Earth, Parts A/B/C*, 30(8), 485-496.

Gomes, E. P. C., Mantovani, W., & Kageyama, P. Y. (2003). Mortality and recruitment of trees in a secondary montane rain forest in southeastern Brazil. *Brazilian Journal of Biology*, 63(1), 47-60.

Gómez-Aparicio, L., Zamora, R., Gómez, J. M., Hódar, J. A., Castro, J., & Baraza, E. (2004). Applying plant facilitation to forest restoration: A meta-analysis of the use of shrubs as nurse plants. *Ecological Applications*, 14(4), 1128-1138.

Google Maps. (2010). Suthep-pui National Park, Chiang Mai, Thailand : Street map. Available: https://www.google.co.th/maps/@18.8453622,98.9096995,13z?hl=th

Goosem, S. & Tucker, N.I.J. (2013). *Repairing the Rainforest (second edition*). Wet Tropics Management Authority and Biotropica Australia Pty. Ltd. Cairns.

Gorchov, D. L., & Trisel, D. E. (2003). Competitive effects of the invasive shrub, *Lonicera maackii* (Rupr.) Herder (Caprifoliaceae), on the growth and survival of native tree seedlings. *Plant Ecolog*y, 166(1), 13-24.

Greacen, E. L., & Sands, R. (1980). Compaction of forest soils. A review. *Soil Research,* 18(2), 163-189.

Hammer, Ø, Harper, D.A.T., & Ryan, P.D. (2001). PAST: Paleontological statistics software package for education and data analysis. *Palaeontologia Electronica* 4(1): 9.

Hammer, Ø. & Harper, D.A.T. (2006). *Paleontological Data Analysis*. Blackwell.

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), 203-214.

Harms, K. E., Wright, S. J., Calderón, O., Hernández, A., & Herre, E. A. (2000). Pervasive density-dependent recruitment enhances seedling diversity in a tropical forest. *Nature*, 404(6777), 493.

Harrington, R. A., & Ewel, J. J. (1997). Invasibility of tree plantations by native and non-indigenous plant species in Hawaii. *Forest Ecology and Management*, 99(1-2), 153-162.

Harris, C. J., Leishman, M. R., Fryirs, K., & Kyle, G. (2012). How does restoration of native canopy affect understory vegetation composition? Evidence from riparian communities of the Hunter Valley Australia. *Restoration Ecology,* 20(5), 584-592.

Harrison, R. D., & Swinfield, T. (2015). Restoration of logged humid tropical forests: An experimental programme at Harapan Rainforest, Indonesia. *Tropical Conservation Science*, 8(1), 4-16.

Hartman, K. M., & McCarthy, B. C. (2004). Restoration of a forest understory after the removal of an invasive shrub, Amur honeysuckle (*Lonicera maackii*). *Restoration Ecology*, 12(2), 154-165.

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., Reid, J. L., Chaves-Fallas, J. M., Oviedo-Brenes, F., & Zahawi, R. A. (2017). Local tropical forest restoration strategies affect tree recruitment more strongly than does landscape forest cover. *Journal of Applied Ecology,* 54(4), 1091-1099.

Hooper, E. R., Legendre, P., & Condit, R. (2004). Factors affecting community composition of forest regeneration in deforested, abandoned land in Panama. *Ecology,* 85(12), 3313-3326.

Hooper, E., Legendre, P., & Condit, R. (2005). Barriers to forest regeneration of deforested and abandoned land in Panama. *Journal of Applied Ecology*, 42(6), 1165-1174.

Huete, A. R., Restrepo-Coupe, N., Ratana, P., Didan, K., Saleska, S. R., Ichii, K., ... & Gamo, M. (2008). Multiple site tower flux and remote sensing comparisons of tropical forest dynamics in Monsoon Asia. *Agricultural and Forest Meteorology*, 148(5), 748-760.

Hunt, R. (1982). Plant growth analysis: second derivatives and compounded second derivatives of splined plant growth curves. *Annals of Botany*, 50(3), 317-328.

Hutchings, M. J., John, E. A., & Wijesinghe, D. K. (2003). Toward understanding the consequences of soil heterogeneity for plant populations and communities. *Ecology*, 84(9), 2322-2334.

Islam, K. R., & Weil, R. R. (2000). Land use effects on soil quality in a tropical forest ecosystem of Bangladesh. Agriculture, Ecosystems & Environment, 79(1), 9-16.

Jantawong, K., Elliott, S., & Wangpakapattanawong, P. (2017). Above-Ground Carbon Sequestration during Restoration of Upland Evergreen Forest in Northern Thailand. *Open Journal of Forestry*, 7(02), 157.

Janzen, D. H. (1970). Herbivores and the number of tree species in tropical forests. *The American Naturalist*, 104(940), 501-528.

Jinto, P., 2010. *Trees and Ground Flora Diversity of Framework Species Plantation Plots and Dong Seng Forest, Mae Rim District, Chiang Mai Province.* M.Sc. Dissertations. Chiang Mai University. Thailand.

Jones, E. R., Wishnie, M. H., Deago, J., Sautu, A., & Cerezo, A. (2004). Facilitating natural regeneration in Saccharum spontaneum (L.) grasslands within the Panama Canal Watershed: effects of tree species and tree structure on vegetation recruitment patterns. *Forest Ecology and Management*, 191(1), 171-183.

Kaewkrom, P., Gajaseni, J., Jordan, C. F., & Gajaseni, N. (2005). Floristic regeneration in five types of teak plantations in Thailand. *Forest Ecology and Management*, 210(1), 351-361.

Kageyama, P. Y., & Castro, C. D. A. (1989). Sucessão secundária, estrutura genética e plantações de espécies arbóreas nativas. IPEF, *Piracicaba*, 41(42), 83-93.

Kageyama, P. Y., & Gandara, F. B. (2000). Recuperação de áreas ciliares. Matas ciliares: conservação e recuperação, 2, 249-269.

Kalacska, M., Sánchez-Azofeifa, G. A., Rivard, B., Calvo-Alvarado, J. C., Journet, A. R. P., Arroyo-Mora, J. P., & Ortiz-Ortiz, D. (2004). Leaf area index measurements in a tropical moist forest: A case study from Costa Rica. *Remote Sensing of Environment,* 91(2), 134-152.

Kamo, K., Vacharangkura, T., Tiyanon, S., Viriyabuncha, C., Nimpila, S., & Doangsrisen, B. (2002). Plant species diversity in tropical planted forests and implication for restoration of forest ecosystems in Sakaerat, Northeastern Thailand. *Japan Agricultural Research Quarterly: JARQ*, 36(2), 111-118.

Kanowski, J., Catterall, C. P., Wardell-Johnson, G. W., Proctor, H., & Reis, T. (2003). Development of forest structure on cleared rainforest land in eastern Australia under different styles of reforestation. *Forest ecology and Management*, 183(1), 265-280.

Kavinchan, N. (2014). *Soil Carbon Sequestration and Dynamics of Natural Forest Ecosystems and Forest Restoration Plots in Mae Rim District, Chiang Mai Province*. Ph.D. Dissertations. Chiang Mai University. Thailand.

Keenan, R., Lamb, D., Woldring, O., Irvine, T., & Jensen, R. (1997). Restoration of plant biodiversity beneath tropical tree plantations in Northern Australia. *Forest Ecology and Management*, 99(1-2), 117-131.

Khamyong, S., Lykke, A. M., Seramethakun, D., & Barfod, A. S. (2003). Species composition and vegetation structure of an upper montane forest at the summit of Mt. Doi Inthanon, Thailand. *Nordic Journal of Botany*, 23(1), 83-97.

Khamyong, S., Seeloy-ounkeaw, T., Anongrak, N., & Sri-ngernyuang, K. (2014). Water storages in plants and soils in two community forests of Karen tribe, northern Thailand. *Tropics*, 23(3), 111-125.

Khopai, O., Elliot, S., Sim, H. C., Appanah, S., & Durst, P. B. (2003). The effects of forest restoration activities on the species diversity of naturally establishing trees and ground flora. *Bringing back the forests: Policies and practices for degraded lands and forests.* Food and Agriculture Organization (FAO), Bangkok, Thailand, 295-315.

Khurana, E. K. T. A., & Singh, J. S. (2001). Ecology of seed and seedling growth for conservation and restoration of tropical dry forest: a review. *Environmental conservation*, 28(1), 39-52.

Khurana, E., & Singh, J. S. (2001). Ecology of tree seed and seedlings: implications for tropical forest conservation and restoration. *Current Science*, 748-757.

King, D. A. (1990). Allometry of saplings and understorey trees of a Panamanian forest. *Functional ecology*, 27-32.

Kitajima, K. (1994). Relative importance of photosynthetic traits and allocation patterns as correlates of seedling shade tolerance of 13 tropical trees. *Oecologia*, 98(3-4), 419-428.

Kitamura, S., Suzuki, S., Yumoto, T., Poonswad, P., Chuailua, P., Plongmai, K., ... & Suckasam, C. (2004). Dispersal of Aglaia spectabilis, a large-seeded tree species in a moist evergreen forest in Thailand. *Journal of Tropical Ecology,* 20(4), 421-427.

Kobe, R. K. (1999). Light gradient partitioning among tropical tree species through differential seedling mortality and growth. *Ecology*, 80(1), 187-201.

Koonkhunthod, N., Sakurai, K., & Tanaka, S. (2007). Composition and diversity of woody regeneration in a 37-year-old teak (Tectona grandis L.) plantation in Northern Thailand. *Forest ecology and management*, 247(1), 246-254.

Kraft, N. J., Valencia, R., & Ackerly, D. D. (2008). Functional traits and niche-based tree community assembly in an Amazonian forest. *Science*, 322(5901), 580-582.

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

Lan, G., Getzin, S., Wiegand, T., Hu, Y., Xie, G., Zhu, H., & Cao, M. (2012). Spatial distribution and interspecific associations of tree species in a tropical seasonal rain forest of China. *PloS one*, 7(9), e46074.

Lasky, J. R., Uriarte, M., Boukili, V. K., Erickson, D. L., John Kress, W., & Chazdon, R. L. (2014). The relationship between tree biodiversity and biomass dynamics changes with tropical forest succession. *Ecology letters*, 17(9), 1158-1167.

Laurance, S. G., Laurance, W. F., Nascimento, H. E., Andrade, A., Fearnside, P. M., Rebello, E. R., & Condit, R. (2009). Long-term variation in Amazon forest dynamics. *Journal of Vegetation Science,* 20(2), 323-333.

Lee Allen, H., Fox, T. R., & Campbell, R. G. (2005). What is ahead for intensive pine plantation silviculture in the South?. *Southern Journal of Applied Forestry*, 29(2), 62-69.

Lemenih, M., Gidyelew, T., & Teketay, D. (2004). Effects of canopy cover and understory environment of tree plantations on richness, density and size of colonizing woody species in southern Ethiopia. *Forest Ecology and Management*, 194(1), 1-10.

Lieberman, D., & Li, M. (1992). Seedling recruitment patterns in a tropical dry forest in Ghana. *Journal of Vegetation Science*, 3(3), 375-382.

Lieberman, M., Lieberman, D., & Peralta, R. (1989). Forests are not just Swiss cheese: canopy stereogeometry of non-gaps in tropical forests. *Ecology*, 70(3), 550-552.

Lima, T. A., & Vieira, G. (2013). High plant species richness in monospecific tree plantations in the Central Amazon. *Forest ecology and management*, 295, 77-86.

Lindell, C. A., Cole, R. J., Holl, K. D., & Zahawi, R. A. (2012). Migratory bird species in young tropical forest restoration sites: effects of vegetation height, planting design, and season. *Bird Conservation International,* 22(1), 94-105.

Liu, N., Zhu, W., Sun, Z., Yang, L., Yuan, S., & Ren, H. (2014). Canopy size dependent facilitations from the native shrub Rhodomyrtus tomentosa to the early establishment of native trees Castanopsis fissa and Syzygium hancei in Tropical China. *Restoration ecology*, 22(4), 509-516.

Lozada, T., De Koning, G. H. J., Marché, R., Klein, A. M., & Tscharntke, T. (2007). Tree recovery and seed dispersal by birds: comparing forest, agroforestry and abandoned agroforestry in coastal Ecuador. *Perspectives in Plant Ecology, Evolution and Systematics*, 8(3), 131-140.

Lu, X., Zang, R., Ding, Y., & Huang, J. (2016). Changes in biotic and abiotic drivers of seedling species composition during forest recovery following shifting cultivation on Hainan Island, China. *Biotropica*, 48(6), 758-769.

Ludwig, J. A., & Reynolds, J. F. (1988). *Statistical ecology: a primer in methods and computing* (*Vol. 1*). John Wiley & Sons.

Lugo, A. E. (1992). Comparison of tropical tree plantations with secondary forests of similar age. *Ecological monographs*, 62(1), 1-41.

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

Lugo, A. E., Gonzalez-Liboy, J. A., Cintron, B., & Dugger, K. (1978). Structure, productivity, and transpiration of a subtropical dry forest in Puerto Rico. *Biotropica*, 278-291.

Madsen, P. (1994). Growth and survival of Fagus sylvatica seedlings in relation to light intensity and soil water content. *Scandinavian Journal of Forest Research*, 9(1-4), 316-322.

Markesteijn, L., & Poorter, L. (2009). Seedling root morphology and biomass allocation of 62 tropical tree species in relation to drought-and shade-tolerance. *Journal of Ecology*, 97(2), 311-325.

Martinez-Ramos, M., & Soto-Castro, A. (1993). Seed rain and advanced regeneration in a tropical rain forest. *Vegetatio*, 107(1), 299-318.

Maxwell, J.F., & Elliott. S. (2001). *Vegetation and vascular flora of Doi Suthep-Pui National Park, Northern Thailand*. CMU Herbarium, Department of Biology, Faculty of Science, Chiang Mai University, In Thai Studies in Biodiversity Project of BRT. 250pp.

McLachlan, S. M., & Bazely, D. R. (2003). Outcomes of longterm deciduous forest restoration in southwestern Ontario, Canada. *Biological Conservation*, 113(2), 159-169.

McNabb, D. H., & Froehlich, H. A. (1983). *Minimizing soil compaction in Pacifc Northwest forests.*

McNamara, S. (2009). *Natural Regeneration and Management of Secondary Forests in Lao PDR and Vietnam.* (Doctoral dissertation, The University of Queensland)

Metz, M. R., Sousa, W. P., & Valencia, R. (2010). Widespread density-dependent seedling mortality promotes species coexistence in a highly diverse Amazonian rain forest. *Ecology,* 91(12), 3675-3685.

Milberg, P., & Lamont, B. B. (1997). Seed/cotyledon size and nutrient content play a major role in early performance of species on nutrient-poor soils. *The New Phytologist,* 137(4), 665-672.

Mokhtaruddin, A. M., Majid, N. M., Yusoff, M. K., Hanum, I. F., Azani, A. M., & Kobayashi, S. (2001). Soil factors affecting growth of seedlings in logged-over tropical lowland forest in Pasoh, Negeri Sembilan, Malaysia. In *Rehabilitation of degraded tropical forest ecosystems: workshop proceedings, 2-4 November 1999, Bogor, Indonesia.* CIFOR, Bogor, Indonesia.

Montgomery, R. A. (2004). Effects of understory foliage on patterns of light attenuation near the forest floor. *Biotropica*, 36(1), 33-39.

Montgomery, R. A., & Chazdon, R. L. (2001). Forest structure, canopy architecture, and light transmittance in tropical wet forests. *Ecology*, 82(10), 2707-2718.

Murphy, P. G., & Lugo, A. E. (1986). Ecology of tropical dry forest. *Annual review of ecology and systematics,* 17(1), 67-88.

Nagai, M., & Yoshida, T. (2006). Variation in understory structure and plant species diversity influenced by silvicultural treatments among 21-to 26-year-old Picea glehnii plantations. *Journal of Forest Research*, 11(1), 1-10.

Nagaike, T., Hayashi, A., Abe, M., & Arai, N. (2003). Differences in plant species diversity in Larix kaempferi plantations of different ages in central Japan. *Forest ecology and management*, 183(1-3), 177-193.

Nagaike, T., Kamitani, T., & Nakashizuka, T. (2005). Effects of different forest management systems on plant species diversity in a Fagus crenata forested landscape of central Japan. *Canadian Journal of Forest Research*, 35(12), 2832-2840.

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

Nichol, J. E., Abbas, S., & Fischer, G. A. (2017). Spatial patterns of degraded tropical forest and biodiversity restoration over 70-years of succession. *Global Ecology and Conservation*, 11, 134-145.

Nicotra, A. B., Chazdon, R. L., & Iriarte, S. V. (1999). Spatial heterogeneity of light and woody seedling regeneration in tropical wet forests. *Ecology*, 80(6), 1908-1926.

Niskanen, A. (1998). Financial and economic profitability of reforestation in Thailand. *Forest Ecology and Management*, 104(1), 57-68.

Oberhauser, U. (1997). Secondary forest regeneration beneath pine (*Pinus kesiya*) plantations in the northern Thai highlands: a chronosequence study. *Forest Ecology and Management,* 99(1), 171-183.

Parrotta, J. A., Knowles, O. H., & Wunderle Jr, J. M. (1997). Development of floristic diversity in 10-year-old restoration forests on a bauxite mined site in Amazonia. *Forest Ecology and Management*, 99(1-2), 21-42.

Petit, B., & Montagnini, F. (2006). Growth in pure and mixed plantations of tree species used in reforesting rural areas of the humid region of Costa Rica, Central America. *Forest Ecology and Management*, 233(2-3), 338-343.

Poorter, L. (1999). Growth responses of 15 rain-forest tree species to a light gradient: the relative importance of morphological and physiological traits. *Functional ecology*, 13(3), 396-410.

Popma, J., & Bongers, F. (1988). The effect of canopy gaps on growth and morphology of seedlings of rain forest species. *Oecologia*, 75(4), 625-632.

Powers, J. S., Haggar, J. P., & Fisher, R. F. (1997). The effect of overstory composition on understory woody regeneration and species richness in 7-year-old plantations in Costa Rica. *Forest Ecology and Management,* 99(1), 43-54.

Raman, T. R., Mudappa, D., & Kapoor, V. (2009). Restoring Rainforest Fragments: Survival of Mixed-Native Species Seedlings under Contrasting Site Conditions in the Western Ghats, India. *Restoration Ecology*, 17(1), 137-147.

Ray, G. J., & Brown, B. J. (1995). Restoring Caribbean dry forests: evaluation of tree propagation techniques. *Restoration ecology*, 3(2), 86-94.

Reid, J. L., Holl, K. D., & Zahawi, R. A. (2015). Seed dispersal limitations shift over time in tropical forest restoration. *Ecological Applications*, 25(4), 1072-1082.

Ribbens, E., Silander, J. A., & Pacala, S. W. (1994). Seedling recruitment in forests: calibrating models to predict patterns of tree seedling dispersion. *Ecology*, 75(6), 1794-1806.

Richards, M. (1996). Protected areas, people and incentives in the search for sustainable forest conservation in Honduras. *Environmental Conservation*, 23(3), 207-217.

Righelato, R., & Spracklen, D. V. (2007). Carbon mitigation by biofuels or by saving and restoring forests?. *SCIENCE-NEW YORK THEN WASHINGTON-*, 317(5840), 902.

Royo, A. A., & Carson, W. P. (2006). On the formation of dense understory layers in forests worldwide: consequences and implications for forest dynamics, biodiversity, and succession. *Canadian Journal of Forest Research*, 36(6), 1345-1362.

Rüger, N., Huth, A., Hubbell, S. P., & Condit, R. (2009). Response of recruitment to light availability across a tropical lowland rain forest community. *Journal of Ecology*, 97(6), 1360-1368.

Saha, S. (2001). Vegetation composition and structure of Tectona grandis (teak, Family Verbanaceae) plantations and dry deciduous forests in central India. *Forest ecology and management*, 148(1), 159-167.

Sangsupan, H. A., Hibbs, D. E., Withrow-Robinson, B. A., & Elliott, S. (2018). Seed and microsite limitations of large-seeded, zoochorous trees in tropical forest restoration plantations in northern Thailand. *Forest Ecology and Management*, 419, 91-100.

Sansevero, J. B. B., Prieto, P. V., de Moraes, L. F. D., & Rodrigues, P. J. P. (2011). Natural regeneration in plantations of native trees in lowland Brazilian Atlantic Forest: community structure, diversity, and dispersal syndromes. *Restoration Ecology*, 19(3), 379-389.

Sansevero, J. B. B., Prieto, P. V., de Moraes, L. F. D., & Rodrigues, P. J. P. (2011). Natural regeneration in plantations of native trees in lowland Brazilian Atlantic Forest: community structure, diversity, and dispersal syndromes. *Restoration Ecology*, 19(3), 379-389.

Sawyer, J. O., & Chermsirivathana, C. (1969). A flora of Doi Suthep-Doi Pui, Chiang Mai, North Thailand. *Nat. Hist. Bull. Siam Soc*, 23, 99-132.

Selaya, N. G., Anten, N. P. R., Oomen, R. J., Matthies, M., & Werger, M. J. A. (2007). Above-ground biomass investments and light interception of tropical forest trees and lianas early in succession. *Annals of Botany*, 99(1), 141-151.

Sharp, A. (1995) Seed Dispersal and Predation in Primary Forest and Gap on Doi Suthep. M.S.c Dissertations. Chiang Mai University. Thailand.

Sheil, D., & May, R. M. (1996). Mortality and recruitment rate evaluations in heterogeneous tropical forests. *Journal of ecology*, 91-100.

Sheil, D., Burslem, D. F., & Alder, D. (1995). The interpretation and misinterpretation of mortality rate measures. Journal of Ecology, 331-333.

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.

Silvertown, J., & Charlesworth, D. (2009). Introduction to plant population biology. John Wiley & Sons.

Sinhaseni, K. 2008. *Natural establishment of tree seedling in forest restoration trials at Ban Mae Sa mai, Chiang Mai Province*. M.S.c Dissertations. Chiang Mai University. Thailand.

Slot, M., & Poorter, L. (2007). Diversity of tropical tree seedling responses to drought. *Biotropica*, 39(6), 683-690.

Tabarelli, M., & Peres, C. A. (2002). Abiotic and vertebrate seed dispersal in the Brazilian Atlantic forest: implications for forest regeneration. *Biological Conservation*, 106(2), 165-176.

Tang, H., Dubayah, R., Swatantran, A., Hofton, M., Sheldon, S., Clark, D. B., & Blair, B. (2012). Retrieval of vertical LAI profiles over tropical rain forests using waveform lidar at La Selva, Costa Rica. *Remote Sensing of Environment*, 124, 242-250.

Tangprasertsri, S., (2005). *Secondary succession of tropical seasonal evergreen forest: species composition and dispersal at Mo Singto, Khao Yai Nation Park, Thailand.* Thesis submitted for the degree of master of science (environmental biology), Mahidol University

Terborgh, J., Nuñez-Iturri, G., Pitman, N. C., Valverde, F. H. C., Alvarez, P., Swamy, V., ... & Paine, C. E. (2008). Tree recruitment in an empty forest. *Ecology*, 89(6), 1757-1768.

Thaiying, J. (2003) *The Diversity of Small Mammal Communities in FORRU's Reforested Areas*. B.S.c. Dissertations. Chiang Mai University. Thailand.

Tucker, N. I., & Murphy, T. M. (1997). The effects of ecological rehabilitation on vegetation recruitment: some observations from the Wet Tropics of North Queensland. *Forest ecology and management*, 99(1), 133-152.

Tucker, N. I., & Simmons, T. (2009). Restoring a rainforest habitat linkage in north Queensland: Donaghy's Corridor. *Ecological Management & Restoration*, 10(2), 98-112.

Turner, I. M. (1990). The seedling survivorship and growth of three Shorea species in a Malaysian tropical rain forest. *Journal of Tropical Ecology*, 6(4), 469-478.

Twedt, D. J., & Wilson, R. R. (2002). Development of oak plantations established for wildlife. *Forest Ecology and Management*, 162(2-3), 287-298.

Valladares, F., & Guzmán, B. (2006). Canopy structure and spatial heterogeneity of understory light in an abandoned Holm oak woodland. *Annals of Forest Science*, 63(7), 749-761.

Valladares, F., & Pearcy, R. W. (2002). Drought can be more critical in the shade than in the sun: a field study of carbon gain and photo-inhibition in a Californian shrub during a dry El Niño year. *Plant, Cell & Environment*, 25(6), 749-759.

Valladares, F., Dobarro, I., Sánchez-Gomez, D., & Pearcy, R. W. (2004). Photoinhibition and drought in Mediterranean woody saplings: scaling effects and interactions in sun and shade phenotypes. *Journal of Experimental Botany*, 56(411), 483-494.

Van Breugel, M., Bongers, F., & Martínez-Ramos, M. (2007). Species dynamics during early secondary forest succession: recruitment, mortality and species turnover. *Biotropica,* 39(5), 610-619.

Van Kuijk, M., Anten, N. P. R., Oomen, R. J., Van Bentum, D. W., & Werger, M. J. A. (2008). The limited importance of size-asymmetric light competition and growth of pioneer species in early secondary forest succession in Vietnam. *Oecologia*, 157(1), 1-12.

Veenendaal, E. M., Swaine, M. D., Agyeman, V. K., Blay, D., Abebrese, I. K., & Mullins, C. E. (1996). Differences in plant and soil water relations in and around a forest gap in West Africa during the dry season may influence seedling establishment and survival. *Journal of Ecology*, 83-90.

Viani, R. A. G., Vidas, N. B., Pardi, M. M., Castro, D. C. V., Gusson, E., & Brancalion, P. H. (2015). Animal-dispersed pioneer trees enhance the early regeneration in Atlantic Forest restoration plantations. *Natureza & Conservação*, *13*(1), 41-46.

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

Walker, L. R., Walker, J., & Hobbs, R. J. (2007). *Linking restoration and ecological succession* (No. Sirsi) i9780387353029). Springer London.

Waller, P. A., Anderson, P. M., Holmes, P. M., & Newton, R. J. (2015). Developing a species selection index for seed-based ecological restoration in Peninsula Shale Renosterveld, Cape Town. *South African Journal of Botany*, 99, 62-68.

Wang, Z., Yang, H., Dong, B., Zhou, M., Ma, L., Jia, Z., & Duan, J. (2017). Regeneration response to canopy gap size in a Chinese pine plantation: Species diversity patterns, size structures and spatial distributions. *Forest Ecology and Management,* 397, 97-107.

Watcharamusic, M. (1998). Preliminary study on vegetation and floral of Doi Suthep-Pui national park, Changwat, Chiang Mai. *Thai National AGRIS Centre*. Thailand.

Welden, C. W., Hewett, S. W., Hubbell, S. P., & Foster, R. B. (1991). Sapling survival, growth, and recruitment: relationship to canopy height in a neotropical forest. *Ecology*, 72(1), 35-50.

White, E., Tucker, N., Meyers, N., & Wilson, J. (2004). Seed dispersal to revegetated isolated rainforest patches in North Queensland. *Forest Ecology and Management*, 192(2), 409-426.

Whitmore, T. C. (1978). Gaps in the forest canopy. *Tropical trees as living systems*., 639-655.

Whitmore, T. C., & Sayer, J. A. (1992). Deforestation and species extinction in tropical moist forests. *Tropical deforestation and species extinction*, 1-14.

Wright, J. S. (2002). Plant diversity in tropical forests: a review of mechanisms of species coexistence. *Oecologia*, 130(1), 1-14.

Wunderle, J. M. (1997). The role of animal seed dispersal in accelerating native forest regeneration on degraded tropical lands. *Forest Ecology and Management*, 99(1), 223-235.

Wydhayagarn, C., Elliott, S., & Wangpakapattanawong, P. (2009). Bird communities and seedling recruitment in restoring seasonally dry forest using the framework species method in Northern Thailand. *New Forests*, 38(1), 81-97.

Yelenik, S. G., DiManno, N., & D'Antonio, C. M. (2015). Evaluating nurse plants for restoring native woody species to degraded subtropical woodlands. *Ecology and evolution*, 5(2), 300-313.

Yirdaw, E. (2002). *Restoration of the native woody-species diversity, using plantation species as foster trees, in the degraded highlands of Ethiopia.* University of Helsinki, Viikki Tropical Resources Institute (VITRI).

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., & Luukkanen, O. (2004). Photosynthetically active radiation transmittance of forest plantation canopies in the Ethiopian highlands. *Forest Ecology and Management*, 188(1), 17-24.

Yoshifuji, N., Kumagai, T. O., Tanaka, K., Tanaka, N., Komatsu, H., Suzuki, M., & Tantasirin, C. (2006). Inter-annual variation in growing season length of a tropical seasonal forest in northern Thailand. *Forest ecology and management*, 229(1), 333-339.

Zanne, A. E., & Chapman, C. A. (2001). Expediting reforestation in tropical grasslands: distance and isolation from seed sources in plantations. *Ecological Applications*, 11(6), 1610-1621.

APPENDIX A

List of Adult Tree Species

Table A presents the list of adult tree species from nearby remnant forest, and tree density (stems/ha) recorded during adult tree survey including successional status, seed-dispersed mode, and seed size (Family-alphabet order). References refer to sources of secondary data to determine successional status and dispersal mode. (*Note: A= Animal-dispersed species, N-A = Non-animal-dispersed species, P = Pioneer species, N-P = Non-pioneer species, na= not available, DNP - Forest Herbarium of the Department of National Parks, FOC - Flora of China, FORRU - Forest Restoration Research Unit, KEW - Kew Herbarium, QSBG - Queen Sirikit Botanic Garden*)

Table A: Adult tree species from nearby remnant forest, and tree density (stems/ha) recorded during adult tree survey including successional status, seed-dispersed mode, and seed size (Family-alphabet order). References refer to sources of secondary data to determine successional status and dispersal mode. (*Note: A= Animal-dispersed species, N-A = Non-animal-dispersed species, P = Pioneer species, N-P = Non-pioneer species, na= not available, DNP - Forest Herbarium of the Department of National Parks, FOC - Flora of China, FORRU - Forest Restoration Research Unit, KEW – Kew Herbarium, QSBG - Queen Sirikit Botanic Garden*)

Family	**Scientific name**	**Successional status**	**Dispersal mode**	**Seed Size**	**Density(per ha)**	**Reference**
ACTINIDIACEAE	*Saurauia roxburghii* Wall.	na	A	small	56.9	DNP-2, QSBG
ADOXACEAE (Caprifpliaceae)	*Viburnum inopinatum* Craib	P	A	small	6.3	QSBG, FOC (Vol. 19 Page 573, 574, 595)
ANACARDIACEAE	*Choerospondias axillaris (*Roxb.) B.L. Burtt & A.W. Hill	N-P	A	medium	5.0	FORRU, QSBG, FOC (Vol. 11 Page 341)
	Gluta obovata Craib	P	N-A	small	0.6	QSBG
	Mangifera sylvatica Roxb.	N-P	A	Big	1.9	FOC (Vol. 11 Page 338, 339)
	Rhus rhetsoides Craib	P	A	small	0.6	FORRU, Kew
	Semecarpus cochinchinensis Engl.	na	A	medium	15.0	DNP-2, QSBG
	Spondias pinnata (L. f.) Kurz	N-P	N-A	Big	10.6	FORRU, QSBG, FOC (Vol. 11 Page 339)
ANNONACEAE	*Goniothalamus laoticus (*Finet & Gagnep.) Bân	N-P	A	small	0.6	DNP-2, FOC (Vol. 19 Page 685, 687)
	Polyalthia sp.	na	na	na	1.9	
APOCYNACEAE	*Alstonia rostrata Fisch.*	P	N-A	Big	3.8	FOC (Vol. 16 Page 155)
	Alstonia scholaris (L.) R. Br.	P	N-A	small	0.6	DNP-2, FOC (Vol. 16 Page 155)
AQUIFOLIACEAE	*Ilex umbellulata* (Wall.) Loes.	N-P	A	small	10.0	DNP-2, FOC (Vol. 11 Page 360, 382), QSBG
ARALIACEAE	*Brassaiopsis ficifolia* Dunn	na	A	small	10.6	QSBG, FOC (Vol. 13 Page 447, 450)
	Heteropanax fragrans (Roxb.) Seem.	P	A	small	0.6	FOC (Vol. 13 Page 474), QSBG
	Trevesia palmata (Roxb. ex Lindl.) Vis.	na	A	small	18.8	DNP-2, QSBG, FOC (Vol. 13 Page 438)
ASTERACEAE (Compositae)	*Vernonia volkameriifolia* DC.	P	N-A	medium	16.9	QSBG, FOC (Vol. 20-21 Page 355, 359)
BIGNONIACEAE	*Markhamia stipulata* (Wall.) Seem. ex Sch. *var. stipulata*	N-P	N-A	small	35.6	DNP-2, FOC (Vol. 18 Page 224)
	Oroxylum indicum (L.) Bth. ex Kurz	P	N-A	Big	2.5	DNP-2, FOC (Vol. 18 Page 215)
	*Stereospermum colais (*B.-H. ex Dillw.) Mabb.	N-P	N-A	small	4.4	QSBG, FOC (Vol. 18 Page 217)

Table A: (count.)

Family	**Scientific name**	**Successional status**	**Dispersal mode**	**Seed Size**	**Density (per ha)**	**Reference**
BURSERACEAE	*Canarium subulatum* Guillaumin	P	A	Big	1.9	FORRU, FOC (Vol. 11 Page 108, 109)
	Garuga pinnata Roxb.	P	A	small	17.5	DNP-2, FORRU, FOC Vol. 11 Page 107, 108
	Protium serratum (Wall. ex Colebr.) Engl.	N-P	A	medium	11.3	DNP-2, QSBG, FOC (Vol. 11 Page 106)
CANNABACEAE (Ulmaceae)	*Celtis tetrandra Roxb.*	p	A	small	0.6	FOC (Vol. 5 Page 18)
	Trema orientalis (L.) Bl.	P	A	small	1.3	DNP-2, FOC (Vol. 5 Page 13)
CELASTRACEAE	*Euonymus cochinchinensis* Pierre	N-P	A	small	0.	DNP-2, QSBG
CLUSIACEAE	*Garcinia merguensis* Wight	P	A	small	3.8	DNP-2, QSBG
COMBRETACEAE	*Anogeissus acuminata* (Roxb. ex DC.) Guill.	N-P	A	small	1.3	FOC (Vol. 13 Page 314), Kew
CORNACEAE (Alangiaceae)	*Alangium kurzii* Craib var. kurzii	P	A	small	30.0	FOC (Vol. 13 Page 304, 306), QSBG
DILLENIACEAE	*Dillenia parviflora* Griff.var. *kerrii (*Craib) Hoogl.	P	A	small	13.1	DNP-2, QSBG
DIPTEROCARPACEAE	*Dipterocarpus turbinatus* C.F.Gaertn	N-P	N-A	Big	0.6	DNP-2, QSBG, FOC (Vol. 13 Page 48, 49)
	Shorea roxburghii G.Don	N-P	N-A	small	13.8	FORRU, QSBG
EBENACEAE	*Diospyros glandulosa* Lace	N-P	A	small	13.1	DNP-2, FORRU, QSBG
ELAEOCARPACEAE	*Elaeocarpus floribundus* Bl. var. *Floribundus*	N-P	A	medium	16.9	DNP-2, Kew
ERICACEAE	*Vaccinium sprengelii (*G. Don) *Sleumer*	N-P	A	small	2.5	QSBG, DNP-1 and -2, Kew
EUPHORBIACEAE	*Balakata baccata* (Roxb.) Esser	P	A	small	1.9	DNP-2, FOC (Vol. 11 Page 284), FORRU
	Cleidion spiciflorum (Burm.f.) Merr.	N-P	A	medium	3.8	DNP-2, FOC (Vol. 11 Page 244, 245)
	Macaranga denticulata (Blume) Müll.Arg.	P	N-A	small	27.5	DNP-2, FOC (Vol. 11 Page 237, 238)
	Mallotus philippensis (Lmk.) M.-A.	P	N-A	small	13.1	DNP-2, FOC (Vol. 11 Page 226, 229)
	Mallotus paniculatus (Lam.) Müll.Arg.	P	A	small	6.9	DNP-2, FOC (Vol. 11 Page 227, 236)
	Ostodes paniculata Bl. var. *paniculata*	N-P	N-A	medium	33.8	DNP-2, FOC (Vol. 11 Page 168, 271)
	Baccaurea ramiflora Lour.	P	A	medium	5.6	DNP-2, FOC (Vol. 11 Page 216), FORRU
FABACEAE	*Adenanthera microsperma* Teijm & Binn.	P	N-A	small	1.9	FOC (Vol. 10 Page 51), JSTOR
	Pterocarpus macrocarpus Kurz	P	A	medium	2.5	DNP-2, FORRU, QSBG
FABACEAE (Leguminosae, Caesalpinioideae)	*Acrocarpus fraxinifolius* Wight ex Arn.	P	N-A	medium	1.9	DNP, FOC (Vol. 10 Page 39)
	Bauhinia variegata Linn.	N-P	N-A	small	20.6	DNP-2, FORRU, QSBG, FOC (Vol. 10 Page 7, 9, 10)
FABACEAE (Leguminosae, Mimosoideae)	*Albizia chinensis* (Osbeck) Merr.	P	N-A	small	28.8	FOC (Vol. 10 Page 62, 65)
	Albizia odoratissima (L.f.) Benth.	P	N-A	small	12.5	FOC (Vol. 10 Page 62, 64)
	Archidendron clypearia (Jack) Niels. ssp. clypearia var. *clypearia*	P	A	small	7.5	DNP-2, FOC (Vol. 10 Page 66, 67, 68)

Table A: (count.)

Family	Scientific name	Successional status	Dispersal mode	Seed Size	Density (per ha)	Reference
FABACEAE (Leguminosae, Papilionoideae)	*Dalbergia cultrate* Graham ex Benth.	P	N-A	small	7.5	FOC (Vol. 10 Page 122, 127)
	Dalbergia ovata var. glomeriflora (Kurz) Thoth.	P	A	medium	78.8	FORRU
	Erythrina subumbrans (Hassk.) Merr.	N-P	N-A	medium	30.0	FORRU, FOC (Vol. 10 Page 238)
FAGACEAE	*Castanopsis acuminatissima* (Blume) A.DC.	N-P	A	medium	95.6	DNP-2, FORRU
	*Castanopsis calathiformis (*Skan) Rehder & E.H.Wilson	N-P	A	medium	1.9	FORRU, FOC (Vol. 4 Page 320)
	Castanopsis diversifolia (Kurz) King ex Hook.f.	N-P	A	Big	166.2	DNP-2, FORRU
	Castanopsis tribuloides (Sm.) A.DC.	N-P	A	medium	53.7	FORRU, QSBG
	Lithocarpus sp.	na	na	na	11.9	
	Lithocarpus elegans (Blume) Hatus. ex Soepadmo	P	A	medium	21.9	FOC (Vol. 4 Page 364)
	Lithocarpus polystachyus (Wall. ex A.DC.) Rehder	P	A	medium	53.7	DNP-2, FORRU
	Lithocarpus thomsonii (Miq.) Rehder	P	A	medium	8.8	DNP-2, FOC (Vol. 4 Page 346)
	Quercus semiserrata Roxb.	N-P	A	medium	1.3	FORRU, FOC (Vol. 4 Page 386)
JUGLANDACEAE	*Engelhardia spicata var.integra* (Kurz) Mann.	P	N-A	small	20.0	FOC (Vol. 4 Page 279)
LAMIACEAE (Verbenaceae)	*Callicarpa arborea* Roxb.	N-P	A	small	6.3	FORRU, FOC (Vol. 17 Page 6)
	Gmelina arborea Roxb.	P	A	medium	7.5	DNP-2, FORRU, FOC (Vol. 17 Page 33)
	Vitex peduncularis Wall. ex Schauer	P	N-A	small	5.6	DNP-2, FOC (Vol. 17 Page 31)
LAURACEAE	*Actinodaphne henryi* Gamble	P	A	small	14.4	FOC (Vol. 7 Page 161, 164), JSTOR
	Beilschmiedia aff. intermedia Allen	N-P	A	medium	43.1	FOC (Vol. 7 Page 233, 242)
	Cinnamomum iners Reinw. ex Blume	N-P	A	small	3.8	FORRU, QSBG, FOC (Vol. 7 Page 113, 168, 183)
	Cryptocarya amygdalina Nees	P	A	medium	2.5	FORRU, FOC (Vol. 7 Page 248, 250)
	*Lindera caudata (*Nees) Hook. f.	na	A	small	5.0	QSBG, FOC (Vol. 7 Page 159)
	Litsea cubeba (Lour.) Pers.	N-P	A	small	2.5	DNP-2, FOC (Vol. 7 Page 118, 122)
	Litsea monopetala (Roxb.) Pers.	N-P	A	small	60.6	DNP-2, FORRU, FOC (Vol. 7 Page 119, 132)
	Litsea salicifolia (Roxburgh ex Nees) Hook. f.	N-P	A	small	60.6	FOC (Vol. 7 Page 119, 131)
	Machilus gamblei King ex Hook. f.	N-P	A	small	153.1	FOC (Vol. 7 Page 204, 213), QSBG
	Neolitsea sp.	na	na	na	1.3	
	Nothaphoebe umbelliflora (Blume) Blume	N-P	A	Big	21.3	QSBG, Kew
	Phoebe lanceolata (Wall. ex Nees) Nees	N-P	A	small	130.0	FORRU, FOC (Vol. 7 Page 190, 192)
	Phoebe cathia (D. Don) Kosterm.	N-P	A	medium	4.4	Kew
MAGNOLIACEAE	*Magnolia garrettii* (Craib) Figl. & Noot.	N-P	N-A	medium	5.6	FOC (Vol. 7 Page 51, 53, 55), QSBG
	Magnolia rajaniana (Craib) Figlar	N-P	A	small	3.8	DNP-2, QSBG
	Magnolia baillonii Pierre	N-P	N-A	medium	20.6	DNP-2, QSBG, FOC (Vol. 7 Page 48, 51, 78, 80)
MALVACEAE	*Pterospermum acerifolium* (L.) Willd.	N-P	N-A	Big	8.1	DNP-2, FOC (Vol. 12 Page 327, 328), QSBG

Table A: (count.)

Family	**Scientific name**	**Successional status**	**Dispersal mode**	**Seed Size**	**Density (per ha)**	**Reference**
MALVACEAE (Bombacaceae)	*Bombax anceps* Pierre	P	N-A	small	0.6	DNP-2, FORRU
MALVACEAE (Sterculiaceae)	*Sterculia balanghas* L.	N-P	N-A	Big	5.6	DNP-2, QSBG, Kew
	Sterculia villosa Roxb.	N-P	N-A	medium	7.5	DNP-2, FOC (Vol. 12 Page 305)
MALVACEAE (Tiliaceae)	*Colona floribunda* (Kurz) Craib	P	N-A	small	10.6	DNP-2, FOC (Vol. 12 Page 250)
	Grewia hirsuta Vahl	N-P	A	small	0.6	FOC (Vol. 12 Page 253, 256)
MELIACEAE	*Aglaia sp.1*	P	A	na	6.3	
	Chisocheton siamensis Craib	N-P	A	medium	6.3	QSBG, FOC (Vol. 11 Page 130)
	Heynea trijuga Roxb. ex Sims	P	N-A	medium	5.6	FORRU, FOC (Vol. 11 Page 120, 121)
	Melia azedarach L.	P	A	small	1.9	DNP-2, QSBG, FOC (Vol. 11 Page 130)
MORACEAE	*Antiaris toxicaria* Lesch.	N-P	A	Big	3.1	FOC (Vol. 5 Page 37)
	Artocarpus lacucha Buch.-Ham. ex D.Don	P	A	medium	30.0	FOC (Vol. 5 Page 34), FORRU
	Artocarpus nitidus Trécul	P	A	medium	20.6	FOC (Vol. 5 Page 33)
	Broussonetia papyrifera (L.) L'Hér. ex Vent.	N-P	A	small	0.6	FOC (Vol. 5 Page 26)
	Ficus benjamina L.	P	A	small	0.6	FORRU, DNP-2
	Ficus fistulosa Reinw. ex Bl. var. fistulosa	P	A	small	3.8	FORRU, FOC Vol. 5 Page 50
	Ficus auriculata Lour.	P	A	small	0.6	FORRU, DNP-2
	Ficus callosa Willd.	P	A	small	0.6	FORRU, DNP-2
	Ficus concinna (Miq.) Miq.	P	A	small	1.3	FORRU, FOC (Vol. 5 Page 40)
	Ficus heterostyla Merr.	P	A	small	1.9	FORRU, DNP-2
	Morus macroura Miq.	P	A	small	7.5	DNP-2, FOC (Vol. 5 Page 25)
MYRISTICACEAE	*Horsfieldia amygdalina (*Wall.) Warb. var. Amygdalina	P	A	Big	8.1	DNP-2, FORRU, FOC (Vol. 7 Page 100)
	Knema erratica (Hook. f. & Thomson) J. Sinclair	N-P	A	medium	3.8	DNP-2
MYRTACEAE	*Eugenia albiflorum (*Duthie ex Kurz) Bahadur & R.C.Gaur	N-P	A	Big	26.3	DNP-2, FORRU
MYRTACEAE	*Eugenia fruticosum* DC.	N-P	A	small	35.6	DNP-2, FOC (Vol. 13 Page 338, 339, 355)
OLEACEAE	*Chionanthus ramiflorus* Roxb.	N-P	A	medium	13.1	FOC (Vol. 15 Page 294)
OLEACEAE	*Olea salicifolia* Wall. ex G.Don	N-P	A	small	3.8	FOC (Vol. 15 Page 298)
PANDACEAE	*Microdesmis caseariifolia* Planch. ex Hook.	N-P	A	small	3.8	DNP-2, QSBG, FOC (Vol. 11 Page 162)
PENTAPHYLACACEAE	*Ternstroemia gymnanthera* (Wight & Arn.) Sprague	N-P	A	small	4.4	DNP-2, FOC (Vol. 12 Page 430, 432)
PENTAPHYLACACEAE (Theaceae)	*Eurya acuminata* DC.	P	A	small	34.4	DNP-2, FORRU, (FOC Vol. 12 Page 450, 461, 464)
PHYLLANTHACEAE	*Antidesma bunius* (L.) Spreng.	N-P	A	small	18.8	DNP-1&2, FOC (Vol. 11 Page 209, 210, 211)
(Euphorbiaceae)	*Antidesma sootepense* Craib	N-P	A	small	1.9	FOC (Vol. 11 Page 210, 213)

Table A: (count.)

Family	Scientific name	Successional status	Dispersal mode	Seed Size	Density (per ha)	Reference
PHYLLANTHACEAE	*Aporosa octandra* (B.-H. ex D. Don) Vick. var. *octandra*	P	A	small	45.6	FORRU, QSBG, Kew
(Euphorbiaceae)	*Aporosa octandra* (B.-H. ex D. Don) Vick. *var. yunnanensis* (Pax & K. Hoffm.) Schot	P	A	small	3.8	Schot (2004), FOC (Vol. 11 Page 215)
	Bischofia javanica Blume	P	A	small	5.6	FORRU, FOC (Vol. 11 Page 217, 218)
	Bridelia retusa (L.) Spreng.	P	A	small	35.6	FORRU, FOC (Vol. 11 Page 175)
	Glochidion eriocarpum Champ. ex Benth.	p	A	small	29.4	Kew, FOC (Vol. 11 Page 194, 198)
	Glochidion sphaerogynum (Müll.Arg.) Kurz	N-P	A	small	8.1	DNP-2, FORRU, FOC (Vol. 11 Page 195, 200)
	Phyllanthus emblica L.	P	A	small	7.5	FORRU, FOC (Vol. 11 Page 163, 180, 182)
PINACEAE	*Pinus kesiya* Roy. ex Gord.	N-P	N-A	small	26.3	DNP-2, FOC (Vol. 4 Page 15)
POLYGALACEAE	*Xanthophyllum flavescens* Roxb.	N-P	A	medium	0.6	QSBG, FOC (Vol. 11 Page 140)
PROTEACEAE	*Helicia nilagirica* Bedd.	N-P	A	Big	53.8	FORRU, FOC (Vol. 5 Page 196)
	Helicia sp.	na	na	na	1.3	
RHAMNACEAE	*Hovenia dulcis* Thunb.	N-P	A	small	0.6	FOC (Vol. 12 Page 115, 118), QSBG
RHIZOPHORACEAE	*Carallia brachiata* (Lour.) Merr.	N-P	A	medium	1.3	FORRU, FOC (Vol. 13 Page 298)
ROSACEAE	*Docynia indica* (Wall.) Decne.	N-P	A	Big	3.1	FOC (Vol. 9 Page 171), QSBG
	Prunus cerasoides B.-H.ex D.Don	P	A	small	3.1	DNP-2, FORRU, QSBG
RUBIACEAE	*Ixora sp.*	na	na	na	0.6	
	Rothmannia sootepensis (Craib) Bremek.	na	A	medium	16.9	DNP-2, QSBG
	Tarennoidea wallichii (Hk.f.) Tirv.& Sastre	N-P	A	small	4.4	FORRU, DNP, FOC (Vol. 19 Page 345)
	Wendlandia tinctoria (Roxb.) DC. *ssp. tinctoria*	N-P	N-A	small	146.2	FORRU, DNP-2, FOC (Vol. 19 Page 355, 364)
RUTACEAE	*Acronychia pedunculata* (L.) Miq.	P	A	small	0.6	DNP, FOC (Vol. 11 Page 77)
	Melicope pteleifolia (Champ. ex Benth.) T.Hart.	P	A	small	1.3	QSBG, FOC (Vol. 11 Page 70, 72)
	Micromelum integerrimum (B.-H. ex DC.) Wight & Arn. ex M. Roem. *var. intergerrimum*	N-P	A	small	6.3	FOC (Vol. 11 Page 79)
SABIACEAE	*Meliosma simplicifolia* (Roxb.) Walp.	na	A	small	2.5	QSBG, FOC (Vol. 12 Page 34, 37, 38)
SALICACEAE (Flacourtiaceae)	*Casearia grewiifolia* Vent. *var. gelonioides* (Bl.) Sleum.	N-P	A	small	3.1	BLUMEA (Volume 30, No. 2, Page 247-248)
	Homalium ceylanicum (Gardner) Benth.	N-P	A	small	1.9	Kew, FOC (Vol. 13 Page 128, 130, 131)
SAPINDACEAE	*Dimocarpus longan* Lour. var. *longan*	N-P	A	medium	4.4	DNP-2, FOC (Vol. 12 Page 6, 15)
	Sapindus rarak DC.	N-P	A	medium	10.6	DNP-2, FORRU, FOC (Vol. 12 Page 11)

Table A: (count.)

Family	**Scientific name**	**Successional status**	**Dispersal mode**	**Seed Size**	**Density (per ha)**	**Reference**
SAPOTACEAE	*Madhuca dongnaiensis* (Pierre) Baehni	N-P	A	medium	11.9	Kew
	Sarcosperma arboreum Hook.f.	N-P	A	medium	18.1	DNP-2, FORRU, FOC (Vol. 15 Page 214)
SIMAROUBACEAE	*Picrasma javanica* Bl.	N-P	A	small	0.6	DNP-2, QSBG
STAPHYLEACEAE	*Turpinia pomifera* (Roxb.) Wall. Ex DC. Var. *pomifera*	N-P	A	medium	125.6	FORRU, FOC Vol. 11 Page 501, 503
STYRACACEAE	*Styrax benzoides* Craib	P	A	small	108.1	DNP-2, QSBG, FOC (Vol. 15 Page 260)
SYMPLOCACEAE	*Symplocos cochinchinensis* (Lour.) S.Moore *ssp.laurina* (Reyz.) Noot.	N-P	A	small	10.6	DNP-2, FOC (Vol. 15 Page 248)
	Symplocos macrophylla Wall. ex DC. *ssp.* sulcata (Kurz) Noot. var. *sulcata*	N-P	A	small	10.6	DNP-2, FOC (Vol. 15 Page 243)
THEACEAE	*Gordonia dalglieshiana* Craib	na	N-A	small	1.9	QSBG, Kew
THEACEAE	*Pyrenaria diospyricarpa* Kurz var. *diospyricarpa*	P	A	medium	0.6	QSBG, FOC (Vol. 12 Page 413)
THEACEAE	*Schima wallichii* (DC.) Korth.	N-P	N-A	medium	136.2	FORRU, FOC (Vol. 12 Page 420, 422)
VERBENACEAE	*Vitex quinata* (Lour.) Will. *Var.puberula* (Lau) Mold.	P	N-A	small	2.5	DNP-2, FOC (Vol. 17 Page 29)

APPENDIX B

List of Recruit Native Tree Species

The table B presents the list of the recruit native species that found in the seedling communities of FWS restoration forests age 6, 10 and 14 years (R6, R10 and R14, respectively). References are sourced by Maxwell and Elliott (2000), Ashton *et al.* (2001) and secondary data used to determine each species' successional status and dispersal mode. (*Note: A= Animal-dispersed species, N-A = Non-animal-dispersed species, P = Pioneer species, N-P = Non-pioneer species, na= not available, DNP - Forest Herbarium of the Department of National Parks, FOC - Flora of China, FORRU - Forest Restoration Research Unit, KEW - Kew Herbarium, QSBG - Queen Sirikit Botanic Garden*)

Table B: Recruit species in the seedling communities of FWS restoration forests by family in alphabetical order. References are sources of secondary data used to determine each species' successional status and dispersal mode. (*Note: A= Animal-dispersed species, N-A = Non-animal-dispersed species, P = Pioneer species, N-P = Non-pioneer species, na= not available, DNP - Forest Herbarium of the Department of National Parks, FOC - Flora of China, FORRU - Forest Restoration Research Unit, KEW - Kew Herbarium, QSBG - Queen Sirikit Botanic Garden*)

Family	**Scientific name**	**Successional status**	**Dispersal mode**	**Seed Size**	**Recruit seedling individual**			**Reference**
					R6	**R10**	**R14**	
ANACARDIACEAE	*Gluta usitata* (Wall.) Hou	P	N-A	Big			1	DNP-2, QSBG
	Mangifera caloneura Kurz	N-P	A	Big		1		DNP-2, QSBG
APOCYNACEAE	*Alstonia rostrata* Fisch.	P	N-A	Big			2	FOC (Vol. 16 Page 155)
	Alstonia scholaris (L.) R. Br.	P	N-A	small			4	DNP-2, FOC (Vol. 16 Page 155)
BIGNONIACEAE	*Oroxylum indicum* (L.) Bth. ex Kurz	P	N-A	Big		3		DNP-2, FOC (Vol. 18 Page 215)
	Stereospermum colais (B.-H. ex Dillw.) Mabb.	N-P	N-A	small	10	12	3	QSBG, FOC (Vol. 18 Page 217)
BURSERACEAE	*Canarium strictum* Roxb.	P	A	small		1		FOC (Vol. 11 Page 108, 110)
	Garuga pinnata Roxb.	P	A	small		1		DNP-2, FORRU, FOC Vol. 11 Page 107, 108
CELASTRACEAE	*Euonymus cochinchinensis* Pierre	N-P	A	small		1		DNP-2, QSBG
CORNACEAE (Alangiaceae)	*Alangium kurzii* Craib var. *kurzii*	P	A	small	4	3		FOC (Vol. 13 Page 304, 306), QSBG
ELAEOCARPACEAE	*Elaeocarpus floribundus* Bl. var. *Floribundus*	N-P	A	medium			3	DNP-2, Kew
EUPHORBIACEAE	*Mallotus philippensis* (Lmk.) M.-A.	P	N-A	small	3	1	8	DNP-2, FOC (Vol. 11 Page 226, 229)
FABACEAE	*Albizia chinensis* (Osbeck) Merr.	P	N-A	small		1		FOC (Vol. 10 Page 62, 65)
(Leguminosae, Mimosoideae)	*Archidendron clypearia* (Jack) Niels. ssp. *clypearia var. clypearia*	P	A	small				DNP-2, FOC (Vol. 10 Page 66, 67, 68)
FABACEAE	*Dalbergia cultrata* Graham ex Benth.	P	N-A	small	2		16	FOC (Vol. 10 Page 122, 127)
(Leguminosae, Papilionoideae)	*Dalbergia ovata* var. *glomeriflora* (Kurz) Thoth.	P	A	medium		3		FORRU
FAGACEAE	*Lithocarpus polystachyus* (Wall. ex A.DC.) Rehder	P	A	medium			1	DNP-2, FORRU
JUGLANDACEAE	*Engelhardia spicata* var.*integra* (Kurz) Mann.	P	N-A	small	6	3	1	FOC (Vol. 4 Page 279)
LAMIACEAE	*Callicarpa arborea* Roxb.	N-P	A	small		1		FORRU, FOC (Vol. 17 Page 6)
(Verbenaceae)	*Vitex peduncularis* Wall. ex Schauer	P	N-A	small	2		1	DNP-2, FOC (Vol. 17 Page 31)
LAURACEAE	*Actinodaphne henryi* Gamble	P	A	small	1	4		FOC (Vol. 7 Page 161, 164), JSTOR
	Beilschmiedia aff. intermedia Allen	N-P	A	medium		1		FOC (Vol. 7 Page 233, 242)

Table B: (count.)

Family	Scientific name	Successional status	Dispersal mode	Seed Size	Recruit seedling individual			Reference
					R6	R10	R14	
LAURACEAE	*Cinnamomum caudatum* Nees	N-P	A	small	6	38	35	FOC (Vol. 7 Page 187)
LAURACEAE	*Litsea glutinosa* (Lour.) C.B. Rob. var. *glutinosa*	N-P	A	small	1	2	5	DNP-2, FOC (Vol. 7 Page 126)
	Litsea cubeba (Lour.) Pers.	N-P	A	small	1			DNP-2, FOC (Vol. 7 Page 118, 122)
	Litsea salicifolia (Roxburgh ex Nees) Hook. f.	N-P	A	small	19	86	84	FOC (Vol. 7 Page 119, 131)
	Neolitsea zeylanica (Nees & T. Nees) Merr.	N-P	A	small		1		DNP-2, FOC (Vol. 7 Page 108, 114)
	Phoebe cathia (D. Don) Kosterm.	N-P	A	medium	1			Kew
MALVACEAE (Sterculiaceae)	*Sterculia lanceolata*	N-P	N-A	small	2	5	2	DNP-2, FOC (Vol. 12 Page 305, 309, 310)
	Sterculia villosa Roxb.	N-P	N-A	medium		1		DNP-2, FOC (Vol. 12 Page 305)
MALVACEAE (Tiliaceae)	*Colona floribunda* (Kurz) Craib	P	N-A	small		1		DNP-2, FOC (Vol. 12 Page 250)
MORACEAE	*Artocarpus lacucha* Buch.-Ham. ex D.Don	P	A	medium	49	48	36	FOC (Vol. 5 Page 34), FORRU
	Artocarpus nitidus Trécul	P	A	medium	45	2	30	FOC (Vol. 5 Page 33)
OLEACEAE	*Olea salicifolia* Wall. ex G.Don	N-P	A	small	2	4	2	FOC (Vol. 15 Page 298)
PHYLLANTHACEAE (Euphorbiaceae)	*Antidesma bunius* (L.) Spreng.	N-P	A	small	4	94	1	DNP-1&2, FOC (Vol. 11 Page 209, 210, 211)
	Aporosa octandra (B.-H. ex D. Don) Vick. *var. octandra*	P	A	small	3	3	61	FORRU, QSBG, Kew
	*Aporosa villosa (*Wall. ex Lindl.) Baill.	P	A	small	1		8	FOC (Vol. 11 Page 215)
	Glochidion acuminatum M.-A. var. *siamense* A.S.	P	A	small			2	FORRU, FOC (Vol. 11 Page 197, 198)
	Glochidion hongkongense M.-A.	P	A	small	6			Kew, FOC (Vol. 11 Page 194, 195, 197)
	Glochidion rubrum Bl.	P	A	small		1		FORRU, FOC (Vol. 11 Page 194, 197)
	Phyllanthus emblica L.	P	A	small			14	FORRU, FOC (Vol. 11 Page 163, 180, 182)
PROTEACEAE	*Helicia formosana* Hemsl. Var.*oblanceolata Sleum.*	P	A	small	1			FOC (Vol. 5 Page 19ถ)
RHIZOPHORACEAE	*Carallia brachiata* (Lour.) Merr.	N-P	A	medium	1			FORRU, FOC (Vol. 13 Page 298)
RUBIACEAE	*Tarennoidea wallichii* (Hk.f.) Tirv.& Sastre	N-P	A	small	1	2		FORRU, DNP, FOC (Vol. 19 Page 345)
	Wendlandia tinctoria (Roxb.) DC. ssp. *tinctoria*	N-P	N-A	small		16	2	FORRU, DNP-2, FOC (Vol. 19 Page 355, 364)
RUTACEAE	*Melicope pteleifolia* (Champ. ex Benth.) T.Hart.	P	A	small	3			QSBG, FOC (Vol. 11 Page 70, 72)
	Micromelum integerrimum (B.-H. ex DC.) Wight & Arn. ex M. Roem. var. *intergerrimum*	N-P	A	small	5	137	103	FOC (Vol. 11 Page 79)
SANTALACEAE	*Scleropyrum pentanadrum* (Penn.) Mabb	N-P	A	small	1			DNP-2, FOC (Vol. 5 Page 210)
SAPINDACEAE	*Mischocarpus pentapetalus* (Roxb.) Radlk.	N-P	A	small	1			DNP-2, FOC (Vol. 12 Page 19)
STAPHYLEACEAE	*Turpinia pomifera* (Roxb.) Wall. Ex DC. Var. *pomifera*	N-P	A	medium		2		FORRU, FOC Vol. 11 Page 501, 503

APPENDIX C

The Framework Species

Table C: FWS species in the seedling communities of FWS restoration forests by Family in alphabetical order

Family	Scientific name	Successional status	Dispersal mode	Seed Size	Density of adults trees (per ha)	Recruit Stems		
						R14	**R10**	**R6**
ANACARDIACEAE	*Choerospondias axillaris* (Roxb.) B.L.Burtt & A.W.Hill	N-P	A	mediun	76.			3
	Rhus rhetsoides Craib	P	A	small	1.0		1	
BIGNONIACEAE	*Markhamia stipulata* (Wall.) Seem. ex Sch. var. s*tipulata*	N-P	N-A	small	43.0	8	1	38
	Stereospermum neuranthum Kurz	N-P	N-A	mediun	4.0		1	3
DIPTEROCARPACEAE	*Shorea roxburghii* G.Don	N-P	N-A	small	17.0		1	
EBENACEAE	*Diospyros glandulosa* Lace	N-P	A	small	13.0			
ELAEOCARPACEAE	*Elaeocarpus lanceifolius* Roxb.	N-P	A	mediun	1.0		1	
EUPHORBIACEAE	*Baccaurea ramiflora* Lour.	P	A	mediun	6.0			2
	Balakata baccata (Roxb.) Esser	P	A	small	5.0		1	
	Macaranga denticulata (Blume) Müll.Arg.	P	N-A	small	35.0		12	1
FABACEAE	*Albizia odoratissima* (L.f.) Benth.	P	N-A	small	16.0	1		
	Bauhinia variegata Linn.	N-P	N-A	small	49.0		53	459
	Erythrina stricta Roxb.	P	N-A	small	11.0	104	4	
	Erythrina subumbrans (Hassk.) Merr.	N-P	N-A	mediun	37.0	2		
	Pterocarpus macrocarpus Kurz	P	A	mediun	10.0	1	1	5
FAGACEAE	*Castanopsis acuminatissima* (Blume) A.DC.	N-P	A	mediun	99.0		3	1
	Castanopsis calathiformis (Skan) Rehder & E.H.Wilson	N-P	A	mediun	20.0	6326	1	
	Castanopsis diversifolia (Kurz) King ex Hook.f.	N-P	A	Big	170.0	20	16	28
	Castanopsis tribuloides (Sm.) A.DC.	N-P	A	mediun	57.0	3	16	1
	Lithocarpus garrettianus (Craib) A. Camus	N-P	A	Mediur	1.0			1
FAGACEAE	*Quercus semiserrata* Roxb.	N-P	A	mediun	5.0	1	5	
LAMIACEAE	*Gmelina arborea* Roxb.	P	A	mediun	11.0		3	
LAURACEAE	*Alseodaphne andersonii* (King ex. Hk. f.) Kosterm.	N-P	A	mediun	7.0	5		

Table C: (count.)

Family	Scientific name	Successional status	Dispersal mode	Seed Size	Density of adults trees (per ha)	Recruit Stems		
						R14	R10	R6
LAURACEAE	*Cinnamomum iners* Reinw. ex Blume	N-P	A	small	4.0	14	10	4
	*Litsea monopetala (*Roxb.) Pers.	N-P	A	small	71.0	238	1073	4027
	Machilus gamblei King ex Hook. f.	N-P	A	small	167.0	4	57	9
	Phoebe lanceolata (Wall. ex Nees) Nees	N-P	A	small	134.0	212	123	53
MAGNOLIACEAE	*Magnolia garrettii* (Craib) Figl. & Noot.	N-P	N-A	medium	30.0	1		
	Magnolia baillonii Pierre	N-P	N-A	medium	31.0	5	5	1
MELIACEAE	*Aphanamixis polystachya* (Wall.) R. Parker	N-P	N-A	Big	1.0		2	
	Heynea trijuga Roxb. ex Sims	P	N-A	medium	20.0	346	9	11
MORACEAE	*Artocarpus lacucha* Buch.-Ham. ex D.Don	P	A	medium	37.0	36	48	49
	Ficus benjamina L.	P	A	small	7.0			1
	Ficus hispida L.f.	P	A	small	1.0			
MYRISTICACEAE	*Horsfieldia amygdalina* (Wall.) Warb. var. *Amygdalina*	P	A	Big	8.0		1	
MYRTACEAE	*Eugenia albiflorum* (Duthie ex Kurz) Bahadur & R.C.Gaur	N-P	A	Big	26.0	5	1	1
	Eugenia fruticosum DC.	N-P	A	small	39.0	42	3	3
PHYLLANTHACEAE	*Bischofia javanica* Blume	P	A	small	9.0		2	
	Bridelia retusa (L.) Spreng.	P	A	small	39.0			9
PHYLLANTHACEAE	*Glochidion sphaerogynum (*Müll.Arg.) Kurz	N-P	A	small	12.0	1	2	1
PROTEACEAE	*Helicia nilagirica* Bedd.	N-P	A	Big	54.0		1	
RHAMNACEAE	*Hovenia dulcis* Thunb.	N-P	A	small	15.0			1
ROSACEAE	*Prunus cerasoides* B.-H.ex D.Don	P	A	small	56.0	43	74	37
SAPINDACEAE	*Sapindus rarak* DC.	N-P	A	medium	14.0	22	5	
SAPOTACEAE	*Sarcosperma arboreum* Hook.f.	N-P	A	medium	25.0	13	7	
STYRACACEAE	*Styrax benzoides* Craib	P	A	small	112.0	3	4	1
THEACEAE	*Schima wallichii* (DC.) Korth.	N-P	N-A	medium	168.0	41	9	4
VERBENACEAE	*Vitex quinata* (Lour.) Will. Var.*puberula* (Lau) Mold.	P	N-A	small	3.0		1	

APPENDIX D

Comparative Research Studies

Researches and studies of trees, seedlings and plant diversity (including biodiversity) in both plantation and restoration forests, are listed from many region of the world between year 1992 to 2017. The study of trees and seedlings in the FWS restoration forest in BMSM, Chiang Mai, Thailand are listed in the last section.

Table D: Literature and reference research of restoration and plantation forests

Location	**Citation**	**Restoration Method**	**Age (yr.)**	**Target**	**Study Topics**
Northern America					
USA	Twedt & Wilson (2002)	1. Plantations: monoculture of *Quercus* sp. 2. Plantations: mixture of *Quercus* sp.	14−18	1. Planted trees 2. Invasive plants	Development of oak plantations established for wildlife.
Hawaii - Dry forest restoration	Cabin, *et al.* (2002)	1. Outplanting: native species 2. Seeding: native species	−	Seedling	Experiment of effects of light, alien grass on native species additions for forest restoration
Hawaii - Subtropical dry forest	Yelenik *et al.* (2015)	Restoration planting: native species	−	1. Nurse plants 2. Seedling	Evaluating nurse plants for seedlings naturally recruiting of native woody species
Canada, Ontario - Deciduous forest	McLachlan & Bazely (2003)	Restoration planting:	35	1.Canopy cover 2.Soil moisture	Long-term impact of management activity that promoted forest succession on understorey plant communities of deciduous forest restoration
Southern America					
Brazil - Lowland Atlantic forest	Sansevero, *et al.* (2009)	Restoration planting: mixture of 8-11 native species (*Inga edulis* Mart., *Pseudobombax grandiflorum* (Cav.) A. Robyns, and *Citharexylum myrianthum* etc.)	9−11	Floristic plant	The natural regeneration of plant composition, diversity, and dispersal syndromes in different treatment of plantations.
Brazil - Semi-deciduous Forest	De Souza & Batista (2004)	Plantation: mixture of pioneer and non-pioneer tree species	5, 9, 10	1. Woody plants (≥50 cm-tall to 4.8 cm-DBH) 2. Overstorey (≥4.8 cm in DBH)	Evaluate influence of age and restoration design on forest structure of semi-deciduous forest restoration
Brazil - Seasonally tropical dry forest	Viani *et al.* (2015)	Restoration planting: 80 native tree species	5, 6, 8	Tree seedlings (≥10 cm-tall)	The role of animal-dispersed tree species in catalyzing the regeneration of woody species in the understorey of restoration plantings

Table D: (count.)

Location	Citation	Restoration Method	Age (yr.)	Target	Study Topics
Southern America					
Central Amazon	Lima & Vieira (2013)	1. Plantations: monoculture of indigenous species (*Simarouba amara, Dipteryx odorata, Bagassa guianensis, Jacaranda copaia* and *Dinizia excelsa*) 2. Primary forest	35	Plant species diversity	Evaluated plant species richness of regenerating forest community beneath forest plantations and primary forests
Costa Rica	Powers *et al.* (1997)	Plantations: monoculture of 7 tree species (*Hyeronima alchorneoides, Inga edulis,* and *Pinus tecunumanii* etc.)	7	Understory woody (0-30 cm and 30-200 cm)	Effect of overstorey composition on understory woody regeneration and species richness among plantations and abandoned pasture
Costa Rica	Petit & Montagnini (2006)	1. Plantation: monoculture of native tree species 2. Plantation: mixture of native tree species (*Terminalia amazonia, Vochysia guatemalensis, and Inga edulis etc.*)	10−11	Native trees	To compares productivity and growth of tree in monoculture and mixtures species plantations
Costa Rica	Lindell *et al.* (2012)	Restoration planting: 4 native species (*Terminalia amazonia, Vochysia guatemalensis, Erythrina poeppigiana,* and *Inga edulis*)	1, 2, 3	Migratory bird species	Migratory bird species in young tropical forest restoration sites: effects of vegetation height, planting design, and season.
	Reid *et al. (*2015)		6−9	Seed rain	Seed rain limitation and influence of restoration treatments on seed community structure and composition.
Costa Rica - Tropical premontane rain forest zone	Holl *et al.* (2017)	1. Plantation: 2 natives (*Terminalia amazoni*a (J.F.Gmel.) Exell and *Vochysia guatemalensis* Donn. Sm.) and 2 naturalized softwoods (*Erythrina poeppigiana* (Walp.) Skeels and *Inga edulis* Mart.) 2. Applied nucleation 3. Secondary forest	9−10	Sapling and seedling plants	To evaluate the relative influence of local tropical forest restoration treatments and landscape forest cover on tree recruitment.

Table D: (count.)

Location	Citation	Restoration Method	Age (yr.)	Target	Study Topics
Southern America					
Costa Rica	Powers *et al.* (1997)	Plantation: monocultures of 11 tree species (*Vochysia guatemalensis, Inga edulis,* and *Pinus tecunmanii etc.*)	7	Understorey plant	Overstorey composition of plantation effect on the regeneration of species richness of understorey trees
Costa Rica - Neotropical rain forest	Cusack, & Montagnini (2004)	1. Plantation: monocultures 2. Plantation : mixed species		Understorey woody	Role of native species plantations in recovery of understory woody diversity in degraded pasturelands
Mexico - Cloud forest	Alvarez-Aquino, *et al.* (2004)	Restoration planting: native tree species	–	Native Tree Seedling	Experiment of seedling growth and survival of 4 native tree species (*Fagus grandifolia var. mexicana, Carpinus caroliniana, Symplocos coccinea,* and *Quercus acutifolia*) for forest restoration plan
Panama - Moist tropical forest	Jones *et al.*(2004)	Restoration planting: 14 tree species	3	Understorey plants Bird species	1. Investigates understory recruitment patterns in plantation 2. Test the hypothesis that regeneration rates vary significantly by overstorey tree species and overstorey tree structure
Panama - Moist tropical forest	Coll *et al.* (2008)	Restoration planting: 14 tree species	3	Plants' root	Investigated biomass allocation and root architecture of eight tropical species with different successional status
Perto Rico - Neotropical rain forest	1. Lugo, (1992) 2. Lugo, (1997)	1. Plantaion: monoculture (*Pinus caribaea* and *Swietenia macrophylla*) 2. Secondary forests	4–49 3–50	Demographic, production and nutrient cycling of trees	1. Comparison of tropical tree plantations with secondary forests of similar age. 2. Paradox of reestablishing species richness on degraded lands with tree monocultures 3. Biomass and nutrient restoration dynamic

Table D: (count.)

Location	Citation	Restoration Method	Age (yr.)	Target	Study Topics
Africa					
South Africa	Geldenhuys (1997)	Plantation: monoculture of Pines (*Pinus patula, P. elliottii,* and *P. taeda*) and Eucalypts (*Eucalypt grandis)*	20−50	Understorey plants	Native forest regeneration in pine and eucalypt plantations
Ethiopia, Southern region - Dry tropical afromontane forest	Lemenih *et al.* (2004)	1. Plantation: monoculture of coniferous trees (*Pinups patula and Cupressus lusitanica)* 2. Plantation : broadleaved trees (*Cordia africana* and *Eucalyptus saligna*)	28	Woody plant (size ≥1cm-DBH) Leaf area index Soil temperature and moisture	Determine gap of plantation influences recruitment of native woody plant and investigate leaf area index, soil temperature and moisture
Ethiopia - Highlands forest	1.Yirdaw (2001) 2.Yirdaw (2002) -Thesis	Plantations: monoculture of Pine and Eucalyptus (*Eucalyptus globulus, Pinus patula, Cupressus lusitanica, Grevillea robusta,* and *Juniperus procera etc.*)	14−16 11, 37	Trees and shrubs Woody plants Canopy gap	1. Diversity of naturally-regenerated native woody species 2. Effect of canopy gaps to plant diversity and growth of native woody species
Ethiopia - Highlands forest	1. Yirdaw, & Luukkanen, (2003) 2. Yirdaw & Luukkanen, (2004)	Plantations: monoculture of *Eucalyptus globulu* Labill. ssp. *globulus*	2−11	Understorey woody species	1. Investigate the indigenous woody species diversity in *Eucalyptus globulus* Labill. ssp. *globulus* plantations 2. Determine photo-synthetically active radiation transmittance of forest plantation canopies
Kenya, Western region	Farwig *et al.* (2009) (Thesis)	1. Plantation: monoculture of exotic sp. (*Bischofia javanica*) 2. Plantations: monoculture of indigenous sp. (*Maesopsis eminii*) 3. Plantations: mixture indigenous sp. (*Prunus africana, Olea capensis*, etc.) 4. Secondary forest 5. Natural primary forest		Seedling recruitment	Plantation types and age influences tree composition, diversity and seedling height

Table D: (count.)

Location	Citation	Restoration Method	Age (yr.)	Target	Study Topics
Southern America					
Uganda - Semideciduous forest - Evergreen forest	Zanne & Chapman (2001) Duncan & Chapman (2003)	Plantation: monoculture of Pine (*Pinus patula* and *P. caribaea*) and Cypress (*C. Iusitanica*)	*ca.*10‒20	Tree regeneration Seeded animal-dispersed species	Evaluated tree species richness and stem density in pine plantations
Australia					
Australia - Rainforest	Kanowski *et al.* (2003)	1. Plantations: monoculture (*Araucaria cunninghamii*, *Agathis robusta*, *Flindersia brayleyana* and *Toona ciliata*) 2. Plantations: mixed-species of Australian rainforest and Eucalyptus trees 3. Restoration plantings: mixture of trees and shrubs (generally 20–100 species, mostly local species and provenances) 4. Secondary forest: Old-field regrowth 5. Old-gorwth forest: Remant of degraded forests by selective logging	5–15 5–10 6–22 Tropics: 10–20 Subtropics: 30–40	Woody plants	Develop and apply a method for quantifying the structural attributes of reforestation sites in tropical and subtropical.
Australia, Queensland - Tropical Forest	Goosem &Tucker (1995, 2003); Tucker & Murphy (1997)	Restoration planting: native trees FWS	5, 7	Biodiversity Native forest tree seedlings	Effects of ecological rehabilitation on vegetation recruitment

Table D: (count.)

Location	Citation	Restoration Method	Age (yr.)	Target	Study Topics
Australia					
Australia, Queensland - Tropical Forest	White, *et al*. (2004)	Restoration planting: native trees FWS	ca.10	Seed sources distance to the restoration sites	Study seed dispersal revegetated in isolated patches and leaf area index (LAI)
Australia, Northern Tropical Forest	Keenan, *et al.* (1997)	1. Plantation: monocultures of the exotic *Pinus caribaea* and 2. Restoration: natives species of *Araucaria cunninghamii, Flindersia brayleyana* and *Toona ciliata*	5-63	Understorey plant	Restoration plant Biodiversity beneath tree plantations and natives species restoration forests
Australia, NSW - Riparian forest	Harris, *et al.* (2012)	Restoration planting: native trees FWS		Seed sources distance to the restoration sites	Restoration native canopy affect understory composition of Riparian forest
Australia, Queensland - Tropical Forest	Florentine *et al*. (2016)	1. Plantation: pioneer sp. monoculture (*Homalanthus novoguineensis)* 2. Restoration planting: "Homalanthus Framework" a group of pioneer species (*H. novoguineensis* + 8 species) 3. Restoration planting: "Alphitonia Framework" (*A. petriei* and 8 species) 4. Restoration planting: "Maximum Diversity" using pioneer species, middle-phase species, and mature-phase species	8, 13, 21	1. Habitat structure 2. Planted and recruited seedlings	Effectiveness of different planting frameworks and the restoration Ecology for recruitment of tropical rainforest species on ex-rainforest land

Table D: (count.)

Location	**Citation**	**Restoration Method**	**Age (yr.)**	**Target**	**Study Topics**
Asia					
Japan	Nagaike *et al.* (2003) Nagaike *et al.* (2005)	Plantation: *Lari kaempferi*	15−60	Plant species diversity	Plant species diversity, richness, and composition among stand age of plantations Effects of different forest management on Ficus species diversity
	Nagai & Yoshida (2006)	Plantation: *Picea glehnii*	21−26	Plant species diversity	Variation of understorey structure and species diversity
China, Hebei - Deciduous broadleaf and coniferous forest	Wang *et al*. (2017)	Plantation: monoculture of Chinese pine (*Pinus tabuliformis* Carr.)	*ca.* 30	Canopy gap Pine seedlings (height < 100 cm) Pine saplings (height > 100 cm)	Evaluate gap creation to the recruitment of canopy trees for sustainable management
Hong Kong	Nichol, *et al.* (2017).	1. Plantations: *Pinus massoniana* 2. Secondary forest	>70	Plant structure	Spatial patterns of degraded tropical forest and biodiversity restoration over 70-years of succession
India, Central India - Dry deciduous	Saha (2001)	1. Plantations: Teak (*Tectona grandis*) 2. Secondary forest	16−18 30−35	Seedlings and adult trees	Compare seedlings and adult trees between teak plantations and secondary forest
India, Western Ghats - Rain forest	Raman, *et al.* (2009)	Restoration : native rain forest tree species	1−4	Planted seedlings	Survival of planted seedlings under different site conditions such as level of degradation and plantation treatment.

Table D: (count.)

Location	Citation	Restoration Method	Age (yr.)	Target	Study Topics
Southeast Asia					
Indonesia, Harapan - Tropical rainforest	Harrison & Swinfield (2015)	1. Direct seeding 2. Enrichment planting	−	Planted trees Restoration treatment design	Experiment restoration design activities of thinning, direct seeding and enrichment planting to test the economic viability of large-scale restoration treatments.
Malaysia, Sabah - lowland dry dipterocarp forests	Cosset *et al* (2017)	1. Enrichment planting: with a variety of native dipterocarp (95%) and wild fruit (5%) species 2. Secondary forests with logged activities 3. Primary forests	ca.14	Avian species	Impact of restoring logged forests on avian phylogenetic and functional diversity
Lao - Seasonally dry tropical forests	McNamara (2009) (Thesis)	1. Plantation: monocultures of native and exotic species (*Acacia mangium, Ailanthus malabarica,* and *Irvingia malayana*) 2. Plantation: mixture of native and exotic species (*Eucalyptus camaldulensis and Tectona grandis*)	7−10 7−10	Seedlings	Seedlings growth and survival rates and the regeneration potential of different restoration designs in Lao and Vietnam and
Vietnam - Seasonally dry tropical forests	McNamara (2009) (Thesis)	3. Restoration planting: 16 spp. enrichment planting 4. Secondary forest: Old-field regrowth	4−9 ca. 7−10		
Thailand					
Thailand, Sakaerat	Kamo *et al.* (2002)	1. Plantation: monocultures of exotics species (*Acacia sp.* and *Eucalyptu sp.*) 2. Plantation: indigenous species (*Dalbergia cochinchinensis, Xylia kerrii* and *Pterocarpus macrocarpus*)	12	Understorey plant	To compared understorey plant species in plantations of indigenous species and the single exotics
Thailand, Chiang Mai	Oberhauser (1997)	Plantations : monoculture of Pine (*Pinus kesiya*)	7, 12, 21, 28	Seedlings (≥1 cm-DBH)	Investigate structure of plantation and rate of forest succession (seedling density, species richness) under pine plantation.

Table D: (count.)

Location	Citation	Restoration Method	Age (yr.)	Target	Study Topics
Thailand					
Thailand, Lampang	Kaewkrom *et al.* (2005)	1. Plantation: monoculture of Teak (*Tectona grandis*) 2. Plantation : mixture of Teak species	12	Species of understorey plant	The influence of the number of species in nurse plantations on the diversity and number of seedlings representative of mature native forests
Thailand, Lampang / mixed deciduous forest	Koonkhunthod *et al.* (2007)	Plantation : monoculture of Teak (*Tectona grandis*)	37	Woody trees (size ≥4.5 cm-DBH)	Understorey structure and composition of woody tree in plantation
Thailand, Northern region	Asanok *et al*. (2013)	1. Enrichment plantation: planting of native species in secondary forest 2. Secondary forest 3. Primary forest	*ca.*15	Trees, seedling and sapling	Relationships between functional traits and the ability of forest tree species to reestablish in secondary forest and enrichment plantations
Thailand : Lampang / mixed deciduous forest	Yoshifuji, *et al.* (2006)	Plantation : Teak (*Tectona grandis*)	37	Forest soil	Variations in the canopy duration and transpiration period as measures of growing season length using time series data of radiative transmittance and sap flux in a teak plantation in Thailand

Table D: (count.)

Location	Citation	Restoration Method	Age (yr.)	Target	Study Topics
Thailand, Chiang Mai, BMSM village - Evergreen seasonal forest					
Thailand, Chiang Mai, BMSM village	Khopai & Elliot 2003	Restoration planting: 20-30 native trees FWS	1, 2	Ground flora and trees (> 1 m tall) species	1) The effects of forest restoration activities and the recruitment of non-planted tree species 2) The increase of species diversity or changing of ground flora
	Elliott *et al.* (2003)	Restoration planting: 20-30 native trees FWS	–	Framework tree species	1) Selecting framework tree species for restoring forest habitat and diversity. 2) Test 37 native forest tree species to accelerate the recovery
	Singhaseni (2008) -Thesis	Restoration planting: 20-30 native trees FWS	2, 4, 8	Plant seedlings	1) Seedling recruit under FWS restoration 2) Investigate density diversity of seedlings
	Wydhayagarn *et al.* (2009)	Restoration planting: 20-30 native trees FWS	8	Bird species Plant seedlings	1) Survey Bird visiting FWS trees in the restoration 2) Survey seedling establish underneath FWS trees
	Jinto (2010) - Thesis	Restoration planting: 20-30 native trees FWS	9	FWS trees	Forest structure and vegetation diversity in restoration plots and natural forest
	Kavinchan (2014)	Restoration planting: 20-30 native trees FWS	3, 7, 10	Litter, belowground carbon	Litter, belowground carbon of restoration forest and DSF

APPENDIX E

Secondary Information

The online information sources and references (alphabet order) for the secondary information of

Copy Right to/ Organization	Online available at;
DNP - Forest Herbarium of the Department of National Parks, Wildlife and Plant Conservation of Thailand	1. http://www.dnp.go.th/botany/detail_wordsci.aspx 2. http://biodiversity.forest.go.th
FOC - Flora of China (eFloras, 2008)	http://www.efloras.org
FORRU - Forest Restoration Research Unit at Biological Department of Science Faculty, Chiang Mai University, Thailand	http://www.forru.net and Maxwell and Elliott (2000)
KEW - Kew Herbarium Catalogue of the Royal Botanic Gardens	http://apps.kew.org/herbcat/navigator.do?
QSBG - Queen Sirikit Botanic Garden; plant database	http://www.qsbg.org/Database/plantdb/herbarium/index.asp

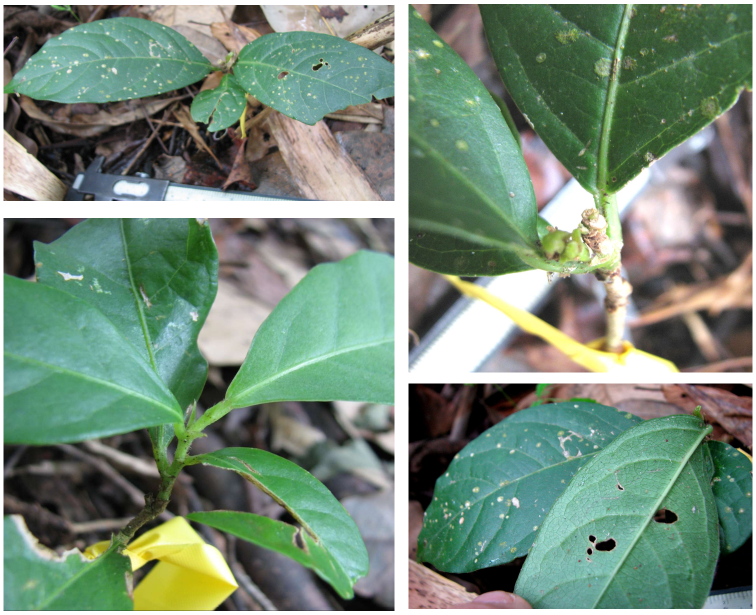
APPENDIX F

Picture of Seedlings

Appendix F.1. Picture of top 5 Recruit Native Species (alphabet order)

1) *Antidesma bunius* (L.) Spreng: Seedling has alternate-arranged simple leaves. Leaf blade is leathery or thick papery oblong, elliptic ((5-)10-23(-32) × (2-)3-10 cm). Upper leaf color is shiny dark green (sometimes grayish or reddish) when under leaf color is lighter green. Leaf base and apex are acute to round. Midvein is flat to impressed adaxially with (5-)7-11 pairs of lateral veins. Petioles are green and thick 3-10 mm-long (eFloras, 2008).

2) *Aporosa octandra* (B.-H. ex D. Don) Vick. var. *octandra*: Seedling has thick, elliptic leaf blade with bright dark green upper leaf and lighter green underneath (2-8 × 5-23 cm). Leaves are simple, alternately arranged, with obliquely ovatelanceolate stipules (4–6 mm). Petioles are 5–12 mm-long with 2 glands at apex. (QSBG and FORRU).



3) *Artocarpus lacucha* Buch.-Ham. ex D.Don: Seedling has broadly elliptic to elliptic leaf blade (25-30 × 15-20 cm) densely covered with yellow bristles. Leaves are distichous with petioles length about 2-3 cm covered with yellow bristles. Leaf base is broadly cuneate to obtuse. Leaf margin is entire or with small teeth (eFloras, 2008).

4) *Litsea salicifolia* (Roxburgh ex Nees) Hook. f.: Seedling has alternate leaves with long thin petiole (1-1.5 cm). Leaf blade is long elliptic (9-19 × 3-5.5 cm) with base acute, apex acuminate or acute. Upper leaf color is yellowish-green with and yellow-brown puberulent when young abaxially. Under leaf color is lighter green. Leaf base is acute and leaf apex is acuminate or acute (eFloras, 2008).

5) *Micromelum integerrimum* (B.-H. ex DC.) Wight & Arn. ex M. Roem. var. *intergerrimum*: Seedling has alternate to subopposite compound-leaves with 7-15-foliolate, but it looks like simple leaf when young. Leaflet are elliptic, lanceolate, or ovate blades, with dark green surface, entire or undulate margin, and acuminate apex (eFloras, 2008)

Appendix F.2. Picture of 2 Super-abundant Species

1) *Castanopsis calathiformis* (Skan) Rehder & E.H.Wilson: Seedling has spiral-arranged simple leaves. Leaf blade is oblong to obovate-elliptic, thickly papery, very variable in size, often 15-25 × 5-9 cm. Upper leaf color is yellowish green covered with brown-red slightly adnate small lamellate waxy scalelike trichomes when young. Leaf base cuneate with undulate and crenate margin, and acute acuminate or rounded apex, (eFloras, 2008)

2) *Litsea monopetala* (Roxb.) Pers.: Seedling has alternate-arranged simple leaves. Leaf blade is broadly ovate or obovate to ovate-oblong (8-20 × 4-12 cm), densely covered with ferruginous pubescent. Leaf is rounded or acute base, and obtuse or rounded, rarely acute apex. Petiole length is about 1-3 cm (eFloras, 2008).