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RESTORATION ECOLOGY OF MIXED DECIDUOUS FOREST ECOSYSTEM WITH TEAK IN NORTHERN THAILAND

Miss Puangpaka Kaewkrom

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By	Miss Puangpaka Kaewkrom
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Thesis Advisor	Associate Professor Jiragorn Gajaseni, Ph.D

Accepted by the Faculty of Science, Chulalongkorn University in Partial Fulfillment of the Requirements for Doctor's Degree.

....Dean of Faculty of Science

(Professor Piamsak Menasveta, Ph.D)

Thesis Committee

....Chairman

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....Member

(Associate Professor Thaweesakdi Boonkerd, Ph.D)

....Member

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การลดลงของพื้นที่ป่าไม้ในประเทศไทยเป็นปัญหาที่ได้ทวีความรุนแรงขึ้น เนื่องจากการเพิ่มขึ้นของประชากรและความต้องการพื้นที่สำหรับทำการเกษตร และการใช้ประโยชน์จากไม้โดยเฉพาะไม้สัก ทำให้พื้นที่ป่ามีปริมาณลดลงเป็นจำนวนมากและพื้นที่ส่วนใหญ่ได้กลายเป็นพื้นที่เสื่อมโทรม ดังนั้นจึงได้มีการพัฒนาระบบหมู่บ้านป่าไม้ ซึ่งพัฒนามาจาก Taungya system เพื่อนำมาใช้ปรับปรุงพื้นที่เสื่อมโทรมในเขตภาคเหนือของประเทศ โดยมีเป้าหมายเพื่อฟื้นฟูพื้นที่เสื่อมโทรมให้มีลักษณะเป็นธรรมชาติหรือใกล้เคียงธรรมชาติ และอนุรักษ์ความหลายหลากทางชีวภาพ ดังนั้นงานวิจัยนี้จึงได้พัฒนากลยุทธ์ในการฟื้นฟูต่อเนื่องจากการพัฒนาระบบหมู่บ้านป่าไม้ จุดประสงค์ของการวิจัยเพื่อศึกษาอิทธิพลและสาเหตุของความหลากหลายของชนิดพันธุ์พืชในระบบวนเกษตรต่อชนิดและความหลากหลายของพืชที่เกิดตามธรรมชาติ ศึกษาการกระจายของเมล็ดไม้เนื้อแข็งสู่พื้นที่ทดลอง และศึกษาอิทธิพลของแปลงทดลองที่มีความหลากหลายทางวนเกษตรต่อพืชที่ปลูกเสริม โดยใช้แปลงทดลองที่ปลูกพืชต่างกัน 5 แบบ คือ 1. สัก 2. สักและมะขาม 3. สักและซ้อ 4. สัก, มะขาม และซ้อ 5. สัก, มะขาม และมะม่วงหิมพานต์ ผลจากการศึกษาความแตกต่างจำนวนชนิดและความหลากหลายของพืชที่เกิดตามธรรมชาติ พบว่าแปลงปลูกป่าแบบผสมที่มี 3 ชนิดคือ สัก, มะขามและมะม่วงหิมพานต์ มีความหลากหลายของชนิดของพืชที่เกิดตามธรรมชาติมากที่สุดและน้อยที่สุดในแปลงที่ปลูกสักเพียงชนิดเดียว นอกจากนี้ยังพบว่าความหนาแน่นของหญ้าคาและสาบเสือ มีค่าสูงที่สุดในแปลงปลูกสัก และน้อยที่สุดในแปลงที่ปลูกป่าแบบผสมที่มี 3 ชนิด แสดงให้เห็นว่าแปลงที่มีความหลากหลายของพืช 3 ชนิดช่วยส่งเสริมความหลากหลายของพืชที่เกิดเองตามธรรมชาติและมีผลยับยั้งการเจริญของหญ้า ในขณะเดียวกันพืชในสังคมที่เจริญเต็มที่ (climax species) สามารถเจริญได้ในแปลงทดลองที่มีความหลากหลายของชนิดสูง และมีผลต่อการนำมาใช้ในกลยุทธ์การฟื้นฟู ผลของการสำรวจการกระจายเมล็ดไม้เนื้อแข็ง พบว่าเมล็ดมีการแพร่กระจายจากป่าธรรมชาติเข้าสู่พื้นที่แปลงทดลองที่มีการปลูกแบบผสมน้อยมากทั้งในด้านชนิด และจำนวน การทดสอบความมีชีวิตของเมล็ดไม้ พบว่ามากกว่าร้อยละ 65 มีอัตราการงอกของเมล็ดต่ำกว่า 50 % ผลของการกระจายและอัตราการงอกที่ต่ำ แสดงให้เห็นว่าอัตราการทดแทนตามธรรมชาติเกิดได้ช้า และผลของการศึกษาอิทธิพลของแปลงทดลองที่มีการปลูกพืชที่ต่างกันทั้ง 5 แบบ ต่ออัตราการอยู่รอดและการเติบโตของพืชที่ปลูกเสริมในกระบวนการฟื้นฟู พบว่าพืชในสังคมที่เจริญเต็มที่มีอัตราการรอดตายสูง และมีความสามารถในการยึดครองพื้นที่และเติบโตในระยะแรกของกระบวนการเปลี่ยนแปลงแทนที่แบบทุติยภูมิ การเพิ่มขึ้นของมวลชีวภาพของพืชที่ปลูกเสริม สูงที่สุดในแปลงทดลองการปลูกแบบผสมที่มี 2 ชนิดคือ สักและซ้อ ดังนั้นแปลงทดลองนี้จึงมีปฏิสัมพันธ์ในเชิงส่งเสริม ทำให้มีการเพิ่มขึ้นของมวลชีวภาพของพืชที่ปลูกเสริมได้ดีที่สุด จากผลการทดลองสรุปได้ว่า ความสามารถในการเกิดแทนที่ตามธรรมชาติของพืช และการยับยั้งการเจริญเติบโตของหญ้าและวัชพืชจะเกิดได้ดีที่สุดในแปลงปลูกป่าแบบผสมที่มี 3 ชนิด นอกจากนี้มีการกระจายของเมล็ดไม้เนื้อแข็งเข้ามาในพื้นที่แปลงทดลองในปริมาณน้อย และผลการสำรวจปริมาณของพืชในสังคมที่เจริญเต็มที่ ที่เกิดทดแทนตามธรรมชาติมีน้อยในแปลงปลูกป่าแบบผสม ดังนั้นการปลูกเสริมด้วยพืชในสังคมที่เจริญเต็มที่ จึงเป็นกลยุทธ์ที่เหมาะสมและมีประสิทธิภาพในการฟื้นฟูพื้นที่เสื่อมโทรม และแปลงปลูกป่าแบบที่ปลูกสักและซ้อมีประสิทธิภาพสูงที่สุดในการส่งเสริมการเติบโตของพืชที่ปลูกเสริมและเร่งกระบวนการทดแทนตามธรรมชาติในกระบวนการฟื้นฟูระบบนิเวศ

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สาขาวิชา..........วิทยาศาสตร์ชีวภาพ........ ลายมือชื่ออาจารย์ที่ปรึกษา...............................

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Forest degradation is proceeding at an unprecedented rate in Thailand, due to population growth, agricultural land demand, as well as timber production demand especially Teak. Large areas of forest have been disturbed leaving behind with a less production area. The implementation of the Modified Forest Village System which is a modified from Taungya system is an attempt to solve this problem in Thailand's northern regions, by established Teak plantation combined with various economic tree species. The future restoration expected goal is to maintain species diversity by converting degraded areas back to natural or near natural forest. Thus, development of restoration strategies must follow advances in research. The objectives of this study were to investigate the effects and casual factors of different mixed-species plantations on floral composition and diversity regenerating in the understorey. Seed input to the plantation via both seed dispersal and the soil seed bank was studied. Finally, the study focused on the effects of five mixed plantation stands on survivorship and productivity of enrichment seedlings in restoration processes. This study have 5 stands of mixed-species plantation included i) *Tectona grandis* ii) *T. grandis and Tamarindus indica* iii) *T. grandis* and *Gmelina arborea* iv) *T. grandis*, *T. indica* and *G. arborea* and v*) T. grandis*, *T. indica* and *Anacadium occidentale*. The results of the study showed that the highest floristic diversity regenerating in the understorey of the five mixed plantation types was found in the three-species plantation. Diversity indices gradually decreased from three-species to single-species plantations when considering both woody and non-woody species. However, the density of herb (*Imperata cylindica* and *Chromolaena odorata*) in single-species plantations was higher than in the two and three-species plantations. This indicates that the three-species plantation enhanced species diversity and suppressed the growth of grass in the understorey better than the single-species plantation. Dominant and climax species were found in single as well as mixed-species plantations, but at a low density. This suggests that climax species can establish during early stages of succession, and is useful to enhance developing and adjusting restoration strategies. For this reason enrichment planting by selecting some of climax tree species has higher potential to enhance restoration mechanisms. Observations on seed inputs showed low seed dispersal and soil seed banks of native woody species being transferred to the mixed plantation area. The number of species and their density decreased from natural forest, through the ecotone to the mixed-species plantation. Observations on seed germination indicated that 65% of woody seeds species had a germination rate of less than 50% under nursery conditions. These factors, of low woody-seed input and low germination rate for some dominant tree species, may retard the recovery process in mixed tree species plantations. The high survival rate of pre-climax enrichment seedlings has proved the success of this enrichment-technique restoration strategy. Teak (*Tectona grandis*) and Gmelina (*Gmelina arborea*) showed synergistic interactions and encouraged higher productivity accumulation in enrichment seedlings than in single and three-species plantations. Results suggest that the mixed-species plantation of Teak and Gmelina is a good foster combination for restoration by planting techniques. In conclusion, mixed-species plantation has more effective to accelerating natural succession better than single-species plantation by enhancing species diversity in their understorey. Due to low primary native tree seedling recruitment and seed sources in mixed plantations, enrichment planting is a useful technique of the restoration strategy. In addition this research clearly indicates that enrichment planting of pre-climax species has the capability to increase restoration mechanisms by accelerating successional processes. Moreover, it can be suggested that mixed-species plantations, especially Teak and Gmelina, can be managed sustainably and can be useful for restoring degraded land.

Department…………….-…………..… Student's signature…………………………....
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ABBREVIATIONS

AGR	Absolute growth rate
ANOVA	One-way Analysis of Variance
B	Biomass
C	Celsius
C	Sorensen similarity index
CA	Correspondence Analysis
cum	Cubic meter
cm	Centimeters
D_0	Diameter at ground level
DBH	Diameter at breast height
E	East
E	Evenness Index
ECO	Ecotone
FAO	Food and Agriculture Organization
FIO	Forest Industry Organization
g	Gram
GIS	Geographic Information systems
h	Height
H	Shannon-Wiener's index of species diversity
H_{max}	Maximum for Shannon-Wiener's index
ha	Hectare
kg	Kilogram
km	Kilometers
LSD	Least –Significant Difference
m	Meters
m^2	Square meter
m^3	Cubic meter
mm	Millimeter
MDF	Natural mixed deciduous with teak forest
mg	Milligram
MP	Mixed species plantation
N	North
OM	Organic matter
r^2	Coefficient of determination
RGR	Relative growth rate
S	Shrub
SD	Standard deviation
Sig	Significant
sq. km	Square kilometer
T	Tree
Temp	Temperature
UTM	Universal Transverse Mercator
WC	Woody climber
Wb	Branch dry weight
Wl	Leaf dry weight
Wr	Root dry weight

ABBREVIATIONS (Cont.)

Ws	Stem dry weight
Yr	Year
%	Percentage

CHAPTER ONE

INTRODUCTION

1.1 General Introduction

Mixed deciduous forest or monsoon forest (Ogawa, Yoda and Kira, 1961, Royal Forest Department, 1962) is the dominant forest type in northern Thailand (Figure 1.1). Teak (*Tectona grandis*) is the dominant tree species in this forest ecosystem, though in some cases it may be absent (Smitinand, 1966). Boontawee, Plengklai and Kao-sa-ard, (1994) and Sukwong (1994) has described ‘mixed deciduous forest with Teak’ or ‘Teak forest’ as a mixed deciduous forest subtype.

Mixed deciduous forest with Teak is the most valuable and important forest ecosystem in northern Thailand. Moreover, this forest ecosystem is rich in species diversity and composition with a complex structure. The complex community of this forest ecosystem provides a rich habitat for wild animals and insects. The complexity of trophic levels in this forest ecosystem preserves native species and genetic diversity. In addition, huge benefits have been raised from the export of deciduous timber products. A report by the Royal Forestry Department (2001) showed that *Tectona grandis* and *Pterocarpus macrocarpus*, which are dominant tree species of the Teak forest, generated more than 1.02 billion Baht from exports of logs and sawn timber products in 2001 for Thailand. This highly valuable forest ecosystem has supported the income of Thai people and produced economic gains for the country. In addition, the Teak forest ecosystem contains vast resources of non-timber products such as bamboo shoots, mushrooms and so on. These products have generated a healthy income for local people and are also a good food resource.

Past and present increasing demand for goods and services from the structure, function, diversity and dynamics of the Teak forest ecosystem has caused major disturbance, which has resulted in rapid forest degradation. Thus, destruction of deciduous forests in Thailand (Table 1.1), especially of mixed deciduous Teak forest, is widely acknowledged to be a serious problem, causing large losses of biodiversity.

This has affected the life and living style of local Thais who where, in the past, supported by the forest.

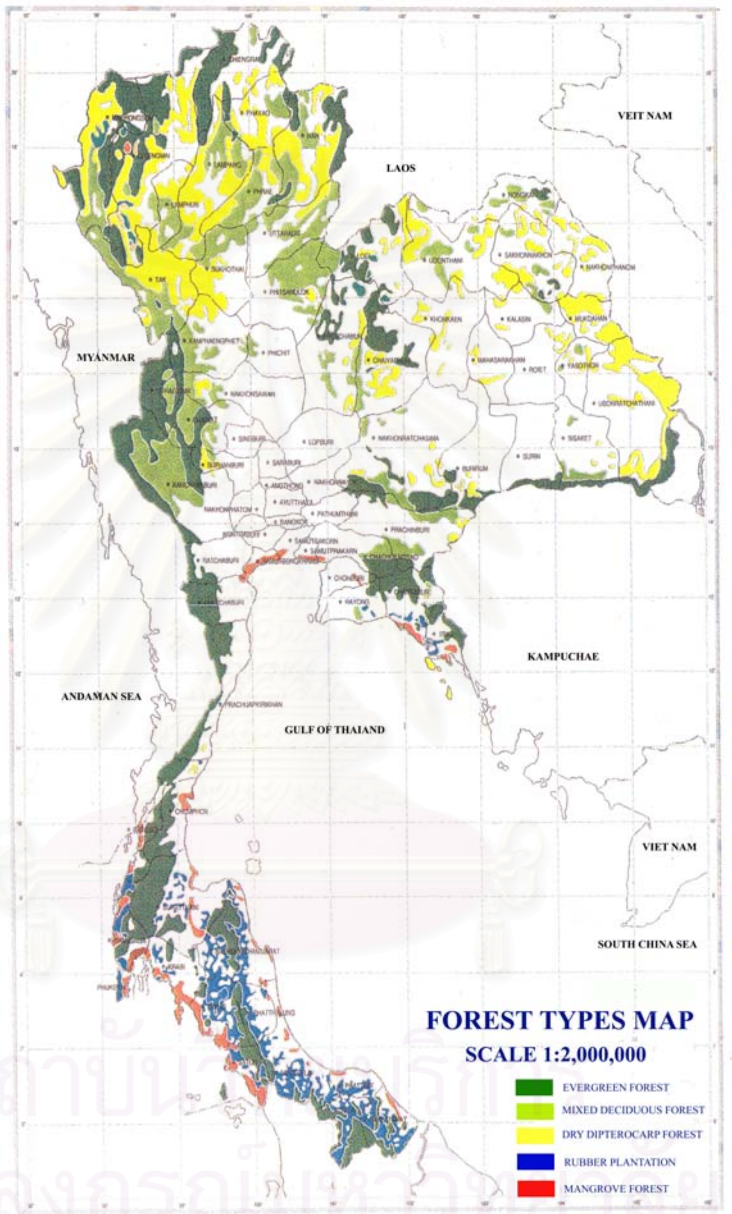

Figure 1.1: Map of forest types in Thailand

Source: Modified from Royal Forestry Department (1988)

Table 1.1: Change in forest area from 1990-2000.

Region	Total forest area in 1990 (1000 ha)	Total forest area in 2000 (1000 ha)	Forest cover changed 1990-2000	
			Annual change (1000 ha/yr)	Annual rate of change (%)
Thailand	15,886.00	14,762.00	- 112.00	- 0.70
Asia	551,448.00	547,793.00	- 364.00	- 0.10
World	3,963,429.00	3,869,455.00	- 9,391.00	- 0.20

Source: Modified from FAO (2003)

Rapid economic expansion in Thailand is one of the causes of increasing demand for industrial wood products and fuelwood. Ogawa, Yoda and Kira (1961) has suggested that mixed deciduous forest with Teak has been disturbed throughout the area due to selective logging for Teak. After the forest has been cleared, secondary thickets of tall bamboos often grow in their place. This species has the potential to suppress woody regeneration (Smitinand and Chumsri, 1985). Therefore, the structure and function of the forest is changed. Gajaseni and Jordan (1990) reported that factors such as shifting cultivation and intensification of modern agriculture have effectively increased deforestation in northern Thailand. For these reasons, the forest ecosystem in northern Thailand is in danger, as indicated by the large percentage of highly degraded land.

The report on forest depletion by FAO (2003) presented in Table 1.1 shows that global forests are decreasing by more than nine million ha, or 0.2% of forest cover per year. Serious problems of destruction are found mainly in tropical forests. In Thailand, which has good resources of Teak, forest area has decreased more than 112,000 ha, or 0.7% per year from 1990 to 2000. If the rate of destruction does not decrease, it will cause serious problems for the country. The destruction of Teak forest has created large areas of unused and abandoned land. These areas are unproductive for local Thais and the nation.

When forest ecosystems are disturbed, natural regeneration processes occur. The major sources of natural regeneration are stump sprouts, root sprouts and seeds from the soil seed bank and/or seed dispersal from adjacent mature forest and/or remnant trees. More than one method of regeneration may be employed. The rate of regeneration depends on many factors such as available seeds, frequency and intensity of disturbance, recruitment resources, seed dispersers, soil organic matter and so on. Therefore, the first species occupying the niche may have a greater chance of establishment and persistence in secondary succession. A species capable of establishment in the early stages of succession is a pioneer species. Natural recovery will then develop based on the classical successional theory (Clements, 1916).

A study of biological characteristics and the dynamics of regenerating species is necessary in order to improve the rate of success in restoration and reforestation projects. Meanwhile, the complex processes of tree regeneration and growth are an important consideration. The scale and intensity of invasion determines whether regeneration will proceed with immigrant species from adjacent areas of natural forest via seed dispersal and the soil seed bank. Woody seeds play an extremely important role in regeneration and it is likely that a lack of available seed is a limitation to the process of natural recovery.

With the destruction of mixed deciduous Teak forest, comes a loss of both biotic components, such as species diversity, structure and species composition, and also abiotic components such as loss of soil fertility, organic matter and/or moisture. Structure and species composition are changed. In this highly degraded environment, aggressive and pioneer grass species (*Imperata cylindrica*) can establish and persist because of their highly competitive potential in degraded areas (Eussen and Wirjahardja, 1973). These species often occupy and establish themselves in the early stages of secondary succession and suppress the natural regeneration processes of native tree species. This is one problem that impedes natural succession. The lack of native tree species seeds is another major problem for succession. Native tree species have an especially low potential for distribution and germination in abandoned areas. This is another factor retarding recovery processes. Moreover, occasional forest fires are a serious threat to the establishment of tree seedlings. Therefore, if we allow natural processes to repair degraded forest communities it would take a long time because natural successional trends and the dynamics of forest ecosystems are extremely slow (Jordan III, Gilpin and Aber, 1987).

Solutions to the slow natural recovery processes of the Teak forest ecosystem, the attempt at reforestation in terms of forest restoration, the problems of forest depletion, and socioeconomic factors for the people of Thailand have to be developed. The sustainability of restoration strategies, increased species diversity and the return of a natural forest ecosystem are major goals in forest management. Many reforestation programmes have been developed in Thailand. The first attempt to restore highly degraded land in Thailand was the Taungya system used by the Royal Forest Department. This is a forest plantation system (Jordan, Gajaseni and Watanabe, 1992) and is a specialized form of reforestation that aims to increase Teak production alongside local crops. The next method of forest restoration that proved successfully in the north of Thailand are the so-called “Forest Village System” (FVS), a modified version of the Taungya system and a modified of Forest Village System. FVS has been developed by Forest Industrial Organization (Corvanich, 1974). Meanwhile a modified FVS was developed by Gajaseni (1988). A modified FVS plants multi-purpose tree species, based on ecological and economic criteria, in combination with crops in the early stages. The Forest Village System uses multi-purpose species because it makes better use of the ecological niches, restores soil fertility and increases economic benefits to local people. Locals receive economic benefits from their crops in the initial stages, followed by income from the economically valuable trees and timber as the plantation ages. The success of the modified Forest Village System has been clearly demonstrated by Gajaseni (1988). However, when considering species diversity, this system does not show any advantages. The Taungya system, FVS and the modified FVS are management strategies that can sustainably improve productivity in degraded areas of northern Thailand. Moreover, the increased supply of wood from this type of management has the potential to reduce pressure on the natural forest, as well as contributing to increased social, economic and environmental benefits for landholders. Another effort to restore degraded land in Thailand is the “Framework species method” (Blaskesley et al., 2002; Elliott et al., 2002). This method plants mixtures of 20-30 pioneer and climax tree species in a single step. The diversity of native species enhances the complexity of the forest community, but provides no economic returns to local people. Because the method has different forest management goals from that of the Forest Village System, the modified FVS or the Tuangya system.

In other countries, many restoration techniques have been used to recover forest communities, such as the mixed-species plantation, the framework species method in Australia, the Miyawaki method in Malaysia (Miyawaki, 1993; Goosem and Tucker, 1995; Lamb, 1998). All methods have the same goal of restoring highly degraded forest areas. Each method has to be modified and developed for particular sites. In some areas one method may be successful but in other sites, it may not. It depends on the biotic and abiotic resources and intensity of disturbances.

The current study focuses on enrichment planting strategies and has the advantage of using an area previously rehabilitated by the modified Forest Village System to restore species diversity and forest structure. Moreover, the usefulness of the multi-purpose species plantation is used to enhance restoration mechanisms in this research. Enrichment planting is the best technique to take shortcuts through some of the early stages of succession. This technique can increase diversity, improve nutrient cycling and convert degraded forest areas to a natural or near-natural forest ecosystem. This research has tried to develop and modify alternative restoration strategies for application in northern regions of Thailand. The conceptual framework of the research is to explore species diversity and composition of vegetation at the understorey level of five multi-purpose species plantations. Moreover, the research also investigates woody seed input invading from adjacent natural mixed deciduous forest with Teak through the ecotone to the multi-purpose species plantations, in particular, on seed dispersal and the soil seed bank. This provides basic knowledge for developing a restoration strategy at the next stage. Finally, this study investigates the effects of different mixed tree species plantations as foster species for the early growth and survival of selective tree species in the restoration process. The conceptual framework for this study is illustrated in Figure 1.2. Specific details are shown in the methodology.

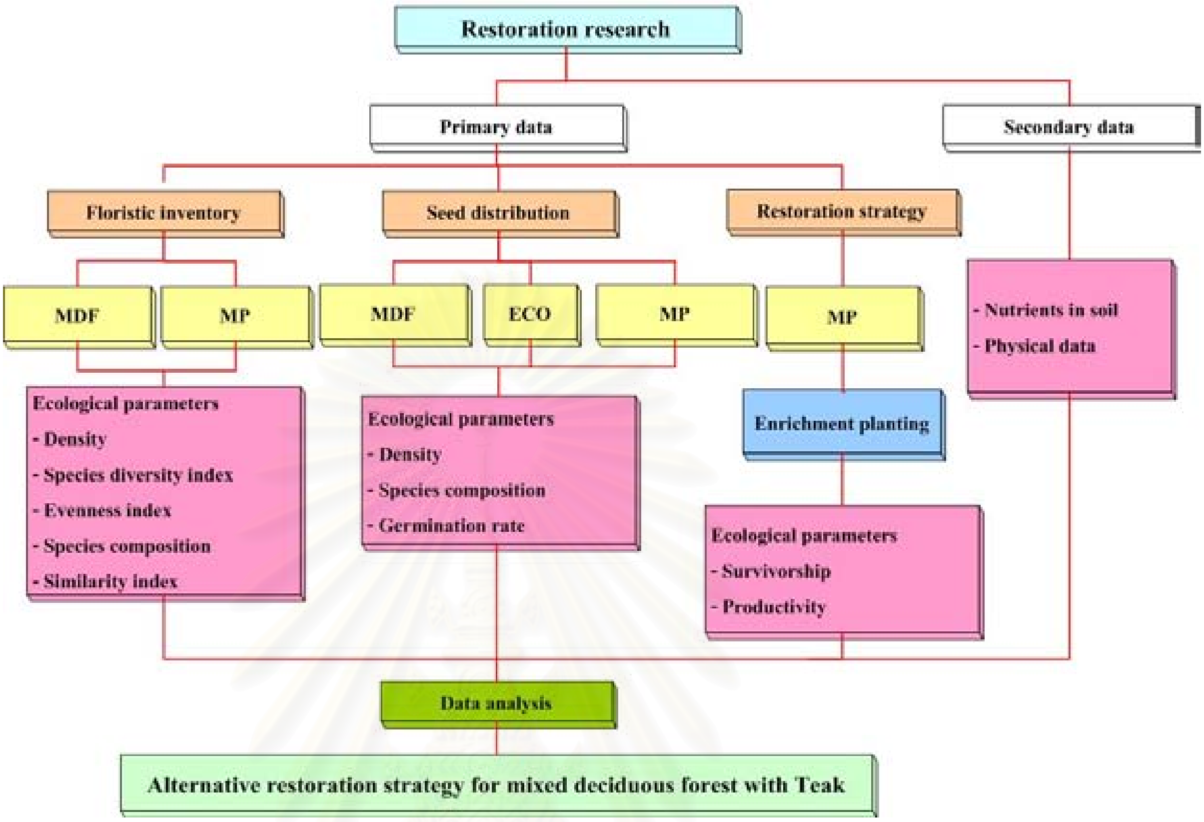

Figure 1.2: Conceptual model for the research

Note; MDF : Natural mixed deciduous forest with Teak

ECO : Transitional area between natural mixed deciduous forest with Teak and multi-purpose species plantation

MP : Multi-purpose species plantation

1.2 Objectives of the study

The research aims to investigate the effects and casual factors of different mixed-species plantations on floral diversity and composition regenerating in the understorey. Secondly, seed inputs, via dispersal and the soil seed bank, from adjacent natural forest via the ecotone to mixed-species plantations are investigated. Finally, the research looks at the use of mixed-species plantations as a foster to encourage enrichment planting techniques as a restoration mechanism for accelerating successional processes. The specific objectives are;

(i) To test the hypothesis that a greater number of native forest species will establish in mixed-species plantations than in single-species plantations.

(ii) To determine which native forest species are most successful in establishing themselves within the plantations.

(iii) To monitor dispersal of woody seeds and the seed bank invading from adjacent mature mixed deciduous Teak forest via ecotone areas to the mixed-species plantations. This data will be used to develop a restoration strategy.

(iv) To test the hypothesis that mixed tree species plantations encourage survival and productivity of enrichment seedling to a greater degree than pure plantations in the restoration strategy.

1.3 Study Hypothesis

The three specific hypotheses tested are,

(i) Single species plantations provide more available resources for regenerative species than mixed-species plantations. Therefore, the single-species plantation has more species diversity than the mixed-species plantation.

(ii) Due to limitations on seed distribution from adjacent natural forest to mixed-species plantations, species diversity and species composition of woody seeds in mixed-species plantations are lower than in natural forest.

(iii) Mixed-species plantations can maintain and enhance survival rate and productivity of restorative species to a higher level than the single-species plantation.

1.4 Anticipated benefits of the research

Due to the serious problem of deforestation of mixed deciduous forest with Teak in northern Thailand many restoration and rehabilitation approaches were developed to resolve this issue. This study focuses on the regeneration processes, which provides basic ecological data on restoration methods, as well as suggesting strategies for developing restoration. Therefore, the expected benefits of this research includes:

(i) An understanding of the interactions between mixed-species plantations and mature mixed deciduous with Teak forest ecosystems will enhance basic knowledge of ecological succession.

(ii) A new effective restoration strategy can be employed to accelerate the natural recovery processes.

(iii) The mixed-species plantations can be used as an effective restoration strategy.

1.5 Organization of the study

The thesis consists of five chapters as outlined below.

(i) **Chapter one**: This chapter gives a general introduction to the mixed deciduous forest with Teak. The importance of the issues, and reasons to develop the research are reported. Finally the specific objectives, the hypotheses and the expected outcome are described.

(ii) **Chapter two**: This chapter presents literature review and overview of the study. The characteristics of the mixed deciduous forest with Teak, overview of deforestation, regeneration, seed and regeneration, succession, impediment of succession and finally forest management and restoration are discussed.

(iii) **Chapter three**: The study area and methodology for this research are presented in this chapter.

(iv) **Chapter four**: The results and discussions on the four objectives of the research are presented. Results on community analysis in natural forest and mixed species plantation are reported as well as data on seed study. Finally the results of restoration strategy is presented. In this chapter, among result is following with discussion and it is clearly and end in individual section.

(v) **Chapter five**: This chapter summarizes the results of the study and provides recommendations for restoration management.

CHAPTER TWO

LITERATURE REVIEW

2.1 Characteristics of the mixed deciduous forest ecosystem with Teak

Mixed deciduous forest or monsoon forest (Ogawa, Yoda and Kira, 1961, Royal Forest Department, 1962) is composed mainly of deciduous tree species that remain virtually leafless throughout the dry season. Within mixed deciduous forest, three subtypes have been classified (Smitinand, 1966).

(i) Moist upper mixed deciduous forest
(ii) Dry upper mixed deciduous forest
(iii) Lower mixed deciduous forest

Teak (*Tectona grandis*) is the dominant species in the first two forest types. It is not found in lower mixed deciduous forest (Smitinand, 1966). Stands of forest that have Teak as the dominant species may also be called mixed deciduous forest with Teak or Teak forest (Boontawee, Plengklai and Kao-sa-ard, 1994, Sukwong, 1994). This forest ecosystem is a type of tropical seasonal forest. Mixed deciduous forest occurs naturally in Southeast Asia and is found throughout Viet Nam, Laos, Cambodia, Thailand, Burma and India (Bunyavejchewin, 1983). Mixed deciduous forest ecosystem with Teak encompasses large areas of peninsular India, Burma, Thailand and northwestern Laos (Kadambi, 1972, Kao-sa-ard, 1977). In Thailand, this forest type occurs mainly in the northern region (Table 2.1) where the elevation is 150-650 m above sea level (Bunyavejchewin, 1983). Annual rainfall ranges from 1,000 to 1,500 mm per year (Bunyavejchewin, 1983). Soils are typically medium texture sandy loam and sandy clay loam (Bunyavejchewin, 1985).

The mixed deciduous forest ecosystem with Teak shows three strata of tree layer. Common species in these layers are *Xylia xylocarpa* var. *kerrii*, *Pterocarpus macrocarpus* and *Terminalia mucronata* (Ogawa, Yoda and Kira, 1961, Bunyavejchewin, 1983, Gajaseni and Jordan, 1990). Other deciduous tree species such as *Spondias pinnata*, *Schleichera oleosa*, *Terminalia bellerica*, *Diospyros mollis* are found occasionally (Bunyavejchewin, 1983, Gajaseni and Jordan, 1990). The understorey is dominated by *Gigantochloa albociliat*, which replaces other deciduous species after disturbance by forest fire during the dry season (Kutintara, 1994). Fire is a strongly influencing factor that helps maintain species composition and structure in this type of forest ecosystem (Stott, Goldammer and Werner,1990, Sukwong,1994).

2.2 Overview of deforestation in Thailand

There are many causes of deforestation in Thailand. However, most are due to the increased demand for timber products, especially of Teak, which is the most valuable of all timber species in northern Thailand. Economic expansion and the need for Teak products are important factors in the depletion of Teak forests. The decrease in forest area is caused primarily by selective logging, and secondly by slash and burn cultivation (Ogawa, Yoda and Kira, 1961; Gajaseni, 1988). The demand for Teak timber from renewable Teak forest resources has not been optimised; therefore the high rate of deforestation has increased. Socioeconomic factors have also played a role. Thailand is still a developing country where people have a low income, especially in the north. This pressure has heightened both illegal and legal Teak logging. Moreover, a report on population projections for Thailand from 2000-2020 commissioned by the Royal Forestry Department (2001) has indicates an increase to 70 million people by 2020. Meanwhile, the Royal Forestry Department (2001) has also reported that the need for timber products is still high (more than 1.8 million m^3 in 2001), and that since the population is still increasing, it must put pressure on the forest ecosystem. The problem of forest destruction is serious.

The Royal Forestry Department (1998) reported that the annual decrease of forest area in Thailand from 1985-1995 was 1,938.10 km^2 per year. The highest rate of decrease was found in the northern region (1,024 km^2 per year). A global survey of deforestation by the FAO (2003) discovered that forest areas were being depleted at a rate of 112,400 ha per year between 1990-2000. The frequency of disturbance, both

Table 2.1: Area of forest types in 5 regions of Thailand.

Unit: sq.km

Forest types	Central	East	North	Northeast	West	South	TOTAL
Tropical Evergreen Forest	0.03	612.92	0.00	0.24	1.86	14,395.27	15,010.32
Dry Evergreen Forest	81.51	5,600.88	8,582.21	6,628.46	1,961.47	239.09	23,093.62
Hill Evergreen Forest	0.04	0.00	11,651.29	0.69	2.164.64	0.00	13,816.66
Pine Forest	0.00	0.00	93.16	22.35	0.00	0.64	116.15
Swamp Forest	3.20	1.04	0.00	0.00	1.40	286.67	292.31
Mangrove Forest	0.00	227.42	0.00	0.00	117.34	2,093.62	2,438.38
Inundated Forest	0.00	0.00	38.57	331.53	0.00	3.87	373.97
Beach Forest	0.00	448.38	0.00	0.00	0.00	113.89	562.27
Mixed Deciduous Forest	827.22	715.74	63,956.59	3,051.65	03,563.70	12.02	82,126.92
Dry Dipterocarp Forest	239.56	19.02	9,176.16	3,285.56	826.22	0.00	13,546.52
Bamboo Forest	0.00	106.60	129.83	185.32	782.77	10.48	1,215.00
TOTAL	**1,151.56**	**7,732.00**	**93,627.83**	**13,505.80**	**19,419.40**	**17,155.55**	**152,592.10**

Source: Royal Forestry Department (2003)

natural and man-made, has often occurred in the Teak forest ecosystem, leaving them as highly degraded and abandoned areas. This has become a serious problem for the nation. It has affected the economic returns to local people who have traditionally received benefits from the forest. Thus, the major concern for Thailand is the ever-increasing area of highly degraded land, with its negative ecological, environmental, social and economic implications.

2.3 Regeneration

After a natural forest ecosystem has been disturbed, natural successional process should occur by regeneration from plant resources, such as stumps, roots, seed input and/or seed bank. Thus in degraded land, one approach towards regeneration is to let nature heal itself. When ecosystems become disturbed, forest regeneration is influenced by four sets of limiting factors: disturbance, site resources, weed competition and lack of established plants or propagules (Hardwick, Healey and Blakesley, 2000). Due to these limiting factors, it can be seen that the ability of forests to regenerate quickly towards their original state is unfounded. A predictable suite of secondary species plants and animals also commonly dominates the early stages of succession, but latter stages are more stochastic in nature and chance events have a greater role in determining the species able to colonize (Webb, Tracey and Williams, 1972). Therefore, it is seen that successful regeneration is complicated. Required elements are seeds, seed dispersers, mycorrhizae and/or soil organic matter, as well as influencing physical factors such as light, temperature and moisture. Forest regeneration occurs by four pathways; advanced regeneration (seedlings), sprouts, seed bank, and seed dispersal (Uhl et al., 1990).

Regeneration is most successful if the soil has not been highly degraded and if fire can be controlled (Janzen, 1988). Research on the processes of natural regeneration after shifting cultivation indicates that trees begin to regenerate in the first and second year and grow to a size able to cover the understorey by the sixth year. Variable regeneration may occur within the same region (Van Son, 2000). In the tropics, rapid recovery by natural regeneration of the structural characteristics of secondary forest occurs in approximately 40 years (Zimmerman et al., 1995, Finegan, 1996, Guariguata et al., 1997, Aide et al., 2000).

Research on tree regeneration by Hartshon (1995) suggests that regeneration of trees in naturally disturbed areas can serve as a guide for the design of regeneration methods by promotion of early establishment, survival, and growth of desire tree species. Many researches have suggested that specialized forms of reforestation such as mono or mixed plantations and many types of agroforestry can encourage regeneration processes. Reports on forest plantations and agroforestry show the potential to accelerate floristic regeneration in the understorey (Lugo, Parrotta and Brown, 1993; Parrotta, 1995; Guariguata, Rheingans and Montagnini, 1995; Harrington and Ewel, 1997; Carnevale and Montagnini, 2002). Moreover, the number of successfully regenerative species varies with the species of tree planted. Upperstorey species (plantation species and agroforestry species) have a positive influence on the understorey microclimate, the forest floor and on soil development that help enhance understorey recruitment. Pre-climax canopy tree species create the habitat for fauna and the environment for many sub-canopy plants of the mature rain forest that require particular conditions for their early growth and establishment (Raich and Gong, 1990). Differences in the size of canopy gaps within rainforests promote differences in regeneration, survival and growth of pre-climax canopy tree species (Raich and Gong, 1990; Turner, 1990). This data coincides with Oberhauser (1997), in that the recovery processes of woody vegetation are faster in 3-needled pine (*Pinus kesiya*) plantations than in other afforested areas in northern Thai highlands. The strategies on plant regeneration are including; seeds, regeneration and resprouting.

2.3.1 Seeds and Regeneration

More extensive basic ecological information is needed for reforestation and restoration, including studies of natural regeneration and the factors that limit regeneration (Hardwick et al., 1997). Seeds are the main resource that determines the trends of regenerative processes. Seed distribution, via seed dispersal and the soil seed bank from adjacent forest to degraded areas influence plant dynamics in deforested areas (Bradshaw and Chadwick, 1980, Garwood, 1989, Nepstad et al., 1996). Seed availability can be characterized by three components: the presence, the gains, and the losses (Figure 2.1). The presence is confined to the viable seed bank.

Gains are made by seed rain and by the availability of seed sources. Finally, loss of viable seeds may be caused by a variety of factors such as longevity of the seed, germination processes and/or predation (Uhl et al., 1981, Wunderle, 1997, Hau, 1997). Chambers and MacMahon (1994) emphasize seed movement and fate because it is essential for ecosystem restoration and conservation efforts. It is clear that seedling establishment in disturbed areas is limited foremost by a lack of dispersal, high seed predation and by low germination rates in some species (Holl et al., 2000, Wijdeven and Kuzee, 2000). Therefore, the most important question in studying recovery of any ecosystem is whether propagules of target species are present in the disturbed area.

According to the model of movement and fates of seeds (Chambers and MacMahon, 1994) the first step is the movement of a seed from the parent to another surface, while the second step includes subsequent horizontal or vertical movements (Figure 2.1). Abiotic factors influence seed dispersal; the distance and type of movement depends on seed morphology, surface attributes, and the nature of physical forces. Biotic factors (animals) move seeds to new sites either passively on body surfaces or by ingestion, or actively by hoarding seeds or consuming fruits (Krefting and Roe, 1949; Temple, 1977; Ellison et al., 1993). Arrival at microsites suitable for germination and establishment is critical and is affected not only by abiotic and biotic factors but also by seed morphology and germination responses. This possibly affects the pattern of regeneration. Therefore, dispersed seed and persistence are important factors for natural recovery processes.

Since woody species are the main structural component of the forest ecosystem, they play an important role in the regeneration and dynamics of the forest because they act as a potential pool of propagules for regeneration (Pakeman and Hay, 1996). It is found that some woody species in temperate and tropical forests have a long viability in soil (Thomson and Grime, 1979). Similarly, in a slash and burn site, 95% of woody seedlings originate from the seed bank (Garwood, 1989). Elliott et al., (1997) has reported that the maximum dormancy for tree seeds is 253 days for *Euodia meliifolia* Benth.(Rutaceae). In contrast, some reports indicate that many tropical forest seeds have an extremely short viability and are therefore not present in the seed bank (Uhl, 1987; Nepstad et al.,1996). Seed dispersal, the soil seed bank and the potential to germinate have strong effects on regeneration dynamics.

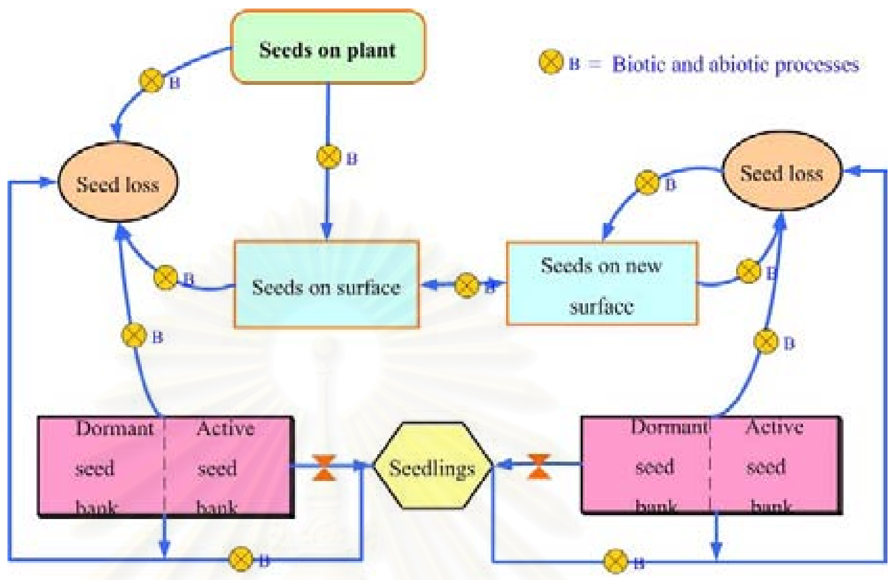

Figure 2.1: Conceptual model of the movements and fates of seeds.

Source: Chambers and MacMahon (1994)

2.3.2 Resprouting

Other than seeds, alternative plant resources for the regeneration process are stumps, shoots, coppice and roots, which are all non-sexual resources. The ability of woody species to resprout following disturbance affects floristic establishment. Many researches, such as that by Snedaker (1970) on tropical trees, has indicated that resprouting behaviour is the main method of plant regeneration in slash and burn sites. Stocker (1980) showed that 90% of tree species in slash and burn areas in Australia had the ability to resprout. In Amazonia, Uhl and Jordan (1984) found that tree species (individuals ≥ 2 m tall) established within the third year, but went on to conclude that the role of resprouting in the regeneration process was dependent upon the severity of disturbance. Plant species that regenerate from sprouts have a clumped dispersion and they have the ability to increase biomass quickly. Uhl and Jordan (1984) clearly showed that regeneration was rapid and that it should eventually reach and form the canopy layer. Therefore, resprouting is an important factor in plant regeneration in disturbed forest ecosystems. Seeds require several stages before establishment in other areas i.e. dispersal, germination and a mechanism for persistence, thus, in slash and burn areas, non-sexual resources play an important role in regeneration. Most woody species produce vigorous resprouts and re-emerge in early and intermediate successional stages.

2.4 Succession

The classical theory of succession as a deterministic process, with the community moving towards a climax state after passing through a series of distinct seral stages is no longer universally applicable. It is shown that both species composition and the physical structure of the community typically change over time (Clements, 1916). Natural successional processes take a long time to return degraded land back to a mature ecosystem. Time scales for biological development of ecosystems (Dobson, Bradshaw and Baker, 1997) are presented in Table 2.2. Cornell and Slatyer (1977) observed that several processes occur in secondary succession and have defined the three following models of successional processes.

(i) The facilitation model: The early successional species promote the establishment and growth of later species.
(ii) The tolerance model: Slow-growing species invade and mature in the presence of fast-growing species.
(iii) The inhibition model: The primary colonizers effectively withhold site resources from invading species.

Meanwhile, secondary succession is shown to be a series of complex processes. These processes are unable to predict the outcome of the change over time in any particular ecosystem because development passes through several stages of increasing structural and biological diversity. Thus, these differing ideas about secondary succession indicate that it is unnecessary to start with pioneer species in the initial stages of restoration, because recovery processes depend on propagule resources i.e. seeds and sprouts. Therefore, intermediate or pre-climax species may have more chance of establishment if they can distribute and occupy niches in the early stages of succession. Thus, chance and the potential of floristic species to occupy niches in early stages of succession may have changed trends of succession. These factors have encouraged the development of a restoration paradigm.

Table 2.2: Time scales for biological processes involved in the development of ecosystems.

Time scale (yrs)	Biological processes
1-50	Immigration and establishment of appropriate plant species
1-10	Accumulation of fine materials captured by plants
1-100	Accumulation of nutrients by plants from soil minerals
1-100	Accumulation of Nitrogen by biological fixation and from atmospheric inputs
1-20	Immigration of soil flora and fauna supported by accumulated organic matter
1-20	Changes in soil-structure and organic matter turnover due to plants soil micro-organisms, and animal activities
1-20	Improvements in soil water-holding capacity due to changes in soil structure
1-1000	Reduction in toxicities due to accumulation of organic matter

Source: Modified from Dobson, Bradshaw and Baker (1997)

2.5 Impediment of succession

Many factors are known to impede natural succession; frequency and intensity of disturbance, lack of nutrients, impacts of soil structure (Lamb, 1989). Seed availability is clearly a major limiting factor in natural recovery processes in Costa Rica (Wijdeven and Kuzee, 2000), whilst Zimmerman, Pascarella and Aide, (2000) reports a negative effect of distance from forest edge on the number of species in the seed rain but not on the number of seeds in abandoned pasture in Puerto Rico. Many reports have suggested that seed availability is a major limiting factor in forest recovery because tree seed density is low, the immigration of new recruits is hampered and seed predation seriously limits the available pool of species

(Holl, 1999; Nepstad et al., 1996, Wijdeven and Kuzee, 2000). Most pioneer species have a long viability and/or dormancy and are major competitive species to pre-climax trees, in contrast to the majority of mature forest tree species (Whitmore, 1983), so adding another competitive factor to primary tree establishment. In addition, larger seeds have a higher energy content and are more attractive to seed predators (Hammond, 1995). Another factor that may limit recovery is the low seed germination seen in a few species that may be caused by a lack of appropriate triggers, such as the required quantity or quality of light (Metcalfe, 1996). Therefore, these factors all limit floristic regeneration and retard successional processes.

2.6 Forest management and restoration

2.6.1 Definition and goals of forest management

Large areas of forest have been disturbed by an increasing demand for forest products, leading to the situation where the forest ecosystem can no longer heal itself by natural regeneration in deforested or highly degraded land. Natural recovery processes are no longer sustainable due to the great demand and need for timber and non-timber products. This situation is of major concern for afforestation in tropical regions including Thailand. Thus forest management is a strategy proposed to conserve and preserve the integrity of the forest ecosystem and ameliorate degraded areas.

Management practices need to contribute to the long-term supply of forest goods and services for the needs of the people. Forest ecosystem management integrates ecological concepts on a spatial scale. Furthermore, ecosystem management is evolutionary in nature rather than being a static set of perceptions (Franklin, 1997). The topic of forest ecosystem management has been covered by much research, all of which emphasizes that the goals of management are to sustain ecosystem composition, structure, and function over the long term (Grumbine, 1994; Christensen et al., 1996, Reid and Rice, 1997, Fox, 2000). A remarkably similar concept presented by Jordan (undated) defined forest ecosystem management as an investment of labour and resources to ensure regeneration of the forest, in order that supply could meet the demands and needs of the future. Its objective is sustainability with a reasonable margin of profit.

By inference, a major goal of forest management is the sustainability of both goods and services in the long term. There is currently no universally accepted definition of sustainable forestry (Fox, 2000). The most complete one to date is, "***Management should aim at forest structures which keep the rain forest ecosystems as robust, elastic, versatile, adaptable, resistant, resilient and tolerant as possible; canopy openings should be kept within the limits of natural gap formation; stand and soil damage must be minimized; felling cycles must be sufficiently long and tree marking so designed that a selection forestry canopy structure and a self regulating stand are maintained without, or with very little, sivilcultural manipulation, production of timber should aim for high quality and versatility. The basic principle is to mimic nature as closely as possible to make profitable use of the natural ecosystem dynamics and adaptability, and reduce costs and risks***." (Bruenig, 1996).

2.6.2 Definition of ecological restoration

In developing countries, most forest clearance is due to the great demand for agricultural land and from pressure of logging. Thus, large areas of forest in this region have been degraded with a consequent loss of biological diversity. Restoration is the core of forest management and aims to reconstruct the prior ecosystem. This includes not only the reestablishment of former functions but also the characteristic species, communities and structure (Grootjans et al., 2001). Likewise, Lamb (1998) has described forest restoration as a management paradigm that can return degraded areas to something approaching their original condition whilst improving landscape and biodiversity beyond that likely to occur naturally.

The conceptual basis for restoration is that the restored site should be self-sustaining (Jackson, Lopoukhine and Hillyard, 1995) and that the goal of restoration efforts is species diversity (Ehvenfeld and Toth, 1997; Lamb, 1998; Grootjans et al., 2001). According to the Society for Ecological Restoration, "***the present state of***

ecological restoration is the process of assisting the recovery and management of ecological integrity. Ecological integrity includes a critical range of variability in biodiversity, ecological processes and structures, regional and historical context and sustainable cultural practices" (Diggelen, Grootjans and Harris, 2001). The conclusive model of restoration is presented in Figure 2.2.

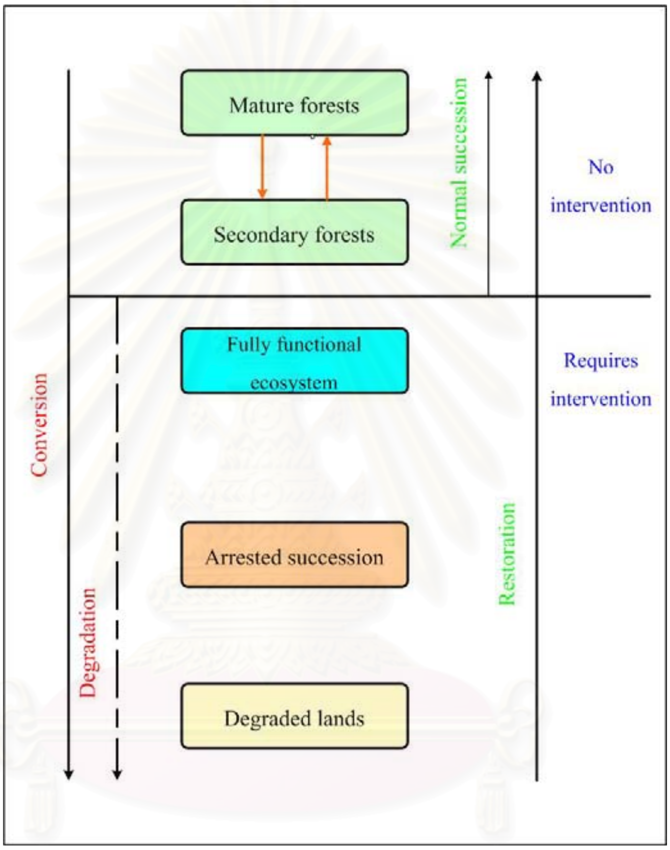

Figure 2.2: A model of human-based restoration/degradation and natural succession, and their modes of repair.

Source: Modified from Brown and Lugo (1994)

2.6.3 Restoration as a practice of forest management

There has been an increased global consumption of wood. Data on the consumption of forest products compiled by the Food and Agriculture Organization (1999, 2001, 2003) shows that demands for forest products in Asia increased during 1970 to 1996 (Figure 2.3). This has caused high rates of deforestation. Concern has been raised about the intensive exploitation of the forest ecosystem, the vast scale of forest degradation and rapid deforestation, all of which are widely recognized as major threats to environmental and economic stability, social welfare and biodiversity. Since degraded land cannot effectively contribute to sustaining economic activity, ecological restoration is a necessary step for increasing the chances of attaining sustainability. Furthermore, practical ecological restoration management should make use of ecological processes in order to maintain ecosystem structure, composition and function with minimum human intervention. Restoration is perhaps the only way to meet the increasing demand for forest products, as well as improving degraded sites or biodiversity for conservation and preservation purposes in the long term.

Restoration of forests on highly degraded land is becoming increasingly important as the total area of natural forest shrinks due to rapid conversion of the forest ecosystem without replacement. Parrotta (1993) has suggested that restoration strategies are necessary in the tropics where regenerative factors such as fire, low seed dispersal and harsh microclimates often slow or prevent natural successional processes in highly degraded areas. Therefore, restoration practices need human intervention in areas that have arrested natural succession (Figure 2.2). Several forms of reforestation have been used, such as the Taungya system, the Forest Village Plantation System (a modified Tuangya system), intensive and extensive plantation management (Gajaseni, 1988; Jordan, Gajaseni and Watanabe., 1992; Lamb, 1998), and conventional logging and sustainable timber management (Pearce, Putz and Vanclay, 2003).

2.6.4 Ecological restoration in Thailand

As a result of the rapid population increase in Thailand, an increased demand for Teak log and timber products (Table 2.3), as well as demand for agricultural land has been noted. This has led to the last areas of mixed deciduous forest ecosystem with Teak being lost or degraded. Forest management practices by the Royal Forest Department using the Taungya system were among the first efforts to increase Teak production in Thailand. The Taungya system was a one-forest

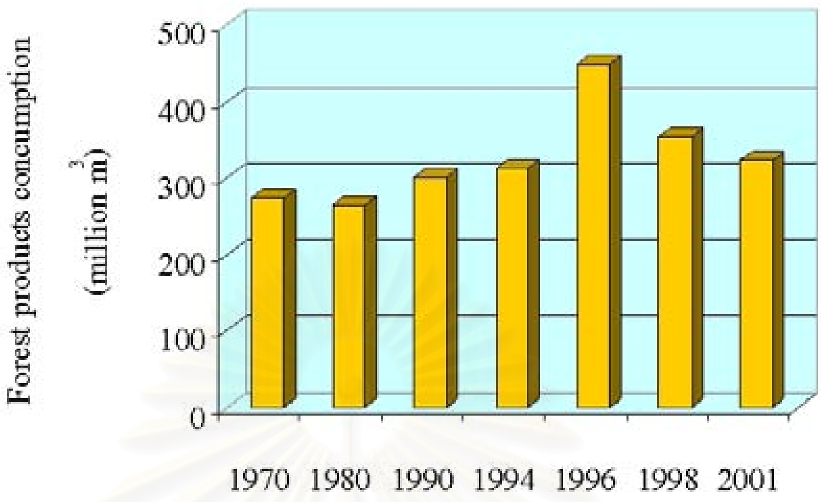

Figure 2.3: Forest product consumption in Asia (industrial round wood and sawnwood) from 1970 to 2001.

Source: FAO (1997, 1999, 2001, 2003)

management system that combined forest trees and cultivated crops (Jordan, Gajaseni and Watanabe., 1992). By this system farmers would receive benefits from their crops in the first few years of planting. The cultivation of crops also reduces costs for weed control. These were the advantages of the Tuangya system. However, many disadvantages were also noted i.e. no further beneficial returns to the farmer after harvest of his crop until the harvest of the forest trees many years later, the increased cost for weed control after cropping and high competition from the same species. The next method employed was a modified version of the Taungya system, which was developed in Thailand by the Forest Industry Organization and was known as the “Forest Village System”. The main objectives were designed to solve the rural socio-economic problems as well as the contributing to the Teak production of the country (Corvanich, 1974). The lastest strategy, which was developed in northern Thailand, is a modified version of the Forest Village System (Gajaseni, 1988). This type of forest management encourages a combination of Teak and other economic trees such as *Tamarindus indica*, *Anacardium occidentale*, *Gmelina arborea* and *Artocarpus heterophyllus* in combination with cultivated crops.

Economic species are introduced in order to increase the value of the plantation: species are chosen that do not compete with Teak, but rather occupy complimentary niches. An advantage over the Taungya system is the continual income from the cultivated crops, economic trees and Teak timber, respectively. In addition, the Forest Village Plantation System receives more return from natural land resources by species that occupy different niches. Moreover, this system provides an opportunity to study the effectiveness of different types of plantations in facilitating the establishment of a diverse community of native forest species. It has been seen that plantations can facilitate the establishment of a diverse forest community for two reasons: 1) environmental conditions under a "nurse" forest are similar to conditions under the canopy of a mature forest; 2) weeding helps eliminate early successional weeds that can out-compete, or cause a fire hazard to desirable mature forest species.

Table 2.3: Apparent domestic wood consumption in Thailand (1986 to 2001)

Year	Apparent Domestic Wood Consumption (Unit: 1,000 m^2)
1986	2334.20
1987	2762.20
1988	2990.30
1989	3373.70
1990	3783.90
1991	3454.50
1992	3888.70
1993	3179.30
1994	4065.60
1995	3418.00
1996	3150.30
1997	2338.60
1998	1186.30
1999	1542.20
2000	1182.10
2001	1843.60

Source: Royal Forest Department (2001)

If the long-term goal of forest management is to re-establish natural mature forest species, either for biodiversity reserves or for the economic purposes, it may be more economically and ecologically feasible to allow these forests to regenerate under plantations than to try to establish them in degraded areas where fire and competition from early successional species could inhibit their establishment. Therefore, these degraded forest areas are being restored by means of multi-purpose Forest Village Plantations supplemented by enrichment planting of native species. This method is quick and has a good chance of converting highly degraded land back to a natural or near-natural forest ecosystem. Moreover, it can help conserve species diversity, restore forest productivity and nutrient cycles that maintain the stability of forest ecosystems.

2.6.5 Examples of restoration projects

Many strategies have been applied in restoration processes, for example, staggered planting of primary forest species (Knowles and Parrotta, 1995), planting of catalytic monoculture that subsequently encourage natural regeneration (Parrotta, 1993, Lugo, 1997), assisted natural regeneration (Hardwick, Healey and Blakesley, 2000) and the Miyawaki method in Malaysia (Miyawaki, 1993). The staggered planting of primary tree is a technique that combines a mixture exposure-tolerant species, with subsequent enrichment with shade loving species. An alternative restoration strategy is the use of plantations as catalysts of natural regeneration in order to make commercial tree species more attractive to wildlife. Meanwhile, assisted natural regeneration (ANR) usually involves no or minimal tree planting, but instead encourages the natural processes of forest succession. Finally, the Miyawaki method involves the direct planting of up to 42 forest-climax tree species, to return the forest to its primary condition as quickly as possible. In addition to these management techniques, Dugan (2000) has suggested that in some natural forests successful acceleration includes the use of fire prevention and weed control by cutting or pressing. Likewise, Holl et al., (2000) has reported that establishment of native tree seedlings or early-successional shrubs is successful when enhancing seed dispersal and by shading out pasture-type grasses. Moreover, the planting of native tree seedlings may be useful as a strategy for accelerating forest recovery at many disturbed sites in the tropics and highlights the importance to testing any restoration strategy on a small scale at each site to evaluate appropriate species. Research on forest regeneration in tropical abandoned pastures by Aide et al., (2000) shows that enrichment planting is necessary to restore the original composition of Puerto Rican secondary forests. Although species richness recovers rapidly, species composition is quite different in comparison with old growth forest sites. Protection from fire and the allowance of natural regeneration are also used to facilitate restoration.

Goosem and Tucker (1995) have developed the "framework species method" for selecting plantation species. The criteria are based on propagation, tolerance of the harsh conditions in deforested sites, regenerative ability, growth rate, canopy density, appeal to wildlife and limited seed-dispersal mechanisms. Meanwhile, Knowles and Parrotta (1995) have only three criteria: seed germination treatment requirements, alternatives for the production of planting stock, and early growth and survival. Lamb et al., (1997) use "framework species" which are fast growing, with dense spreading canopies, and provide wildlife resources, to accelerate the return of biodiversity.

Use of Caribbean pine (*P. caribaea*) as a nurse for establishing pre-climax canopy tree species, together with pine canopy removal, results in enhanced growth and dry mass for planted trees (Aston et al., 1997). Elliot et al., (1997) use *Gluta usitala*, a common native deciduous tree species, for inclusion in tree planting programs to restore deciduous forest. Aston et al., (1997) recommends that canopy tree species of tropical forests in south-western Sri Lanka grow best when seedlings are planted within openings created by the removal of three rows of Pinus canopy. Where planting without canopy removal is required, seedlings that are shade tolerant are selected. In addition, the results from this research demonstrate that a plantation species (*P. caribaea*) can play an important role in the establishment of pre-climax tree species in Mesue-Shorea rainforest. Wanawong, Belt and McKetta, (1991) has improved productivity and financial returns to the farmer in Thailand by the use of mixed plantations of *Eucalyptus camaldulensis*, *Leucaena leucocephala* and *Acacia auriculiformis* inter-cropped with cassava.

CHAPTER THREE

METHODOLOGY

3.1 Site description

The research project was conducted at the Mae Moh forest plantation station, managed by the Forest Industry Organization (FIO) in Lampang Province, northern Thailand (18^{o} $25^{/}$ N, $99^{o}43^{/}$ E). It was 600 km from Bangkok, the capital city of Thailand (Figure 3.1). Topographically, the site was located in a mountain valley at an elevation of 300-350 m above sea level. North of the basin were quartzite mountains. To the east and west limestone and mudrock mountains bounded the site, whereas to the south basalt flows overlaid a limestone mountain (Gajaseni, 1988). The area was surrounded by natural mixed deciduous forest with Teak, an important ecosystem in the region (Figure 3.2). The distance between the edge of the plantations and the natural forest was approximately 50–100 m. Topsoil was 15 cm deep. Both topsoil and subsoil were clay loam (Srisuksai, 1990). The area was influenced by tropical monsoons. In contrast to the majority of species in the native natural mixed deciduous forest, Teak was an early successional tree, and was fire tolerant. Gaps in the forest created by occasional disturbances such as fires had permitted Teak to maintain a presence in the community of otherwise pre-climax species (Sukwong, 1994). Therefore, in this area, Teak (*Tectona grandis*) had maintained its presence as a dominant species in natural forest.

3.2 Climatic regime

Northern Thailand exhibits a strongly seasonal monsoon climate. There are 3 distinct seasons, a cool dry season (November-February), a hot dry season (March-May) and a warm wet season (May-October). The rainy season usually runs from June-October. Meteorological data gathered between 1979-2000 from Mae Moh Plantation Station (unpublished) reported that annual rainfall normally exceeded 1,234.89 mm per year and mainly fell between May and September. The average maximum rainfall was

252.79 mm in September and the average minimum was 1.48 mm in January. The mean monthly temperature was 26.66 ^{o}C, with a maximum in April and a minimum in December. (Figure 3.3) (Appendix A.2, Mae Moh Plantation Station, Unpublished).

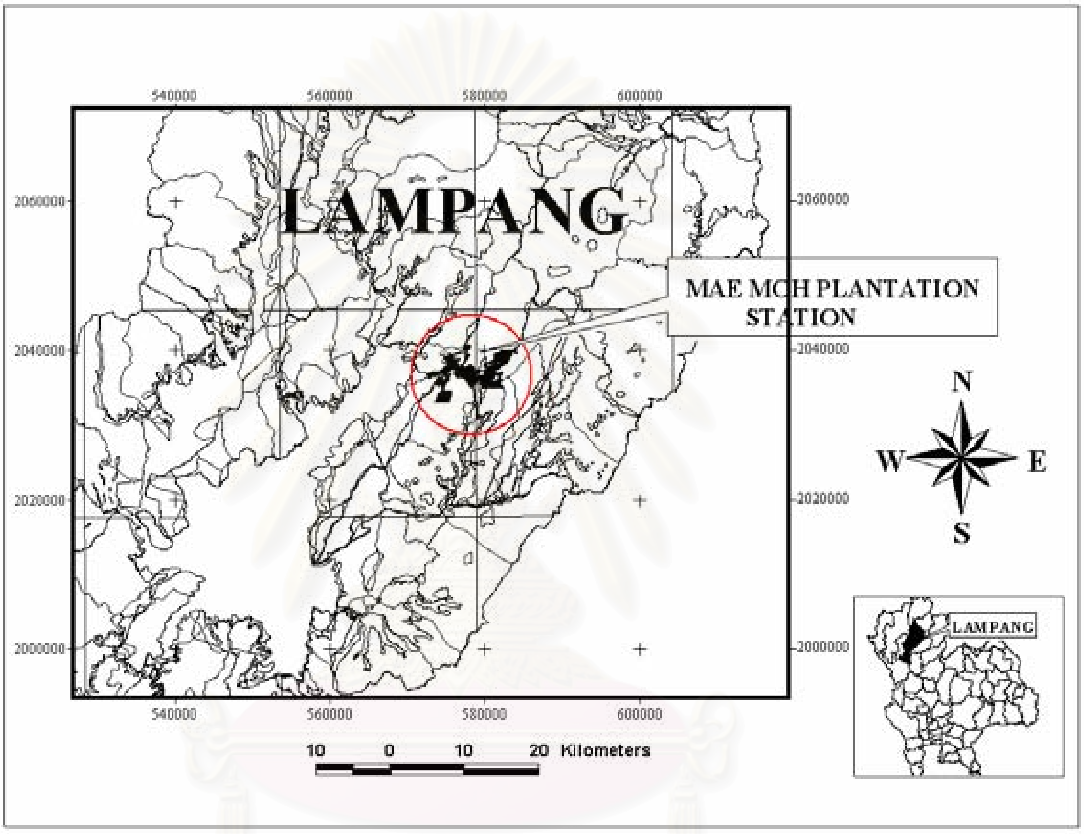

Figure 3.1: Map of Mae Moh plantation station in Lampang Province, northern Thailand

a) Rainy season

b) Dry season

Figure 3.2: Natural mixed deciduous forest ecosystem with Teak in the dry and rainy seasons, Lampang Province, northern Thailand.

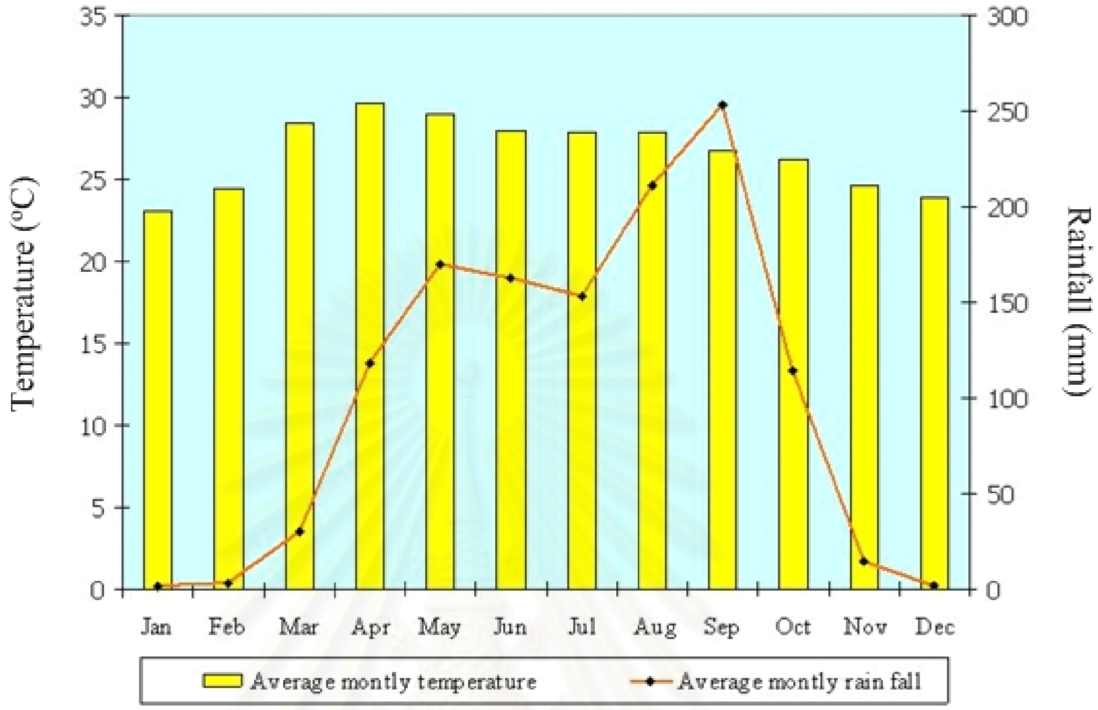

Figure 3.3: Average monthly temperature and average monthly rainfall in Mae Moh plantation station, Lampang province, Thailand from 1997-2000.

3.3 History of site study

Historically, Teak forest has been the dominant ecosystem in northern Thailand. The study site was representative of this type of ecosystem. In the past, the site had been disturbed mainly by anthropogenic influences. Selective Teak logging was perhaps the most important factor in the destruction of this natural forest ecosystem, closely followed by shifting cultivation for agricultural land. Much of the area became highly degraded. The Taungya system, using a pure Teak monoculture, was implemented in northern Thailand by the Royal Forest Department (RFD) to resolve the deforestation issue. An advantage of this method was the ease of management. But this type of plantation was found to be continuously attacked by insects and pathogens. However, due to the increased social and economic benefits to landless people, the Forest Industry Organization (FIO) developed the Forest Village System (FVS) in northern Thailand during 1967. The plantation farmer received increased income from their crops, and weed control was of positive benefit to the FIO by decreasing costs. However, the method had a few weaknesses, including the fact that the farmers were still not authorized to occupy plantation land. To reduce other problems within the monoculture plantation, a modified FVS method was initiated in this area. A multi-purpose species plantation was planted in 1988. The spacing of trees within the plantation was established at 4 m x 4 m. In each stand weed control was conducted every year and cattle were kept out, however no fire prevention methods were employed. For two years after the trees had been planted, plantation farmers were allowed to cultivate annual or biennial crops between the trees. After two years, the shade created by the spreading tree canopy became too deep and crops could not be cultivated. To date, the objective of the study has focused on converting highly degraded land back to near-natural and/or natural conditions, through rehabilitation and improvement restoration strategies. The study of understorey regeneration and restoration has been conducted at the site since 1999.

3.4 Experimental plot and background

The plantations at Mae Moh are part of what is called the “Forest Village System” (Gajaseni, 1988). The objective of the system is to reforest degraded lands, whilst at the same time giving landless locals an opportunity to cultivate their own crops. Peasants living in the government-subsidized village planted the Teak plantations, and for social and economic reasons, cultivated their own crops within them for 2 years until the canopy began to close. During the first few years, the Teak plantation benefited from the weeding carried out by the peasants as they cultivated their agricultural crops.

The experimental plantations were established in 1988, as part of a long-term ecological study (Figure 3.4) (Gajaseni, 1988). They were established primarily for the production of Teak. Other species were introduced to produce economic crops that would increase the value of the plantations. Species were chosen that would not compete with Teak, but rather occupy complimentary niches. These plantations offer an opportunity to study the effectiveness of different types of plantations in facilitating the establishment of a diverse community of native forest species. Plantations can facilitate the establishment of a diverse forest community for two reasons: 1) environmental conditions under a “nurse” forest are similar to conditions under the canopy of a mature forest; 2) the weeding conducted by local villagers helps to eliminate early successional weeds that may out-compete, or cause a fire hazard to the desirable mature forest species. If the long-term goal of forest management is to re-establish native mature forest species, either for biodiversity reserves or for the economic value of these species, it may be more economically and ecologically feasible to allow these forests to regenerate under plantations than to try to establish them in degraded areas where fire and competition from early successional species may inhibit their establishment. The experimental plots (Figure 3.4 and 3.5) were designed as different mixed-species plantations as follows:

(**A**). *Tectona grandis* and *Tamarindus indica* (TT)
(**B**). *T. grandis* and *Gmelina arborea* (TG)
(**C**). *T. grandis* (T, control)
(**D**). *T. grandis, T. indica* and *G. arborea* (TTG)
(**E**). *T. grandis, T. indica* and *Anacardium occidentale* (TTA)

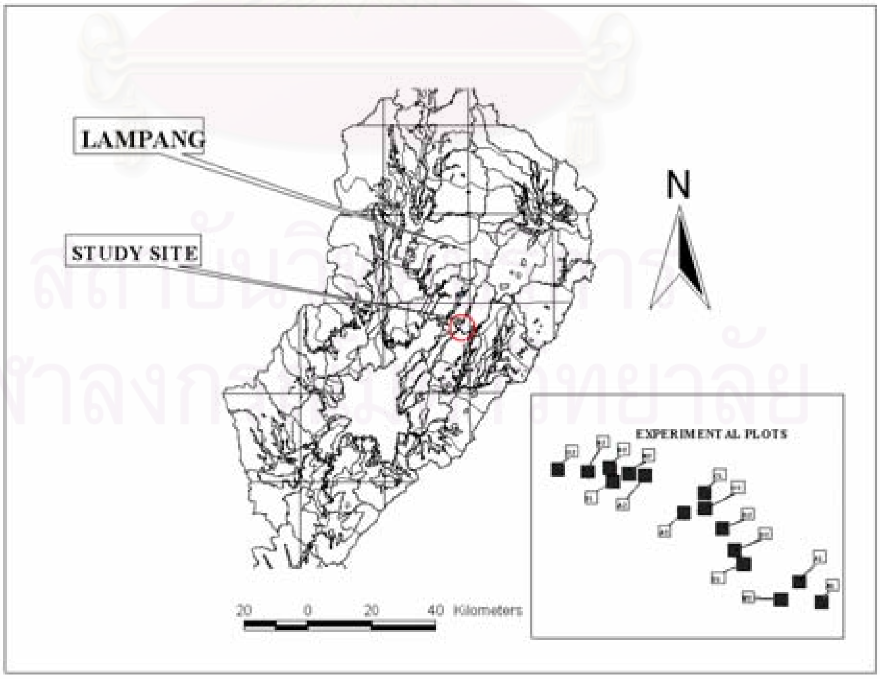

Figure 3.4: Map of experimental plots in multi-purpose species plantations, Lampang Province, Thailand.

a) TTG

b) TTA

c) TG

Figure 3.5: Five experimental plots of multi-purpose species plantations in Lampang province, northern Thailand.

Figure 3.5 (Continued)

d) TT

e) T

3.5 Primary data collection

3.5.1 Floristic inventory

3.5.1.1 Multi-purpose species plantation

The model illustrating methods for the floristic inventory is shown in Figure 3.6. Three study plots of 20 m x 20 m were set up in each of five plantation stands (T, TG, TT, TTG and TTA). Each plot was subdivided into quadrats of 5 m x 5 m. Five quadrats in each plot were randomly selected for floristic inventory. The number and size of quadrats required was calculated using a species area curve (Figure 3.7), following the procedure of Mueller-Dombois and Ellenberg, (1974). All plants were counted and identified to species level. Data was collected between September 1999 and January 2000. The following values were calculated: density (individuals per unit area), Shannon-Wiener's index of species diversity (H') (Shannon and Weaver, 1963) and evenness index (Pielou, 1969; Brower and Zar, 1977). These ecological values were used for comparison between the five multi-purpose species plantations. The Sorensen similarity index was used to calculate species similarity between natural forest and the diversified species plantations (Bray and Curtis, 1957; Magurran, 1988). All of ecological indices equation was presented in Appendix B.1. The diameter at breast height (DBH) and crown width (measured twice perpendicularly) of upperstorey trees was measured. The average basal area and average canopy cover for each stand was then calculated. Values were analysed by One-Way Analysis of Variance (ANOVA) by use of SPSS/PC version 11.5 computer software. Least-Significant Different (LSD) tests were used for comparisons among means (Hair et al., 1998).

3.5.1.2 Natural mixed deciduous forest ecosystem with Teak

Systematic sampling was used for surveying the flora in an adjacent natural mixed deciduous forest with Teak. However, surveying methods between the mixed-species plantation and the natural forest ecosystem were different, due to differences in topography. To optimise data collection, a random sampling method was used in the multi-purpose species plantation, whilst systematic sampling was used in the natural forest. Consequently, errors may arise when comparisons are made between the plantation and the natural forest. However, the focus of this study was not on quantitative ecological parameters (density, basal area, diversity index and so on) but on similarity of species. Data was collected between October and December 2000. Sampling lines ran from west to east, in parallel with all experimental plots. Each sampling point was at 20 m intervals and plot size was established at 10 m x10 m for woody trees (DBH ≥ 4.5 cm) and 5 m x 5 m for saplings (DBH < 4.5cm, height > 1.30 m). Plants species were identified, with both vernacular and scientific names, in accordance to Smitinand (2001). Density was then calculated.

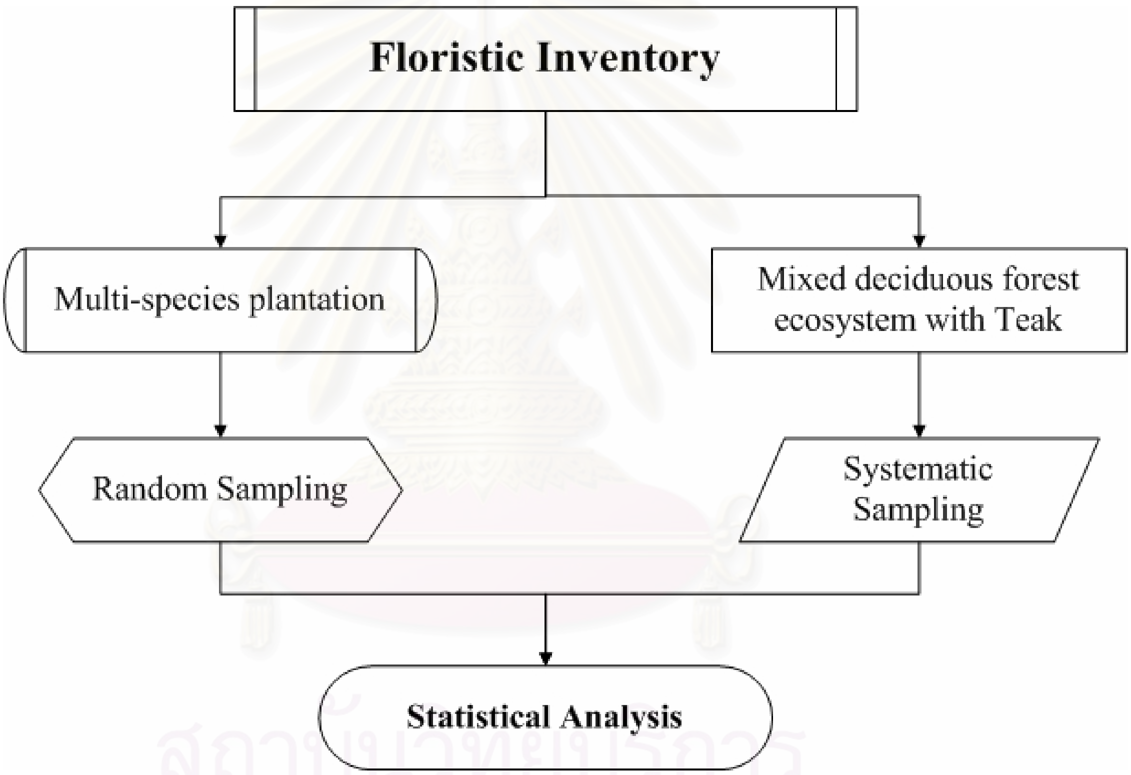

Figure 3.6: Model of floristic inventory in mixed deciduous forest with Teak and multi-purpose species plantations

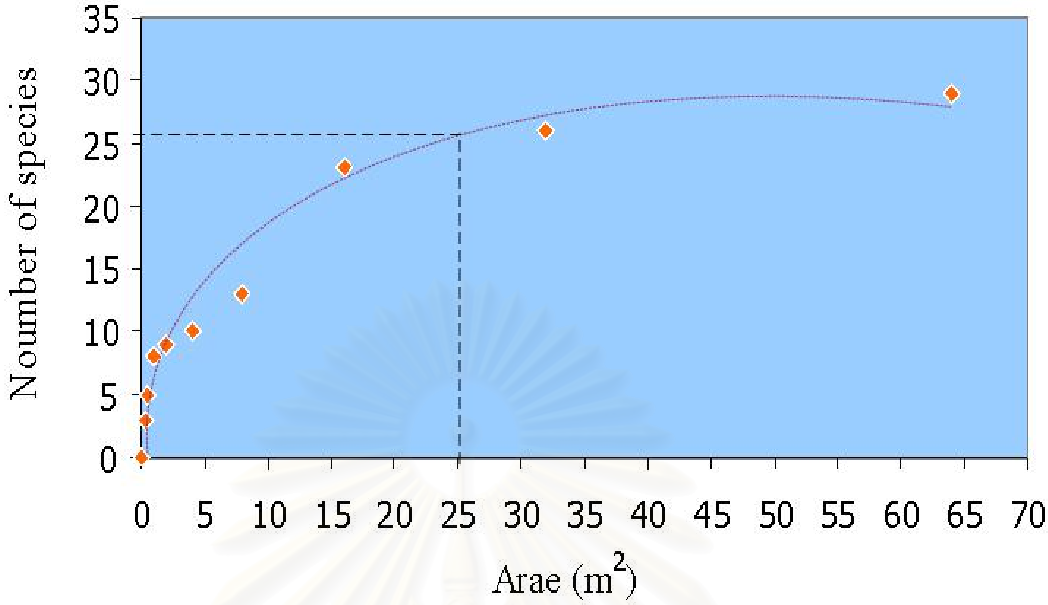

Figure 3.7 Species area curve in multi-purpose species plantation

3.5.2. Seeds study

3.5.2.1 Seed dispersal

A stratified random sapling method was applied for the collection and study of woody seed dispersal. Study areas were divided into three zones; multi-purpose species plantation (MP), ecotone (ECO) and mixed deciduous forest ecosystem with Teak (MDF). This method of dividing 3 contiguous areas for study could therefore show patterns of seed distribution from natural forest, through the ecotone, to mixed plantations. Seed traps were circular and 0.80 meters in diameter, giving each a surface area of 0.5 m^2. They were constructed of 2 mm nylon netting secured to a steel-wire frame, with three 1-meter steel legs supporting each trap. Seed traps were run west to east at 20 m intervals. A direct counting technique was used to monitor abundance of mature seed species. Those seeds that were mature, ripe and undamaged were counted; viability was tested using germination tests (Sutherland, 1996). Seed traps were set up in November 2001. Woody seeds were collected at the beginning of each month over a 1-year period from November 2001 to October 2002.

3.5.2.2 Soil seed bank

The study of woody seed bank dynamics was monitored in MDF, ECO and MP at the same study site, by a stratified random sampling method. Soil samples (0.04 m^2 to a depth of 5 cm) were collected at 20 m intervals. A report by Young, Ewel and Brown (1987) concluded that the majority of viable seeds in secondary forest were found within the top soil (0-10 cm). Seed samples were cleaned by a sieving technique (Robert, 1981, Ter Heerdt et al., 1996). Mature, ripe and undamaged tree seeds were counted by direct counting techniques; viability was tested using germination tests (Sutherland, 1996). Soil seed bank data was collected at the same time as seed dispersal, at the beginning of every month for a 1-year period from November 2001 to October 2002. Figure 3.8 models the processes followed for the study of seed distribution in natural forest areas and multi-purpose species plantations. Correspondence Analysis (CA) was conducted to analyse the statistical relationships between the spatial distribution of seeds in the multi-purpose species plantation, the ecotone and the mature mixed deciduous Teak forest ecosystem (Hair et al., 1998). Model for the monitoring of seed distribution is shown in Figure 3.8.

3.5.3 Restoration study

3.5.3.1 Development of planting design

A model for restoration strategy is presented in Figure 3.9. Selective tree species for enrichment in mixed-species plantations were chosen based on four criteria: (i) canopy species (ii) economically valuable indigenous timber species (iii) indigenous species and/or (iv) nitrogen-fixing species. This research selected 11 enrichment species based on the above four criteria. The number of seedlings per species for planting in the experimental plot is shown in Table 3.1.

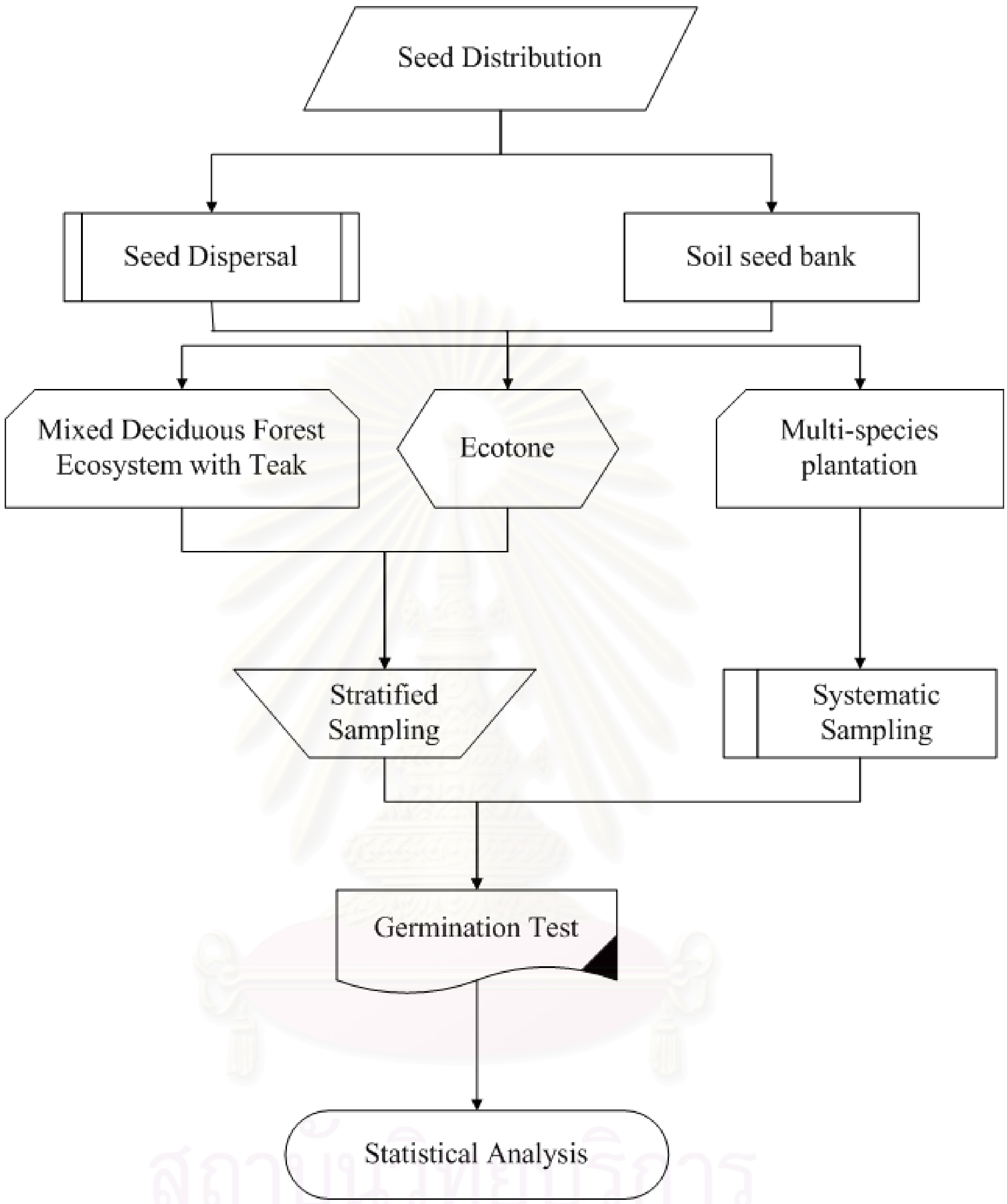

Figure 3.8: Model for the monitoring of seed distribution

The proportion of enrichment species was calculated from the density of primary tree species encountered in natural mixed deciduous forest ecosystem with Teak. The number of enrichment seedling was calculated from the mean of the remaining gap in multi-purpose species plantations divided by the spacing for planting (1.6 m x 1.6 m). A total of 73 seedlings were randomly planted in the mixed plantation during the rainy season in June 2001.

The space between seedlings was approximately 1.6 m x 1.6 m (Forest Restoration Research Unit, 2000). Management practices included fertilization (once after initial planting), controlled weeding and forest fire protection. The first period of weeding control and fire protection was conducted during the dry season between late December 2001 and early January 2002. The second period for weeding control was during the end of the rainy season (September 2002). The model for plantation design is shown in Figure 3.10.

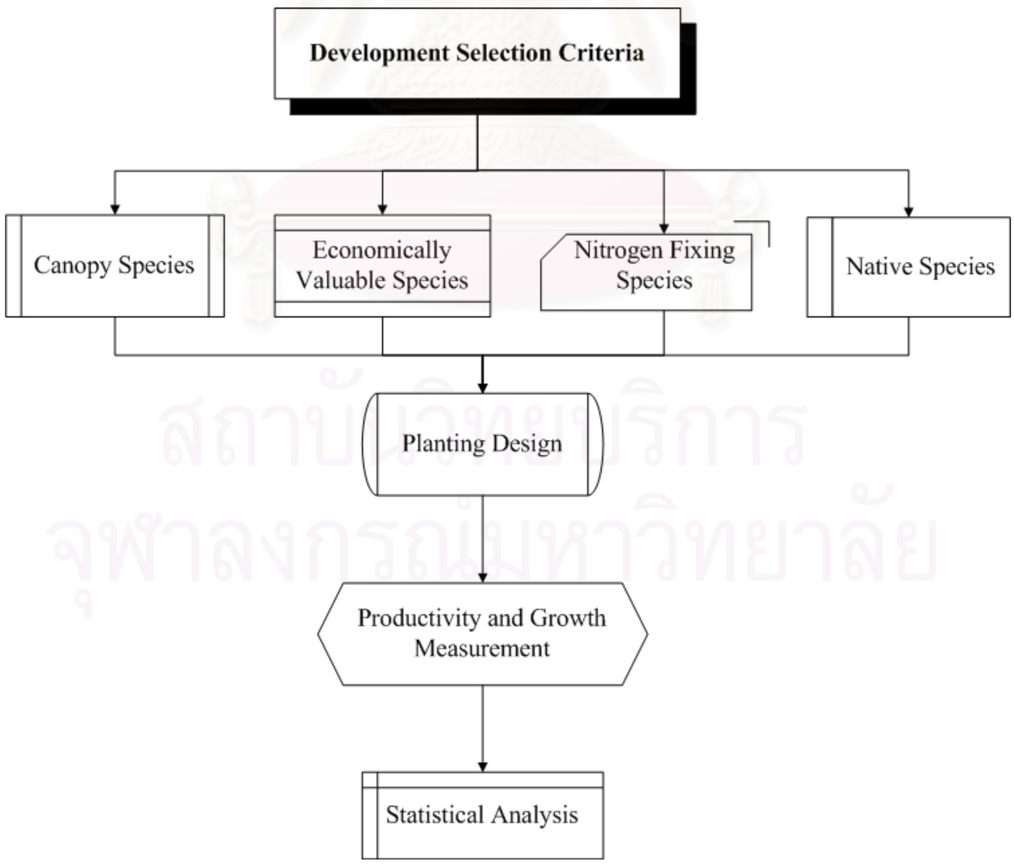

Figure 3.9: Model of restoration strategy followed in the research

Table 3.1: The ratio of enrichment seedlings for 11 species planted in the multi-purpose species plantation.

Species	**Family**	**Percentage density (%)**	**Individuals/Plot (20 m x 20 m)**
Pterocarpus macrocarpus	Leguminosae	45	30
Xylia xylocarpa var. *kerrii*	Leguminosae	18	13
Largeratroemia floribunda	Lythraceae	18	13
Schleidhera oleosa	Sapindaceae	5	3
Chukrasia velutina	Meliaceae	2	2
Dalbergia oliveri	Leguminosae	2	2
Spondia pinnata	Anacardiaceae	2	2
Afzelia xylocarpa	Leguminosae	2	2
Diospiros mollis	Ebenaceae	2	2
Toona cilliata	Meliaceae	2	2
Millettia leucantha	Leguninosae	2	2
Total		**100**	**73**

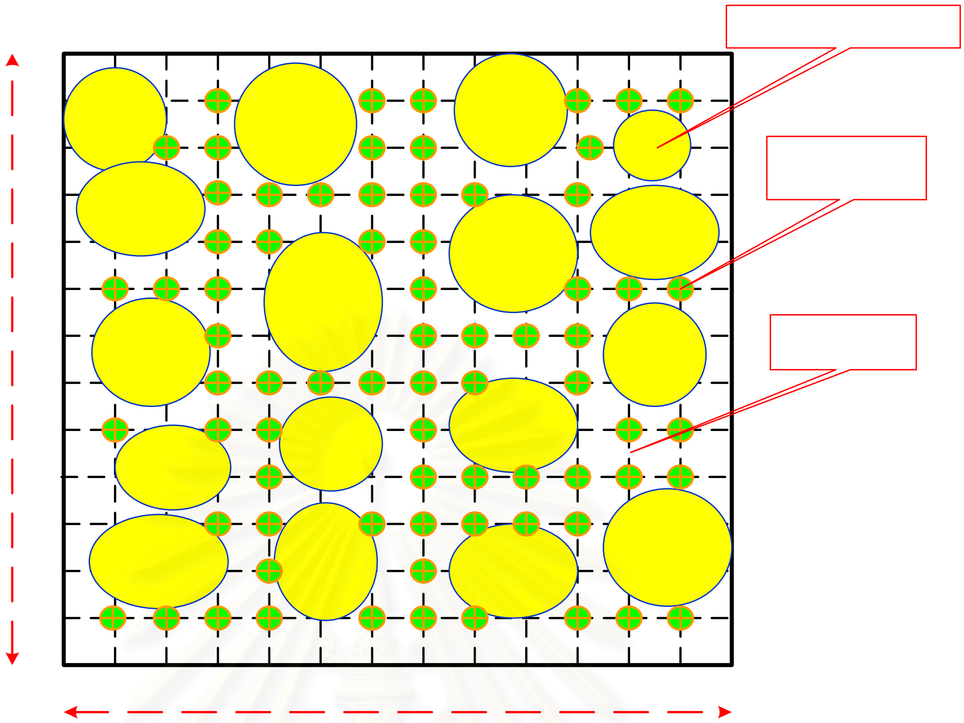

Figure 3.10: Model of plantation design for 11 selective enrichment seedling in five multi-purpose species plantations.

3.5.3.2 Survival and productivity measurements

Diameter at ground level (D_0) and the height of all seedlings was recorded one year after planting (May, 2002). Increments in diameter and height were calculated. The total biomass of enrichment species was calculated by use of an allometric method (Kira and Shidei, 1967, Ogawa et al., 1967). The component biomass (leaves, branches, stem and roots) of each individual seedling per species was calculated using allometric equations being developed exclusively for this research. The total biomass of seedlings in each category was calculated to total biomass per hectare. Further details of the procedures and methods used for estimation seedling biomass and development of allometric equations are shown in Appendix B.2. Increments for absolute growth rate of stand biomass above ground (AGR) and relative growth rate of stand biomass above ground (RGR) were calculated (Appendix B.3). One-Way Analysis of Variance (ANOVA) was used to examine statistical differences of all ecological parameters between the five different multi-purpose species plantations.

3.6 Secondary data collection

3.5.1 Available nutrients in mixed-species plantations was gathered from Rattanasinganlachan (1996).

3.5.2 Meteorological data was obtained from Mae Moh plantation station, Lampang Province.

CHAPTER FOUR

RESULTS AND DISCUSSIONS

4.1 COMMUNITY ANALYSIS IN NATURAL FOREST AND MULTI-PURPOSE SPECIES PLANTATIONS

4.1.1 Community analysis of mixed deciduous forest with Teak

Results

Investigations studied the stand characteristics of trees and saplings in natural mixed deciduous forest with Teak, the most extensively distributed forest type in northern Thailand. Species composition, and the density of primary trees and saplings are presented in Table 4.1. Within the stand, 54 known species and 5 unidentified species were found in the tree category and 49 known species and 10 unidentified species were found in the sapling category (Appendix A.3). Teak was not the dominant species in this stand. Results showed that *Pterocarpus macrocarpus* had the highest density followed by *Tectona grandis*. The prior removal of large trees by selective logging was the reason for the current low abundance of Teak (Gajaseni and Jordan, 1990). Investigation of DBH distribution (Figure 4.1) showed that few large trees were found at the site but that the overall density of trees was higher than quoted values (Table 4.2).

Other common climax tree species were found, such as *Shorea siamensis*, *Grewia elatostemoides*, *Xylia xylocarpa* var. *kerrii*, *Lannea coromandelica*, *Lagerstroemia duperreana* and *Nephelium hypoleucum*. The dominant tree species composition in this forest type was similar to previous studies (Bunyavejchevin, 1983, Gajaseni and Jordan, 1990). In the sapling group the dominant species was *Phyllanthus cf. orientalis*, with *Dalbergia glomeriflora* and *Croton oblongifolius* as the secondary dominant species. Other common sapling species in the stand were *Albizia lebbeck*, *Wrightia tomentosa*, *Croton longissimus*, *Pterocarpus macrocarpus*, *Antidesma ghaesembilla* and *Largerstroemia dupperreana*. These results suggested that the characteristics of this forest stand, in term of species composition, could be described as mixed deciduous forest with Teak, conforming to Bunjavejchewin (1983) and Gajaseni and Jordan (1990), though overall tree density was high when compared to quoted values.

Table 4.1: Species composition and average density of some predominant trees and saplings in mixed deciduous forest with Teak in Lampang Province, northern Thailand.

	Tree			Sapling	
No	Scientific name	Average density (Individuals/ha)	No	Scientific name	Average density (Individuals/ha)
1	*Pterocarpus macrocarpus*	172	1	*Phyllanthus orientalis*	178
2	*Tectona grandis*	94	2	*Dalbergia glomeriflora*	167
3	*Shorea siamensis*	89	3	*Croton roxburghii*	150
4	*Grewia elatostemoides*	78	4	*Albizia lebbeck*	144
5	*Xylia xylocarpa* var. *kerrii*	67	5	*Wrightia tomentosa*	122
6	*Lannea coromandelica*	67	6	*Croton longissimus*	122
7	*Lagerstroemia duperreana*	64	7	*Pterocarpus macrocarpus*	87
8	*Nephelium hypoleucum*	56	8	*Antidesma ghaesembilla*	78
9	*Sterculia villosa*	42	9	Unidentified sp. 2	78
10	*Croton oblongifolius*	39	10	*Largerstroemia duperreana*	61
11	*Wrightia tomentosa*	36	11	*Dalbergia sp.*	50
12	*Dipterocarpus obtusifolius*	33	12	*Albizia odoratissima*	50
13	*Antidesma ghaesembilla*	31	13	*Canarium subulatum*	50
14	*Terminalia mucronata*	28	14	*Grewia elatostemoides*	50
15	*Canarium subulatum*	28	15	*Tectona grandis*	50
16	Unidentified sp. 2	22	16	*Vitex peduncularis*	45
17	*Vitex peduncularis*	22	17	*Millettia brandisiana*	45
18	*Schleichera oleosa*	19	18	*Sterculia villosa*	44
19	*Phyllanthus orientalis*	19	19	*Xylia xylocarpa* var. *kerrii*	39
	Other species (40 spp.)	256		Other species (39 spp.)	215
	Total	**1,261**		**Total**	**2,028**

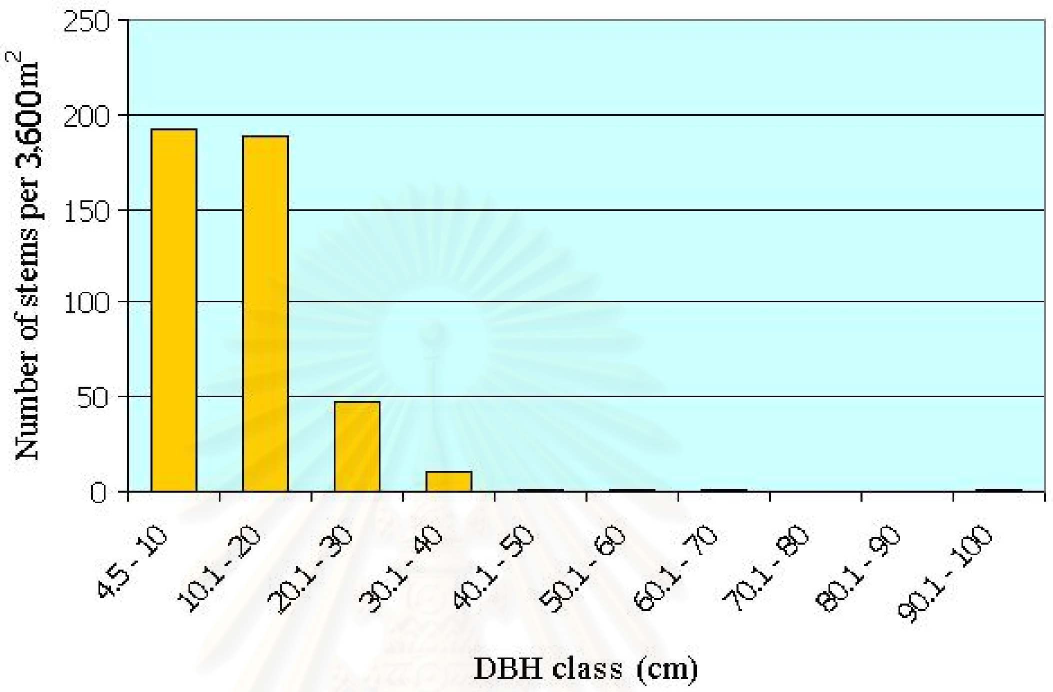

Figure 4.1: DBH distribution of trees in natural mixed deciduous forest with Teak in Lampang Province, northern Thailand.

Table 4.2: Tree density in mixed deciduous forest with Teak in Thailand (DBH > 4.5 cm).

Forest ecosystem	Density (No. trees/ha)	Author
Mixed deciduous forest with Teak	262-395	Bunyavejchewin (1983)
	433	Gajaseni and Jordan (1990)
	1,261	Current study

Discussion

In natural mixed deciduous forest with Teak (MDF) 59 tree species were recorded. When considering species density, it was found that *Pterocarpus macrocarpus* had the highest value (172 trees per ha) and was clearly the dominant species, followed by *Tectona grandis* (94 trees per ha). However, a previous inventory by Gajaseni (1988) showed that Teak was then the dominant species (303 trees per ha). This suggests that in this particular forest, *T. grandis* may have been partially disturbed by external disturbances. Observations by Gajaseni and Jordan (1990) showed that previous selective logging in the region had caused the absence of Teak and other large deciduous trees. Since Teak had been disturbed by selective logging, then *P. macrocarpus* had a greater competitive potential to occupy niches and dominate the upperstorey, and thus showed the highest density. In Gajaseni's (1988) study *P. macrocarpus* was not the dominant species. If Teak was present as a dominant species, then *P. macrocarpus* was absent or showed a low density in the natural forest. When Teak was eliminated, *P. macrocarpus* occupied the niche and increased in abundance and density. Therefore, the structure and composition of the forest stand had changed. The competition exclusion principle supports this observation.

If considering species composition and structure, then the stand could still be described as mixed deciduous forest ecosystem with Teak or Teak forest, when compared to previous observations (Bunyavejchewin, 1983, Gajaseni and Jordan, 1990). However, some characteristics such as dominant species and stem density differed from quoted values due to the effects of selective logging. The stand also exhibited a small area of dry dipterocarp forest on the ridge-top, with species including *Shorea siamensis* in the upperstorey, and *Phyllanthus orientalis*, *Dalbergia glomeriflora*, *Croton oblongifolius*, *Albizia lebbeck* and others in association (Table 4.1). Bunyavejchewin, (1983) recognized 2 association subtypes (*Tectona grandis-Xylia kerrii* and *Tectona grandis-Xylia kerii-Terminalia mucronata*) in mixed deciduous forest ecosystems in Thailand. Species composition observed in this study could not be assigned to subtype according to Bunyavejchewin's classification. However, when considering species composition of upperstorey trees, the results showed the continuous layer of canopy trees was rich in species, which does coincide with observations by Ogawa, Yoda and Kira (1961).

The variety in species composition exhibited by Teak forest ecosystems is regulated by environmental conditions. Many factors, including topography, soil properties and elevation, determine the distribution and establishment of important species in the mixed deciduous forest, as reported by Bunyavejchewin (1985). For example, *Tectona grandis* reaches its ecological optimum with high levels of calcium and phosphorus. In contrast, *Largerstroemia calyculata* achieves its optimum with low levels of calcium, phosphorus and potassium content. Thus, variable species association in Teak forest is common. Obviously, species composition in mixed deciduous forest with Teak is controlled by several complex environmental factors. And it is these environmental factors that control the variation in species seen in this study when compared to other studies at different sites and under different physiological conditions.

Generally, in mature natural forest ecosystems with little or no disturbance, class distribution of size is best described by the negative exponential curve or reverse J-shaped distribution, which implies constant rates of mortality from one size-class to the next (Muller, 1982). In Thailand, undisturbed natural mixed forest ecosystems with or without Teak, commonly shows reversed J-shaped distribution for diameter (Bunjavejchewin, 1983; Gajaseni, 1988; Marod et al., 1999). The results of this study coincide with these observations, in that the diameter of trees in the natural mature stand closely followed a negative exponential decrease from the lowest size class (4.5–10.0 cm) to the largest size class (around 91.0-100.0 cm) (Figure 4.1), suggesting that the reverse J-shaped distribution was valid for this community. However, results also revealed that some size-classes were absent. The number of trees with a DBH of 40 cm or more was lower than that found in previous surveys (Bunyavejchewin, 1983; Gajaseni, 1990) and may have resulted from intensive selective logging of Teak about 35 years ago. This has affected the structure and species composition of the natural forest stand. DBH distribution in natural MDF showed that most of the trees in this stand were small with a diameter of less than 20 cm.

The estimated stem density was 1,261 individuals/ha. In comparison with other studies this figure was high. Bunyavejchewin (1983) and Gajaseni and Jordan (1990) quote values well below this. This indicated that the community was not a mature forest ecosystem. Based on the dynamics and stability found within communities, as presented by the succession theory (Clements, 1916), this forest stand should follow the process of succession and develop towards a climax state if further serious disturbances are reduced. Important climax species for this forest type were found in the sapling category, such as *Pterocarpus macrocarpus*, *Largerstroemia duperreana*, *Canarium sabulatum*, *Tectona garndis* and *Xylia xylocarpa* var. *Kerrii*. Therefore, these species should mature and eventually become dominant in the upper-storey layer. The dominant species of saplings found in the natural forest included; *Phyllanthus orientalis*, *Dalbergia glomeriflora*, *Croton longissimus*, *Croton roxburghii*, *Albizia lebbeck*, and *Wringtia tomentosa*. Most showed characteristics of fast-growing species. Natural recovery processes in disturbed forests typically occurs in four sequential stages; stand initiation and regeneration, thinning or stem exclusion, transition or understorey regeneration and lastly steady-state or old-growth; (Kozlowski, 1971; Oliver and Larson, 1996). Each of these stages exhibits unique characteristics. If considering species composition and structure, the stand currently shows characteristics of a pre-climax community and is in a state of transition, with obvious regeneration of the understorey. Fast-growing species are establishing and occupying niches better than slower ones. As the community develops the slower-growing species will eventually occupy more space and become dominant in the canopy.

4.1.2 Community analysis of understorey plants in multi-species plantations

Results

4.1.2.1 Density and species composition

Average density and species composition of both woody and non-woody species in the understorey of five multi-purpose species plantations are shown in Tables 4.3 and 4.4. Of the non-woody species (herbs, climbers and bamboo), *Imperata cylindrica* had the highest density and was clearly the dominant species, followed by *Chromolaena odorata* (Table 4.4). These two species were more than twice as abundant in the single-species plantation as in the two-species plantations, and more than three times as abundant than in the three-species plantations. Of the woody species, *Croton longissimus* and *Cratoxylum formosum* had the highest density, followed by *Dalbergia cana* and *Antidesma ghaesembilla*. Some dominant trees of primary forest were found, such as *Tectona grandis*, *Pterocarpus macrocarpus, Mitragyna brunonis, Lagerstroemia duperreana*, *Albizia lebbeck, Lannea coromandelica* and *Xylia xylocarpa* var. *kerrii*.

Within the multi-purpose species plantations, most of the dominant species, by basal area, were intermediate successional species and pre-climax species. Maximum basal area for woody species (Table 4.5) was 1,357.03 m^2/plot in TT, followed by TTG, T and TG. The minimum was 390.28 m^2/plot in TTA. In the TT plot, the dominant species was *Cratoxylum formosum*, followed by *Dalbergia cana* and *Dalbergia cutrata*. The TTA plot was dominated by *Dalbergia cutrata*. The TTG combination was dominated by *Morinda coreia*, whilst *Pterocarpus macrocarpus* dominated the T plot. In the TG plantation, the dominant species was *Aporusa villosa*, followed by *Dalbergia cutrata* and *Millettia sp*. The dominant species in TTA was *Dalbergia cutrata,* followed by *Dalbergia cana*, *Millettia sp*. and *Harrisonia perforate* respectively. Hence, in multi-purpose plantations some species, such as *Dalbergia cana*, *Dalbergia cutrata* and *Millettia sp*., occupied niche space better than others. Of the primary tree species found in natural mixed deciduous forest with Teak, *Pterocarpus macrocarpus* had the highest average basal area (T).

Average density and average basal area (Tables 4.3 and 4.5) indicated that four species, *Cratoxylum formusum, Croton longissimus*, *Dalbergia cana* and *Antidesma ghasembilla*, occupied niches better than others. All of them showed characteristics of pioneers, in that they were short-lived, fast-growing species that required disturbed sites in which to grow and reproduce. Only *Pterocarpus macrocarpus*, a climax species found in natural mixed deciduous forest, had a maximum value of average basal area in the multi-purpose species plantations. However, other climax species such as *Irvingia malayana, Dalbergia cutrata, Siphonodon celastrineus, Artocarpus lacucha, Largerstroemia duperreana* and *Vitex peduncularis* managed to establish and persist spatially in the plantations. A total of 101 species were found in the five multi-species plantations. Of these, 59 were trees, 14 were shrubs, 15 were herbs, 11 were climbers, two were bamboo and nine were unidentified (Appendix A.4). A total of 36 families were represented. Leguminosae was the most common with 18 species, followed by Euphobiaceae with 8 species, Combratceae with 6 species and Labiatae with 5 species (Appendix A.4).

Table 4.3: Species composition and average density of some woody species in multi-purpose species plantations in Lampang Province, northern Thailand

Species	**Average density of woody species (Individuals/125 m^2)**				
	T	**TT**	**TG**	**TTG**	**TTA**
Cratoxylum formosum	78	144	100	60	58
Croton longissimus	52	159	162	117	143
Dalbergia cana	28	83	49	40	93
Antidesma ghaesembilla	22	29	58	23	21
Tectona grandis	7	26	7	4	14
Dalbergia sp.	13	30	35	11	43
Pterocarpus macrocarpus	14	11	3	15	18
Dalbergia glomeriflora	19	11	98	29	6
Mitragyna brunonis	4	3	1	3	11
Barringtonia acutangula	2	4	4	11	4
Wrightia pubescens	10	10	2	13	3
Lagerstroemia duperreana	4	3	6	1	7
Antidesma bunius	4	5	7	3	5
Albizia lebbeck	7	9	0	8	1
Grewia elatostemoides	1	3	2	1	3
Lannea coromandelica	1	4	2	3	2
Millettia brandisiana	0	4	0	0	0
Berrya ammonilla	1	3	9	4	4
Harrisonia perforata	1	1	24	16	112
Xylia xylocarpa var *kerii*	1	2	1	6	0
Colona siamica	1	9	2	15	31
Phyllanthus cf. orientalis	18	9	11	17	4
Holarrhena antidysenterica	2	9	19	4	0
Albizia odoratissima	1	3	1	5	5
Aporusa sp.	8	6	30	3	3
Other species	33	19	64	27	69
Total	**328**	**602**	**696**	**440**	**656**

Table 4.4: Species composition and average density of some non-woody species in multi-purpose species plantations in Lampang Province, northern Thailand

Species	Average density of non-woody species (Individuals/125m^2)				
	T	TT	TG	TTG	TTA
Imperata cylindica	1,495	559	914	498	319
Chromolaena odorata	19	16	16	18	13
Millettia sp.	32	12	29	19	17
Commelina sp.	10	23	3	34	28
Other species	58	50	48	60	68
Total	**1,614**	**660**	**1,010**	**630**	**444**

Table 4.5: Average basal area of woody species in five multi-purpose species plantations in Lampang province, northern Thailand

No	Species	Average basal area (m^2/400 m^2)				
		T	TT	TG	TTG	TTA
1	*Pterocarpus macrocarpus*	111.45	28.99	2.88	14.80	5.98
2	*Aporusa villosa*	69.98	40.40	70.13	38.67	0.71
3	*Dalbergia cana*	68.68	100.09	28.73	72.71	61.65
4	*Barringtonia acutangula*	51.75	18.04	20.99	39.06	4.05
5	*Antidesma ghaesembilla*	45.87	20.15	24.59	36.53	13.17
6	*Croton longissimus*	43.57	51.93	34.30	45.01	17.71
7	*Irvingia malayana*	37.94	0.00	12.14	0.00	0.00
8	*Morinda coreia*	37.18	63.65	16.67	238.00	28.94
9	*Cratoxylum formosum*	35.74	677.49	18.36	20.28	10.00
10	*Wrightia pubescens*	35.54	17.52	1.36	33.17	0.60
11	*Phyllanthus cf. orientalis*	23.38	5.28	3.90	12.44	2.41
12	*Dalbergia cutrata*	20.22	99.89	66.71	19.09	87.71
13	*Markhamia stipulata*	15.99	2.40	12.03	32.54	1.27
14	*Siphonodon celastrineus*	12.93	0.00	11.31	0.21	0.69
15	*Getonia floribunda*	12.87	0.95	2.34	0.32	0.05
16	*Holarrhena antidysenterica*	12.38	2.20	32.24	9.62	0.00
17	*Mitragyna rotundifolia*	10.85	10.18	0.85	13.73	22.63
18	*Artocarpus lacucha*	8.23	1.20	2.39	8.62	5.06
19	*Largerstroemia duperreana*	8.05	0.71	5.18	0.77	4.22
20	*Microcos paniculata*	6.13	0.00	0.00	0.00	4.79
21	*Vitex peduncularis*	5.96	0.26	2.65	4.84	2.27
22	*Millettia sp.*	4.95	26.38	57.89	40.30	36.63
23	*Antidesma bunius* var. *bunius*	4.53	5.74	4.37	0.98	6.13
24	*Dalbergia oliveri*	4.23	0.32	0.00	3.09	0.88
25	*Dalbergia glomeriflora*	3.85	0.69	1.69	2.57	0.92

Table 4.5 (Continuted)

26	*Randia longispina*	3.46	0.00	0.00	0.13	0.30
27	*Terminalia triptera*	1.98	0.00	0.00	0.00	0.00
28	*Vitex canescens*	1.91	0.00	0.00	0.00	3.64
29	*Diospyros ehretioides*	1.97	3.20	1.64	2.13	0.00
30	*Litsea glutinosa*	1.71	0.77	0.00	0.26	0.00
31	*Croton roxburghii*	1.51	0.00	0.00	1.64	0.00
32	*Bombax anceps* var. *cambodiense*	1.48	0.04	0.15	0.97	0.66
33	*Lannea coromandelica*	1.16	0.86	0.19	0.64	0.33
34	*Berrya cordifolia*	0.85	0.80	3.67	3.11	1.32
35	*Tectona grandis*	0.78	2.57	0.51	2.60	0.76
36	*Albizia lebbeck*	0.75	0.47	0.00	0.13	0.00
37	*Phyllanthus emblica*	0.71	0.98	0.76	0.00	0.21
38	*Xylia xylocarpa*	0.68	4.17	0.00	8.09	0.00
39	*Pterospermum semisagittatum*	0.44	0.21	0.00	0.00	0.21
40	*Ficus hispida*	0.43	0.55	0.11	0.47	0.26
41	*Albizia odoratissima*	0.33	0.33	0.08	1.24	5.88
42	*Hymenodictyon orixense*	0.24	0.09	0.00	1.30	0.74
43	*Sterculia guttata*	0.24	0.87	0.00	3.46	5.93
44	*Grewia eriocarpa*	0.07	1.17	0.00	0.21	0.90
45	*Berrya mollis*	0.05	0.00	0.03	0.00	0.09
46	*Chukrasia tabularis*	0.01	0.00	0.13	0.05	0.09
47	*Harrisonia perforata*	0.01	0.02	1.33	5.10	30.38
48	*Cassia fistula*	0.01	0.00	0.34	0.30	0.00
49	*Millettia leucantha* var. *buteoides*	0.00	0.02	0.00	0.00	5.15
50	*Gmelina arborea*	0.00	0.00	0.14	0.00	0.00
51	*Canarium subulatum*	0.00	1.05	0.00	0.17	0.00
52	*Millettia brandisiana*	0.00	164.34	0.00	0.00	6.40
53	*Terminalia alata*	0.00	0.00	0.39	0.00	0.00
54	*Garuga pinnata*	0.00	0.07	0.13	0.00	0.15
55	*Memecylon scutellatum*	0.00	0.00	2.37	0.00	0.00
	Other species	0.00	0.00	2.61	36.21	8.39
	Total	**713.02**	**1,357.03**	**448.28**	**755.55**	**390.28**

The highest number of species was found in the three-species plantations (Table 4.6). The total number of species was higher in the three-species plantations (TGA = 75 species) than in the two-species (TT = 60 species) and the single-species plantation (T = 61 species). Of the three-species plantations, TTA had the highest number of understorey woody species (55 species), followed by TG of the two-species plantation (Table 4.6). The number of primary trees and primary saplings was high in the three-species plantation (TTG), but there were no clear trends showing a decrease in these numbers in relationship to the number of planted species in the other plantations (Table 4.6).

Table 4.6: Total number of species found in natural forest and in five multi-purpose species plantations.

Ecological values	Multi-purpose species plantation				
	T	TT	TG	TTG	TTA
Total number of species	61	60	68	64	75
Number of woody species	47	41	51	46	55
Number of non woody species	14	19	17	18	20
Number of primary tree	33	30	33	37	33
Number of primary sapling	33	35	31	38	34

Note: A double line indicates a maximum value; a single line indicates a minimum value.

The three-species plantation had the highest number of woody species (TTA = 55 species) regenerating in the understorey (Table 4.6). Likewise, when the species considered were limited to those occurring in the primary forest, the three-species plantations still had the highest number of primary trees (TTG = 37 species) and saplings (TTG = 38 species) (Table 4.6). There were statistically significant differences in the density of woody species among the five multi-purpose species plantations ($F=4.81$, $df=4$, $p<0.05$), with TTA, TG and TT showing significantly higher density than the single-species plantation (T). The TTG plantation had a mean density of woody species higher than the T plot, though the differences were not significant at the 0.05 level.

When considering non-woody characteristics, the differences between mean densities of non-woody species were statistically significant for the five plantation types ($F=4.24$, $df=4$, $p<0.05$). The results showed that the single-species plantation had the highest density (1,614 Individuals/125 m^2) and also had the highest density ratio of woody to non-woody species (1:4.91). The density of non-woody species gradually decreased from the single-species plantation to the three-species plantations respectively (Table 4.7).

For total density (woody and non-woody species), the differences were not statistically significant over the five multi-purpose species plantations. However total density in the three-species plantations (TTG and TTA) was significantly lower than in the single species plantation (Table 4.7).

4.1.2.2 Ecological indices

Both Shannon-Wiener’s diversity index and the evenness index were highest in the three-species plantation (TTA) whether including ($H^{/}$ = 2.83 and Evenness = 1.63) or excluding ($H^{/}$ = 3.44 and Evenness = 1.83) non-woody species. When comparing all floristic components (woody and non-woody species), the trend in ecological values clearly decreased from three-species to single-species plantations (Table 4.7). However, when considering only woody species, though both diversity and evenness indices were still highest in the three-species plantations, the trends in ecological values were not related to the number of planted species in other plantations. The lowest indices (1.35) were found in the two-species plantation (TT).

4.1.3 Floristic similarities of natural forest and multi-purpose species plantations

Results

Observations showed that climax trees and saplings of natural mixed deciduous forest with Teak also occurred in mixed and single-species plantations. These results were clearly confirmed by similarity indices (Table 4.8). For primary tree species found in mixed plantations the indices ranged from 0.63-0.70. The highest similarity index was found in the TTG plantation. These high values indicated that species found in the two areas were indeed similar.

Likewise, a comparison between primary saplings and woody species in the five plantations resulted in high value indices (0.58-0.65). The three-species plantation (TTA) and the two-species plantation (TT) showed the highest values (0.65). Similarity indices showed high values within the multi-species plantation and lower values when compared to the primary species. However, there was no difference in woody species composition between the five plantation types and primary species as indicated by their equally high similarity indices in all plots.

Table 4.7: Ecological values in the five multi-purpose plantations in Lampang province, northern Thailand.

Ecological values	Mixed species plantation				
	T	TT	TG	TTG	TTA
Average density of woody species (Individuals/125 m^2; ±SD; n=3)	328 ± 25.03^{b}	601 ± 182.96^{ac}	696 ±94.04^{a}	440 ± 18.56^{bce}	659 ± 57.01^{ae}
Average density of non-woody species (Individuals/125 m^2; ±SD; n=3)	1,614 ± 343.80^{b}	660± 443.08^{a}	1,010±276.62^{ab}	630 ± 510.59^{a}	442 ± 325.58^{a}
Average density ratio of woody and non-woody species	1:4.91	1:1.10	1:1.45	1:1.43	1:1.49
Total average density (125 m^2; ±SD; n=3)	1,942 ±361.40^{bc}	1,260 ±79.74^{a}	1,705 ±3.26^{ac}	1069 ± 495.10^{a}	1,100 ± 359.73^{a}
Shannon-Wiener Index (H')*	1.27	2.26	2.08	2.41	2.83
Shannon-Wiener Index (H')**	2.79	2.40	2.67	2.76	3.44
Evenness Index *	0.76	1.40	1.56	1.45	1.63
Evenness Index **	1.56	1.35	1.46	1.53	1.83

Values with the same letter in the same row are not significantly different (LSD test; $p < 0.05$)

* Shannon-Wiener's diversity index for all species (woody and non-woody species)

** Shannon-Wiener's diversity index for woody species

Table 4.8: Sorensen's Index of Similarity between woody species in five multi-purpose species plantations and primary trees and saplings in natural mixed deciduous forest with Teak (MDF).

AREA	MDF tree	MDF sapling	T	TT	TG	TTG	TTA
MDF tree	–	0.65	0.64	0.63	0.63	0.70	0.60
MDF sapling		–	0.58	0.65	0.63	0.63	0.65
T			–	0.80	0.84	0.82	0.73
TT				–	0.74	0.83	0.74
TG					–	0.80	0.72
TTG						–	0.72
TTA							–

Note: A double line indicates a maximum value; a single line indicates a minimum value

Discussion

Plant community development

In many parts of northern Thailand, forest plantations have been established on highly degraded agricultural or pastoral soils. The productivity of these soils has degraded to a point where they no longer economically support the production of crops. In these areas, where no ameliorative treatments are applied, soil properties such as compaction, organic matter, and nutrients etc., can significantly reduce the survival and growth of seedling (Reisinger, Simmons and Pop, 1988). The rate of natural recovery may also be slow (Greacen and Sands, 1980). However, from a sustainability and productivity standpoint, intensive forest management strategies are used to ameliorate limiting soil properties in order that long-term productivity is maintained. The plantation directly causes an increase in soil quality that is most clearly illustrated with the application of fertilizer. Addition of nutrients in amounts that are relatively large in comparison to the pool of available soil nutrients can have a long–term impact on productivity. Nowak, Downard and White (1991) demonstrated a long-term improvement in soil quality with a single application of potassium fertilizer on a site that had suffered years of degradation caused by crop production. An indirect impact of intensive mixed-plantation forest management is that the accelerated stand growth improves the accumulation of organic matter and nutrient cycling in the soil (Fox, 2000). Many case studies have indicated that intensive management in forest plantations increase rates of nutrient cycling, improves the condition of the forest floor and boosts organic matter (Harding and Jokela, 1994; Richter et al., 1995; Van Lear and Kapeluck, 1995; Nambiar, 1996; Byard, Lewis and Montagnini, 1996). Therefore, forest plantations have both direct and indirect effects upon soil properties, which are related to forest regeneration. In the present study the multi-species plantations acted as a 'nursery' for regenerative plants and facilitated the growth of enrichment species in the restoration mechanism.

The regenerative processes that occur in the understorey of the nurse plantations are interesting. To study this we look at the performance of the selected restoration strategy and how it affects the presence or absence of important native species. Natural recovery or revegetative processes within disturbed forests typically occurs in four sequential stages (Kozlowski, 1971; Oliver and Larson 1996).

1. A stand initiation and regeneration stage. Following tree harvesting or disturbance, forest stands may regenerate from propagules, seeds in the seed bank or those dispersed to the site, sprouting or remnant trees, depending upon soil and climatic conditions.
2. A thinning or stem exclusion stage. The most prevalent feature of this stage is accelerated mortality and changes in the dominant trees in monocultures or variations in species in mixed-species stands.
3. A transition or understorey regeneration stage. The dominant feature of this stage is the death of some of the upperstorey trees, resulting in gaps in the canopy and the reintroduction of understorey vegetation. The formation of gaps allows more solar radiation to reach the forest floor and enhances the growth of trees that were suppressed in the previous (thinning) stage.

4. A steady state or old-growth stage. The salient characteristic of this stage is a continuation of the successional processes that eventually culminates in an old-growth climax forest.

Each stage shows specific and unique characteristics, though several factors determine the regenerative processes seen in forest ecosystems; for example, propagule dispersal, the distance from external seed sources, previous land-use and the seed bank were the most important features acting at this site (Robinson and Handel, 1993; Guariguata, Rheingans and Montagnini, 1995; Strykstra, Bekker and Bakker, 1998). According to Kozlowski (1971), and Oliver and Larson (1996), multi-species plantations are at the transition or understorey regeneration stage.

All plantation types in this study provided favourable understorey environmental conditions for the recruitment and regeneration of a variety of native secondary forest woody and non-woody species. However, no differences were noted at the understorey regeneration stage between the single-species plantation and the four mixed-species plantations, though differences in understorey density (for both species and total density) and species diversity were found. The results showed that all five plantation types were capable of catalysing native forest succession on highly degraded or deforested land in northern Thailand, where natural regeneration was seriously impeded by the absence of seed dispersal, seed distribution and soil seed bank, as well as limited by the number of seed dispersal agents and the impediment by invasive and aggressive grasses (i.e. *Imperata cylindrica*) and herbs (i.e. *Chromolaena odorata*). Species composition (both woody and non-woody) beneath the five plantation types showed that most were native species of natural mature forest ecosystems, though a few exotics were found, such as *C. odorata*. Therefore, this study suggests that multi-purpose plantations show high potential to return the community back to as near a natural state as possible. This is in agreement with Parrotta, Turnbull and Jones (1997) who stated that for the purposes of forest restoration and rehabilitation on highly degraded areas in tropical forest, plantations under appropriate management can help accelerate natural recovery of a species-rich ecosystem and provide much-needed woody products for a population faced with shortages of fuel, timber and other products. Appropriate management includes, thinning of the canopy to increase available light, removing excessively even-spacing and providing external interventionist action only when serious disturbances occur. Likewise, in another restoration project in Point Pelee National Park, Mclachlan and Bazely (2001) reported that there were no significant differences in native plant diversity between restored and relatively undisturbed control sites. This suggested that restored sites recover to natural forest, though in their case it took more than 30 years. However, at the community level, measures of diversity to indicate habitat disturbance and recovery can be criticized. If one species disappears from a site and is replaced by a new species, then no overall change in diversity is recorded. Differences in native versus exotic succession are not reflected in the overall measure of diversity. Mclachlan and Bazely (2001) suggest it is important to examine any underlying changes in species composition. The present study supports this conclusion, since although some pre-climax and climax species were found in the understorey of the plantations, many of the primary species that still persisted in the surrounding forest were absent. Therefore, when developing a restoration strategy this crucial point should be considered.

One point to consider is that of ‘vulnerable’ native species, which often seem to be replaced by fast growing, shade-intolerant, weedy species. These vulnerable species have ephemeral flowering and restricted seed dispersal, so disappear quickly from disturbed habitats (Bratton, Hapeman and Mast, 1994; McLachlan and Bazely, 2001). The early phase of succession is often dominated by wind-dispersed species (Dzwomko, 1993), and long-distance dispersers seem to be the most effective colonizers of newly disturbed habitat (Willson, 1992). As well as seed dispersal and flowering phenology, other factors appear to affect the recovery of species in restored sites, such as dense thickets of early successional shrubs (e.g. *Cornus spp*.) (Kollman, 1994). In this study, it was the grass *Imperata cylindrica* that seemed to deter establishment of native species. Changes in soil fertility, pH and compactness that accompany long-term human use have also been shown to hinder and prevent recolonization (Peterken and Game, 1984). McLanchlan and Bazely (2001) state that plant species with restricted seed dispersal and/or ephemeral flowering were the most vulnerable; indeed the most highly vulnerable species were those that showed both traits. Therefore, highly vulnerable species are likely to remain absent from restored sites and will need to be reintroduced. The reintroduction of native plants is now routinely employed in many restoration projects and has also been suggested as a way of supplementing declining natural populations (Reinartz, 1995). Thus, reintroduction of vulnerable species should be recommended. This can be achieved passively by changing habitat conditions in order to facilitate natural reintroductions, or actively by replanting.

The rapid increase in aggressive grasses on disturbed sites is an important issue that should be discussed. The density of *Imperata cylindrica* increased in the plantations due to its regeneration strategy (resprouting), its tolerance and its need for intensive disturbances. The multi-purpose species plantation received intensive disturbance via weeding control and forest fire, thus increasing the density of *I. cylindrica*. A characteristic of *I. cylindrica* is its tolerance and ability to grow in highly degraded areas, thus growing to a higher density in the single-species plantation than in the mixed-species plantations. Soil fertility in the mixed-species plantations was improved by the increase in organic matter derived from leaf-litter decomposition. A study by Rattanasinganlachan (1996) at the same site supports this argument, in that average soil organic matter in the mixed-species plantations was much higher than in single-species plantation. Many factors influenced regeneration under nurse trees, with understorey conditions in mixed-species plantations more suitable for regenerative plant species than under single-species plantation. Factors may have been related to the amount of canopy cover, amount of litter, and decomposition. Research on leaf litter decomposition indicated that four different nurse species had different decomposition rates; 120 days for Gmelina (*Gmelina arborea*), 146 days for Teak (*Tectona grandis*), 204 days for Cashew (*Anacardium occidentale*) and 277 days for Tamarind (*Tamarindus indica*) (Saeheng, 1993). This indicated that the three-species plantation had a continuous and synchronized release of nutrients to the soil that was probably greatly influencing natural tree regeneration. However, there were no significant differences between the five plantation types.

Another factor that may be linked to the abundance of *I. cylindrica* in multi-purpose plantations is the amount of shade provided by the nurse trees. In some plantations the shade may not have been sufficient to suppress competing understorey herbs. Results on canopy cover indicate that shade was high in mixed-species plantations (TG = 308.95 ± 93.23 m^2/ plot and TTA = 231.74 ± 77.92 m^2/ plot) and low in single and mixed plantations (T = 172.09 ± 50.69 m^2/ plot, TT = 187.80 ± 49.01

m^2/plot and TTG = 162.99±11.85 m^2/ plot). The increased shade in the TG and TTA plantations was probably more effective in shading out grasses than in T, TT and TTG, thus giving a competitive advantage to woody seedlings. The problem of abundant grasses due to the amount of shade provided by the canopy was also reported by De Souza and Batista (2004), who commented that persistence of grasses may be related to the opening of the canopy in the dry season, which is a very common event in seasonal semi-deciduous forest ecosystems. Although it was not quantified, it is possible that the number of deciduous, rather than evergreen, species used in the plantations was excessive for successful forest restoration purposes. Thus, the massive shedding of leaves in the dry season probably allowed enough light through the canopy to provide grasses and other light demanding species the means for survival. This supports the findings of this study, which indicated that the highest density of *I. cylindrica* was found in the pure Teak plantation. The spacing of individual plants may also be another factor that contributes to the establishment and persistence of the grasses (De Souza and Batista, 2004). However, this was neither confirmed nor denied by our study.

The study indicated that the high density of *Imperata cylindrica* affected the establishment of other pre-climax species. The high density of *I. cylindrica* in the single-species plantation may be partly attributed to inter-specific competition (underground root competition), hence indicating that *I. cylindrica* can obstruct establishment by other pre-climax species (Guariguata et al., 1997; Parrotta, Knowes and Wunderke, 1997). The high competition by grasses and herbs in disturbed communities has been reported in several studies (Putz and Canham, 1992; Hill, Canham and Wood, 1995; Li and Wilson, 1999; Peltzer et al., 2000). Commonly, grasses and herbs will compete with tree seedlings for water, nutrients and light, to such an extent that they can slow the growth of the seedlings and slow the recovery of dominant canopy species. However, mixed-species plantations can solve this problem. Mixed plantations can reduce the growth of *I. cylindrica* and accelerate the regenerative processes of pre-climax species. This study has revealed that the number of species and the species diversity index increased from single to two-species plantations, with the highest ecological values found in the three-species plantation (Table 4.6 and 4.7). This matched results of other research (Otsamo, 1998; Carnevale and Montagnini, 2002), and suggested that if the objective of management is to maximize biodiversity then mixed-species plantations are preferable. Since grasses and other weeds are a problem on degraded land, especially during the initial stages of succession or during the first year of establishment, the cost of maintenance increases. De Souza and Batista (2004) suggested that the deciduousness of planted species seemed to favour the persistence of grasses, which is why the proportion of deciduous trees deserves attention in restoration projects. Moreover, alternative techniques should be used to prevent the survival of these grasses and other weedy invaders, such as incorporating trees that have a low degree of shedding and reducing the spacing between trees.

The results of species composition reveal that climax slow-growing species (i.e. long-lived, slow-growing tree species) that can survive and reproduce in mature forest were found in the pure plantations as well as the mixed-species plantations. These species, which included *Lannea coromandelica*, *Spondias pinnata*, *Garuya pinnata*, *Terminalia mucronata*, *Diospyoros mollis*, *Irvingia malayana*, *Xylia xylocarpa* var. *kerrii*, *Pterocarpus macrocarpus*, *Schleichera oleosa* and *Tectona grandis* have been found to be dominant in natural mixed deciduous forests in both the present and earlier studies (Ogawa, Yoda and Kira, 1961; Bunyavejchewin, 1983; Gajaseni and Jordan, 1990). This suggests that enrichment planting to establish climax forest may be successful in pure plantations as well as diverse plantations. The results of this study showed that secondary succession did not necessarily begin with pioneer species and that some pre-climax species could establish during the early stages of succession at the same time as pioneer species. Some of these pre-climax species could therefore be useful for restoration of forest on highly degraded land, by bypassing some of the early stages of natural successional processes. The aggressive weed *Imperata cylindrica* occurred in the understorey of all plantations. *I. cylindrica* is a typical pioneer species (i.e., short-lived, fast-growing and light demanding) which establishes rapidly within the first few years after disturbance in deciduous forest ecosystems.

Results on density and basal area provided strong evidence that some climax species, such as *Pterocarpus macrocarpus*, *Irvingia malayana, Dalbergia cutrata, Siphonodon celastrineus, Artocarpus lacucha, Largerstroemia duperreana* and *Vitex peduncularis* were able to occupy niches in the multi-purpose species plantations. This supports the concept that climax species can grow and establish in the early stages of succession, and that secondary succession does not have to begin with pioneer species. However, not all these species were able to establish in all the experimental plots. This was primarily due to different distribution mechanisms. For example, *P. macrocarpus*, which is a wind disperser, showed a much higher basal area in the T plot than in other plantation types. It could therefore disperse to areas far from the parent tree. A second reason was the physical distance between adjacent trees and the plots. It may be that some of the parent trees were close to the T plantation. Finally, the forest fires common to the area helped destroy the hard seed coat and promote germination. Many other primary tree species, such as *Irvingia malayana*, *Siphonodon celastrineus*, *Dalbergia oliveri*, *Vitex canescens*, *Albizia lebbeck*, *Xylia xylocarpa* var. *kerrii*, *Millettia leucantha* var. *buteoides*, *Gmelina arborea*, *Canarium subulatum*, *Millettia brandisiana*, *Terminalia alata* and *Garuga pinnata* were absent from some of the experimental plots (Table 4.5) for the above reason. Species such *Irvingia malayana*, *Siphonodon celastrineus*, *Gmelina arborea*, *Canarium subulatum* had big fruits and hard seeds, which could not be dispersed far from the parent trees without the action of large birds or mammals and may also have been prone to attack by seed predators.

Reasons for the low number of primary tree species found in the plots included both dispersal mechanisms and germination processes. In natural ecosystems the heavy seeds of *Irvingia malayana* were affected by rodent attack (personal observation) thus, they were unable to germinate and establish under natural conditions. Moreover, the germination rates of some of the primary trees were low. Under nursery conditions, germination rates for *Spondias pinnata*, *Chukrasia tabularis*, *Afzelia xylocarpa*, and *Irvingia malayana* were 20.95%, 35%, 48.62% and 13.70% respectively (Mae Moh plantation station, unpublished data). To improve the germination rate, these species needed special treatment such as trimming of the seed coat, boiling in water or acid for a few minutes and so on. Therefore, poor distribution mechanisms and low germination rates caused a low density of primary tree species in the plantation. This is a point that should be considered when developing restoration strategies.

The original floristic diversity and complexity of the forest is still far from being reached at the site. The low number of regenerating species originating from external seed sources suggested that seed dispersal limited species enrichment within multi-species plantation areas. The distribution of seeds from the surrounding forest will be observed in the next stage of this study. The presence of a seed source in the neighbourhood may be crucial for re-establishing species-rich communities. The floristic diversity of reforested sites may also be greatly enhanced through propagules that remain in primary and secondary remnants of forest or from isolated trees in the area (Wijdeven and Kuzee, 2000; Carnavale and Montagnini, 2002; Carriere, Letourmy and Mckey, 2002), though the degree of isolation will affect diversity.

In this study, flooding and fires provided natural disturbance models to observe regeneration methods. Different regeneration strategies affected the potential for occupation and establishment in multi-purpose species plantations. The study indicated that species such as *Croton roxburghii, Croton longissimus, Cratoxylum formusum, Barringtonia acutangala, Dalbergia cana* and *D. glomeriflora* regenerated vigorously by means of root and coppice resprouting after wild fire disturbances (personal observation). Therefore, these species increased in number and occupied spaces more quickly than other species. This suggested that non-sexual reproduction by means of root and coppice sprouting may be of greater relative importance in plantations than in the adjacent natural primary forest ecosystem. Previous intensive management, i.e. weed control, as well as natural and unnatural disturbances, such as annual forest fires and the hunting and collecting of wild products, influenced non-sexual reproduction in this plantation. In many disturbed ecosystems some species regenerate by sprouting from stumps or roots (Hoffmann et al., 2000). The sprouts that arise from root collars and the lower parts of stems emanate from dormant buds that grow outward just under the bark. They typically develop into shoots only when growth of the tree crown is disturbed. Such sprouts account almost entirely for the reproduction of some broad-leaved forest species. Nevertheless, some species regenerate by both sexual and vegetative reproduction. Naveh (1974) divided reproduction by woody plants into 2 types.

1. Obligatory resprouters that depend entirely on vegetative propagation.
2. Facultative resprouters that regenerate by both sprouting and seed germination.

A report from the Mediterranean region has stated that woody plants survive fire by sprouting from dormant buds on the root crown, and also from adventitious buds on lateral roots, stems and old shoots. In some species wildfire actually stimulated seed germination. A similar study indicated that resprouting was an important restoration method in tropical moist forest after slash-and-burn agriculture. This is a common recovery strategy found in most deciduous tropical forest species (Lugo, 1992, Rico-Gray and Garcia-Franco, 1992). However, primary forests have fewer species that can resprout compared to secondary forests (Kammersheidt, 1998). Ewel (1977) reported that adventitious sprouting from roots and stems was an important response to disturbance in many dry-forest ecosystems, though Dickinson, Whigham and Hermann (2000) noted that shade-intolerant species showed the highest degree of root sprouting. Paciorek, et al. (2000), demonstrated that the majority of moist tropical forest species (from small shrubs to large trees and of any age) showed a resprouting ability. This suggested that the resprouting ability might be an important component of a species' life history, affecting mortality and growth rates, and being likely to be important in forest community dynamics.

Another similar survey in temperate hardwood forest of floodplains, found that woody species relied on vegetative mechanisms regeneration (Deiller, Walter and Tremolieres, 2003). Their survey found various strategies of vegetative regeneration, such as aerial layering (*Prunus padus* and *Ligustrum vulgare*), production of root suckers (*Cornus sanguinea*, *Ulmus laevis* and *Popular alba*), production of adventitious root (*Hedera helix*) and resprouting from shoots (*Clematis vitalba*, *Cornus sanguinea*, *Corylus avellana*, *Crataegus monogyna*, *Euonymus europaeus*, *Fraxinus excelsior*, *Prunus padus*, *Ouercus robus* and *Ulmus laevis*). Vegetative reproduction was an important mechanism in this environment due to the consequences of flooding, which was particularly dramatic for seedlings due to their high sensitivity to prolonged anoxic conditions. Due to the environmentally restrictive conditions that limited their establishment and growth, the plants of the area had to develop strategies that ensured successful regeneration in spite of the constraints. Deiller, Walter and Tremolieres (2003), also noted that for sexual reproduction of woody species in the floodplain, the timing of seed release and germination, i.e. flowering, fruiting phenology, duration and level of flooding seemed to be critical.

I. cylindrica had an especially high potential for resprouting. The stands also received intensive disturbances by natural forest fire and clear cutting of weeds that affected the natural reproductive mechanism for this grass species. The seeds of *Antidesma sp*. had the ability to germinate rapidly under suitable environmental conditions, increasing in number and occupying niches and space to become the dominant species in multi-purpose plantations during this stage of succession. This observation is supported by the inter-specific competition theory, which states that species that regenerate quickly can occupy niches and space more quickly than other plant species. This could therefore be one reason for the reduced rate of establishment by other plant species. *Dalbergia cana* and *Cartoxylum formusum* showed a clumped pattern of dispersal since they regenerated from rootstock. The densities and basal areas of theses species were particularly high in the TT plot, since they sprouted quickly and grew fast, thereby becoming dominant. Their clumped distribution created strong root competition with other species. Therefore, it may take a long time for the ecosystem to recover if left to its own devices. The regeneration strategies of pioneer and intermediate species therefore affected the establishment of dominant natural climax tree species.

The importance of resprouting in terms of forest dynamics depends on the frequency of severe damage and the percentage survival and growth rate following resprouting (Paciorek et al., 2000). This is supported by the results of our study in that within the multi-purpose species plantations (that received annual disturbance from weeding and fire), the dominant species showed enhanced ability to resprout. The high rate and ability to resprout could have been caused by the high rate of damage. If differences in resprouting rates are caused by differences in the amount of damage, then resprouting rate is likely to be positively correlated to the mortality rate of undamaged individuals. Since a high mortality rate for a species might be indicative of high levels of damage that cause either mortality or resprouting. Sampling on Barro Colorado Island in Panama by Guariguata (1998), revealed that 3% of saplings of four shade-tolerant species (1-2.5 m tall) were severely damaged each year, though all survived until the end of the 2-year study, presumably via resprouting. Several studies have reported that trees resprouting from coppiced or logged stumps could survive for decades (Johnson, 1975; Murphy and Lugo, 1986). This study suggested that the fast-growing species *Morinda coreia* had a low density in the multi-purpose species plantation, but had a high basal area due to its growth performance. Since this species occupied space quickly, especially in TTG, it may have obstructed establishment of other regenerative species by means of dense canopy shade and root competition. This is indicated by the diversity index. Under natural regeneration strategies, species that do not resprout may be at a competitive disadvantage, since severe damage to an individual could cause mortality.

The density of woody species regenerating individual in plot T was low compared to other forests undergoing restoration (though a few species showed greater density values). The densities of woody species in the five plantations were 26,240 individuals per ha in plot T, 48,080 individuals per ha in plot TT, 55,680 individuals per ha in plot TG, 35,200 individuals per ha in plot TTA and 52,720 individuals per ha in plot TTG. Each individual woody species was counted. In a study of the density of regenerating species in Brazil, Durigan and Dias (1990) found 140,650 individuals per ha (for plants ranging from 5-200 cm in height) in a 17-year-old, mixed-species reforestation site. This value is very high compared to this study. Another report, from Parrotta, Knowles and Wunderke (1997) indicated a density of 28,800 individuals per ha (up to 200 cm) in a 10 year-old restored site in the state of Para, Brazil. Though the area was highly degraded by bauxite mining, the whole site was surrounded by primary forest, which contributed significantly to the regeneration process. Grombone-Guaratini (1999) recorded a density of 27,500 individuals per ha (50-400 cm in height but less than 4.8 cm in DBH) at a seasonal semi-deciduous forest site in Sao Paulo. Despite some differences in sampling criteria and the ages of the forest. Thus, forest physiognomy and dominance of species are insufficient indicators of the success of restoration projects, since in the long run the apparently successful community may disintegrate (Ewel, 1987). The assessment of other important factors such as soil components and dynamics, biotic interactions, wildlife colonization, seed dispersal and others (Ewel, 1987, Ehrenfeld and Toth, 1997; Parrotta, Knowles and Wunderke, 1997; Block et al., 2001) has been proposed in order to more accurately evaluate the success of restoration and improve our comprehension of restoration processes.

Consideration of densities for shrubs and trees found that TTA had the highest value for shrubs and that TG had the highest value for trees. The lowest values were found in the single-species plantation. This indicated that the mixed-species plantations enhanced the establishment of tree and shrub species. Likewise, Shannon-Wiener diversity indices and the evenness indices were highest in the three-species plantations, and lowest in the single species plantation. These results did not support the hypothesis of the research. The research found that diversity of regenerative plants in the understorey of three-species plantations was greater than that in the single-species plantation. Although the single-species plantation had more vacant niches than the mixed species plantations, the species regenerating in the understorey of the single-species plantation could not occupy all available niches. The study clearly showed that the density of the grass *Imperata* negatively affected the establishment of other species. Plot T had the highest density of grass and showed the lowest species diversity index. For these reasons, species diversity in the single-species plantation was lower than in the mixed species plantations. These results suggest that if the objective of forest management is maximizing biodiversity, then mixed-species plantations are the preferable choice for restoration projects, though they should be adapted for each area. Restoration efforts often involve a focus on multi-species assemblages, but since they consist of populations of co-existing species, they must be understood not only in terms of species interactions but also in terms of population processes, habitat and resource dynamics and disturbance theory. Since a feedback exists between species composition and ecosystem processes; processes that evolve over different time scales. Therefore, practical restoration may involve the establishment of sequential, multi-step goals to restore desired community structure or species richness, monitor the development of community structure and verify that links between community structure and function have been established (Palmer, Ambrose and LeRoy Poff, 1997). To develop a restoration strategy requires not only data of existing plant diversity, but also further information regarding ecosystem dynamics and the seed input necessary to achieve the desired results of the project.

The trends found in this study were similar to those of others studies that compared diversity of regenerative species in forest plantations. Carnevale and Montagnini (2002) showed that regeneration of native tree species, both in terms of number of species and number of individuals, was higher in mixed plantations than in pure plantations. Parrotta (1999) showed that species richness was higher in mixed plantations of Eucalyptus/Casuarina than in pure Casuarina plantations. Our study showed that mixed-species plantations had a high potential for accelerating the process of natural succession and establishing a stand of ecologically and economically desirable trees. Other studies also showed that mixed-tree plantations could be an effective tool for arresting site degradation, acting as a catalyst for forest regeneration and rehabilitation (Lugo, 1988; Parrotta, 1995).

Modes of seed input for natural plant species

Many of the tree species occurring in the adjacent natural mixed deciduous forest needed some form of animal assistance to disperse their seeds. Reports by the Forest Restoration Research Unit (2000) and Elliott et al. (1994) showed that some woody species found in deciduous forests, such as *Ficus sp*., *Gmelina arborea*, *Phyllanthus emblica*, *Phyllanthus orientalis*, *Antidesma sp*. and *Aporosa sp*. utilised animal seed dispersers. Other naturally occurring species such as *Spondias pinnata*, *Irvingia malayana* and *Canarium subulatum* could not disperse far from the fruiting tree due to their seed morphology and weight (no wings and large heavy seeds). Therefore, these species required a large frugivorous animal to act as a seed dispersal agent. In the multi-purpose plantations some species, such as *Anacardium occidentale* (red coloured, fleshy fruit) and *Gmelina arborea* were more attractive to animal dispersers than others. These planted species therefore induced frugivorous animals into the plantation, which were then able to disperse tree seeds from the natural forest ecosystem to the multi-purpose species plantation. This was one reason for the increased diversity of plant species in the mixed-species plantations.

Ecological factors involved

There were probably many ecological factors that contributed to the higher diversity of primary forest trees in the multiple-species plantations, such as amount of litter accumulation, amount of canopy shading, and stocks of soil nutrients (Carnevale and Montagnini 2002). Ewel (1980) reported that in the development of successional vegetation, nutrient stocks were an important factor, influencing quantity and diversity of species establishment. Horn and Montagnini (1999) showed that litter fall and accumulation fostered the establishment of pre-climax species. For the plantations used in this study, Saeheang (1993) showed that the supply of nutrients in the soil of multiple-species plantations was higher than in the soil of pure stands of *Tectona grandis*. Leaf litter, and thus stocks of nutrients that were returned to the soil, may have been higher in mixed-species plantations because they (with several realized niches) were better able to utilize available resources of light, nutrients and water than single-species plantations (where only one niche was realized). The theory of competition can help explain why the mixed-species plantations had the highest species diversity. Inter-specific competition among understorey species in the three-species plantation was high due to decreased niche width, which raised the floristic diversity of understorey species. This theory is supported by the ecological niche concept (Kimmins, 1996).

Ecological succession

According to Perry (1995), successional processes are influenced by the two following factors:

1. The species that comprise the initial colonists at the disturbed site.
2. The impacts of the initial and succeeding dominants on subsequent conditions and events.

Regulation of establishment and persistence of seral plants during succession is complex. It typically includes aspects of facilitation and inhibition and is affected by interactions for available resources, as well as other environmental factors such as climate, animals, pathogens, mycorrhizal fungi and other microbes. The nature of the interactions also changes over time (Connell and Slatyer, 1977).

The inhibition model is based upon the changes in available resources during successional processes. Once the early colonizing species have occupied space and obtained available resources, they either arrest invasion of subsequent species or suppress the growth of those already present. Sometimes early successional species comprise a dense monoculture that slows or precludes invasion by other species. Tree species in South America do not readily invade abandoned pastures due to inadequate seed dispersal, consumption of seeds and seedlings by rodents, competition between grasses and trees for solar radiation, mineral nutrients and water, and mortality of seedlings by recurrent burning of grasslands (Nepstad, Uhl and Serrao, 1990).

The facilitation pathway modifies the site and renders it more suitable for establishment and growth of later successional plants. Site modification is accomplished by the weathering of rocks, accumulation of mineral nutrients and the provision of energy that allows plants to build soils and absorb mineral nutrients. An example of the facilitation model is the 'island effect', which is characterized by establishment and persistence of plant species near nurse trees and shrubs. Nurse trees provide seedlings with shelter from severe environmental factors, as well as being centres for seed dispersal via visiting birds and mammals and also providing local sites with enriched soil. Plant islands also may facilitate establishment of late successional species by influencing soil chemistry, biology and structure. Early successional species may also facilitate colonization of succeeding species by providing a legacy of mycorrhizal fungi (Perry, 1995).

The different regeneration mechanisms found in each species may have affected establishment and persistence in the understorey of the mixed-species plantation. These mechanisms are complex and may have affected the outcome of the successional process. Successful regeneration following disturbance, particular of woody species, is a complex process. Methods of regeneration and growth are important considerations. Physical factors such as light, temperature and moisture also influence woody regeneration, as do other plants, animals and microbes. Fungi, bacteria and other microorganisms can slow regeneration by destroying seeds and seedlings. Non-woody species, such as *Imperata*, are often shade-intolerant and tend to regenerate immediately after disturbance. Meanwhile, woody species include both shade-tolerant species (that grow well under deep shade, but respond less well to increased light) and intermediate species (that grow more slowly than intolerant species in full sunlight, but grow more quickly in shade). The primary route towards regeneration in degraded or disturbed areas is resprouting from roots and stumps, as well as seed germination. This study indicated that resprouting was a common and important mechanism in the stands, as shown by the clumped distribution of regenerating plant species in the understorey of the multi-species plantations. After serious disturbance, these resprouts could grow more rapidly than vegetation that developed from seeds, due to the stored energy available in the roots and stumps. Seeding was a secondary regeneration strategy in this area, probably due to the extensive slash and burn agriculture of the past.

Early successional species are usually better represented than late successional species in the seed bank (Leck, 1995), though composition of the early colonists depends on the severity of disturbance. Therefore, early colonizing species, which are fast growing and usually short-lived, initially dominate a site but are later replaced by species that were dominants prior to disturbance. The latter species may develop from seeds, sprouts or residual plants that survived the disturbance. Succession in severely disturbed tropical forests back to the state of primary forest is very slow. Following the clear-cutting of primary forest a dense growth of weeds, shrubs, vines and young trees typically emerges. Fast-growing, very-short-lived trees then become dominant and are succeeded by slower-growing, longer-lived species, sometimes only after hundreds of years. If the course of succession is interrupted by additional disturbances, more mineral nutrients are lost from the soil, thus rendering the site unsuitable for even the early successional tree species (Kimmins, 1996).

Much research has indicated that in slash and burn forest communities, secondary forest species or certain grass species become established within the first few years after disturbance (Sukwong, 1978). Results on density and basal area indicated that pioneer and intermediate successional species occupied and dominated large areas in all five multi-purpose species plantations at this successional stage. Species included *Imperata cylindrica*, *Cratoxylum formusum*, *Croton longissimus*, *Dalbergia cana*, *Morinda coreia*, *Wringtia pubescens*, and *Phyllanthus orientalis*. In the understorey of the multi-purpose species plantations, just one grass species, *I. cylindrica*, colonized, established, and grew vigorously to be solely dominant, especially in the Teak plantation (control). However, some climax species found in natural mixed deciduous forest with Teak became established in the understorey of the mixed tree plantations.

Species included *Irvingia malayana*, *Siphonodon celastrineus*, *Artocarpus lucuctha*, *Tectona grandis, Xylia xylocarpa* var. *kerrii*, *Vitex peduncularis*. So long as the area receives no further disturbances, the floristic characteristics of the understorey will change over time to become dominated by these climax species. Therefore, the multi-purpose species plantations were acting as a transitional zone in terms of successional processes.

Succession is a complex task, and it was unclear whether the facilitation, tolerance or inhibition model was the dominant process at this stage. Results on the number of primary tree saplings that became established in the understorey of the multi-species plantations found that the three-species plantations (TTA and TTG respectively) had the highest number of species. Thus, compared to the other multi-purpose species plantations, the three-species plantations should have a greater potential to develop into a climax community than the single and two-species plantations. In addition, the understorey of the multi-purpose species plantation had a greater diversity of plant species than the single plantation. The density of grass was also lowest in these plots. This suggested that, due to successional processes, mixed-species plantations had greater potential to recover to a near-natural or natural forest ecosystem. Results also suggested that some stages in ecological succession were not necessary for reforestation or afforestation projects in the area. This is a useful point to consider when developing a restoration strategy to accelerate germination and colonization of primary tree species. Therefore, enrichment planting of some pre-climax and climax species would seem to be advisable in future restoration processes.

The study investigated which of the principles of ecological successional could explain the phenomenon observed at the site. During the early stages of succession, we surprisingly found that a few climax species, such as *Xylia xylocarpa* var. *kerrii*, *Lannea coromendelica*, *Spondias pinnata*, *Terminalia mucronata*, *Garuya pinanta*, *Irvingia malayana* and *Pterocarpus macrocarpus*, had become established in the understorey of the multi-purpose species plantation. These tree species are slow growing and long-lived, their seeds are hard to germinate and they have characteristics suitable for establishment in the environment of a mature forest ecosystem. The tolerance principle, one of Cornell and Slatyer’s (1977) three classical successional models, could therefore explain this phenomenon since these species were tolerant of the harsh conditions in the multi-purpose plantations. They also did not require the facilitating effect of pioneer species to condition the environment for them. In fact, some pioneer species such as *Imperata cylindrica* and *Chromolaena odorata* obstructed or inhibited their regeneration potential. Thus, the facilitation principle was not supported by this study. The third ecological successional concept is that of inhibition. This concept was difficult to study due to the regeneration phenomenon observed in the plantation. However, some of our observations may be explained by the inhibition successional concept. A few pre-climax species were seen to occupy niches, but the majority of those had low numbers of individuals. That would indicate that some factors had inhibited their regeneration. The first factor was probably competition from other species, especially *I. cylindrica,* which grew rapidly in the plots. The second factor may be related to the harsh conditions observed in the multi-species plantations, such as low levels of soil nutrients, high light intensity or other soil properties. Therefore, the inhibition concept may possibly explain the regeneration processes in the multi-species plantation. However, neither the tolerance nor inhibition principles of succession could clearly support the regeneration processes observed in the stands.

Analogy between species in the understorey of plantations and the natural forest ecosystem

All multi-purpose species plantations in this study introduced an element of structural complexity to highly degraded land, though they were considerably less complex than the adjacent natural forest and lacked many of the unique life forms typical of the ecosystem (both upper and understorey plants). The observations recorded by this study have implications to the design and management of restoration projects, in that the structure and complexity of a mixed deciduous forest ecosystem with Teak cannot be restored in a short time (a few decades). Fortunately, the suite of species in a multi-purpose species plantation helps promote a diverse species composition within a short time, when considering similarity indices. Therefore, if given enough time, multi-purpose species plantations may also develop a complex forest in the understorey layer. This is similar to timber plantations in other areas that catalyse the restoration of rainforest on cleared and highly degraded lands (Lugo, 1997; Parrotta, Knowles and Wunderke, 1997; Lamb, 1998; Ashton et al., 2001). However, the plantations in this study showed variable recruitment to the understorey. The species composition and number of recruited species varied between the species and number of nurse trees. Natural regeneration of secondary forest in the understorey, without intensive human intervention, might be expected to provide relatively poor habitat for specialist wildlife, i.e. without fire protection or thinning, and with only weed control. Thus, design and management may need to be modified to accelerate the development and maintenance of an understorey ecosystem.

Results on numbers and diversity indices of both woody and non-woody species showed that mixed-species plantations encouraged and sustained understorey floristic establishment. Although the number of primary tree species was highest in the three-species plantation (TTG) and in TTA and TT for primary saplings, this trend was not related to the number of mixed tree species, since the ecological values were similar in all experimental plots. This indicated that single-species plantations did not provide an environment more suitable to encouraging primary species from natural forest than mixed plantations. However, similarity results in the five multi-purpose species plantations showed that they supported a richness of species similar to the neighbouring natural forest ecosystem. The restored site (the multi-purpose species plantation) had substantially recovered since the modified version of Forest Village System program was initiated in 1988. The presence of forest tree species in the understorey of the plantation indicated that the species pool for recolonization was made available by stumps or rootstocks or by seed dispersal via animals and winds from the neighbouring forest ecosystem.

An external factor that had an important influence on the dispersal of native seeds from natural forest ecosystems to highly degraded areas was seed dispersers. Many reports have indicated that some tree species in the tropics require animal assistance in order to disperse their seeds (Howe and Smallwood, 1982; Elliott, Promkutkaew and Maxwell, 1994; Wunderle, 1997). The results of this study showed that there was little difference in woody species composition between the plantations and the natural forest ecosystem. An advantage of the mixed-species plantation is that fruit of trees such as Gmelina and Cashew attract animal dispersers from the natural forest ecosystem. This therefore increased the similarity of woody plant species between mixed plantations and adjacent natural forest. However, there are many factors that affect plant colonization and establishment in highly degraded areas. Studies have indicated that structural complexity of vegetation affects natural seed distribution (Wunderle, 1997). Similarly, Reeders (1985) concluded that bird recolonization of highly degraded land in northern Australia was associated with the structure of existing vegetation at the site.

If the purpose of restoration is the conversion of highly degraded areas back to near-natural or natural forest, then a diversified nurse tree plantation is suitable. This is indicated by the high index value of primary tree species found in the three-species mixed plantations (TTG, TTA and TT). These values suggest that successional trends in the understorey of mixed-species plantations will move towards a mature forest community faster than other plantations. Since trees form the main structure of the forest ecosystem, then if primary native tree species have the chance to occupy niches in the initial stages of succession, they seem able to grow and develop into a mature forest structure and composition. Mixed-species plantations are therefore the preferable choice for use in any restoration strategy. Factors that influence distribution of native flora to plantation areas include dispersal agents, distance between the natural forest and the plantation, remnant trees. Carnevale and Montagnini (2002) suggest that canopy shade and deep litter provide favourable conditions for the arrival and germination (recruitment) of tree seeds. Thus, the various canopy layers found in mixed plantations may positively affect understorey conditions, thereby encouraging sustainability of species diversity.

Similarity values between size categories (trees and saplings) in the natural mixed deciduous forest ecosystem were high (0.65). Kiratiprayoon et al. (1994) showed similar results (0.54) in a study in Ngao district, Lampang province. Results of species density and composition showed that the dominant sapling species were not different from the tree species dominating the canopy. This suggests that the proportion of saplings is related to the proportion of trees. Since this forest ecosystem had been disturbed it had therefore lost the large and previously dominant species. A replicate group of species in the sapling category would exhibit similar patterns of inter-specific dominance, thereby being directly available to replace the upperstorey.

The high similarity values (Table 4.8) indicate that the natural forest has a high capability of reverting to a stable and climax community after disturbance, or that it provides a relative measure of resilience to encroaching selective logging or severe destruction. This resilience will therefore be of benefit to future restoration strategies, since it can provide practical guidelines necessary for developing appropriate sustainable management via selection of plant species that help accelerate natural succession.

However, the results of the present study indicated that the dominant species occupying niches in the understorey were different from dominant species in the upperstorey of the primary forest ecosystem. Therefore, initial species composition after disturbance can influence forest stand composition and structure. For example, in a survey in Northern Ontario, Carleton and Maclellan (1994) found that conifers were more widespread in post-fire stands, and hardwoods such as Aspen and tall broadleaved shrubs were more frequent in logged areas. Peltzer et al. (2000), likewise found high densities of spruce and reduced Aspen growth in burned sites in southern boreal forest near Prince Albert National Park, Canada. Peltzer et al. (2000), also noted that shoot mass, density and basal area of Aspen tended to be lower in the more intensely prepared treatment sites than in naturally regenerated sites. The accumulation and mineralization of soil organic matter is a key process maintaining the productivity of forest ecosystems, and roots are one of the major sources of organic matter. Therefore, disturbed areas, which show a decrease in the abundance of early successional species, suffer resulting nutrient leaching and long-term alterations in soil structure (Vitosek et al., 1979; Ezell and Arbour, 1985). This suggested that site preparation decreased overall Aspen density, rather than decreasing the growth and resource demand of individual Aspen stems. Moreover, human disturbances often increase the rate of invasion of herbaceous vegetation (Hobbs and Huenneke, 1992), and may also be the case in our multi-purpose species plantation, especially the pure Teak stand. Many reasons support this conclusion; for example, disturbance increases soil moisture levels and micro-site availability for invasive species, such as rhizomatous grasses, resulting in a different initial community structure. Therefore, when forests and other ecosystems become increasingly affected by human activities, integration of biological diversity and resource management objectives is needed to maintain the processes necessary for the continued productivity of managed systems. For this reason, restoration using native enrichment climax species would seem to be a strategy that supports initial species composition and eventually succession towards a primary forest ecosystem.

In contrast, Healey and Gara (2003) showed that the establishment of a Teak plantation on abandoned pasture in southwestern Costa Rica had impeded the regeneration of several native species that otherwise may have become established. Their control was an abandoned, unplanted agricultural site, and it demonstrated that the incoming native tree species to the Teak plantation were limited in terms of abundance, height, growth, diversity and growth form. The factor that impeded the establishment of native species in the area was aggressive competition by local grasses and ferns. In our study, the aggressive grass *Imperata cylindrica* was a dominant competitor in pure Teak and mixed-species plantations, similar to Healey and Gara (2003), but similarity indices for trees and saplings in the Teak plantation and natural forest were high (0.64 and 0.58). Therefore, natural native species had the potential to establish and persist beneath the pure Teak plantation, though the potential was less than in the mixed-species plantations. For species similarity between the five multi-species plantations and the natural forest ecosystem, the Teak plantation showed the lowest value. This might be related to the shade produced by Teak's large leaves. Varying crown shapes and canopy densities can have a strong effect on the amount of available light reaching the understorey and on recruitment (Kabakoff and Chazdon, 1996; Chazdon et al., 1996; Healey and Gara, 2003). Another factor limiting recruitment might have been that the pure Teak plantation lacked appeal to native seed dispersers. Its flowers are hermaphroditic and contain little nectar (Tewari, 1992), and the seed is enclosed in a small dry buoyant fruit that depends chiefly upon water for dispersal (White, 1991). Finally, foliar allelopathic leachates may have limited the number of seedlings growing in the Teak plantation. Research on leaf litter components has indicated that phenolic acids are significantly concentrated in the foliage of Teak (Murugan and Kumer, 1996). Thus, phenols have been implicated in the regeneration failure of many forest types (Pelissier and Souto, 1999; Mitzutani, 1999).

Litter depth and texture also affect seedling emergence and growth. For example, the coarse texture of Teak and grass litter is a barrier to seedling growth due to physical obstruction. In mixed deciduous forest ecosystems, most seeds are dispersed after the leaves have fallen. Thus, they become trapped in the inhospitable upper strata of the litter. In addition, the physical composition of the litter may impede the growth of radicals (Facelli and Pickett, 1991). Likewise, solar radiation can create extremely hot and dry conditions in the upper strata. These factors have the effect of obstructing seeding germination.

The similarities of tree and sapling groups between the five plantations and the natural forest were almost equal, except for that between MDF saplings and those in the Teak plantation. Likewise, the ecological values between the plantation types were high and not very different. This suggests that the regenerating plant species originated from the same place, namely the adjacent natural forest. In addition, the remnant trees of the area were an important source of seeds. The relatively successful recolonization by native species was likely associated with the remnant forest habitat. A characteristic regenerative strategy observed in the plots was resprouting from root and/or stem. Therefore, existing plants could regenerate and increase in number to occupy niches. These results suggest that the floristic composition of the understorey of nurse tree plantations has the potential to develop into a near-natural forest ecosystem.

4.2 SEED STUDY

4.2.1 Seasonal and annual variation in seed production

Results

Seed production of native woody species in the natural forest occurred mainly in April (before the rainy season), though a second peak was seen in July (rainy season) (Figure 4.2 and 4.3). In the ecotone area, the greatest seed production was in the late dry season and early rainy season (April), with a second seeding for some species during the rainy season in July, similar therefore to the natural forest. In the mixed-species plantations, seed dispersal and/or production by native woody species was observed from January to June. At other times the number of seeds produced was very low (Figure 4.2). Likewise, the pattern of seasonal change in the soil seed bank was similar to seed dispersal (Figure 4.3).

Average annual density of dispersed seed was 36,182 seeds/ha in MDF, 20,434 seeds/ha in ECO and 17,073 seeds/ha in MP respectively. In the natural forest, average individual density ranged from 66 seeds/ha for *Colona siamica* (Tiliaceae) to 12,791 seeds/ha for *Vitex peduncularis* (Labiatae). In the ecotone area, average density of individual species varied from 88 seeds/ha for *Croton oblongifolius* (Euphobiaceae) to 303,000 seeds/ha for *Grewia elastostemoides* (Tiliaceae). Average individual density for species in the mixed-species plantation ranged from 416 seeds/ha (*Albizia lebbeck*, Leguminosae) to 37,986 seeds/ha for *Tectona grandis* (Labiatae). The number of woody species that dispersed seeds increased from 9 in MP, to 17 in ECO and 24 in MDF (Appendix A.6).

The soil seed bank comprised of 21 species in MDF, 19 species in the ecotone and only 9 species in MP. Average annual soil seed bank density was 17,037 seeds/ha in MP, 20,434 seeds/ha in ECO and 36,182 seeds/ha in MDF. Among individual seed species in MDF, mean density ranged from 119-5,555 seeds/ha. The minimum value was represented by *Spondias pinnata* (Anacardiaceae) and the maximum value was for *Getonia floribunda* (Combretaceae). In the ecotone, the minimum average density among species was 88 seeds/ha for *Croton longissimus* (Euphorbiaceae) and the maximum was 15,387 seeds/ha for *Tectona grandis* (Labiatae). Finally in MP, average density varied from 139 (unidentified sp. 23) to 28,264 seeds/ha (*T. grandis*) (Appendix A.6).

When considering species individually, the results showed that production of dispersed seed of dominant tree species varied throughout the year (Figure 4.4). For example, *Lagerstroemia duperreana*, *Afzelia xylocarpa* and *Xylia xylocarpa* var. *kerrii* produced seeds in the late dry season (April). Meanwhile, *Irvingia malayana* produced seeds in the rainy season and the seeds of *Pterocarpus macrocarpus* were found in both the dry season (January) and the rainy season (September). *Tectona grandis* shed their seeds over nearly 8 months from October to June. The period for seed production and seed fall of dominant native species in natural MDF forest of this region therefore varied among species.

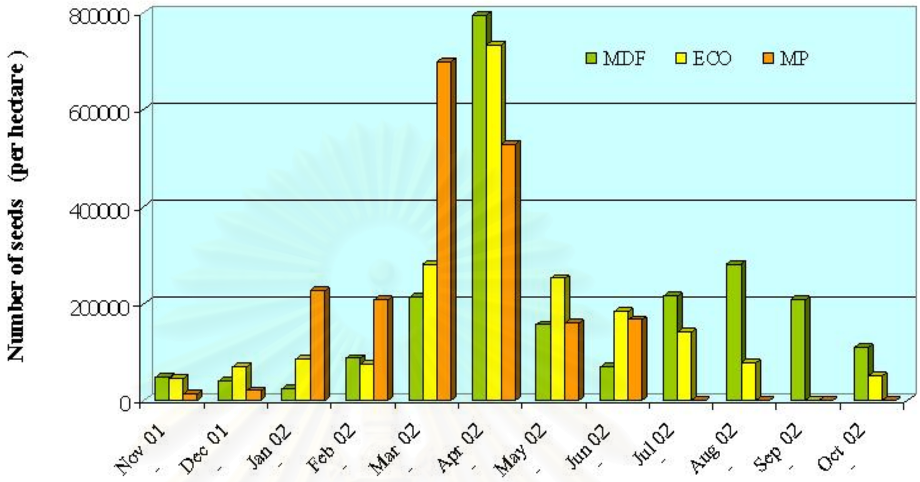

Figure 4.2: Monthly variation in the average density of dispersed seeds in three areas; natural mixed deciduous forest with Teak (MDF), ecotone (ECO) and multi-purpose species plantation (MP) from November 2001-October 2002.

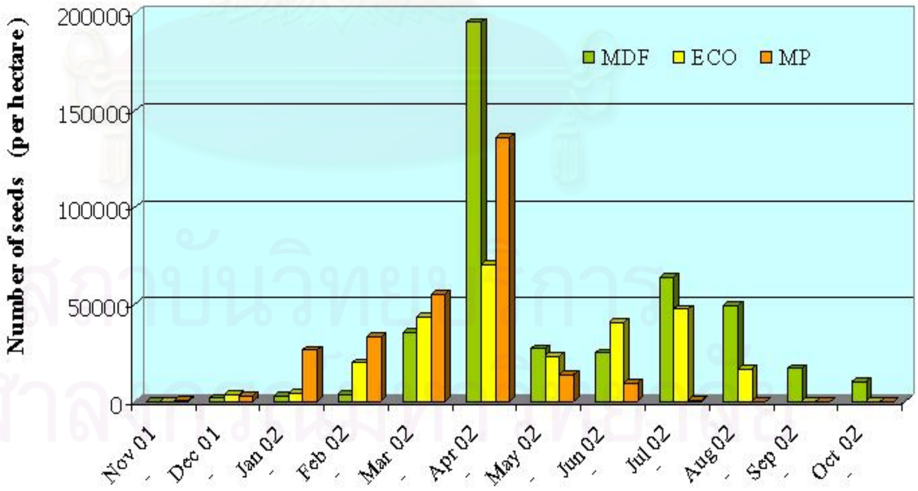

Figure 4.3: Monthly variation in the average density of the soil seed bank (at a depth of 1-5 cm) in three areas; natural mixed deciduous forest with Teak (MDF), ecotone (ECO) and multi-purpose species plantation (MP) from November 2001-October 2002.

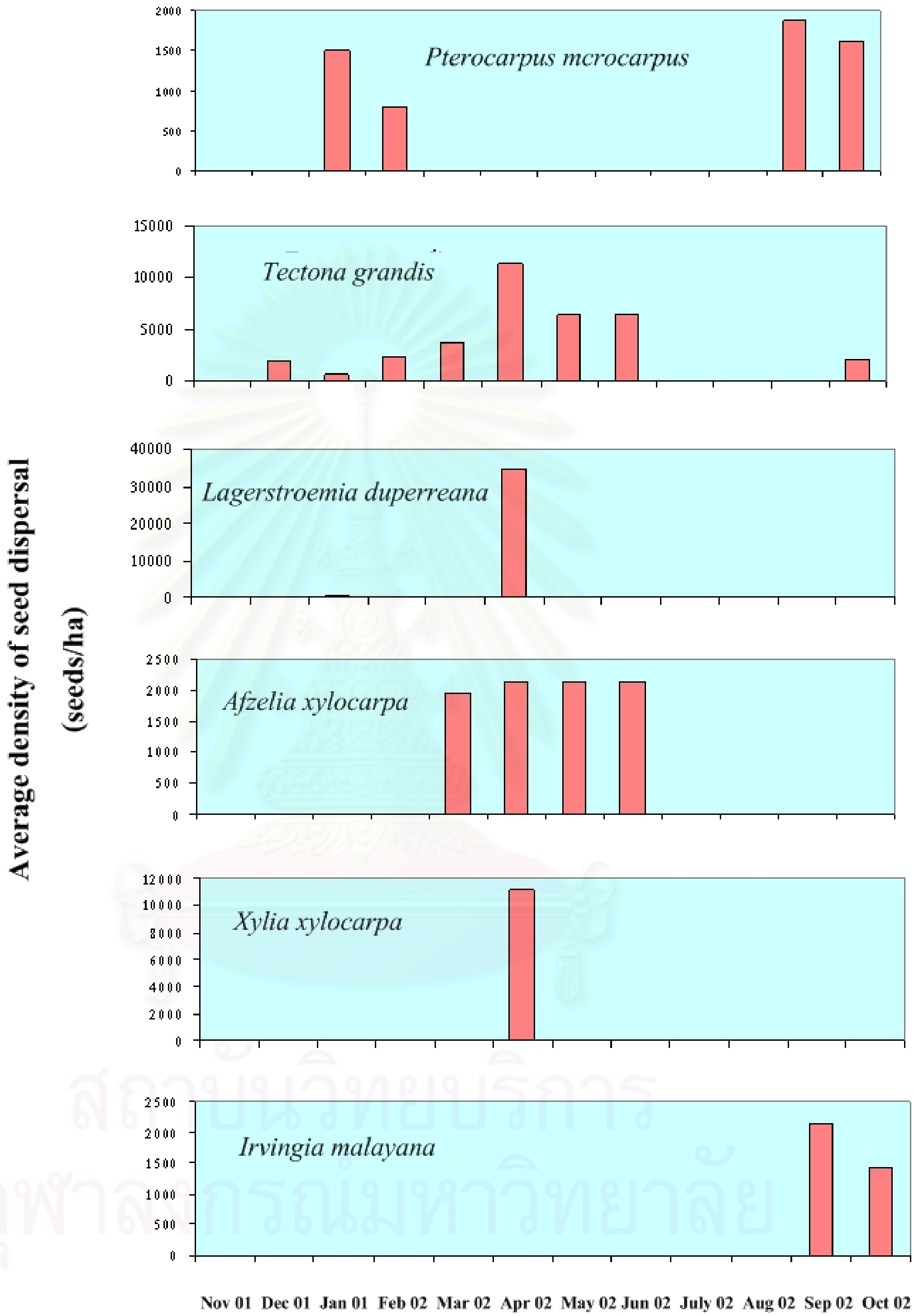

Figure 4.4: Seasonal trends in seed production for selected dominant native tree species in natural mixed deciduous forest with Teak in Lampang province, northern Thailand from November 2001-October 2002.

Discussion

Variation in seed dispersal, and implications to the soil seed bank

In northern regions of Thailand, an important characteristic of the mixed deciduous forest ecosystem is that it is mostly leafless from December to April during the dry, and later hot season. At the end of this period most trees disperse their seeds, allowing them to germinate in favourable environmental conditions during the rainy season. Fruits and seeds are dispersed over time as well as space (Harper, 1977), and are affected by climatic variables. Factors such as wind velocity may determine the rate at which ripe fruits and seeds disperse form the parent plant. Observations in natural MDF showed variation and fluctuation in tree seed production throughout the year, though the highest peak for seed production occurred before the rainy season, due to ecologically favourable conditions for germination. There was a second peak in both seed dispersal and the soil seed bank in the middle of the rainy season (July). This double peak of seed production was also observed in a study by Sukwong et al. (1995) in northeastern Thailand. A double peak in fruiting was also reported for dry tropical forest in Doi Suthep-Pui National Park in northern Thailand (Elliott, Promkutkaew and Maxwell, 1994). These results can be attributed to differences in seed phenology. The first phase of seed production mostly concerned wind dispersers, due to the strong, gusting winds at this time of year. The second phase of seed production was designed to primarily attract frugivores. Thus, these animals played an important role as seed dispersal agents.

Plants growing in highly unpredictable environments can either release fruit and seeds over a long period or concentrate dispersal into shorter periods under more favourable conditions. Some species adopt both strategies with linked patterns of dispersal and germination (Venable, Dyreson and Morales, 1995; Mandak and Pysek, 2001). For the genus *Atriplex* (Chenopodiaceae) the period of germination is extended by the gradual release of fruits, as well as by varying levels of dispersability, germinative ability and dormancy (Khan and Ungar, 1986). These patterns probably maximize the chances of survival by ensuring the presence of appropriate fruits and seeds in the soil when conditions for germination are favourable (Mandak and Pysek, 2001). Many factors are related to seed dispersal and germination, especially at the end of the dry season and before the rainy season. Some seed species, such as *Tectona grandis* need external factors like fire to stimulate germination processes. Other species must avoid severe impacts such as drought and forest fire. In addition, this peak period for seed dispersal could be related to wind speed, which is highest at this time of year. This would allow their seeds to be distributed over a greater area, increasing successful invasion and occupation of available niches. Moreover, the ripe seeds are dispersed in time for the following rainy season, when environmental conditions are suitable for germination. Differences in seed phenology, arising from both genetics and evolution, affect seedling optimisation in natural forest. This provides greater efficiency in germination and establishment, helping maintain species diversity, composition and forest structure.

However, the present study noted that some dominant tree species in natural mixed deciduous forest with Teak did not disperse their seeds during the first peak of seed dispersal, but instead formed a second peak, during the rainy season, by dispersing their seed after the dry season had ended. Some mature ripe seeds, such as those of *Canarium subulatum*, *Vitex peduncularis*, *Getonia floribunda* and *Grewia eriocarpa* are not designed for wind dispersal. The phenology of these species can be attributed to several ecological reasons, the first being related to forest fire. Tree seeds that fall during the rainy season avoid damage to their seeds from forest fire. This improves their rate of germination. Most seeds that fall in the early dry season are killed by occasional forest fire. However, the seeds of a few species survive forest fires and show a considerable increase in germination rate. Another factor may be related to the movement of seed, since many are large and heavy, thus requiring animal vectors e.g. bats, birds and mammals in order to disperse from the parent tree. During the rainy season many of these large frugivores are present due to the availability of food.

Phenological patterns of seed production varied among species (Figure 4.4). For example, *Tectona grandis* shed their seeds from January to June. *Pterocarpus macrocarpus* produced seeds twice a year (January-February and September–October) and *Xylia xylocarpa* var. *kerrii* shed their seeds only in April. This variation may have an influence on regeneration in both natural stands and multi-purpose species plantations. However, the research was limited in terms of time. Two primary components affect floristic recruitment, limited spatial seed dispersal and limited parent fecundity (Webb and Peart, 2001). Guevara and Gomez-Pompa (1976) suggested that seasonal variation in seed production was a factor influencing seed bank abundance and species richness. Variation in seed fall between species may be related to regeneration strategy. Some, such as seeds of *T. grandis*, required fire to destroy the thick, hard seed coat and stimulate germination. Without fire, Teak has a low rate of germination, thus the structure and composition of Teak forest changes. *P. macrocarpus* shed seeds over two phases. The first phase, in September to October, was influenced by heavy rainfall, since moisture was absorbed into the seed allowing the germination process to begin. The second phase was during February, when strong winds assisted distribution from the parent tree. Even after accidental fire the seed could still germinate, since the seed coat had been burnt away and the rainy season closely followed. The seeds of *Largerstroemia duperreana*, *Afzelia xylocarpa* and *X. xylocarpa* var. *kerrii* also needed moisture for germination. The absorption of water helped destroy the thick seed coat of the woody seeds. In contrast, *Irvingia malayana* shed its seed at the end of rainy season, possibly to avoid fire. It also needed large animals to spread its seed. These results suggest that the annual variation in seed production helps maintain the structure of the forest ecosystem. Furthermore, differences in the period of seed fall may reduce the probability of attack by biotic disturbances (herbivores and epidemic pathogens) and by abiotic disturbance (fire), and may increase the chance of finding suitable environmental conditions for germination and establishment.

Seed production within the three areas occurred over two phases. During the first period, from January to April, seed production in the mixed-species plantations was higher than in either the ecotone or natural forest, suggesting that the production of Teak seeds in the plantation affected seed density. In contrast the second phase occurred from April until September, and density was higher in the natural forest than in the multi-purpose species plantation. This can be explained by the diversity of tree species in the natural forest.

Figures 4.2 and 4.3 indicate that seed dispersal over one year correlated with the number and density recorded in the soil seed bank. This coincides with the report by

Guevara and Gomez-Pompa (1976) who reported that seasonal and annual variation of seed production was a factor influencing seed bank abundances and species richness in tropical forest. The soil seed bank was highest in the rainy season (during the period of growth) and decreased towards the end of the year. This suggests that some of the seeds germinated during the growing season, and that others were lost to seed predators or were consumed by pathogens or micro-organisms. This would have decreased the abundance of seeds, which were otherwise dormant during the dry season.

4.2.2 Ecology of seeds

<u>Results</u>

Throughout the observation period the number of dispersed seed species, in terms of both seed dispersal and soil seed bank, was lowest in the mixed-species plantations (Table 4.9). The number of seed species in the plantation excluded *Gmelina arborea, Tamarindus indica* and *Anacardium occidentale*, which were planted as multi-purpose species and were non-native species of the natural stand. The number of seed species clearly showed a gradual decrease from the natural forest, to the ecotone and the plantation. Average density for all native woody species in natural forest, ecotone and mixed deciduous forest with Teak (MDF) is presented in Table 4.9. The average density of dispersed seeds in MDF was more than twice the average density in the multi-species plantation. The mean density of dispersed native woody seeds was significantly different between MDF and the multi-purpose species plantation (MP) at the 0.05 level. There were no statistically significant differences for average densities of native woody seeds in the soil seed bank of the three areas, though the number of species in MDF was more than three times the value in the multi-species plantation (MP). Natural mixed deciduous forest with Teak (MDF) showed high values for both seed dispersal and soil seed bank.

Among the 20 species of dispersed seeds of native woody trees in natural forest, *Vitex peduncuralis* (KSP) had the highest average density, followed by *Lannea coromandelica* (KUK) and *Grewia elatostemoides* (PKT), respectively (Figure 4.5). Dispersed seeds of dominant canopy tree species such as *Tectona grandis* (TEAK), *Pterocarpus macrocarpus* (P) and *Albizia lebbeck* (TUT) were found in the multi-species plantation, as well as common species such as, *Morinda coreia* (YOPA) and *Croton oblongifolius* (PYAI). The abundance of *P. macrocarpus* and *A. lebbek* did not differ between MDF and MP (Figure 4.5). In the ecotone zone, Teak had the highest density, but other dominant canopy tree species such as *L. duperreana* (TB), *P. macrocarpus*, *Spondias pinnata* (MKO), *Chukrasia velutina* (YHI2) and *Vitex peduncularis* were also found. Seeds of Teak, the dominant canopy tree in the natural forest stand, were found in the multi-purpose species plantation at the highest density. These viable seeds may have come from trees in the plantation or may have arrived from adjacent forest and/or remnant trees. The seed density of *P. macrocarpus* in the plantation did not differ from natural and ecotone areas. However, when considering the variety of seed species in the multi-purpose species plantation it was apparent that it had a low diversity of native tree seeds.

The results of soil seed bank composition for woody tree species are shown in Figure 4.6. In natural forest, unidentified species 2 (MAKE) had the highest average density, followed by *L. duperrareana* (TB), *Lannea coromandilica* (KUK), *T. grandis* (TEAK), *Vitex peduncularis* (KSP) and *Canarium subulatum* (ML) respectively. Meanwhile *T. grandis* was the dominant seed species in the ecotone, followed by *Erythrina subumbrans* (TL), *Vitex peduncularis* (KSP) and *L. coromandelica* (KUK) respectively. In the multi-purpose species plantation *T. grandis* had the highest average density, whereas *P. macrocarpus* (P), *Terminalia calamansanai* (HAAN), *Morinda coreia* (YOPA) and *Bombax anceps* (NGIU) were found at a very low density.

As with the results for seed dispersal, the seeds of two dominant tree species in natural mixed deciduous forest, *T. grandis* and *P. macrocarpus* were found in the soil seed bank of the ecotone and mixed-species plantations. However, when considering average densities of Teak in the soil seed bank, it was found that the density of viable seeds in the multi-species plantations was higher than in the MDF (Figure 4.6) due to the addition of seed input from plantation Teak. However, the number of *P. macrocarpus* seeds that persisted in the soil was approximately equal for all three areas. Finally, the results of both seed dispersal and soil seed bank showed a low diversity of seed species in the multi-species plantations. Therefore, the multi-species plantations received limited woody seed distribution from adjacent natural forest and/or remnant trees. The seeds of some tree species, such as *T. grandis, P. macrocarpus* and *Albizia lebbeck* dispersed from adjacent forest and/or remnant trees to the multi-species plantation, though in total, 80% of the native tree seed species did not migrate to the multi-species plantations. These results were strongly confirmed by correspondence analysis (CA) (Figure 4.7). CA was used to analyse the presence or absence of plant species within the three study sites. A few native tree seeds persisted in the soils of the muti-species plantations, such as *T. grandis*, *P. macrocarpus* and *Bombax anceps* (Figure 4.6). No viable seeds of other dominant tree species, such as *Afzelia xylocarpa, Lagerstroemia duperreana, Nephelium hypoleucum, Canarium subulatum, Schleichera oleosa*, *Irvingia malayana* and *Albizia lebbeck*, which were found in MDF, persisted in MP. Correspondence analysis again firmly supported this observation (Figure 4.8).

Table 4.9: Number of native woody species and average density (±SD, seed/ha) for seed dispersal and soil seed bank (at a depth 0-5 cm) for the three areas; natural mixed deciduous forest with teak (MDF), ecotone (ECO) and multi-purpose species plantation (MP), from November 2001-October 2002.

Characteristics	**MP**	**ECO**	**MDF**
No. of dispersed seed species	9	17	24
No. of species in soil seed bank	9	19	2
Average density of dispersed seed	17,037.04 ± 1,981.04 [a]	20,434.21 ± 2,276.36 [a]	36,182.32 ± 10,082.08 [b]
Average density in seed bank	10,856.41 ± 2,124.96 [a]	13,295.26 ± 1,501.89 [a]	13,677.42 ± 1675.33 [a]

Note: The non-native species *Gmelina arborea, Tamarindus indica* and *Anacardium occidentale* are excluded.

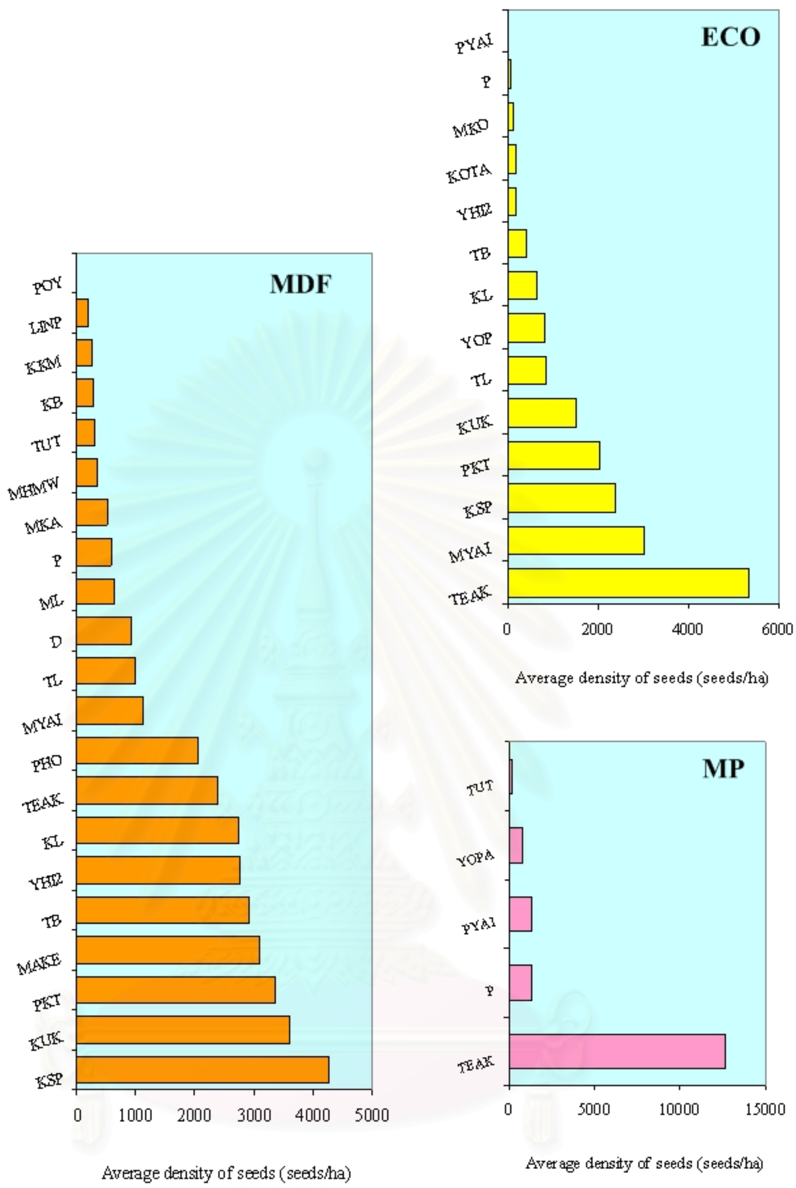

Figure 4.5: Average density of dispersed seed of native woody trees in natural mixed deciduous forest with Teak (MDF), ecotone (ECO) and multi-purpose species plantation (MP).

Key:	KSP - *Vitex peduncularis*	KUK – *Lannea coromandelica*
	PKT – *Grewia elatostermoides*	TB – *Largerstoremia duperreana*
	TEAK – *Tectona grandis*	P – *Pterocarpus macrocarpus*

Note: A full list of abbreviations is presented in appendix A. 8.

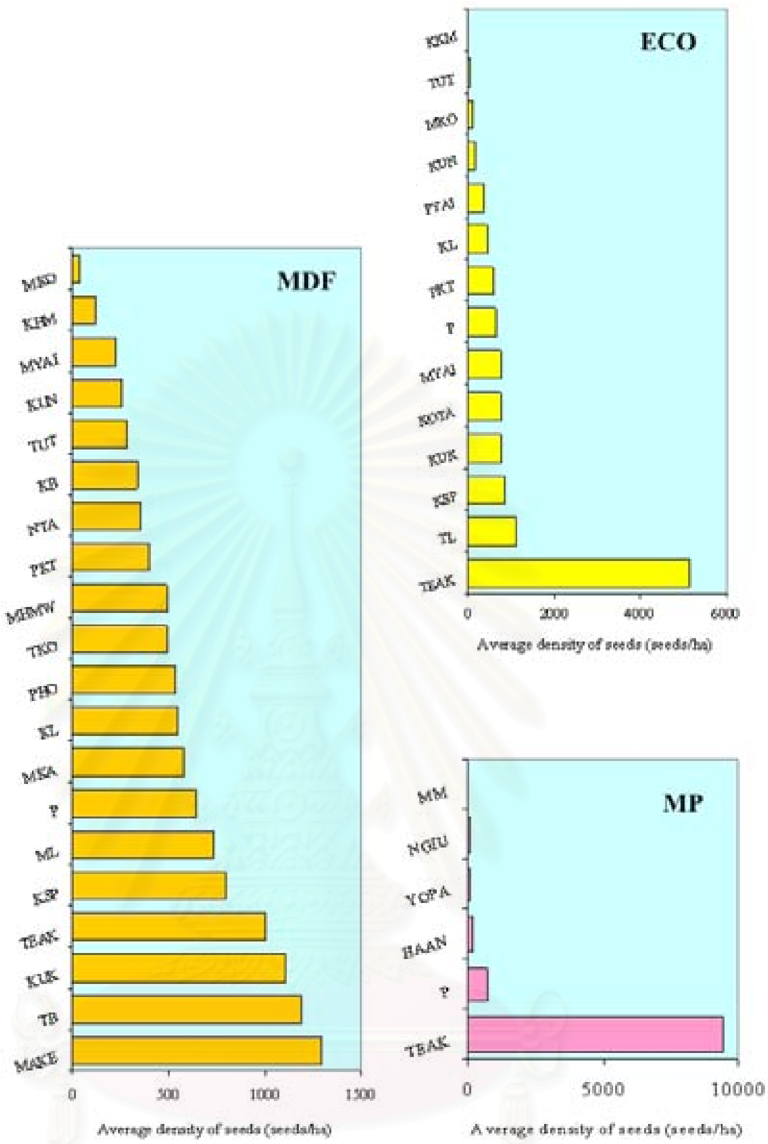

Figure 4.6: Average density of soil seed bank for some native tree species in natural mixed deciduous forest with teak (MDF), ecotone (ECO) and multi-purpose species plantation (MP) at a soil depth of 0-5 cm from November 2001-October 2002.

Key:	TB – *Largerstoremia duperreana*	TEAK – *Tectona grandis*
	P – *Pterocarpus macrocarpus*	MAKE – unidentified sp 2
	KUK – *Lannea coromandelica*	TL – *Erythrina subumbrans*
	KSP – *Vitex peduncularis*	HAAN – *Terminalia calamansanai*

Note: A full list of abbreviations is presented in appendix A. 8.

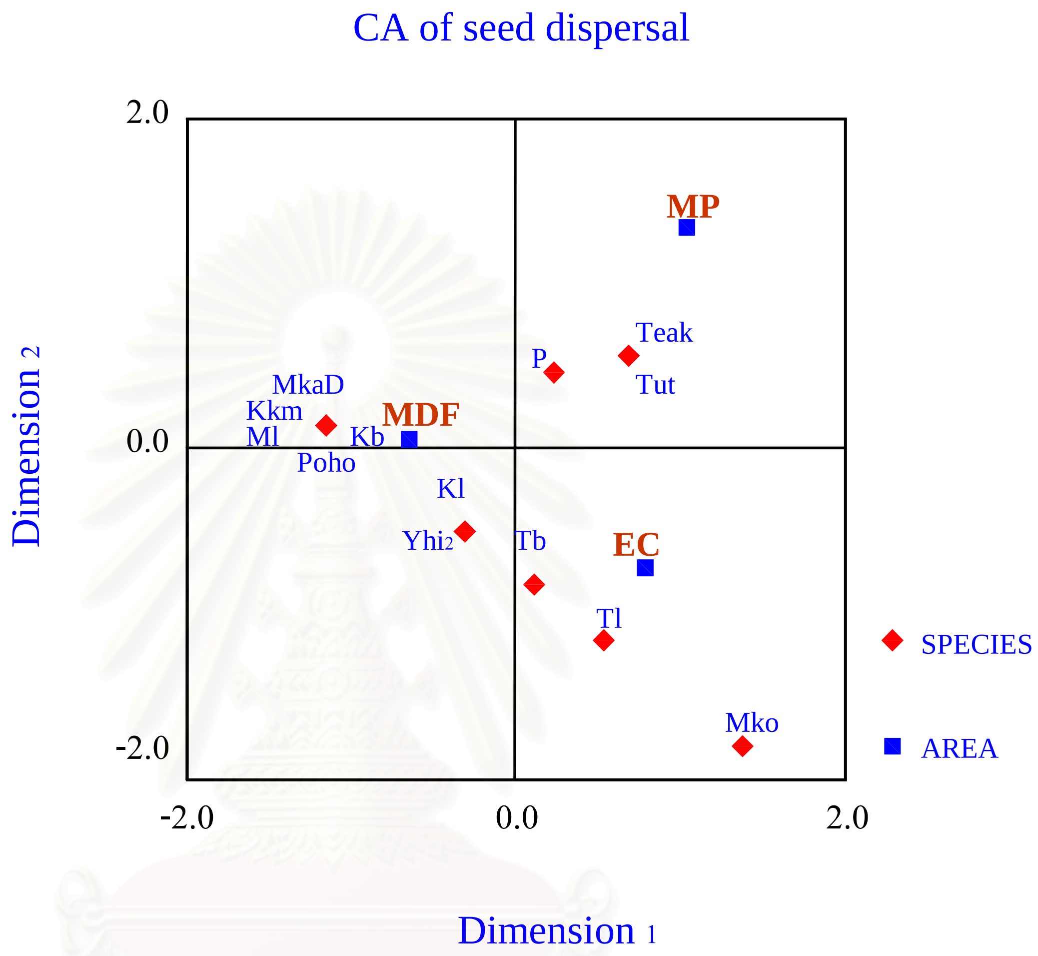

Figure 4.7: Correspondence analysis on seed dispersal of some dominant tree species in three communities; natural mixed deciduous forest with Teak (MDF), ecotone (ECO) and multi-purpose species plantation (MP).

KEY:

Teak - *Tectona grandis*	KL - *Nephelium hypoleucum*
Pohoo - *Sterculia villosa*	P - *Pterocarpus macrocarpus*
ML - *Canarium subulatum*	Mka - *Afzelia xylocarpa*
Kkm -*Albizia odoratissima*	TB - *Lagerstroemia duperreana*
Yhi 2 - *Chukrasia velutina*	Tut - *Albizzia lebbeck*
KB - *Irvingia malayana*	TL - *Erythrina subumbrans*
D - *Xylia xylocarpa*	Mko - *Spondias pinnata*

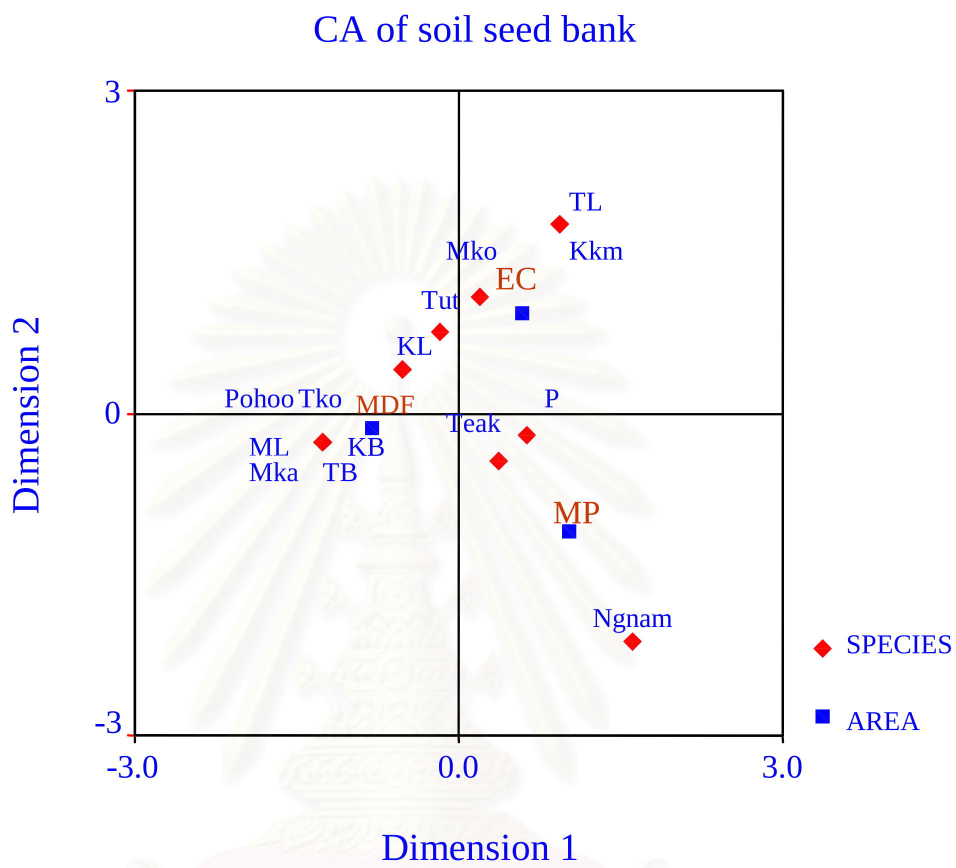

Figure 4.8: Correspondence analysis on soil seed bank of some dominant tree species in three communities; natural mixed deciduous forest with Teak (MDF), ecotone (ECO) and multi-purpose species plantation (MP).

KEY:		
	Teak - *Tectona grandis*	KL - *Nephelium hypoleucum*
	Pohoo - *Sterculia villosa*	P- *Pterocarpus macrocarpus*
	ML - *Canarium subulatum*	Mka - *Afzelia xylocarpa*
	Kkm - *Albizia odoratissima*	TB - *Lagerstroemia duperreana*
	Tut - *Albizzia lebbeck*	KB - *Irvingia malayana*
	TL - *Erythrina subumbrans*	Mko - *Spondias pinnata*
	Tko - *Schleichera oleosa*	Ngnam - *Bombax anceps*

Discussion

Ecological dynamics of seed dispersal and the seed bank

After disturbance, forests regenerate through interactions of propagules with environmental factors i.e. soil and climatic conditions (Kozlowski, 1971). Regeneration of forest stands may occur from seeds recently dispersed from their parent trees or from seeds stored in the soil. Both wind and animals play variable roles in establishing the seed banks in disturbed forest stands. In tropical forests, wind is more important for the dispersal of liana or woody climber seeds than it is for trees. The distribution of seeds by wind also occurs more commonly in dry tropical forests than in wet tropical forests (Leck, 1995). The majority of wind-dispersed seeds in forests are those of canopy trees (Keay, 1957). The present study indicated that animals, as well as wind, influence seed colonization. However, the full potential of seed banks to regenerate disturbed forests is never realized due to the massive losses of seeds and young seedlings. For example, seed losses have been attributed to seed aging, failure of seeds to germinate, pathogens, seed redispersal and seed and seedling mortality (Leck, 1995). The risk of mortality for forest vegetation is greatest during the early stages of germination. Many seeds fail to germinate due to extended dormancy or unfavourable environmental conditions. Mortality of seedlings in the cotyledon stage is particularly high due to their low reserves of carbohydrates and mineral nutrients, when even mild environmental stress often leads to seedling mortality. Many studies, from different areas, have revealed several species that face this situation; e.g. *Psedutsuga menziesii* (Douglas Fir), *Eucalyptus spp*., *Acer saccharum* (Sugar Maple) and *Pinus thunbergii* (Conifer) (Tappeiner and Helms, 1971; Hett and Loucks, 1971; Tazaki, Tshihara and Usijima, 1980).

In the multi-purpose species plantations, results showed that most of the native seedlings originated primarily from asexual reproduction (preliminary observation), and that recruitment from the current year’s dispersed seeds and seed bank were minimal. This conclusion was supported by the low seed distribution to the multi-species plantations, as well as the lack of seedlings germinating from the seed input of the adjacent natural forest. The results of seed germination will be discussed in the next section. As reported by Cubina and Aide (2001) and Zimmerman, Pascarella and Aide (2000) there is a scarcity of seed input, especially of woody species, from adjacent forest to highly degraded and abandoned lands. Thus an understanding of seed dynamics is a basic requirement for the development of restoration strategies. There have already been several reports on the effects of seed availability on forest recovery and diversity (Weijdeven and Kuzee, 2000; Zimmerman, Pascarella and Aide, 2000; Holl et al., 2000; Webb and Peart, 2001), so this discussion will focus on a comparison of seed input by seed dispersal and soil seed bank between natural forest, ecotone and multi-purpose species plantations.

An understanding of the rates and spatial patterns of seed colonization has become increasingly recognized as crucial to an understanding of the dynamics and maintenance of species diversity and structure in mixed deciduous forests of northern Thailand. The seeds of *Canarium sabulatum*, *Irvingia malayana* and so on are large and heavy, therefore having limited dispersal. Some taxon (species and/or genera) need animal vectors for dispersal, such as *Antidesma bunius*, *Aporusa villosa*, *Gmelina arborea*, Diospyros, Memecylon, Canarium etc. (Elliott, Promkutkaew and Maxwell, 1994; Scott et al., 2000, Webb and Peart, 2001).

Forest regeneration in highly degraded areas or abandoned land relies on seeds dispersed from the neighbouring forest ecosystem and/or remnant trees, whilst seed dispersal and the seed bank (density and species) are related to reproductive ability of individuals, regularity of seed production, availability of dispersal agents and distance. However, much research has indicated that annual seed production varies among species. For example, Hardwick et al., (1997) commented that in northern Thailand, *Beilschmiedia* did not produce seeds annually, in contrast to *Prunus* and *Engelhardia* that did. Therefore, annual seed production affected forest regeneration. Moreover, Van Der Pijl, (1982), linked density and number of species to fruit morphology, since it affects the distance that seeds are dispersed. In theory, the mean dispersal distances of 34 wind-dispersed tropical trees ranged from 22 to 194 m (Augspurger, 1986). Results of our study showed that large woody tree seeds were found in very low numbers in the plantation compared to the numbers in natural forest and ecotone.

Seed distribution into multi-purpose plantation areas differed dramatically in both seed density and species composition to that in natural MDF forest. The results of this study showed that the number of species colonizing the multi-purpose species plantations was three times less than in natural forests and two times less than in the ecotone. The low average density and diversity of woody seeds in MP was due to low input. This may well have affected regeneration because it is recognized that seed input is an important source of recruitment in degraded sites besides sprouting. Due to limitations on seed distribution, natural recovery would therefore be slow. Thus a restoration strategy by planting enrichment seedlings may be a quicker and more suitable way of encouraging succession. Factors that influence limitations on seed colonization and abundance in multi-purpose species plantations are seed weight, lack of dispersal vectors, germination, predators and so on. Some species, with large colourful fruit, may attract seed predators as well as seed dispersers. Janzen (1986) explained that large seeds are more likely to be found and consumed by seed predators than small seeds. Research on modes of dispersal has indicated that the majority of tropical tree seeds are dispersed by animal vectors rather than wind, water or other forms of dispersal (Wunderle, 1997). In temperate and neo-tropical forests, seed dispersal by animals predominates over wind dispersal during the early to mid-stages of succession. Others have noted similar trends, in that wind dispersal is ineffective in the forest understorey and that, unsurprisingly, a large proportion of understorey plants disperse their seeds via animal vectors (Wilson, 1986; Foster, Arce and Wachter, 1986). In northern Thailand animal vectors include bats, gibbons, elephants and frugivorous birds. In particular, the larger frugivores have the potential capability of distributing large seeds away from adjacent forest. However, large seed dispersers may be rare in diversified mixed-species plantations since they are unlikely to move from natural habitat to more open areas. In fact, most move higher up the mountain slopes to find food, avoid forest fire (in the dry season) and evade hunters. There are many reasons therefore to explain the limited seed distribution to mixed plantations.

Nepstad et al. (1996) and Hammond (1995) found factors that influenced the lack of seeds in abandoned or degraded areas were low numbers of seed vectors, as well as seed predators. Howe and Smallwood (1982) discovered that most tropical shrub and tree seed dispersers were frugivorous animals and that many avoided large open areas that could expose them to predators, therefore limiting the increase in floristic diversity, especially that of trees, which were a major component of the forest structure. Therefore, techniques such as enrichment planting may be useful in helping to increase species diversity in degraded areas (Wunderle, 1997; Holl et al., 2000).

Animal seed dispersers are important vectors in the spatial dynamics of tropical woody species, especially in wet tropical forest communities. Therefore, the attractiveness of the site to seed dispersers influences natural regeneration. Such structural complexity of vegetation is known to affect the diversity of animals and has been demonstrated to be an important factor in attracting avian seed dispersers. Reeders (1985) showed that bird recolonization of bauxite mines in northern Australia was associated with the existing vegetative structure of the site and that islands of vegetation left undisturbed on the site facilitated bird invasions. Increased avian diversity associated with vegetation complexity was expected to result in increased seed input to the site. Meanwhile, fruiting as an attractant to dispersers, is another trait of many study sites. It is evident that frugivores are especially responsive to spatial and temporal changes in fruit abundance at a variety of levels (Blake and Loiselle, 1991). Similar research into avian dispersers was conducted by Silva, Uhl and Murray, (1996) in an abandoned pasture in the eastern Amazon. They reported that three species of bird were more common during the peak period of fruiting than during the dry season when fruit was absent. Seed dispersers were observed feeding on the fleshy fruits of *Cordia multispictata*. Likewise, Guimaraes Vieira, Uhl and Nepstad (1994) found that the rain of woody seeds dispersed by birds and bats was much greater near *Cordia* than in the open patches of grassland. The seed rain also displayed distinct seasonality, a pattern that corresponded to the fruiting phenology of *Cordia* and the abundance of avian seed dispersers visiting. The ability of a plantation to attract animal seed dispersers is interesting. Wunderle (1997) suggested that the composition of a tree plantation makes certain types of plantation better suited than others to attracting animal seed dispersers. This is due to the characteristics of the trees and their diverse structures. For example, Indian Teak plantations, which lack nectar and fruit resources, had a scarceness of bird species requiring these resources and were visited only by flocks of wandering insectivorous birds (Beehler, Krishna Raju and Ali, 1987). When these resources are absent, many plantations become unattractive to wildlife, which can then retard or even prevent succession. However, some exotic tree species have qualities that attract animal dispersers, as shown by observations of plantations containing variously aged (1, 3, 5 and 7 years) *Albizia falcataria* trees in Sabah, East Malaysia (Mitra and Sheldon, 1993). They demonstrated that 64% of the 162 bird species recorded in the nearby primary forest were recorded in the plantations. They associated the attractiveness of their *Albizia* plantations to four factors:

1. The rapid growth and thin canopy provided the space and light needed for the development of a substantial secondary forest with appropriate food resources.
2. The *Albizia* trees were infested with caterpillars, which attracted birds (including many frugivores).
3. The plantation was adjacent to primary forest and areas of active logging, so had a ready source of birds.
4. The plantation was young, and therefore, birds displaced by prior logging were still alive, and since there was a structured microhabitat in the plantation (stumps and logs) the birds visited.

Finally, they concluded that the attractiveness of plantations to forest species depended upon many factors including the type, age and mixture of cultivated trees, crop rotation, proximity of primary forest, plantation physiognomy (streams, cliffs etc.) and timing and complexity of pest infestations (Mitra and Sheldon, 1993). Unfortunately, our study did not investigate seed dispersal vectors within the five plantation types, though a survey of regenerating plant species showed that the mixed plantations had greater diversity than the pure stand. It can be assumed that the mixed plantations will attract animal dispersers better than the pure Teak stand as supported by the many arguments presented above.

Results on seed density within the three areas indicated that only a few woody species moved to the diversifed plantation, e.g. *Bombax anceps*, *Terminaria calamansanai* and *Pterocarpus macrocarpus* were found in the soil seed bank and *Albizia lebbeck*, *P. macrocarpus* and *Croton roxburghii* were found generally dispersed. Since these species had colonized the degraded area this suggested that most were wind dispersers and that their seeds were small, hence possessing greater seed mobility. Nearly fifty percent of the dispersed seed species required animal dispersers. Much research has indicated a similar proportion. For example, more than 66% of the canopy tree species in neotropical forests and 46% in tropical Asia and Africa have seeds which are dispersed by animals (Howe and Smallwood, 1982). Wunderle (1997) also reported that over half of the tree species had seeds dispersed by animals rather than other forms. Therefore, the lack of animals in highly degraded areas is main reason for the low number of tree seed species in MP. In addition, the multi-purpose species plantation had a less complex canopy, thus being less attractive to frugivorous animals that may have otherwise helped distribute woody seeds to the area. Wijdeven and Kuzee (2000) also observed problems in seed distribution and only found dispersed seed within 20 metres of the forest edge. Away from the edge, colonization was almost absent. Likewise, Cubina and Aide (2001) found that few woody seeds (shrubs or trees) dispersed into pasture even when a rare dispersal event occurred. Therefore, the abundance and diversity of woody tree seeds in the seed rain and seed bank is strongly affected by distance from the forest edge. At Mae Moh plantation station the natural forest was not far away, just 50 metres from the mixed plantation. The low density and diversity of woody seed species in the multi-purpose species plantations clearly retarded natural recovery processes. Meanwhile, the high demand for Teak and timber product is still increasing, and the pathway to a climax community by natural succession is still not fully understood. It is clear that enrichment planting of primary plant species will boost the progression towards MDF with Teak in both structure and composition. Data on seed density indicated that some dominant species within the mixed deciduous Teak forest, such as *Afzelia xylocarpa*, had infrequent seed production, thus their potential to regenerate in both natural and multi-purpose species plantations was limited. Therefore, dominant pre-climax species with low seed production should be planted in multi-purpose species plantations. This should be a consideration of any restoration strategy.

Research has also investigated the effect of isolation on seed dispersal in natural forest and degraded areas. Colonization of primary seeds may be limited due to the long distances from potential seed sources. A study of isolated woodlots in Europe indicated that plants with animal-dispersed seeds were negatively affected by woodlot isolation (Van Ruremonde and Kalkhoven, 1991). Silva, Uhl and Murray (1996) observed bird movements from secondary growth forest into pastures. They found that the maximum distance moved by three frugivorous species into the pasture varied from 2 to 254 m, but that most movements fell between 1 and 80 m. The birds spent from 0.5 to 23 min in the pastures before returning to the forest. Thus, the resulting seed shadow in the abandoned pasture exhibited two characteristics, a general decrease in seed density from the source plant and a localized increase in seed density near existing shrubs/small trees in the pasture, which was closely linked to the perching and defecating behaviour of the birds. The immobility of large seeds is a major factor that regulates the regeneration processes. Large seeds are more likely to be dispersed shorter distances than small seeds. Therefore, they are expected to have a lower rate or likelihood of colonizing sites (Wunderle, 1997). Several studies have revealed limited dispersal of large seeds. For example, a lower mobility of large seeds was evident in a 10-year-old restoration plot in the Brazilian Amazon, where large-seeded trees were slow to arrive. The average size of seeds for species that naturally dispersed to the site was significantly smaller than the seeds of tree species absent from the plot but found in the surrounding primary forest (Parrotta, Knowles and Wunderke, 1997). In addition, large seeded species tend to be rare relative to small-seeded species in seed banks (McClanahan and Wolfe, 1993). Due to limitations in large-seed dispersal, Wunderle (1997) has suggested that many restoration sites will suffer retarded or even arrested successional development, and that they will be overwhelmingly composed of small-seeded pioneer species.

More woody species seeds were found in the ecotone than in MP. This can be explained by spatial distribution. The distance from the natural forest to the ridge of ECO was very close and the transitional topography between the two areas was sloping. Hence, more seeds and a greater diversity of species could easily colonize the area by wind and animal dispersers. Secondary dispersal agents were gravity and floods. It was noted that no tree seeds from MP were found in the ecotone other than Teak, which probably dispersed from both natural forest and MP. Woody seed species found in the ecotone were similar to that in MDF. No tree seeds from the multiple plantation occurred in MDF, indicating that plant species of this area, especially woody plants, invaded and/or colonized from natural forest through the ecotone to the multi-purpose species plantation via seed dispersal. This suggested that after regeneration by remaining plant resources i.e. roots or stumps in the mixed plantation, seeds from the natural forest were a secondary resource for vegetative recovery in this highly degraded area, confirming that adjacent and remnant trees have an effect on regeneration in degraded areas. Moreover, these results coincide with other studies and clearly demonstrate that the distance between the adjacent forest ecosystem and the highly degraded area strongly influences the recovery process. This suggests that before developing a restoration strategy a survey on seed distribution and the effects of nearby natural forest should be undertaken.

Garwood (1983) reported that seeds of tree species in tropical forests were often seasonally dormant. *Albizia* seeds and *Hovea* seeds may remain viable for over 100 years (Osborne, 1980). On the other hand, the reported of longevity of seeds in tropical forest by Kozlowski and Pallardy (1997) revealed that many seed species were very short-lived. Nevertheless, under natural conditions the persistence of seeds in the soil is a strategy that increases the recruitment rate under ephemeral favourable conditions. Because the seeds of most species of woody plants exhibit some degree of dormancy they may not germinate promptly under ostensibly favourable environmental conditions. Long postponement of seed germination often accounts for failure of adequate regeneration of forest ecosystems. Seed may be dormant for a variety of reasons (Kozlowski, 2002);

1. Seed immaturity.
2. Seed coat impermeability to water and oxygen.
3. Seed coat resistance to embryo growth.
4. Metabolic blocks in the embryo.
5. Various combinations of these.

Seed dormancy is most commonly physiological, and involves the failure of morphologically mature embryos germinating. Embryo dormancy appears to be regulated by complex interactions of hormonal and other internal factors in the embryo and surrounding tissues. These interactions are influenced by availability of light, water, and suitable temperature (Battaglia, 1989). Physiological embryo dormancy often cannot be reversed by the same environmental conditions that induced it. In some species, the failure of seeds to germinate may result from restricted embryonic expansion due to a thick pericarp, and/or physiological embryonic dormancy. In natural mixed deciduous forest with Teak, seed dormancy in species such as *Afzelia xylocarpa*, *Irvingia malayana* and *Tectona grandis* has been attributed to their hard seed coats. In another study, the dormancy of *Quercus nigra* acorns was caused by the mechanical strength of the pericarp, chemical inhibition of embryonic growth by the pericarp and slow absorption of water (Peterson, 1983).

Some dominant native tree species such as *T. grandis*, *P. macrocarpus* and *Bombax anceps* persisted in the multi-purpose species plantation, but overall there was a low diversity of tree seed species. Therefore, this would impede natural succession. In addition, research by Whitmore (1983) and Garwood (1989) indicated that tree seed composition in the soil seed bank was mainly limited to pioneer species, due to their long viability and dormancy. In this study, no significant differences in the soil seed bank were noted between the three areas. In the natural forest, seeds were lost to germinating seedlings, predators and so on at twice the rate of seed input. The ratio of inactive seeds to seed input, at the end of season, increased from MDF, ECO, to MP respectively (Table 4.9). This indicated that some of the seeds in the natural forest had germinated or been lost to predators and pathogens, whilst some of those in ECO and MP had become dormant. The majority of the seeds were Teak, which had a hard seed coat and needed external factors such as fire to stimulate germination. Therefore, the seeds in MP and ECO had become inactive and part of the soil seed bank. This illustrates that regeneration by seeding under natural conditions is a slow process, because the soil seed bank waits for suitable environmental conditions before germinating. However, this phase of dormancy is a strategy that avoids hazardous conditions, and is therefore an important source for floristic regeneration. Combined with this period of dormancy, selective logging and deforestation contribute severe problems to the Teak forest ecosystem. This is another reason to use restoration strategies to accelerate natural successional processes.

Accumulation of seeds in seed banks varies appreciably depending on the size of seeds and regularity of seed production by different species. For example, Marod et al. (2002) reported that seed production in natural mixed deciduous forest in Thailand fluctuated considerably from year to year and that patterns varied among species. This variation may have strong influences on the recruitment of plant populations (Crawley and Long, 1995). For example, *Xylia xylocarpa* var. *kerii* could produce seeds twice a year (November-December and February-March), though its period for shedding seeds extended from the late dry season through the rainy season, whereas the majority of other species shed their seeds in one short period (Marod et al., 2002). Therefore, the abundance of dispersing seeds directly affects the abundance in the seed bank. In Alaska, *Betula papyifera* produces large crops of seeds once every 4 years, and *Picea glauca* once every 10-12 years (Perry, 1995).

This study found the number of seed species in the soil seed bank to be almost equal to the seed dispersal seen in the three zones, though the number of species was lowest in the multi-species plantation, due to limited distribution mechanisms. The distribution of inactive seeds in the soil differs from tropical to temperate areas. Thus, Deiller, Walter and Tremolieres (2003), studying temperate hardwood forest of the floodplains in France, reported that most species present were absent from the seed bank, both qualitatively and quantitatively. Thus, the fact that no seeds were stored in the soil indicated that most of the hardwood species of the area were unable to build-up a persistent seed bank. This differs from other studies conducted in temperate areas. Several indicated that the cold winter season was unfavourable for the germination process. Therefore, the seed bank was an important strategy to improve germination rates, in contrast to tropical areas. However, in temperate hardwood forest, many woody species did build transient seed banks, e.g. *Fraxinus excelsior* (Ash), *Acer campestre*, *Acer pseudoplatanus*, *Corylus betulus*, *Corylus avellana* (Hazel) and *Quercus robus* (Pedunculate Oak). Brown and Oosterhuis, (1981), state that shade-tolerant species do not need to develop seed dormancy mechanisms.

From a genetic perspective, the aim of restoration efforts should be the creation a dynamic and expanding forest resource, which has the capacity to evolve in the future and respond to environmental change. Therefore, genetic variety must accommodate the potential for plants to tolerate hazardous environments. The results of seed distribution were low for both density and diversity. Thus, plant species in the understorey of multi-species plantations are extremely vulnerable to extinction. This research clearly showed that only a few woody species colonized highly degraded areas; it found only *T. grandis* (T), *Erythrina subumbrans* (Tl) and *Spondias pinnata* (Mko) in the form of dispersed seed, and *T. grandis* (T), *Pterocarpus macrocarpus* (P) and *Bombax anceps* (Ngnam) in the soil seed bank. These species established at a low density in the plantations (Table 4.3). Dominant species in the five plantation types were not found in the form of distributed seed, indicating that the important resource for plant regeneration in this highly degraded area was resprouting from stumps or roots. A secondary resource was from natural forest and remnant tree seed dispersal, as well as from the soil seed bank, which still held inactive seeds from the previous year. For example, the regeneration of *Albizia lebbeck* and *Pterocarpus macrocarpus* was not limited by seed availability. Seeds of these species were dispersed to the multi-species plantations and seedlings were found. However, regeneration of other species was limited by low seeds dispersal, though this study showed high species diversity and species composition of regenerative plants. More than 60% were native species found in natural primary forest ecosystems. Due to the low seed input to the plots, vegetative mechanisms tended to dominate the regeneration of woody species. However, sexual reproduction still affected the regeneration processes of a few species, but was limited to those that were able to release large numbers of seeds, thereby increasing their chances of germinating (i.e. *Albizia lebbeck*, *Pterocarpus macrocarpus*, *Bombax anceps*, *Morinda coreia* and *Croton oblongifolius*). The low rate of sexual reproduction may well affect the genetic pool. Since there is little genetic exchange within species, adaptive variation within the populations will decline, leaving species vulnerable in the future. Currently, the number of seeds that have managed to migrate to the multi-purpose species plantations is very low, since the parent trees are isolated, relict and also low in density. Genotypic variation is therefore low. For these reasons, this study suggests that one of the tasks in restoration should be to find ways to increase gene flow within populations, perhaps by physically moving seeds or by enrichment planting. This would achieve successful restoration, reduce the risk of low genetic variety and enhance the ability to adapt in the future.

4.2.3 Seed Germination

Results

Seeds were collected in March-April and June–July and tested for germination viability in a nursery (Table 4.10 and Table 4.11). These months were the peak of seed production (Figures 4.2 and 4.3). Statistical analysis was not performed. The dispersed seeds of eighteen woody species were planted in the nursery, fourteen of which germinated. Thirteen species found in the soil seed bank were also tested for percentage germination, two of which did not germinate. There was no correlation between percentage germination and site of origin. Germination of seeds from the soil seed bank ranged from 0-67%. However, there were temporal variations and differences in germination rates within individual species for both dispersed seeds and the soil seed bank. Percentage germination of dispersed seeds ranged from 0 to 100% (Table 4.10). There was variation within species between the three areas, though data remained unclear because several species were not found in each zone.

For dispersed seed, there was variation among species for germination rate. Some native tree species such as *Tectona grandis, Lagerstroemia duperreana, Vitex peduncularis* and *Nephelium hypoleucum* had germination rates lower than 50%. A few of the dominant tree species such as *Xylia xylocarpa* var. *kerrii* and *Afzelia xylocarpa* had a higher value (Table 4.10). Germination rates from the soil seed bank were low ($<$ 50%) for dominant tree species such as *Tectona grandis*, *Lagerstroemia duperreana*, *Nephelium hypoleucum* and *Lannea coromandelica* (Table 4.11).

Discussion

Germination and regeneration

The germination of seeds involves reestablishment of embryo growth and seed coat rupture, followed by seedling emergence. During the germination process the radicle typically elongates first and enters the soil. The thin seed leaves (cotyledons) of most plant species emerge above the ground (epigeous germination). Epigeous cotyledons are important largely as photosynthetic organs and sometimes also for food storage when environmental factors are unfavourable. Even so, their storage function varies appreciably among species. Dormancy is an important mechanism in the process of natural regeneration.

Table 4.10: Percentage nursery germination of dispersed native woody plant seeds.

Scientific name	Family	Percentage of germination for seed dispersal		
		MDF	ECO	MP
Tectona grandis	Labiatae	6.67	0.00	38.00
Xylia xylocarpa	Minosaceae	93.33	ND	ND
Pterospermum acerifolium	Sterculiaceae	33.33	ND	ND
Lannea coromandelica	Anacardiaceae	42.00	51.92	ND
Afzelia xylocarpa	Leguminosae	100.00 *	ND	ND
Lagerstroemia duperreana	Lythraceae	32.00	ND	ND
Oroxylum indicum	Bignoniaceae	100.00	ND	ND
Buchanania latifolia	Anacardiaceae	43.75	ND	ND
Erythrina subumbrans	Leguminosae	75.00	50.00	ND
Phyllanthus cf. orientalis	Euphorbiaceae	90.00	43.00	ND
Vitex peduncularis	Labiatae	24.00	0.00	ND
Nephelium hypoleucum	Sapindaceae	20.00	50.00	ND
Grewia elatostemoides	Tiliaceae	6.00	0.00	ND
Morinda coreia	Rubiaceae	ND	0.00	0.00
Antidesma ghaesembilla	Stilaginaceae	ND	0.00	ND
Croton longissimus	Leguminosae	ND	ND	86.31
Unidentified sp. 2	?	0.00	ND	ND
Unidentified sp. 9	?	0.00	ND	ND

Note: ND: No data, * Trimmed seed coat

Table 4.11: Percentage nursery germination of native woody plant seeds from soil seed bank.

Scientific name	Family	Percentage of germination for seed bank		
		MDF	ECO	MP
Tectona grandis	Labiatae	45.00	19.64	30.83
Lagerstroemia duperreana	Lythraceae	44.00	ND	ND
Nephelium hypoleucum	Sapindaceae	23.07	20.00	ND
Lannea coromandelica	Anacardiaceae	1.39	0.00	ND
Buchanania latifolia	Anacardiaceae	26.67	ND	ND
Erythrina subumbrans	Leguminosae	ND	55.00	ND
Croton oblongifolius	Leguminosae	ND	72.72	ND
Pterospermum acerifolium	Sterculiaceae	48.00	ND	ND
Grewia elatostemoides	Tiliaceae	0.00	0.00	ND
Phyllanthus orientalis	Euphorbiaceae	ND	77.00	ND
Antidesma ghaesembilla	Stilaginaceae	ND	0.00	ND
Croton longissimus	Leguminosae	ND	ND	66.67
Unidentified sp. 9	?	8.33	ND	ND

Note: ND: No data

Results clearly indicated that the seeds of woody plants had a low germination rate for both dispersed seed and for that in the soil seed bank. Germination rates showed spatial differences among species, and also between different species and location. This suggested that variation in seed germination affected establishment and persistence in each ecosystem. A low germination rate could therefore inhibit natural regeneration and/or succession. Some species, especially woody trees that cannot regenerate by themselves, may require external factors to increase numbers and diversity in early and intermediate successional communities. The seeds of pioneer or early successional species can germinate quickly, in contrast to late successional and climax species which germinate later, are long-lived and have slower growing characteristics (Uhl and Jordan, 1984). Several factors are linked to seed germination in natural environments. For example, tolerance of environmental extremes is linked to low seed moisture (Mayer and Poljakoff-Mayber, 1975), and small seeds are at greater risk of predation than large seeds (Osunkoya, 1994). Animals that defecate or regurgitate seeds can sometimes affect the probability of germination. Scarification in an animal's gut contributes to enhanced germination. Seeds may actually rot if not processed by animal dispersers or fail to break dormancy (Noble, 1975; Fleming and Heithaus, 1981). Since many species cannot germinate under natural conditions, restoration by seeding may not be successful, so the use of enrichment planting techniques may be the best restoration strategy.

Results showed that seeds from the current or previous years were found in the soil seed bank and that they could germinate under nursery conditions, though at a low rate. Only three woody species had germination rates of over fifty percent (*Croton oblongifolius*, *Phyllanthus orientalis* and *Croton logissimus*). However, other seed species that did not germinate may simply have been inactive, yet still viable, thus suggesting that the soil seed bank may encourage regeneration processes in the next growing season. Seeds will remain dormant if environmental conditions are unfavourable, and can therefore help maintain the structure of the forest community. However, limitations on seed colonization are still a major problem for natural succession in mixed and pure-species plantations. Furthermore, seed viability under natural conditions may be a factor that retards regeneration. A few woody species have seeds with prolonged viability in the natural environment, though much research has indicated that the seeds of pioneer species are usually better equipped for dormancy than pre-climax species. However, pre-climax trees on the most degraded of sites may adopt relative dormancy under unfavourable environmental conditions, thus affecting regeneration of the area. For example, Marod et al. (2002) reported that *Pterocarpus macrocarpus* and *Berrya ammonolla* are two species that exhibit a seed bank strategy in mixed deciduous forest in western Thailand. These species store viable seeds in the soil seed bank, but also keep many in their canopies. The viable seeds can be disseminated throughout the year, but germinate best in high temperatures of about 40-45^{o}C (Smitinand et al., 1975). This characteristic may be an adaptation that enables them to make use of light gaps and reach open areas that have been created by fire disturbance (Zammit and Westoby, 1988). It has been suggested that *Berrya* forms a persistent soil seed bank, because after disturbance by fire, many seedlings emerge, even though seed fall was not observed prior to the fire. The seed bank would therefore have been formed during a previously good seed mast year.

Most of the woody species collected from dispersed seeds and the seed bank were pre-climax and climax species. Based on ecological concepts, the initial stages of succession are normally favourable to pioneers since they have a wide range of tolerance and are able to occupy niches in degraded or hazardous environments. Pre-climax or climax species have a narrower range of tolerance. Thus, under natural conditions most of them have difficulty in germinating and establishing in early stages of succession when limiting factors such as low soil nutrients, soil compaction, low moisture and so on are prevalent. According to the theory of competitive exclusion, pioneer species exert more competitive pressure on pre-climax and climax species, since they produce a large number of small seeds that germinate quickly and occupy most of the available niches. Hence, pre-climax and climax species with low numbers of large, slow to germinate seeds, have few vacant niches to occupy. This theory is supported by the results on regenerating species in the understorey of the mixed-species plantation. These factors affect the natural recovery process; so selective planting of dominant canopy tree species may greatly speed up or bypass the early stages of forest succession.

Different restoration strategies must be applied to each region where they have varying initial resources and potentiality of seed germination under natural conditions. For example, in the Mediterranean, reforestation by seeding was at one time a widespread large-scale restoration practice. It was the commonest way to restore pine forests until the mid-seventies, when increased possibilities of mechanical site preparation as well as improved seedling production led to a shift towards plantation practices. In our study, the results of seed germination trials suggested that some seed dispersal species e.g. *Lagerstroemia duperreana*, *Vitex peduncularis*, *Nephelium hypoleucum* and *Morinda coreia*, as well as some in the soil seed bank such as *Lannea coromandelica*, *Grewia elatostemoides*, and *Antidesma ghaesembilla* had low potential to germinate. It can predicted that the development of these species will proceed at a slow to medium rate. This is therefore one reason for using a replanting strategy in the restoration process. However, in recent years the need for reforestation has renewed interest in comparing different seeding and planting strategies, particularly in areas that suffer serious and extensive wildfires (Moreno and Vellejo, 1999).

In Thailand, following the framework method of Elliott et al. (2002), germination is one criterion for the selection of species to be planted on degraded sites. Furthermore, nursery tests of the selected species are necessary for the restoration strategy, in order to establish which of them has a high germination rate (Blakesley et al., 2000). In contrast to the framework method, this study suggests that species with a high germination rate should have greater potential to establish and persist upon invasion of disturbed areas, so dominant woody species of the forest canopy with a low capability for germination or those that need special treatment should be selected for enrichment planting in order to accelerate recovery to a complete forest ecosystem. After the dominant species are planted and become established, early and intermediate successional species, which disperse to and germinate in the plantation, will form the undergrowth of the community.

4.3 RESTORATION

4.3.1 Survivorship and productivity

<u>Results</u>

4.3.1.1 Seedling Survival

Enrichment species were planted in five stands of multi-purpose species plantation (T: *Tectona grandis*; TT: *T. grandis* and *Tamarindus indica*; TG: *T. grandis* and *Gmelina arborea*; TTG: *T. grandis*, *T. indica* and *G. arborea* and TTA: *T. grandis*, *T. indica* and *Artocarpus occidentale*). Figure 4.9 shows mean percentage survival in the five different plots, over one growing season, from June 2001 to May 2002. Average seedling survival of all enrichment species, which is one parameter necessary to evaluate the success of restoration, ranged from 81.73% (TTG) to 86.30% (TTA) for the first year after planting. These values are quite high and are acceptable for restoration techniques. However, the high rate of survival can also be partially attributed to intensive management. The area commonly suffered from forest fire during the dry season, though some species were tolerant to this type of disturbance. This could obviously affect survival rate, so cutting of weeds, which were also competitors was conducted twice a year, first in the dry season (to protect against forest fire) and again during the rainy season (to reduce competition). Statistical analysis indicated no apparent statistically significant differences (F=1.445, df=4) in seedling survival between the five different multi-species plantations (Figure 4.9, Table 4.12). This suggests that enrichment planting by the use of 11 indigenous slow-growing canopy trees qualifies as a restoration program in this region. Although *Schleichera oleosa* and *Xylia xylocarpa* var. *kerrii* showed a high mortality rate, the other nine species maintained excellent survival rates after one year (Appendix A.9). Moreover, this illustrates that canopy trees can be established in highly degraded areas during the early successional stage by employing management techniques in the restoration programme.

4.3.1.2 Absolute growth rate

Table 4.12 shows mean absolute growth rate (AGR) of 11 candidate enrichment seedlings at the end of the first growing season. In terms of productivity, the mean absolute growth rate (AGR) of seedlings in the five multi-species plantations did not show statistically significant differences. However, mean AGR in the TG plot was significantly higher, at the 0.05 level, than in the TTA plot. After one year, the two-species TG plot had the highest absolute growth rate (8.54 kg/ha/yr), followed by T. The three-species TTA plantation showed the lowest value (3.18 kg/ha/yr) (Figure 4.10 and Table 4.12). Data clearly showed no relationship between the number of canopy tree species and AGR of seedlings, though it did show fluctuation within the five multi-purpose species plantations. Although, TG had a relatively low survival rate, it showed the highest value of AGR. Surprisingly, survival rates were similar within the five plantations, but growth rates were significantly different.

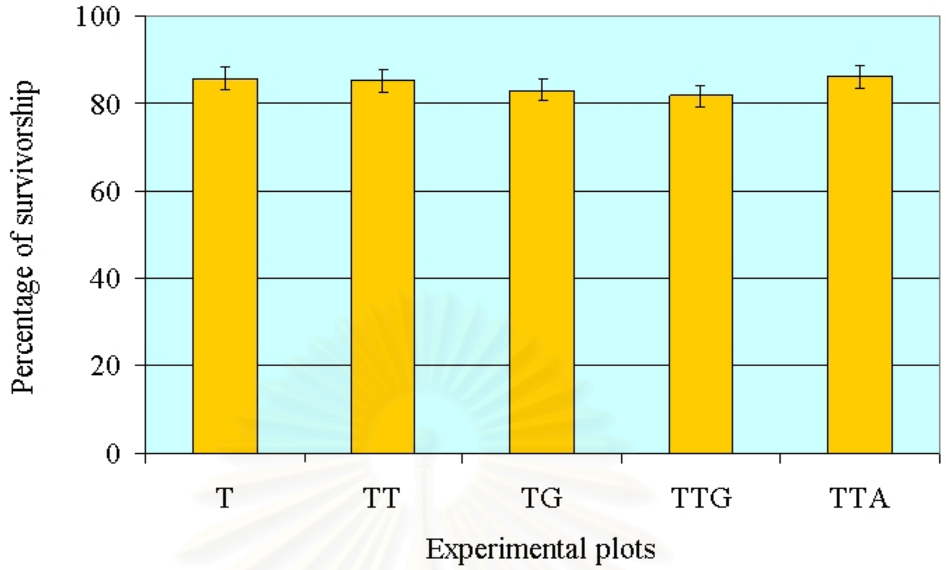

Figure 4.9: Mean percentage survival of enrichment species in five multi-purpose species plantations.

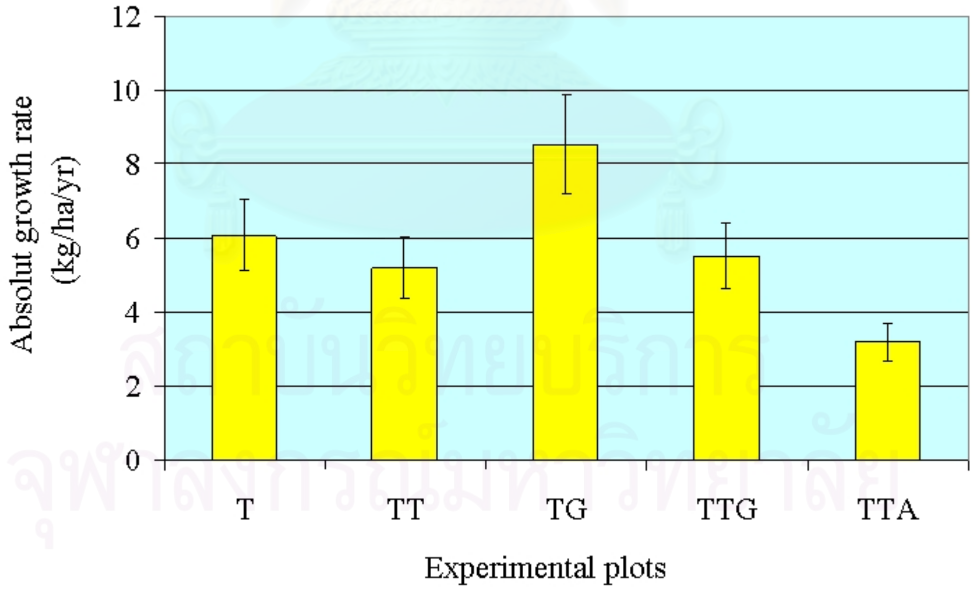

Figure 4.10: Mean absolute growth rate of enrichment species in five multi-purpose species plantations.

4.3.1.3 Relative growth rate

Relative growth rate, which is a modified form of AGR, is an ecological parameter for determining productivity. The trends observed for RGR were similar to AGR. The TG plots had the highest RGR (0.40), whilst the lowest value (0.17) was recorded in TTA (Figure 4.11 and Table 4.12). There were no significant statistical differences among the plantation types, but the mean RGR in the TG plot was significantly higher than in the TTA plot at the 0.05 level. RGR values for three of the mixed-species plantations (T, TT and TTG) showed little variation, with values ranging from 0.26 to 0.30. Evaluation of both AGR and RGR found that productivity of enrichment seedlings fell into three categories; the first, with the highest increment of productivity was TG, the second with moderate increment was T, TT and TTG and the last, with the lowest productivity increment, was TTA. Surprisingly, TTA had the highest rate of survival, yet seedling productivity was lower than in all the other four plantations.

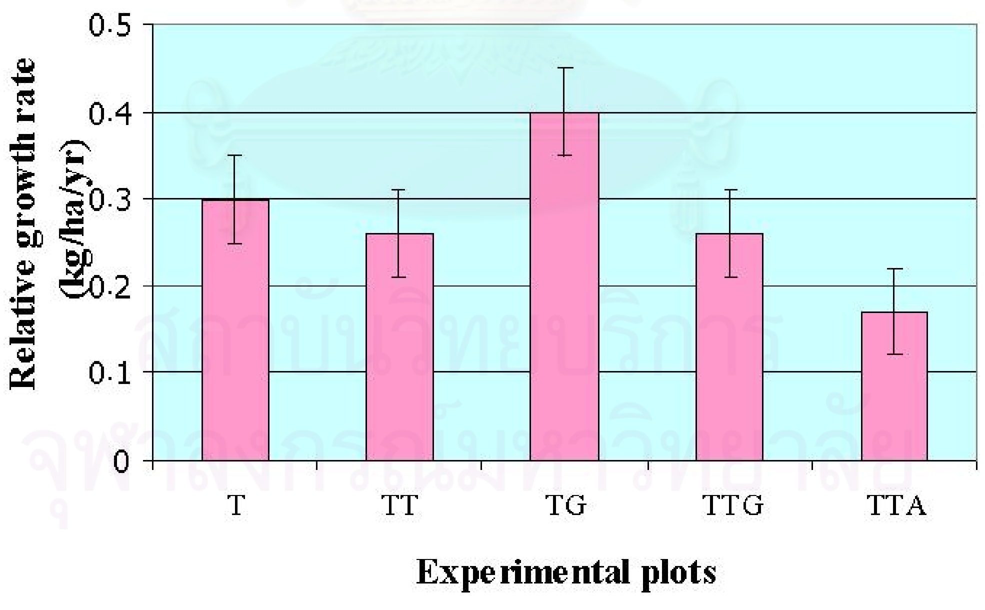

Figure 4.11: Mean relative growth rate of enrichment species in five multi-purpose species plantations.

Table 4.12: Statistical analysis (ANOVA) on seedling survival and productivity.

Experimental plot	Survivorship			Absolute growth rate (AGR)			Relative growth rate (RGR)		
	Mean	**SD**	**Sig.**	**Mean**	**SD**	**Sig.**	**Mean**	**SD**	**Sig.**
T	85.84	1.58	*a*	6.074	1.31	*a*	0.299	0.05	*a*
TT	85.38	2.09	*a*	5.203	0.15	*a*	0.265	0.17	*a*
TG	83.10	4.40	*a*	8.542	4.83	*ab*	0.404	0.09	*ab*
TTG	81.73	3.44	*a*	5.501	0.08	*a*	0.261	0.01	*a*
TTA	86.30	1.37	*a*	3.180	0.29	*ac*	0.174	0.02	*ac*

Note: Differences between plantations are statistically significant ($p<0.05$) when different letters follow their means.

4.3.1.4 Increments of diameter and height

Diameter above ground level (D_0) of enrichment seedlings was significantly different between the five multi-purpose species plantations. The results of statistical analysis fell into two main groups and are shown in Table 4.13. The T and TG plots showed greater increments than TT, TTG and TTA (Figure 4.12). Therefore, within one growing season, the two-species (TG) and single-species plantations (T) clearly showed greater increased diameter than the three-species plantations (TTG and TTA) or TT.

Results clearly showed that the two-species plantations (TG and TT) had greater increments in height over one year than the T, TTA and TTG plots respectively (Figure 4.13). However, there were no statistically significant differences between the five plantation types (Table 4.13). There were no trends in the relationship between diameter and height increment within the five plantations. However, TG showed high increments in both diameter and height. T plot had the highest increment in diameter, but had lower height increments than TT and TG. The three-species plantation TTG had a higher increment of diameter than TTA, but its height increment was less than TTA.

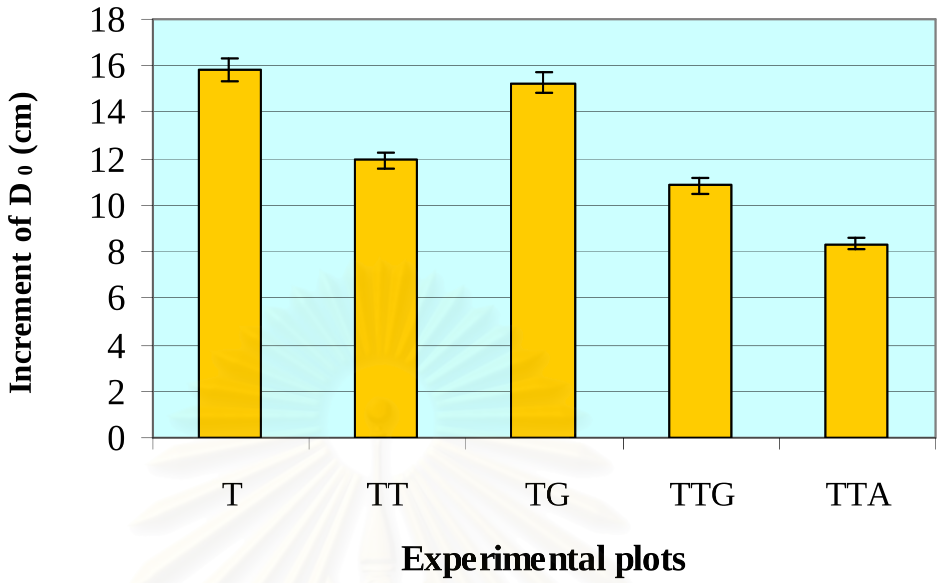

Figure 4.12: Mean increment of diameter above ground (D_0) for enrichment seedlings one year after planting.

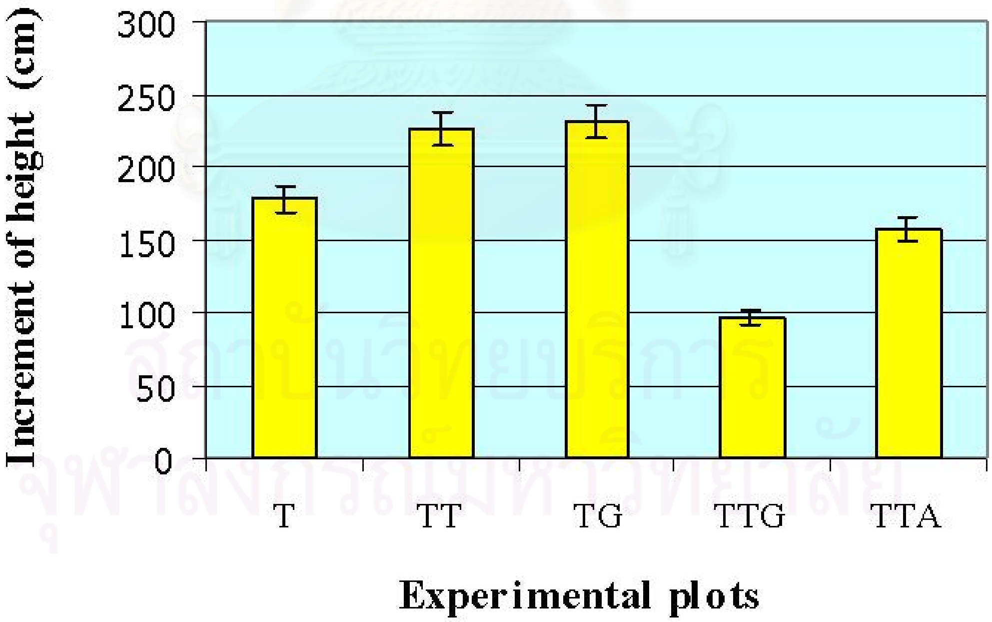

Figure 4.13: Mean increment of height for enrichment seedlings one year after planting.

Table 4.13: Statistical analysis (ANOVA) on increments of diameter above ground and in height for enrichment seedlings one year after planting.

Experimental plot	Diameter above ground (cm)			Height (cm)		
	Mean	SD	Sig.	Mean	SD	Sig.
T	15.80	1.70	*a*	177.90	51.05	*a*
TT	11.93	1.61	*ab*	225.66	99.28	*a*
TG	15.26	3.88	*a*	231.33	78.03	*a*
TTG	10.85	1.91	*ab*	97.00	35.35	*a*
TTA	8.33	0.38	*b*	157.33	70.54	*a*

Note: Differences between plantations are statistically significant ($p<0.05$) when different letters follows their means.

Discussion

Survivorship

Central to community ecology is the study of species diversity, particularly the creation and maintenance of local and regional biodiversity. It is well accepted that restoring biodiversity is desirable for a variety of ecological and applied reasons. Restoration is, in effect, a series of experiments that introduces different numbers and varieties of species to degraded areas, experiments that can be used to test the effects of species richness and the role of species in community recovery and functioning (Palmer, Ambrose and LeRoy Poff, 1997). May (1973), presented mathematical evidence indicating that diverse systems are less stable than simple ones. The more diverse a community, the more complex the web of species interactions and thus the larger the effect that disturbances would have on the system. Several later works, such as that of Naeem et al. (1994) and Tilman (1996), describe how diversity may make individual species more vulnerable to extinction, though overall community or ecosystem properties such as energy transfer or biomass remain stable, since some species functionally compensate for others. Thus, Palmer, Ambrose and LeRoy Poff (1997) suggested that some communities or ecosystems might be more stable if you increase diversity, even though individual species may or may not persist. Therefore, a clearly defined restoration goal is imperative at the outset of each project, such as that of the present study, which aims to recover the diversity of mixed deciduous forest ecosystem with Teak that has been rehabilitated from highly degraded land. The results of this study indicated that some primary species, as well as a few climax species, could regenerate naturally in the understorey of the multi-species plantations. The recovery of climax species is however of more concern than that of primary species. Unfortunately, other biological components, e.g. animals, insects or micro-organisms, were not explored in this study.

The number and variety of enrichment seedlings should reflect that of the primary forest ecosystem, since the restoration strategy is attempting to imitate a natural community and accelerate successional processes. The high diversity of tropical forest ecosystems is usually associated with a large number of species occurring at low densities (Hartshorn, 1980). This suggests that common species, as well as rare species, should be planted in the same proportions as they are found in the natural ecosystem, though this is somewhat dependant upon the criteria of the restoration strategy. If a species seed dispersal is limited, they may be unable to establish new populations in the plantations or restoration areas, and this would be a problem for rare species conservation (Strykstra, Bekker and Bakker, 1998).

"Ecological functioning" was defined by Schulze and Mooney (1993) and Davis and Richardson (1995), and is a loose concept that basically refers to keeping systems "working" i.e. cycling energy and nutrients through trophic levels to retain system integrity. Palmer, Ambrose and LeRoy Poff (1997) suggested that it need-not concern individual species if there is some degree of functional redundancy among the pool of colonists. This would indicate that it might be possible to set minimum levels for restoration of species richness that ensures proper functioning. This would be clearly related to the initial conditions; i.e. how much of the community would need to be initially restored in order for the site to ultimately support the desired community, with its proper structure and function? Our study tried to develop a restoration strategy that selected a diversity of suitable seedling species (climax species) for replanting in the understorey, and which reflected the structure of mixed deciduous forest with Teak.

However, there is a great need for pragmatism and acceptance that restoration of all species will not typically be possible.

From the above discussion, the use of enrichment species is useful in the selection of restoration strategies for this area. Patterns of enrichment planting design in the five multi-purpose plantations were uniform. Random selection was used for establishing the experimental plots. Seedling density, plant spacing (1.6 m x 1.6 m) and ecological methods for randomly planting were the same in all stands. There were no genetic variants because seedlings of each species were taken from the same nursery (in northern Thailand). The five plantations were adjacent and close to each other. Therefore, topographic, climatic and edaphic effects were assumed to be equal, implying that differences in growth and productivity of planted seedlings had arisen from interactions between upperstorey species and the enrichment seedlings.

Enrichment planting involves the germination of seeds in trays or pots, and transplanting the resulting seedlings to the plantations, thereby accelerating the establishment of desirable species. To preserve biological diversity and to accelerate the conversion of highly degraded areas back to forest community, 11 native tree species were selected for underplanting in plantations of multi-purpose tree species. Survival of native enrichment tree species in the five multi-species plantations was high, at over 80% one year after planting, suggesting that canopy and slow-growing enrichment species have the potential to establish and grow between the gaps of multi-purpose species plantations. Other than *S. oleosa* and *X. xylocarpa* var. *kerrii*, these species should be recommended for enrichment planting in this area in spite of their relatively slow growth rates. However, *S. oleosa* and *X. xylocarpa* var. *kerrii* may be useful as selected species for enrichment planting if they receive more intensive nursery treatment before planting and more intensive silvicultural follow-up. These species show co-dominance in natural MDF in northern Thailand (Bunjavechewin, 1983). Moreover, *X. xylocarpa* var. *kerrii* produces highly economic timber. Therefore, when considering future forest management, this species can aid forest structure and provide economic returns to the manager.

Secondary succession may or may not start with either pioneer species or pre-climax species; it depends upon their chances of occupying available niches in the initial stages of succession. Nevertheless, enrichment species still need intensive tending in the form of periodic weeding, cutting and fertilization for at least a year after planting. There are many interrelated factors influencing seedling mortality. In nature, unfavourable conditions such as nutrient availability, soil moisture deficit and drought can limit seedling establishment (Reich and Borchert, 1984). Therefore, the inability of some enrichment seedlings to adapt to the site's microclimate may cause seedling mortality. In this study it was unclear which factors affected mortality.

Enrichment planting of native species was successful in mixed-species plantations after one year. It was found that planted native pre-climax species permitted the coexistence of many other species in the understorey of mixed-species plantations, due to differences in niche occupation and low levels of competition. Moreover, the upperstorey did not seem to have any negative interactions, such as allelopathy, on the growth of the enrichment species. However, according to the theory of competitive exclusion, as enrichment species grow, competition on available resources, such as water, nutrients and light will increase. The theory also predicts that competition between species will reduce niche width, therefore affecting diversity of plant species. In the long term, the coexistence of enrichment plant species results in heterogeneity at the level of the individual. For this study, enrichment species were planted within the gaps of the plantation, thereby reducing competition for light. However at this site, competition between enrichment species, plantation species and regenerative species may have affected growth. The interactions between inter-planted species are of three types; complementary; supplementary and competitive (Filius, 1982). Therefore, if restoration strategies are based on the relationship between upperstorey trees and enrichment seedlings, then the interactions must be positive i.e. complementary or supplementary, rather than negative i.e. competitive.

Two species, in each plot, died within one year of planting (Appendix A.9). These dominant climax species were *Schleichera oleosa* and *Xylia xylocarpa* var. *kerrii*, suggesting that they may or may not be suitably selected for restoration purposes. This would depend upon the objectives and criteria of the restoration strategy, such as the economic returns or the preservation of native species and biodiversity. Multiple factors caused mortality in these species, such as age of seedlings before transplantation, tolerance to the micro-habitat of the plantations, quality of seedlings and so on. However, due to the limited number of seedlings, a final conclusion on the suitability of these two species for establishment in degraded land cannot be made. Mature forest exhibits different abiotic conditions to that in plantations. Hence, mortality may have reflected limited tolerance to poor soil properties, since the other species managed to successfully establish themselves on the degraded soils of the mixed-species plantation. Evans (1992) reported that many factors affect initial survival rate, including;

1. The skill of the planters.
2. Immediate post-planting weather conditions.
3. Condition of seedlings.
4. Poor soil conditions.
5. Insects, such as termites.
6. Weed competition.
7. Animal damage.

Therefore, an important step in restoration is to identify and minimise the factors that limit seedling growth and contribute to mortality. Previous research, in many regions of both temperate and tropical forests, has clearly demonstrated that mammalian herbivores may limit seedling recruitment (Auld, 1995; Asquith, Wright and Clauss, 1997). Protection against seedling damage is an increased cost for restoration projects. Holl and Quiros-Nietzen (1999) reported that to protect against herbivores cost more than 0.30$ per seedling and that additional labour was needed to install fencing. However, our study revealed no indication of herbivorous damage, even one year after planting. After seedlings have been damaged by herbivores or other disturbances, they are usually able to resprout. Even so, options to reduce the problem of herbivorous attack have been put forward in other restoration projects (Aldous and Aldous, 1944; Brenes and Di Stefano, 1997) including;

1. Planting larger seedlings.
2. Planting stem cuttings (stakes), which are sufficiently large to prevent herbivorous damage.
3. Taking measures to control the populations of herbivores.

Seedling survival in the diversified plantations was higher than 80%, may be due to nutrient input from either decomposition or fertilizer (applied once after planting). Weeding (twice per year) and control of forest fire were management policies that increased survival of planted seedlings. Weeding was a management policy to reduce inter-specific competition between enrichment seedlings and grass, especially *Imperata*, which dominated every stand in the multi-purpose species plantation. Due to the annual forest fires that occur every dry season in the northern regions of Thailand, fire protection is a necessary policy in all restoration projects.

The high survival rates seen in the present study confirm that climax and slow-growing species can grow in degraded sites that suffer drought, low nutrients and exposure to sunlight. In contrast, Finegan (1984), suggested that a potential cause of seedling mortality in large clearings was caused by high leaf temperatures, brought on by drought and exposure to full sunlight, thus pioneer species, which are more tolerant of these conditions than forest species, are better suited to open field environments. However, the results of a study by Napstad et al. (1990) strongly support the results of this study. They suggested that large-seeded, shade-tolerant, slow-growing species were more drought resistant than pioneers, since they use the energy reserves in their seeds to develop deeper root systems that can tap deep reserves of moisture. However, full sunlight is the most important limiting factor in the dry season in an exposed site, though nurse trees may reduce this problem, even though some species such as Teak and Gmelina shed their leaves at this time. The shade provided by nurse trees is therefore an advantage of multi-purpose species plantations.

Silvicultural management tends to complement natural regeneration beneath the understorey. The initial weeding and other maintenance for the enrichment species probably favour natural regeneration. The regenerating tree and shrub combination together with enrichment species in the understorey of nurse tree plantations may be a more economically attractive alternative for managers of restoration strategies. Silvicultural treatment is recommended, since it encourages the establishment of enrichment species, as well as favouring regeneration processes, especially of highly important economic tree and shrub species.

The high labour costs of enrichment planting are a problem that should be considered. Therefore, restorationists are concerned with cost/benefit ratios. It is important to determine the minimum numbers and species necessary for proper community functionality (Palmer, Ambrose and LeRoy Poff, 1997). The planting of high-value timber species that grow quickly should be recommended. The value of timber species was one of the criteria used for selecting enrichment species in our study, since the potential financial benefit would compensate the cost of planting. Enrichment planting has other indirect benefits i.e. environmental protection and ecotourism. Alternative economic enrichment species have been planted in many areas. For example, enrichment of depleted dipterocarp forest with fruit trees such as *Dialium* spp., *Garcinia* spp. and *Willughbeia* spp. was an economically and ecologically viable alternative to increasing lumber prices in Indonesia (Korpelainen et al., 1995). Additionally, combining timber trees with species that produce an earlier profitable harvest (i.e. plum, *Euterpe edulis*), accelerate returns on investment and make this technique more economically attractive (Schulze et al., 1994; Montagnini et al., 1997). Therefore, enrichment adds value to degraded lands with low volume forest by increasing the expected harvest.

Enrichment planting of native species is an important criterion in restoration strategies since it relates to the goal of conserving biological diversity. Many studies have indicated that native species, rather than non-native or exotic species, should be used to restored degraded land, since non-native species may have huge impacts on the community. Exotic species may alter species diversity or prevent the reestablishment of native species in restoration sites (Simberloff, 1990; Vitosek, 1990). These species may be particularly difficult to remove, since they are often subject to less pressure from competition or predation than are native species. However, it may be necessary, or at least more practical, to rethink restoration practices that do not exclude exotics that have become well established in a country. For example, the plantations in this study were originally planted with non-native species, such as *Tamarindus indica* and *Anacardium occidentale*, due to their ability to improve nutrient content and provide economical returns to manager. Fortunately, these tree species greatly benefit the farmer and have minimal impact upon the environment. In this case, the exotic species helped heal a degraded environment and were useful to restoration. Gajaseni and Jordan (1992) reported several potential advantages to using interplanting with *T. indica* (Tamarind) in multi-purpose species plantations;

1. Economic advantage; the tamarind produces a fruit which is highly prized in Thailand and which has a good economic value. Income from sales of the tamarind fruit can extend the economic income for perhaps 15 years.

2. Ecological advantage; the tamarind occupies a different niche than the Teak and thus competition is much less and stagnation does not occur. Structural diversity prevents interlocking of canopy branches and competition for light, which is often the cause of stagnation in Teak monocultures.

However, long-term restoration requires the planting of native species rather than non-native species. In long-term plantation forests, where capital input may be sporadic as in developing countries, survival may be better achieved with native species, since they are better adapted to local soil conditions and can be more resistant to local pests. More importantly, they can survive competition with local weed species, whereas weeding is often critical in exotic plantations (Jordan, 1992).

Growth and Productivity

Forest plantations can play a key role in long-term forest ecosystem rehabilitation and restoration goals with near-future socio-economic development objectives (Brown and Lugo, 1994; Lamb, 1998). Several studies have indicated that forest plantations significantly accelerate natural succession by overcoming barriers to natural regeneration under disturbed and degraded land where forest remnants and forest seed-dispersing wildlife are present on the landscape. This is due to influences on understorey microclimate conditions, structural complexity of vegetation, and development of litter and humus layers during the early years of growth. These changes lead to increased seed input from neighbouring native forests by seed dispersing wildlife attracted to the plantations, suppression of grasses or other light-demanding species that normally prevent tree seed germination or seedling survival, and improved light, temperature and moisture conditions for seedling growth (Parrotta, 2000). This coincides with Horn and Montagnini (1999) who indicated that high production and accumulation of litter in plantations contributes to inhibited growth of herbaceous species, thus favouring establishment by trees. Plantations therefore benefit the growth and persistence of enrichment seedlings.

Results for productivity were obtained from observations on the increment of total biomass in each plot over one growing season or one year after planting. The high increased productivity of enrichment species in the TG mixed-species stand cannot be easily explained and may be due to several factors, such as favourable micro-environmental conditions, nutrient availability, light-use efficiency, grass competition and so on. The three-species plantation (TTA) had the lowest rate of enrichment species productivity in terms of both AGR and RGR. Similar research on single and mixed-species plantations in the tropics showed that any consistent productivity advantages of mixed-species plantations were greater than that in mono-specific plantations (Montagnini et al., 1995). Lower pest damage was also noted in mixed plantations in Costa Rica. One of the reasons for greater growth in mixed stands could be that species compliment each other in their nutrient cycling strategies (Montagnini and Sancho, 1994). These results are in contrast to our study, in which we found mixed-species plantations (other than TG) showed lower productivity than pure plantations of Teak. One factor that is related to increased seedling growth rate is the release of nutrients to the soil through litter decomposition. The mixed-species plantation would have a better chance of satisfying the demands of decomposers due to its more diverse chemical and nutrient make-up (Byard, Lewis and Montagnini, 1996). Therefore, nutrient concentration in the soil through decomposition processes should correlate to the growth rate and productivity accumulation of planted seedlings. If considering only this factor, then mixed plantation types should have positive influence on seedling productivity, but since this was not the case in our study, then more than one factor was involved.

Multi-purpose species plantations can encourage survivorship, growth rates and productivity of planted seedlings. Thus the multi-purpose plantation species act as nurse trees and affect the early period of growth for seedlings of enrichment species. One year after planting, the productivity of seedlings was highest in the understorey of the TG plantation, followed by T, TTG, TT and TTA, respectively. The high growth rates of enrichment seedlings in the TG plot suggest that the understorey conditions strongly favoured the early development of enrichment species. The increase of basal area and canopy cover of nurse tree species in TG represents reasonably good growth compared to multi-purpose tree species in other plots. The larger crown area of Teak and *Gmelina* may have affected the quantity of floor litter and understorey conditions, such as light, temperature and moisture at the soil surface, and may have discouraged aggressive weeds, thereby providing suitable conditions for survival and growth of enrichment seedlings. *Gmelina* is a fast growing species, and a study in Sarawak recorded its yearly biomass accumulation as 13 Mg/ha (Halenda, 1993). However, its performance varied in different areas, probably due to various insect pests that attacked the leaves. In our study *G. arborea* grew very well and did not show a lack of performance. In Brazil and Costa Rica, tropical species typically accumulate biomass at rates of 1.6-29.8, though most fall in the range of 6-15 Mg/ha per year (Lugo, Brown and Chapman, 1988).

Growth rates of enrichment seedlings, as presented in increments of diameter and height, are closely related to productivity. Kira and Shidei (1967) used this relationship as a measurement of plant productivity. Growth rates and productivity between species were different. However, TG represented the best overall growth rate. TT, TTG and TTA, which all contained the nitrogen-fixing species *Tamarindus indica*, had a low rate of productivity accumulation for enrichment seedlings one year after planting, in contrast to T and TG. The poor accumulation of biomass in TT, TTG and TTA was due to damage to the Tamarind tree from occasional forest fires. Meanwhile, other nurse trees such as Teak showed tolerance to forest fire. Also, *Gmelina*, which was a fast growing species, could resist periodic disturbances by fire. Therefore, in this study, mixed plantations with nitrogen-fixing species provided poor environmental conditions for enrichment seedlings, in contrast to results from other tropical areas.

Within the multi-purpose species plantations, the addition of the N_2-fixing tree *Tamarindus indica* had no or very little effect on seedling growth and productivity accumulation. This lack of growth response is perhaps surprising. However, other research has recorded similar results in plantations with the N_2-fixing trees, *Stryphnodendron microstachyum* Poepp. Et Endl. and *Albizia guachapele* (H.B.K.) Little (Montagnini, 2000). This is in contrast to the enhanced aboveground productivity of mixed plantations with added N_2-fixing trees in other areas, such as that of *Eucalyptus-Albizia* plantations in Hawaii (Binkley et al., 1992). Binkley et al. (2003) later reported that the sustained growth in the plantations probably resulted from the presence of increased soil nitrogen supply over time (from N-fixation by *Facaltaria moluccana*), and increased dominance of Eucalyptus. In our study, total soil nitrogen content prior to leaf fall did not differ within the three plantation types (Figure 4.16) that had leguminous trees, thus suggesting that the plantations with N_2-fixing trees had not yet developed positive interactions with enrichment seedlings in their understorey. This is possibly why some of the mixed plantations showed lower seedling productivity accumulation than the pure Teak stand, though several other factors such as light interception and light-use efficiency may have been involved. However, a comparison between the pure Teak plantation and the mixed species plantation TG, showed a higher increment of productivity in TG. This coincided with studies by many other researchers who stated that mixed plantations were more advantageous to productivity than pure plantations. This was believed to be related to decomposition, production of litter, rates of nutrient return and so on. Thus, this study suggests that mixed plantations of Teak and Gmelina show greater productivity consistency to enrichment seedlings in the restoration processes than the single species plantation.

The difference in growth rate and productivity between the mixed and pure plantations depended on soil properties, such as availability of nutrient from decomposition processes, and species. Much research has indicated that mixed-species plantations clearly have the potential for out-producing both regenerative species and enrichment species (Binkley et al., 2003). Likewise, DeBell, Cole and Whitesell (1997) recorded similar results and concluded that the productivity in mixed stands of *Eucalyptus* and *Albizia* increased over a 10-year period. Moreover, the mixed-species plantations had greater total wood yields (Kaye et al., 2000), especially after 10 years, due to soil increased carbon and nitrogen levels. Bauhus, Khanna and Menden (2000) reported that 4 years after starting a species–replacement series experiment with *Eucalyptus globules* and *Acacia mearnsii* in Victoria, Australia, stem volume in 1:1 mixtures was about 20% greater than in *Acacia* monocultures, and more than twice that in *Eucalyptus* monocultures.

Another external factor regulating seedling regeneration is moisture and drought. The initial requirement for seed germination is absorption of enough water to catalyse essential physiological processes in the embryo. Water is necessary to soften seed coats and stimulate metabolic processes. The second greatest demand for water is to replace that lost through foliage and transpiration. The effect of drought on seed germination varies with the temperature regime, though the majority of young seedlings die from desiccation during periodic drought (Kozlowski, 2002). Flooding is another factor that influences seeds and seedlings, since inundation of soil deprives them of the oxygen necessary for respiration. Kozlowski and Parllardy (1997), indicated that germination is postponed or prevented by flooding. Solar radiation is another factor, and both seed germination and early seedling development are variously influenced by light intensity, light quality and photoperiod (Kozlowski, 2002), especially in tropical forest ecosystems. These physical factors directly affect both survival and/or growth and productivity of seedlings.

Pollutants and agricultural chemicals are the last external factors to be mentioned. A number of agricultural chemicals such as insecticides, herbicides, antitranspirants and fungicides are useful or even indispensable in regenerating forest ecosystems. However, plant regeneration is sometimes retarded due to the toxicity of excessively high dosages or incorrect application. These chemicals may alter plant metabolism, arrest germination and impede seedling growth, leading to early seedling mortality. Likewise, pollutants sometimes reduce seed production and inhibit seed germination and plant growth (Kozlowski, 1986). Pollutants may also alter mechanisms of flowering and fruiting and injure reproductive structures (Kozlowski, 2000). Plants regenerating in disturbed forest ecosystems often receive a wide array of pollutants and chemicals in the air and soil, such as gaseous and particulate pollutants, herbicides, pesticides, growth retardants and antitranspirants. Arrested stand regeneration may be associated with lowered seed production and inhibition of seed germination and plant growth, as well as accelerated plant mortality. These responses often follow injury to plants by chemicals as well as metabolic dysfunctions. Combinations of chemicals arrest plant growth to a greater degree than single chemicals. The major chemicals and pollutants include sulphur dioxide (SO_2), ozone (O_3), fluorides, oxides of nitrogen (NO_x), peroxyacetyl nitrates (PAN) and particulates (e.g. soot, cement-kiln dusts, lead particles, magnesium oxides, foundry dusts and sulphuric acid aerosols (Kozlowski and Constantinidou, 1986). The interaction of chemicals and pollutants with seeds and seedling performance is known as allelopathy. Some plant species can also release toxic chemicals (allelochems) to the soil. Chou (1990) considers alleopathy to have detrimental effects on plant biochemistry. Allelopathic compounds are released into the environment by means of volatilisation, leaching, decomposition of residues and root exudation. These compounds may adversely affect seed germination and growth of neighbouring plants (Hytonen, 1992). Allelochems include phenolic acids, coumarins, quinines, terpenes, essential oils, alkaloids and organic cyanides. These potentially toxic compounds are released by exudation, decay of plant tissues, leaching and volatilisation (Rice, 1984). The validity and ecological significance of laboratory experiments on the phytotoxicity of allelochems has been debated, since accumulation of these compounds in forests is modified by soil moisture, as well as being broken-down by soil microflora (Kozlowski, 1995). A study on the effects of allelochems released by *Empertrum hermaphroditum* on the regeneration of *Pinus syvestri* in Sweden can be debated, since they occurred in frozen soil and could therefore not be detoxified by micro-organisms until the ground thawed (Nilsson, 1994). Another factor that may regulate seedling growth and survival and affect plant regeneration and succession is salinity. Plants species have different salt tolerances, and since salty soils vary in nutrient balances, toxic concentrations of various ions can build up, hence influencing the soil processes essential for the uptake of nutrients (i.e. nitrogen and phosphorus (Swanson et al., 1998). It is possible that the Mae Moh study site may have suffered from the influences of a nearby sulphur mine, but further investigations will be needed to confirm this.

Many techniques of forest management are required to achieve successful re-establishment of native species in degraded woodland. For example, intervention in form of soil ripping and/or removal of adult trees has a positive influence on the growth of native species planted in fragmented landscapes of the southwestern Australian wheat belt. This technique is relatively inexpensive and simple to apply at scales appropriate to the size of the remnant woodland (Yates, Hobbs and Atkins, 2000). It was also clear that the removal of adult tree competition had the largest impact on new growth. This reflects the findings that regeneration of canopy species occurs most commonly after large-scale disturbances that kill adult trees (Yates, Hobbs and Bell, 1994). Meanwhile, the removal of the overstorey in Jarrah forests resulted in lower soil water deficits, and consequently the Jarrah seedlings had higher leaf water potentials and were less stressed than seedlings growing on sites where the canopy remained intact. To apply these findings to other sites does not necessarily mean creating gaps by killing trees; gaps created by individual windfalls occur sporadically throughout remnant and forest ecosystem. By reducing weeds and planting seedlings in naturally created gaps, unique opportunities for establishing upperstorey tree species and understorey seedlings can be provided (Yates, Hobbs and Atkins, 2000). In our study, seedlings were planted within the gaps to reduce the effects of competition.

The high increase in productivity and high percentage of seedling survival indicated a successful restoration strategy in this area. Conclusions and suggestions for future research on restoration strategies in tropical and subtropical locations, as raised at the international symposium convened in Washington DC in 1996, reflect interesting issues for other regions (Parrotta, 2000), including;

- Relative to initially similar, unplanted sites, plantations generally have a marked positive effect on native forest redevelopment (succession) on severely degraded sites and on sites dominated by grasses and ferns which otherwise preclude colonisation by forest species.
- Choice of plantation species can significantly affect the process of understorey regeneration, several studies having shown that plantations of different species of the same or similar age grown on very similar or identical sites showed marked differences in the density and species composition of their woody understorey community. These differences are due to a combination of factors, including the effect of the overstorey species

on understorey light environments and seasonal regimes, soil chemical and biological characteristics, nutrient cycling processes and their relative value to seed-dispersing wildlife.

- Structure complexity of the planted forest is an important determinant of subsequent biodiversity enrichment due to the importance of habitat heterogeneity for attracting seed-dispersing wildlife and microclimatic heterogeneity required for seed germination for a variety of species. This suggests that broadleaf species yield generally better results than conifers, and that mixed-species plantings are preferable to monocultures, due in part to their increased structural complexity. Future studies were suggested to assess the influence of overstorey (planted) species architecture and phenology on understorey microclimate heterogeneity (spatial and temporal patterns), and aspects of forest floor and soil development that influence recruitment of native forest species, under a variety of site and landscape conditions.
- Wildlife, especially bats and birds, is of fundamental importance as seed dispersers in tropical regions, due to their effectiveness in facilitating plantation-catalysed biodiversity development on degraded sites. Additional research is needed under a variety of ecological conditions to better understand the dynamics of animal seed dispersal in degraded landscapes, to develop appropriate plantation designs to encourage seed transport from remnant forest stands, and to determine the range of distances between seed sources and rehabilitation sites over which seed dispersal by animals is likely to be effective.
- Larger-seeded forest species are far less likely to colonise degraded sites than smaller-seeded species due to seed dispersal limitations, and therefore require management interventions, e.g. enrichment planting to facilitate their establishment, particularly where forest restoration is a major objective.
- Management issues and potential applications relevant to plantation design should also be investigated. Specifically, experimental research is needed to systematically evaluate the effects of site preparation, selection of species, understorey management practices, and stand manipulation techniques on the productivity of both the planted crop and its regenerating understorey,

the associated economic and social costs and benefits, and the environmental impacts of alternative design and management systems related to biodiversity conservation, soil fertility rehabilitation and carbon sequestration.

The use of native versus non-native species is an interesting point to discuss. In degraded ecosystems, native species are usually preferred over exotic species, since they are more likely to fit into a fully functional ecosystem and to be climatically adapted. Unfortunately, indigenous species are often slow growing. In highly degraded areas, with destruction of vegetation and alteration of physio-chemical properties of the soil, exotics may be suitable, however they may have either positive or negative affects upon naturally regenerating vegetation. The role of exotic species in restoration has often been looked at with concern due to their negative impact on soil fertility and biodiversity (Lugo, 1997). However, Dutta and Agrawal (2003) looked at the large-scale use of exotic afforestation on the slag heaps of opencast mines, and found that due to the severe physical and biological damage to the ecosystem, they were of positive benefit, even though the height of shoots was lower than reported in other studies. For example Nath et al. (1991) reported a shoot height of 7.78m for *Eucalyptus sp.*, 7.07m for *Casuarina eqisetifolia* (Beefwood) and 4.51m for *Acacia auriculiformis* (Australian Acacia) in 2 ½ year-old roadside plantations in West Bengal, India. Similar data was reported by Aulokh and Sandhu (1990). When considering biomass, net primary production and stability of exotic plantations, it helps to evaluate nutrient cycling, organic matter and energy transfer. An exotic plantation of *Eucalyptus hybrid* (White Gum) on mine spoil showed the highest water-holding capacity, soil moisture, available nitrogen (NH_4-N and NO_3-N) and phosphorus content in comparison to others. Clearly, this plantation had maximum favourable impact on the soil and thereby enhanced its own productivity. In conclusion, exotic species can be especially recommended for primary rehabilitation on barren coal mine spoil due to their fast growth and establishment (Dutta and Agrawal, 2003). This observation is also supported by Parrotta (1999).

4.3.2 Study of multi-purpose species plantations

Results

4.3.2.1 Basal area of upperstorey canopy trees

The highest average basal area for trees in the five multi-purpose plantations was found in the TG plot (0.150 m^2/tree) followed by TTA, TTG, TT and T respectively (Figure 4.14, Table 4.14). Significant statistical differences were found in the average basal area between the multi purpose plantations ($F=3.51$, $df=4$, $p<0.05$). The mean basal area in TG was significantly higher than in T and TT, confirming that Teak and Gmelina make a good combination in mixed plantations. This combination affected net biomass gain, clearly shown by the increased basal area of both multi-purpose trees. The relationship between these two tree species may have affected litter accumulation, nutrients and the understorey environment.

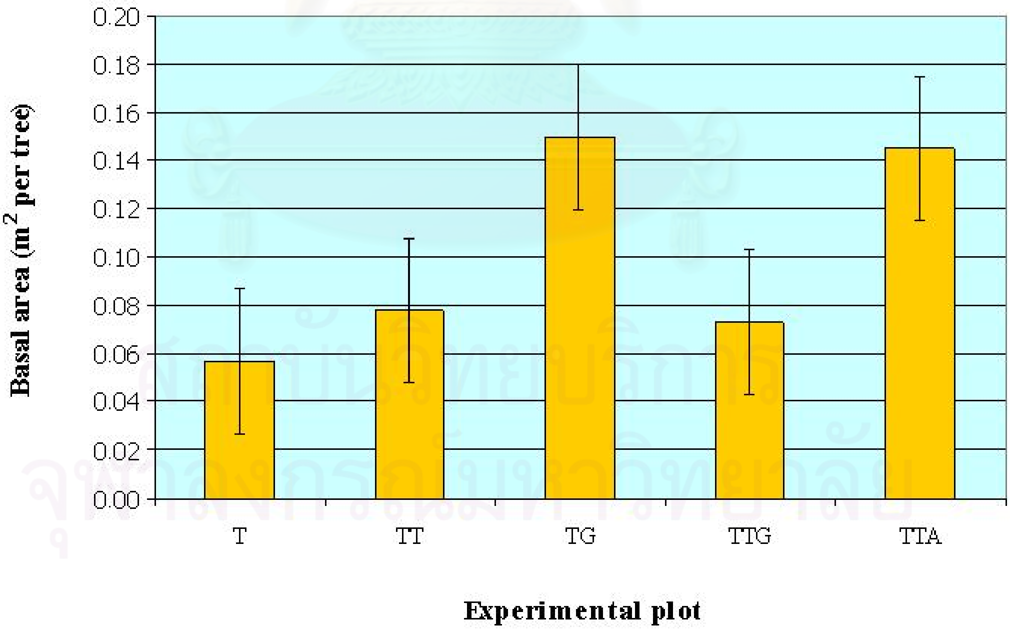

Figure 4.14: Mean basal area of upperstorey canopy trees in five multi-purpose species plantations.

4.3.2.2 Crown width of upperstorey canopy trees

A similar trend was found in the canopy cover (Figure 4.15). There was no statistically significant difference in mean canopy cover of trees in the five mixed-species plantations (F=3.184, df=4), but when comparisons were made among plantation types it was found that the canopy cover of TG was significantly higher than in T, TT and TTG, but not significantly different to TTA (Table 4.14). These results coincide with a study of average total biomass of trees in multi-purpose plantations from 1988 to 1995 by Rattanasinganlachan (1996) (Figure 4.16). The highest value for average total biomass was 39.70 g/m^2 in the TG plot, followed by TTA. Observations showed that 15 years after planting most of the Tamarind trees (*T. indica*) in the TT and TTG combinations were damaged by natural disturbances, such as forest fire and/or weed competition. Thus, Tamarind did not grow well under the hazardous conditions of the plantation. This resulted in a low basal area and canopy cover in the TT and TTG plots. Consequently, this affected accumulation of leaf litter, understorey conditions, organic matter and nutrients.

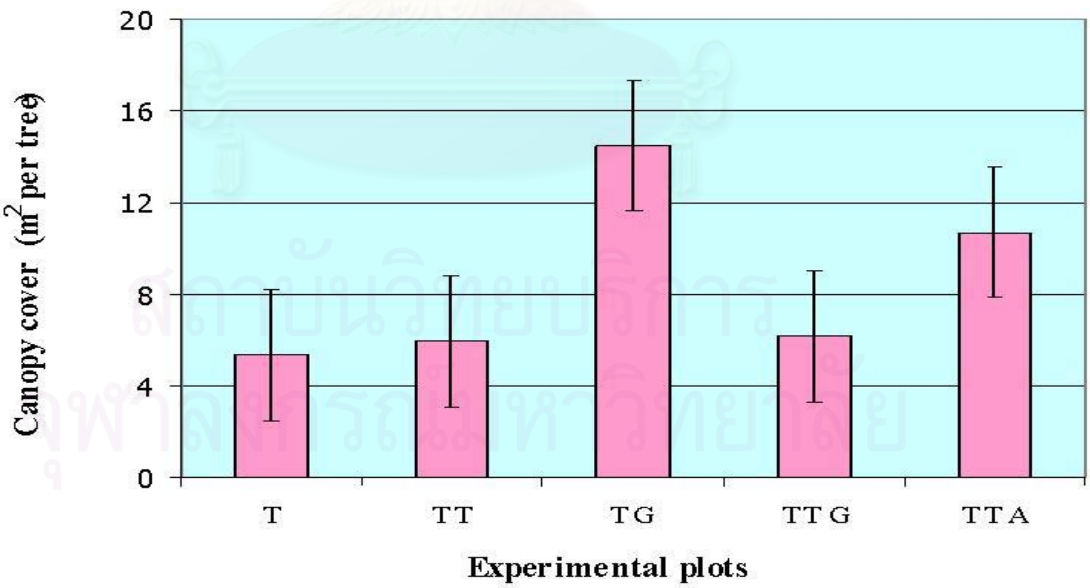

Figure 4.15: Mean canopy cover of upperstorey canopy trees in five multi-purpose species plantations.

Table 4.14: Statistical analysis (ANOVA) on basal area and canopy cover of upperstorey canopy trees in five multi-purpose species plantations.

Experimental plot	Basal area (m^2/tree)			Canopy cover (m^2/tree)		
	Mean	SD	Sig.	Mean	SD	Sig.
T	0.056	0.011	*a*	5.376	0.899	*a*
TT	0.078	0.034	*a*	5.968	0.857	*a*
TG	0.150	0.543	*bc*	14.484	6.186	*b*
TTG	0.073	0.088	*ace*	6.200	0.664	*a*
TTA	0.145	0.057	*be*	10.722	5.743	*ab*

* Differences between plantations are statistically significant ($p<0.05$) when different letters follow their means.

Discussion

Interaction between upperstorey trees and understorey species

All environmental factors (macroclimate, topography and soil properties) affecting enrichment seedling in the five plantation types were similar. Thus, differences in seedling growth and survivorship should have been caused by the diversity of plantation types. The multi-purpose species plantation (a modified Forest Village System) is used for restoring highly degraded land and for increasing the economic value of the stand. It also provides unique opportunities for testing basic ecological theory. Research on restoration strategies can also be viewed as a truly powerful research technique, especially in the case of amelioration on highly degraded land. Interactions of canopy species in the mixed combination may have had both positive and negative effects. The results of this study show that the combination of *Tectona grandis* (Teak) and *Gmelina arborea* (Gmelina) produced positive interactions or synergistic effects. The combination of these two species seemed to be useful for increasing the efficiency of the restoration strategy. The three-species combination of *T. grandis, Tamarindus indica* and *Anacardium occidentale* also produced positive interactions. Species in the mixed plantation that utilized the different niches increased productivity. The increased canopy cover in the TG and TTA plots may have affected light and moisture conditions in the understorey. Thus, the shade provided by the canopy in the mixed-species plantations decreased the rate of colonization of light-demanding pioneer species, especially *Imperata cylindrica*. A diverse mixed tree structure should have positive effects on understorey species composition because it changes environmental conditions and reduces weed competition. This result is supported by Byard, Lewis and Montagnini (1996), who suggested that other than the beneficial effects from mixed design on nutrient cycling, tree species with rapid canopy closure can decrease the growth of weeds after 2-3 years, thus decreasing competition from grass and the cost of weeding during plantation establishment.

Although this study did not estimate total biomass of the mixed tree species, basal area had a strong relationship to biomass. Two reasons may have caused low biomass accumulation in the TT and TTG plots. One is the competitive exclusion principle (Vandemeer, 1992), which suggests that when two species are similar in some respect (i.e. occupy the same niche, interfere with each others activities and so on) it is unlikely that there is enough room in the environment for both. It appears as though the species in the TT and TTG combinations seemed to have a negative interaction with each other, an "antagonistic effect". Due to strong competition, basal area was low in these combinations. The second reason may have been due to the lifespan of the mixed tree species. *Tamarindus indica* is an economic tree, and has a short lifespan in comparison with Teak or Gmelina. Therefore, 15 years after planting some were weak and could no longer persist in the environment. It is also a tree that is easily damaged by the occasional forest fire. For this reason a low value for basal area was found in the TT and TTG combinations.

In terms of restoration, the mixed-species plots TG and TTA seemed to be the most suitable, since they showed positive interactions as indicated by high values for basal area, canopy cover and productivity. Mixed plantations (two and three-species combinations) may have altered poor understorey conditions such as light, litter accumulation, and nutrient concentration in the degraded environment, thus favouring seedling recruitment and increasing the diversity of plants in the understorey. Many reports have suggested that favourable understorey conditions affect arrival and recruitment of seedlings (Byard, Lewis and Montagnini, 1996; Kershnar and Montagnini, 1998; Stanley and Montagnini, 1999; Carnevale and Montagnini, 2002).

If considering regeneration processes, upperstorey trees have the potential to encourage forest regeneration in the understorey. This is supported by Parrotta (1995), who observed that planting of nurse trees such as *Prunus* in open grassy areas, or assisting those that were naturally regenerating by suppression of surrounding weeds would have the double advantage of encouraging bird dispersal and improving the micro-environment for germination and establishment by shading out grass and weeds. Since direct exposure to sun reduces survival of young seedlings, vegetative cover should be allowed to persist towards the end of the rainy season, in order to provide shade during the following dry season. Therefore, multi-species trees can encourage the growth of existing trees and shrubs in their understorey, as well as planted seedlings. Under nurse trees that do not create thick leaf litter such as Gmelina, Tamarind and Cashew, planted seedlings may grow well. However, studies on litter depth and dry mass in the understorey of plantations indicates that deep litter has negative effects upon seed germination and seedling density, since seeds become trapped in the less hospitable upper strata of the litter layer (Parotta, 1995). Successful emergence of many species is reduced when seeds are above or suspended in the litter layer, since solar radiation can create extremely hot and dry conditions, which impedes radicle growth (Facelli and Pickett, 1991, Molofsky and Augspurger, 1992, Peterson and Facelli, 1992).

The results on canopy cover, as presented in previous section, indicated high cover in TG and TTA and low cover in T, TT and TTG. High cover may accelerate both mortality and survival of planted trees and may alter their intended proportions due to competition among plants for light, water and mineral nutrients. Competition for resources and mineral nutrients among individual woody plants, as well as among woody and herbaceous plants is well documented (Van den Driessche, 1984, present study). However, due to the randomness of planting of enrichment seedling as used in this study, the differences among species in depths of rooting and in patterns of absorption of mineral nutrients may be due to variations in availability of resources.

Environmental factors regulate seedling regeneration, especially seedbed and/or quantity of litter, drought, flooding, temperature, solar radiation, pollutants and agricultural chemicals. Extreme differences among seedbeds in terms of physical characteristics, supplies of water and mineral nutrients, and temperature often account for variations in regeneration of disturbed forest ecosystems. The most important determinants of success in reestablishment are the capacity of the surface medium to supply water, and the amount of light that reaches the young seedlings. The denser the soil medium, the better is its capacity to supply water to both seeds and seedlings (Smith, 1995). However, severely compacted soil is a factor that negatively influences seedling germination and growth, often causing mortality. Compaction of soil is common in recreation areas such as picnic sites, timber harvesting sites, fruit orchards, tree nurseries and agroforestry systems (Kozlowski, 1999). Compaction induces a variety of changes in soil structure and hydrology. It increases bulk density (mass per unit volume), decreases total pore volume and the proportion of macroporous space, increases water run off and decreases soil aeration (Anderson and Spenser, 1991, Huang, Lacy and Ryan, 1996, Horn et al., 1995, Miller, Scott and Hazard, 1996). The many changes in soil structure and hydrology associated with compaction often lead to severe physiological dysfunctions in both seeds and seedlings. Water absorption becomes inhibited and leaf water deficits follow. The rate of photosynthesis is then reduced by both stomatal and non-stomatal inhibition (Kozlowski, 1999).

Forest ecosystems are variable due to the impacts of competition on the structure of the plant community during successional processes. These impacts may be influenced to a considerable degree by a variety of environmental factors. Perry (1995) and Kozlowski (2002) categorised competition into 3 types;

1. Species may not need to compete when their growth is largely influenced by predation, disturbance or climatic perturbation (e.g. when herbivores and pathogens increase biodiversity by lowering the capacity of one species to dominate others).

2. Competing species may actually benefit each other by either decreasing or eliminating the adverse effects of plant competition (e.g. by improving the soil or

microclimate, reducing the influence of other competing plants, discouraging predators, catalysing beneficial components in the root zone, and attracting pollinators).

3. Trees may avoid competition through specialization. For example, early-colonizing N_2-fixing plants compete for resources, but later benefit each other by increasing soil fertility.

4.3.3 Concentration of nutrients in the soil

Rattanasinganlachan (1996) conducted soil nutrient analysis in the five multi-purpose species plantations from 1988-1995. Biomass (B), organic matter (OM) and soil nutrients are shown in Figure 4.16. TG had the highest average total biomass (39.70 g/m^2) and highest average organic matter (2.97%), so this plot was able to release more nutrients to the soil than other mixed and pure plantations. TG also had the highest concentration of soil nutrients, i.e. total phosphorus and total nitrogen, whilst fluctuations for these nutrients were seen within T, TT, TTG and TTA (Figure 4.16). Average available phosphorus ranged from 0.92-1.14 mg/100g of soil, with TTA showing the highest value, and TG just 0.96 mg/100g of soil. Available phosphorus showed little variation between TTA and TG.

<u>Discussion</u>

Implications of soil nutrients on restoration strategies

It is envisaged that the growth of tree seedlings does not depend on a single soil property but the interaction of several. Mokhtaruddin et al. (1999) performed stepwise multiple regression analysis between growth and soil parameters and found that tree growth and biomass were strongly and positively influenced by bulk density and thickness of the A-horizon, and concluded that these two factors were limiting to the growth of young seedlings. The A-horizon provides a seat for the development of root systems and many plants tend to develop these surface root systems during the initial phase of root development. A well-developed root system in

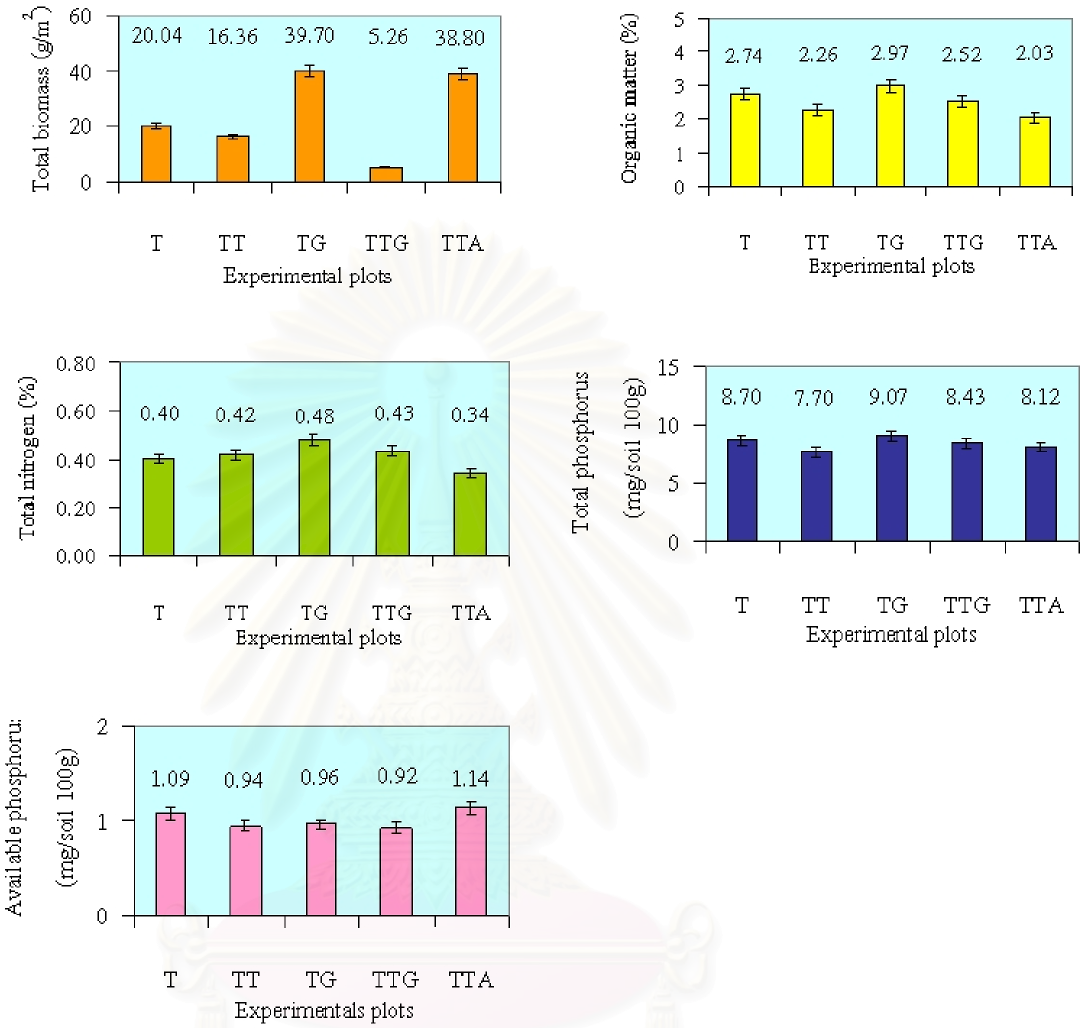

Figure 4.16: Average total biomass, average organic matter, average total nitrogen, average total phosphorus and average available phosphorus in five multi-purpose species plantations from 1988-1995.

the A-horizon also influences the rate of water permeability into deeper soils and therefore affects water availability in the root zone (Mokhtaruddin et al., 1999). Organic matter is an important factor linked to the quality of soil and its nutrients. For example, Alegre, Cassell and Bandy (1986) and Craul (1995) reported that organic matter affected the fertility of soil through its influence on many other properties such as water holding capacity, soil structure and nutrient availability. Barber and Romero (1994) and Kobayashi, Yarwudhi and Suksawang (1996) indicated that poor establishment of seedlings during rehabilitation of logged-over forest was attributed to a decrease in organic matter due to forest harvesting.

The different rates of survival and productivity for the planted species in the five multi-purpose plantations are associated with their response to soil properties, e.g. soil pH, water content, N-mineralization and foliar nutrient composition. However, this study did not survey soil properties or nutrient content, since several previous surveys have shown different rates of nutrient accumulation within the five plantation types. Research has also been conducted in other regions, such as the temperate zone, to explore plantation productivity. In Canada for example, a strong relationship between soil pH and tree growth was recorded for Red Oak (*Quercus rubra* L.), whilst foliar Mg content was important for Bur Oak (*Quercus macrocapa* Michx.) (Cogliastro et al., 2003). In another study, the presence of calcium carbonates at less than 30 cm from the soil surface appeared to be a primary limiting factor for Red Oak (Colin-Belgrand, 1994). Negative plant nutrition effects can also probably be attributed to high pH, which reduces the availability of P (Marschner, 1995). Cogliastro et al., (2003) concluded that a high level of nitrate availability in recently abandoned farmland should be profitable for early or mid-successional species in southwestern Quebec, Canada.

Nutrients derived from leaf litter are also of critical importance to seedlings or regenerating understorey plants. The presence of leaf litter is a powerful tool for restoration. For example, in a report on the relative importance of the litter layer in plantation nutrient conservation, Lugo et al. (1990) revealed that the amount of litter in 10 tropical plantations ranged from 5-28 Mg ha^{-1}. Thus, differences in plantation litter may emphasize its importance to nutrient cycling. Byard, Lewis and Montagnini (1996) suggested that green mulch from decomposing indigenous leaf litter could be a low-cost and effective soil supplement. In the La Selva region, northern Costa Rica, small-scale farmers will remove litter from the plantations to use in their fields or home gardens or to otherwise aid in the growth of subsistence crops. Research conducted on plantation floor litter and nutrient accumulation in pure and mixed plantations of indigenous tree species in Costa Rica, found that during the peak period, litter in pure stands of *H. alchorneoides*, *V. ferruginea* and *G. americana* contained 40% of the nitrogen found in aboveground biomass, whilst litter in the mixed stand had 45%. Furthermore, in the pure stand of *G. Americana*, the litter contained 54% of the K, and 56% of the Mg found in the aboveground biomass (Stanley and Montagnini, 1999).

Moreover, studies have indicated that patterns of understorey diversity and nutrient accumulation are influenced by age and life-history traits in plantations of indigenous species (Hagger, Wightmann and Fisher, 1997, Stanley and Montagnini, 1999). Results of the present study clearly confirm that differences in soil nutrient concentration regulate the productivity of enrichment seedlings in the recovery process. Analysis of soil nutrients clearly indicated that the two-species combination TG had the highest concentrations of both total phosphorus (0.48%) and total nitrogen (9.07 mg/100g soil). The three-species plantation showed a low soil nutrient concentration and a low productivity for enrichment seedlings, similar to the pure plantation. Montagnini (1994) noted that fast decomposition rates affected the release of nutrients from leaf litter to soil, which led to accumulated productivity in enrichment seedlings. She suggested that mixed-species designs could be more advantageous than monocultures for site nutrient rehabilitation if systems were planned so as to complement each species' nutrient demands and effects. This is because mixed litter had differing rates of decomposition and released organic matter over a continuous and synchronized period. In addition, Byard, Lewis and Montagnini (1996) suggested that mixed litter provided intermediate decomposition rates, thus allowing the litter layer to protect the soil. Therefore, the high potential of nurse tree species to aid rehabilitation, via their direct impact on soil fertility of degraded areas, was obvious.

The chemical components of litter also affect decomposition rates (Palm and Sanchez, 1991; Constantinides and Fownes, 1994). A study of leaf litter in mixed-species plantations indicated that of four species studied, each had different decomposition rates; 120 days for Gmelina (*G. arborea*), 146 days for Teak (*T. grandis*), 204 days for Cashew (*A. occidentale*) and 277 days for Tamarind (*T. indica*) (Saeheng, 1993). This indicated that the leaf litter of Teak and Gmelina had a faster decomposition rate, thereby releasing nutrients to the soil faster than those of Cashew and Tamarind. In terms of restoration, the continuous and synchronized release of nutrients to the soil from mixed tree plantations is useful for enrichment planting techniques. This is because the nutrients can increase the ability to convert highly degraded land back to near-natural forest through successful establishment and persistence of enrichment seedlings. The advantages of mixed plantations on nutrient accumulation via litter decomposition have been discussed in other reports. For example, a mixed plantation of *Terminalia amazonia* (Gmell.) Exell. and *Virola koschnyi* Warb. had high foliar Ca content and high rates of annual litter fall (*Terminalia amazonia*: 853 g/m^2, *Virola koschnyi* 620 g/m^2). The plot exhibited consistent litter fall and decomposition throughout the year, and was a good mulch (Kershnar and Montagnini, 1998). Montagnini (2000) observed beneficial effects on soil nutrients under single stands of some species; therefore characteristics of nutrient cycling must also be taken into account when assessing the potential impacts of plantation species on site nutrients. Mixed plots showed intermediate values for the nutrients examined, and even improved soil conditions, as for P in a plantation of *Stryphnodendron microstachyum* Poepp. Et Endl., *Vochysia guatemalensis* D.Sm., *Jacaranda copaia* (Aubl.) D. Don, and *Callophylum brasiliense* Cambess. Thus, in mixed plantations it may take longer to deplete soil nutrients than in mono-specific stands of fast-growing species. Likewise, Byard, Lewis and Montagnini (1996) reported that mixed designs provided intermediate to fast decomposition rates, releasing nutrients to the soil and allowing a litter layer to protect the soil. Therefore, as with the present study, the results indicated that mixed stands encouraged nutrient cycling in understorey, and that knowledge of the nutrient cycling characteristics of enrichment species could help in the selection of management strategies to conserve site nutrients.

Litter decomposition rates are determined by climate, chemical composition of the litter and soil organisms (Swift, Heal and Anderson, 1979). In the humid tropics, where climatic variation tends to be less extreme, much research has indicated that litter chemistry and soil organisms play an increasingly important role in decomposition processes (Tian et al., 1993; Gonzalez and Seastedt, 2001; Gonzalez et al., 2001). Hence, climatic factors and litter chemistry are considered to be the most important factors in the regulation of decomposition rates. Soil invertebrates are another important factor related to decomposition processes. A report by Warren and Zou (2002) indicated that soil macro-invertebrates were associated with litter quality more than litter quantity. Overall forest floor mass did not vary, and litter quantity had no significant correlation with the abundance of soil macro-invertebrates. The quality of the litter substrate has been shown to significantly affect earthworm populations in tropical regions (Tian, Brussaard and Kang, 1995; Gonzalez and Zou, 1999). Furthermore, earthworms, which are the most important faunal species in tropical soils, encourage decomposition and may respond to plantation species over a sufficient period of time, when litter input and differing chemistry affect physical and chemical properties of the soil, such as moisture, pH and organic matter content (Warren and Zou, 2002). Although, this study did not investigate soil fauna, different rates of nutrient accumulation were observed between the five multi-species plantations. One factor that was known to affect nutrient concentrations was litter decomposition rate. However, different species have different litter quality, which may directly affect soil faunal abundance in terms of both species and number. Therefore, decomposition rates depend on litter quality and the presence of soil invertebrates. The quality of litter, and soil invertebrate populations may be an important factor regulating the turnover of nutrients on forest floor, emphasizing the importance of biological systems in the regulation of decomposition processes.

The multi-purpose species plantations were subjected to almost annual wildfire disturbance. Hence, fire was an important factor in this area, releasing nutrients from the litter to soil. This may have had strong influences upon the plant community. Many factors may contribute to maintenance of nutrient levels after burning. As in natural primary forests, multi-species plantations (which are a modified agroforestry system - AFS) have a dense, deep and permanent network of roots, contributing to more efficient nutrient cycling and prevention of losses due to leaching, which are often the main cause for a loss in soil fertility. Tree canopies and leaf litter also protect the soil against erosion and high temperature. Numerous studies have shown an increase in nutrients incorporated into soil from the ashes of slash and burn practices. For example, chemical analysis of soil in three land-use systems in Rodonia, Brazil revealed that even 10 years after the burning of the primary forest for the establishment of agroforestry systems (AFS), the expected increment in soil pH, a reduction in the level of exchangeable Al and increases in the level of exchangeable Ca and Mg were still to be seen. That these higher levels of soil fertility were still apparent after 10 years is remarkable (Alfaia et al., 2004). Recco et al. (2000) observed that older AFS showed a trend in recovery and maintenance of organic C, similar to that in primary forest in western Amazonia. Seubert, Sanchez and Valverde (1977) calculated that the burning of primary vegetation incorporated 67 kg ha^{-1} of N, 6 kg ha^{-1} of P, 38 kg ha^{-1} of K, 75 kg ha^{-1} of Ca and 16 kg ha^{-1} Mg into the soil.

Upperstorey trees have a role as reservoirs for carbon, phosphorus and other elements, particularly soil nitrogen. TG was markedly enriched in total nitrogen and total phosphorus relative to other multi-purpose species plantations. The rise in soil nitrogen and phosphorus may have been related to the efficiency of organic matter decomposition, suggesting that this type of mixed species plantation is optimally suited to conserve and maximize nutrient gains, thus increasing growth rate and productivity more than in other stands. Not only do the increased soil nutrients encourage the growth of enrichment seedlings, but they also provide a favourable microhabitat for germination of natural regenerative plants.

Agroforestry systems do however suffer from nutrient loss due to harvesting. Cravo and Souza (1996) found that the export of K, a macronutrient, in cupuassu fruits was approximately 5 $kgMg^{-1}$; therefore, the annual harvest of cupuassu fruits exported approximately 20 kg of K ha^{-1} per year. The majority of the AFS plantations at the Economic Mixed and Dense Reforestation Project (RECA) require K replacements and would probably show a positive response to K fertilization. But any technology needed to recuperate soil productivity must be economically viable since most smallholders do not have the economic means to practise high input agriculture. Multi-species plantations and mixed AFS plantations may be an alternative. The addition and mineralization of organic matter such as cupuassu fruit rinds, which are rich in K, N and P, pejibaye leaves, which are rich in N, or nitrogen input from leguminous trees such as Tamarind, can be alternative sources of organic manure.

The input of mineral nutrients is an external factor that should be recommended for the future restoration strategies in highly degraded land. However, our study did not focus on this idea, due to the limited experimental plantations and the need to explore the effects of the plantations on productivity of planting seedlings. Therefore, we could not use more fertilizer in this study. Nevertheless, the effects of nutrient input, as well as the costs involved should be explored in other restoration projects. A study by Singh, Jha and Singh (2000) revealed that nutrient enrichment with NPK fertilizer positively promoted the growth of native woody species, including leguminous species (*Acacia catechu*, *Albizia lebbeck*, *Dalbergia sissoo* and *Pongamia pinnata*) and non-leguminous species (*Azadirachta indica*, *Gmelina arborea*, *Phyllanthus emblica*, *Tectona grandis* and *Terminalia bellirica*) in an area that had been disturbed by coal mining. However, the response of the trees to NPK was variable from species to species. The impact of NPK fertilization was comparatively greater for non-leguminous tree species than leguminous species. Nitrogen-fixing species may have overcome the nitrogen deficiency by virtue of their nitrogen fixing ability. Hence, they may remain unaffected by N fertilization but may respond to other fertilizers. Likewise, a study by Tanner, Kapos and Franco (1992) indicated the addition of nutrients resulted in increased growth and productivity of woody species. Singh, Jha and Singh (2000) suggested that the necessity of using chemical fertilizers should depend on the species planted. Finally, since nutrient input accelerates plant growth, it will also help improve the success of restoration, particular in seriously degraded habitats and in plantations that use non-nitrogen-fixing species for enrichment planting, such as in this study.

4.4 INTEGRATED DISCUSSION ON MULTI-PURPOSE SPECIES PLANTATIONS, FLORISTIC REGENERATION, SEEDS AND RESTORATION

In the past, most plantation forests in northern Thailand were established for the purpose of producing commercial Teak timber. However in recent years, there has been an increased appreciation for the value of other native forest species, both for commercial utilization and for preservation of biodiversity. Future plantation managers may wish to establish plantations that will enhance biodiversity as well as increase timber production. Plantations are a good management technique to help restore highly degraded land. The re-establishment of native mixed deciduous forest with Teak may be one of the most important issues that need to be tackled in northern Thailand. It is important for conservation because it would help preserve biological diversity. It is important economically because it has the potential to generate income, especially from non-timber products. To a restorationist, restoration is an attempt to imitate and control succession. Thus, active restoration of a community draws attention to critical factors in the process, especially those that have been improperly handled, and provides countless opportunities to identify and determine more precisely the role they play in the community (Jordan III, Gilpin and Aber, 1987).

Several positive interactive benefits can arise from multi-species systems. However, not all advantages occur in all projects. Possible positive benefits that may arise from multi-species plantations have been suggested by Beer (1987). They include,

1. Stabilized output that permits more efficient use of labour and machinery.
2. Suppression of weeds.
3. Product diversification.
4. Reduced evapotranspiration.
5. Lowered excess soil moisture.
6. Increased moisture input.
7. Reduced temperature extremes.
8. Reduced rain damage.
9. Reduction in disease.
10. Reduced wind.
11. Improved drainage.
12. Provision of mulch.
13. Reduced erosion.
14. Reduced decomposition.
15. Recycling of nutrients.
16. Fixation of nitrogen.
17. Decreased need for chemicals.

Social demands for increased diversity in plantations may also make mixed-species plantations more attractive in the future. The low number of case studies that have been conducted cannot provide the comprehensive insight needed for operational-scale use of mixed-species plantations; many other combinations of species, sites and silvicultural practices could be profitable. Thus, a coordinated, international set of experiments needs to be developed to provide an information base that will allow forest managers to make informed and effective decisions about the overall value of mixed-species and pure plantations.

As well as providing a rehabilitative and sustainable forest ecosystem on degraded areas, multi-purpose plantations can play an important role in restoring productivity, ecosystem stability, and biological diversity (Parrotta, 1992). Therefore, a developed and diversified mixed-species plantation, such as that established by Gajaseni (1988) in Lampang Province, northern Thailand, is an alternative strategy to restore degraded forest areas in northern Thailand. Fifteen years after planting, both tree basal area and canopy cover were significantly higher in the two-species plantation (TG) than in single-species plantation plots. Thus the two species in the TG plot had positive interactions with each other. Teak and Gmelina made a good combination and the plantation canopy may have altered the understorey microclimate and the physical and chemical environment of the soil. Many studies on the development of canopy have indicated facilitation in recruitment, survival and growth of native forest species (Uhl et al., 1982; Soni, Vasistha and Kumar, 1989; Montagnini, 1994; Byard, Lewis and Montagnini, 1996). Guidelines based on these results recommend that mixed-species plantations of Teak and Gmelina, which are fast growing species, should be developed in degraded areas in order to restore soil fertility and facilitate floristic reestablishment in the region. There have been several studies on forest restoration, such as the framework species method (Goosem and Tucker, 1995) and the catalytic monoculture (Lugo, 1997).

Both mono-specific and mixed-species plantations, with either native or exotic species, can help improve soil fertility and change the understorey microclimate to favour natural successional processes. Our study indicated that there was greater floristic species diversity in the understorey of three-species plantations than in those of single or two-species plantations. The diversity of understorey flora may have helped reduce the leaching of nutrients and thus conserve nitrogen within living biomass, decomposing litter and soil organic matter (Parrotta, 1992). When studying only woody life forms, the three-species plantation still showed the highest diversity. This suggested that the three-species mixed plantation was favourable to occupation and establishment of native woody species, and was thus useful for accelerating successional processes. Nevertheless, this plantation had a low abundance and diversity of dominant primary tree species found in the natural forest, so alternative techniques for restoration should be applied for use in this study.

A high floral diversity in the understorey of mixed-species plantations has been observed in other areas (Guariguata, Rheingans and Montagnini, 1995; Parrotta, 1999, Carnevele and Montagnini, 2002). Factors that may impede the establishment of new seedlings in pure plantations may be an invasion of herbaceous vegetation (Carnevele and Montagnini, 2002), differing levels of available light (Guariguata, Rheingans and Montagnini, 1995; Powers, Haggar and Fisher, 1997), and varying nutrient concentration in the soil. Moreover, mixed plantations have low levels of herbivores (Montagnini et al., 1995). The potential to inhibit weeds was clearly shown by our research.

Our research indicated that mixed-species plantations affected species composition and density of, in particular, woody species and to a lesser degree non-woody species of the understorey. However, as only a few primary dominant tree species were naturally present, they were insufficient to stimulate natural recovery processes. This suggests that mixed-species plantations may be more effective in achieving this goal than pure stands.

“A basic tenet of succession theory is that certain species serve as pioneers and give way to succeeding species as site conditions and competitive relations are altered. Virtually no experiments have been carried out specifically to determine the direction of change or the forces influencing it. Under these circumstances forest restoration projects can provide some of the best information about this process, even though they may not have been set up specifically as experiments to test ideas about succession" (Ashby, 1984).

In the case of this study, due to the successional processes that occurred after disturbance, tree and shrub seedlings colonized and established themselves in the understorey of five mixed-species plantations in the northern region of Thailand. Results showed that only a few species, and a low abundance of dominant canopy trees, occurred beneath these plantations. Therefore, this was one reason for using enrichment techniques, in order to accelerate the successional process. The establishment of a few pre-climax species during early and/or intermediate succession indicated that secondary succession did not necessarily have to start with pioneer species. This fact can be used to adjust restoration strategies by bypassing some of the earlier stages of succession, and introducing pre-climax species at an earlier stage. By monitoring seed distribution, it was observed that there was limited seed dispersal into the degraded plantation areas and that this was another reason for human intervention in forest restoration. Research in many abandoned areas has suggested that limitations of seed input for both seed dispersal and soil seed bank have negative effects on species composition, structure and dynamics of forest regeneration (Zimmerman, Pascarella and Aide, 2000; Wijdeven and Kuzee, 2000; Webb and Peart, 2001; Dalling, Hubbell and Silvera 1998; Masaki et al., 1998). If considering only seed distribution, then dispersal by animals has the potential to help restore forest plant diversity on degraded sites in a reasonably short period of time, thereby helping to defray restoration cost. In terms of restoration strategy, Wunderle (1997) suggested that restoration sites should ideally be contiguous with the native forest seed source. If not possible, consideration should be given to the development of forest or plantation corridors through which forest-dwelling seed dispersers might pass to the restoration site. Another option might be to consider an "archipelago" of small restoration patches scattered with intervening open areas less than 50 m across. Thus, the important management concern here is to reduce the site's isolation from seed sources and to ensure adequate seed rain. Options that are available to make plantations attractive to seed dispersers include,

1. Providing a perching or roosting site that has the potential to attract at least some seed dispersers.

2. Increasing the vegetative complexity of the site (i.e. a mixed plantation) increases the attractiveness to more animal species, thereby improving the likelihood of seed dispersal by generalist or opportunistic frugivores.

3. A differential use of plantation edges by seed dispersers, since seedling colonization rates on the periphery of plantations tend to be higher than those in the interior (Parrotta, 1995).

Consideration should also be given to planting fruit-bearing trees, in order to attract seed dispersers into plantations and accelerate seed dispersal and enrich diversity. As demonstrated by Parrotta (1995), species composition of the overstorey can have substantial effects on recruitment rates in the understorey. Attractiveness to potential animal seed dispersers is based on the availability of resources. The diversity and abundance of wildlife in monocultures that lack such resources can be exceptionally low. Fruiting plants attract frugivores to a site, and the seeds that they deposit represent a diversity of plant species. Obviously, care should be taken in selecting appropriate plant species because the seeds of an attractive plant are likely to be thoroughly dispersed throughout the plantation, and if aggressive, the species has the potential to dominate the plantation. Due to the relative immobility of primary forest plants, often characterized by large seeds, Wunderle (1997) suggested that managers cannot rely on animal dispersers to provide adequate dispersal. Therefore, managers should consider planting or seeding these species in restoration sites, if a full return to primary forest diversity is desired.

Many studies have demonstrated the ability of pioneer tree and shrub species (Parrotta, 1995; Sou, 2000) and seedlings of pre-climax canopy species (Ashton et al., 1997) to establish in highly degraded areas. However, the succession theory predicts that late-succession species will tend to fail when introduced into a disturbed or open site. In fact, this is generally the case. For example, late-successional hardwoods generally fail when introduced directly into open sites. These failures are due to unfavourable water and nutrient relations or occasionally to animal damage that might have been ameliorated by pioneer species. However, there have been exceptions, particularly for hardwoods in multi-species plantations. The findings of this study indicated a high survival rate (more than 80%) of selected canopy and slow-growing species. This demonstrated that seedlings of slow-growing canopy species could also become established in highly degraded areas, such as that beneath mixed-species plantations, since the plantations acted as nurse trees. The high survival rate of canopy tree seedlings indicated that they could easily adapt to site conditions in the understorey of mixed plantations due to the improved microclimate and favourable environment conditions that may be related to porosity, since well-drained coarse soils weather to release mineral nutrients. Therefore, the high rate of seedling survival found in this study suggests that selected canopy tree species can be used for restoration at certain sites.

The study of biomass accumulation by enrichment seedlings in diversified multi-purpose species plantations found that TG had the highest increment in total biomass. Therefore, good growth and positive interactions between Teak and Gmelina affected the growth rates of seedlings. Nutrient analysis of the soil can support this, as it was found that TG had the highest concentration of total nitrogen and total phosphorus. Thus the higher nutrient concentration in TG contributed to the high growth rate of seedlings. Moreover, good tree growth in the TG combination provided more litter accumulation in the understorey and favourably altered the microclimate environment for seedlings. Therefore, these results suggest that the planting of slow-growing species between gaps in multi-purpose species plantations can be applied as an alternative restoration technique. Successful species combinations, such as Teak and Gmelina, can ameliorate site conditions and increase the diversity of the understorey with a corresponding increase in canopy shading and litter accumulation.

Restoration ecology in this region is in the early stages of development, with only basic studies available to provide guidance for restoration of degraded landscapes. Specific studies, in order to adapt management practices to local conditions, such as the effects of plantation size, composition, age and isolation from animal seed dispersers on enhancing diversity on degraded sites are recommended. Parrotta, Turnbull and Jones (1997) suggested that restoration strategies in areas other than those for which information is currently available, needs careful selection of sites. Potential managers must determine distance from the nearest natural forest remnants and give priority to sites advantageously disposed to seed vectors. Candidate plantation species should be screened for their potentiality to become problematic weeds in relation to local and regional flora and possible invasion of surrounding ecosystems. If possible, native species should be given preference over exotics, and while most species appear to act as catalysts, broadleaf species seem to be preferable. Mokhtaruddin et al. (1999) suggested that to improve growth and seedling establishment in restoration projects, it is necessary to adopt silvicultural practices that can enhance soil properties. For example, the size of planting holes can be enlarged (e.g. 60 cm x 60 cm x 50 cm) and a mixture of topsoil, organic compost and fertiliser used to refill the holes after planting.

In deciduous forest ecosystems where rates of deforestation are particularly high, and where formerly forested landscapes have been converted into a mosaic of remnant stands, forest restoration has become an increasingly important tool for species preservation and conservation. A multi-disciplinary approach provides one of the best ways of developing a working restoration strategy in order to better understand and recreate such complex ecosystems.

Habitat destruction is the most prevalent cause of species endangerment (Wilson, 1992) and increases in highly degraded lands and overexploited forest. Forest recovery requires techniques such as restoration in order to accelerate this process. Restoration is a field encompassing many techniques suitable for rehabilitating a wild variety of degraded ecosystems (Jordan III, Gilpin and Aber, 1987). A common approach to restoring degraded areas is tree planting or enrichment planting. A major problem with this approach is the expense. However, many restoration projects have been successful, and at low cost, by taking advantage of the services of nature (Jordan, 1995). In northern Thailand, local people and managers are seeking guidelines for selection of species and strategy. The results of the present study provide insights into the use of enrichment species at the restorative stage in highly degraded areas. These results can be modified for use in other areas with similar conditions. Restoration, which is an important and growing discipline within the field of conservation, aims to re-establish or rehabilitate damaged or lost plant and animal populations or species assemblages native to the area of interest (Jordan III, Gilpin and Aber, 1987). This is a conceptual issue in restoration. The planning of restoration with a landscape approach may facilitate the recruitment of organisms and play a decisive role in its success.

Moreover, restoration represents a long-term dedication to natural resources and a substantial financial commitment. Therefore, practitioners need to understand how to allocate this limited funding, labour and time for maximum effect in the design of restoration programs for large-scale projects. Nevertheless, the major goal in any restoration project is the conservation of biological diversity, and ultimately *global* biodiversity. The major threat to world biodiversity comes from habitat destruction, thus preservation of habitat and restoration of highly degraded areas may best achieve the conservation of biodiversity and ecosystem functions.

CHAPTER V

SUMMARY AND RECOMMENDATIONS

5.1 SUMMARY

Forest degradation, especially mixed deciduous Teak forest, which is the dominant forest type in northern Thailand, became serious problem. Plantation management is a strategy to improve degraded land. In the past most plantation forests in northern Thailand have been established for the purpose of Teak timber production. However, in recent years, there has been an increased appreciation for the value of other native forest species, both for commercial utilization and for the preservation of biodiversity. Future plantation managers may wish to establish plantations that will enhance biodiversity as well as wood production. The Forest Village System (which is a modified version of the Taungya system) has been set up in this area in response to the need to produce high-value timber. In order to fulfil the need to preserve biodiversity and ameliorate degraded land, this research has developed a restoration strategy to accelerate natural successional processes and convert highly degraded areas back to natural forest or near-forest communities by use of the Forest Village System.

The aim of this research was to investigate regeneration and dynamics of vegetation, in particular woody species, in the understorey of diversified multi-purpose tree species plantations. Four main aspects were investigated; (i) to study the diversity and species composition of native species in the understorey of multiple tree plantations, (ii) to investigate which native forest species are most successful in establishment within the plantations (iii) to monitor both seed dispersal from adjacent natural forest to mixed tree species plantations and the soil seed bank and (iv) to understand the effect of different multiple tree species plantations on growth rate and survival of enrichment seedlings.

The purpose of this research was to test three hypotheses. (i) Single-species plantations provide more available resources for regenerative species than mixed-species plantations. Therefore, the single-species plantation has greater floristic diversity in their understorey than the mixed-species plantation. (ii) Due to limitations of seed dispersal and seed bank distribution from adjacent natural forest to multi-purpose species plantations, species composition and density of woody tree seeds in multi-purpose species plantations is low. (iii) Three-species plantations provide more resources and have more favourable understorey conditions than single-species plantations. Therefore, mixed-species plantations can maintain and enhance the survival rate and productivity of enrichment species to a higher level than the single-species plantation.

With reference to species composition and species diversity, the data indicated that the three-species plantation had a higher floristic diversity of both woody and non-woody species in their understorey than the single-species plantation. However, focusing only on the density of grasses, the mixed-species plantation was also more effective than the single-species plantation in suppressing aggressive weeds and grasses. In addition, it was more effective in accelerating succession towards mature forest. This suggests that if the management goal is to regenerate forest with a high diversity of tree species similar to those in native mature forests, a multiple-species plantation is more effective than a single-species plantation in providing an environment into which seeds of native species will disperse and germinate. Furthermore, if the management strategy is to enrich plantations with seedlings of native forest species, mixed-species plantations are more effective nurse communities than single-species plantations.

The first hypothesis of this study is therefore not supported. Single-species plantations do not provide more available resources for regenerative species than mixed-species plantations and therefore, the single-species plantation does not have greater species diversity than the mixed-species plantations.

Results of seed inputs (both seed dispersal and soil seed bank) indicated that low levels of viable woody seeds were dispersed from adjacent natural forest to the mixed tree species plantations. Therefore, there was an apparent strong negative effect of distance from natural forest on species composition and density of woody species due to poor seed distribution. Moreover, the variation of woody tree seed products may have affected the structure and composition of the dynamics of vegetation in the understorey of multi-purpose tree species plantations. Furthermore, it was found that a few woody seed species, such as teak, had a great abundance in the mixed plantation, thus this species could act as an important recruitment species affecting structure and composition of vegetating recovery in the area. However, if the purpose of reforeststation is to preserve species diversity, then a high seed input of Teak may decrease species diversity, if they have the chance to germinate.

There was a low input of seeds to the mixed-species plantations from surrounding natural forest. Thus, the lack of seed was a serious limiting factor on the development of natural successional processes. In addition 65% of seeds showed a low seed germination rate especially those of dominant tree species. Based on the results of this study the second hypothesis was supported.

Due to the low seedling recruitment and seed input of dominant species, this suggests that an alternative restoration strategy such as enrichment-planting techniques should be selected to accelerate recovery processes. The mixed-species plantations had positive impacts on enrichment seedlings by improving nutrient concentration in the soil and understorey conditions. The pure plantation had high efficiency and the potential to increase the survival rate of seedlings planted in their understorey as well as mixed tree species plantations. Seedling survival was high in all plantation types. Whilst evaluating productivity of enrichment seedlings, the mixed plantation of Teak and Gmelina helped in drawing up specific management recommendations. They made a good combination and preserved the forest floor, which was a key area for nutrient recycling. Furthermore, their canopy favourably changed understorey conditions to facilitate seedling growth.

The results show that the two-species plantation of Teak and Gmelina may be more effective in achieving restoration goals than other plantation stands in this region. Therefore, the results of survivorship and productivity do not support the third hypothesis. The mixed three-species plantation does not encourage growth and increased productivity of seedlings to a higher degree than double or pure plantations.

In conclusion, mixed plantations were found to be suitable for reducing the density of *Imperata* grass. This therefore affected species establishment and diversity of regenerative plants in the understorey. The research indicated that high species diversity and high woody density of trees and shrubs occurred in the understorey of, especially, TG and TTA. Hence, these particular mixed-species stands were most effective in accelerating succession. This suggests that if the goal of management is to regenerate forest with a high diversity of tree species, similar to those in native mature forests, a multiple-species plantation (TG and TTA) is more effective than a single-species plantation in providing an environment into which seeds of native species will disperse and germinate. Due to low dominant native tree seedling recruitment and seed sources in mixed plantations, enrichment planting is a useful technique in the restoration strategy. In addition, this research clearly shows that enrichment planting by pre-climax species has the capability of increasing restoration mechanisms by accelerating successional processes.Furthermore, if the management strategy is to enrich plantations with seedlings of native forest species then, mixed-species plantations are more effective nurse communities than single-species plantations.

5.2 RECOMMENDATIONS

Future studies need to solve the serious problem of forest degradation in Thailand and other areas by use of alternative restoration strategies. The implication of ecological and economical parameters should be investigated.

- Cost-benefit analysis is a significant indicator. It could be used to evaluate the efficiency of restoration strategies by enrichment-planting techniques compared with alternative approaches. The results of this evaluation could be used for adjusting and improving restoration techniques for the maximum benefit, especially when expanding to a larger scale.

- To measure the success of restoration strategies, socioeconomic factors should be monitored. Socioeconomic evaluation as a parameter indicates the stability of the system, and the efficiency and sustainability of the ecosystem.
- The limited time scale of the study did not allow for sufficient investigation of the dynamics of the ecosystem. Variation in seed production and ecology may affect floristic recruitment. Therefore, a longer time period would help clarify this point.
- Seed germination tests for each species should be performed. The results can be used to select seedlings used in restoration strategies. Species with a high germination rate should be selected.

APPENDICES

APPENDICES A

Appendix A.1 Geographic Information systems (GIS) data for 15 experimental plots in five different mixed tree species plantations.

Plot	Universal Transverse Mercarter (UTM)		
	East	North	Z
A1	577129	2037815	0
A2	576945	2037984	0
A3	576859	2038033	0
B1	576818	2038043	0
B2	576803	2038019	0
B3	576766	2038033	0
C1	576996	2037994	0
C2	576996	2037994	0
C3	576715	2038043	0
D1	577058	2037888	0
D2	577058	2037888	0
D3	577016	2037951	0
E1	577204	2037788	0
E2	577176	2037850	0
E3	576835	2038032	0

Appendix A.2 Average meteorological data at Mae Moh plantation station in Lampang province, northern Thailand, 1997-2000

Month	Mean Temp.	Mean rainfall
Jan.	23.05	1.48
Feb.	24.46	2.90
Mar.	28.43	30.53
Apr.	29.63	118.03
May.	28.97	169.90
Jun.	28.01	163.03
Jul.	27.92	153.53
Aug.	27.92	210.90
Sep.	26.81	252.79
Oct.	26.25	114.50
Nov.	24.62	14.67
Dec.	23.90	1.80

Appendix A.3: Average density of trees and saplings in natural mixed deciduous forest with Teak in Lampang province, northern Thailand (Individuals/ha).

No	Scientific name	Family	Plant habit	Average density for Trees	Average density for Saplings
1	*Xylia xylocarpa var. kerrii*	Mimosoideae	T	67	39
2	*Croton roxburghii*	Euphorbiaceae	S	39	150
3	*Shorea siamensis*	Dipterocarpaceae	T	89	-
4	*Tectona grandis*	Labitae	T	94	50
5	*Pterocarpus macrocarpus*	Papilionoideae	T	172	89
6	*Sterculia guttata*	Sterculiaceae	T	11	17
7	*Sterospermum cylindricum*	Bignoniaceae	T	6	6
8	*Grewia eriocarpa*	Tiliaceae	T	78	50
9	*Harrisonia perforata*	Simaroubaceae	S	3	-
10	*Largerstroemia floribunda*	Lythraceae	T	64	61
11	*Albizia lebbeck*	Mimosoideae	T	14	144
12	*Millettia brandisiana*	Papilionoideae	T	11	44
13	*Wrightia arborea*	Apocynaceae	T	36	122
14	*Schleichera oleosa*	Sapindaceae	T	19	6
15	*Morinda coreia*	Rubiaceae	T	14	6
16	*Dalbergia sp.*	*Papilionoideae*	-	3	6
17	*Chukrasia tabularis*	Meliaceae	T	8	11
18	*Lannea coromandelica*	Anacardiaceae	T	67	22
19	*Dalbergia negrescens*	Papilionoideae	T	3	17
20	*Bambax aanceps*	Bombaceae	T	8	11
21	*Terminalia mucronata*	Vombretaceae	T	28	22
22	*Vitex peduncularis*	Labiatae	T	22	44
23	*Artocarpus lacucha*	Moraceae	T	11	11
24	*Canarium subulatum*	Burseraceae	T	28	50
25	*Sterculia vilosa*	Sterculiaceae	T	39	44
26	*Randia longispina*	Rubiaceae	S	14	-
27	*Nephelium hypoleucum*	Sapindaceae	T	56	-
28	*Mitragyna rotundifolia*	Rubiaceae	T	11	-
29	*Buchanania lanzan*	Anacardiaceae	T	3	6
30	*Markhamia kerrii*	Bignoniaceae	T	3	17
31	*Antidesma ghaesembilla*	Euphorbiaceae	S	31	78
32	*Phyllanthus emblica*	Euphorbiaceae	T	3	-
33	*Careya sphaerica*	Lecythidaceae	T	6	-
34	*Dalbergia cultrata*	Papilionoideae	T	11	50
35	*Strychnos nux-vomica*	Strychnaceae	T	3	6
36	*Phyllanthus cf. orienthus*	Euphorbiaceae	S	19	178
37	*Aporusa vilosa*	Euphorbiaceae	T	14	17
38	*Holarrhena antidysenterica*	Apocynaceae	T	6	-
39	*Croton longissimus*	Euphorbiaceae	S	3	122
40	*Dalbergia cana*	Papilionoideae	T	3	-
41	*Irvingia mayalana*	Irvingiaceae	T	6	-
42	*Diospyros ehretioides*	Ebenaceae	T	3	6
43	*Dipterocarpus obtusifolius*	Dipterocarpaceae	T	33	17
44	*Dalbergia oliveri*	Papilionoideae	T	6	12
45	*Berrya cordifolia*	Tiliaceae	T	8	11
46	*Spondias pinanta*	Anacardiaceae	T	6	-
47	*Milientha suavis*	-	-	6	6
48	*Shorea obtusa*	Dipterocarpaceae	T	19	22
49	*Dipterocarpus tuberculatus*	Dipterocarpaceae	T	3	-
50	*Hymenodictyon orixense*	Rubiceae	T	3	6
51	*Dalbergia ovata*	Papilionoideae	T	3	167
52	*Albizia odoratissima*	Mimosoideae	T	3	50
53	*Largerstroemia macrocarpa*	Lythraceae	T	6	-

Appendix A.3 (Continuted)

54	*Terminalia calamansanai*	Combretaceae	T	6	-
55	*Diospyros sp.*	Ebenaceae	T	-	11
56	*Antidesma bunius*	Euphorbiaceae	T	-	22
57	*Garuga pinnata*	Burseraceae	T	-	6
58	*Terminalia nigrovenulosa*	Combretaceae	T	-	11
59	*Cratoxylum formusum*	Guttiferae	T	-	22
60	*Ficus hispida*	Moraceae	T	-	6
61	*Litsea glutinosa*	Lauraceae	T	-	6
62	*Gmelina arborea*	Labiatae	T	-	6
63	Unidentified sp 1	-	-	-	22
64	Unidentified sp 2	-	T	22	78
65	Unidentified sp 3	-	-	3	6
66	Unidentified sp 4	-	-	3	-
67	Unidentified sp 5	-	-	3	-
68	Unidentified sp 6	-	-	3	-
69	Unidentified sp 7	-	-	-	6
70	Unidentified sp 8	-	-	-	6
71	Unidentified sp 9	-	-	-	17
72	Unidentified sp 10	-	-	-	6
73	Unidentified sp 11	-	-	-	6
74	Unidentified sp 12	-	-	-	6
	Total			1,265	2,036

Appendix A.4: Density of woody species in five multi-purpose species plantations in Lampang province, northern Thailand. (T= Tree, S= Shrub and C= Climber)

No	*Scientific name*	Family	Plant habit	Average density (Individuals/125 m^2)				
				T	TT	TG	TTG	TTA
1	*Pterocarpus macrocarpus*	Paplionoideae	T	14	11	3	15	18
2	*Dalbergia ovata*	Paplionoideae	T	19	11	98	29	6
3	*D. cana*	Paplionoideae	T	28	83	49	40	93
4	*Tectona grandis*	Labiatae	T	7	26	7	4	14
5	*Cratoxylum formusum*	Guttiferae	T	78	144	100	60	58
6	*Antidesma ghaesembilla*	Euphorbiaceae	S	22	29	58	23	21
7	*Mitragyna rotundifolia*	Rubiaceae	T	4	3	1	3	11
8	*Barringtonia acutangula*	Tiliaceae	T	2	4	4	11	4
9	*Wringhtia pubescens*	Apocynaceae	T	10	10	2	13	3
10	*Pterospermum semisagittatum*	Sterculiaceae	T	1	1	-	-	1
11	*Millettia brandisiana*	Paplionoideae	T	-	4	-	-	-
12	*Largerstroemia duperreana*	Lythraceae	T	4	3	6	1	7
13	*Lannae coromandelica*	Anacardiaceae	T	2	4	2	3	2
14	*Antidesma bunius*	Euphorbiaceae	T	4	5	7	3	5
15	*Albizia lebbeck*	Mimosoideae	T	7	9	-	8	1
16	*Grewia eriocarpa*	Tiliaceae	T	1	3	2	1	3
17	*Markhamia stipulate*	Bignoniaceae	T	4	2	4	3	1
18	*Litsea glutinosa*	Lauraceae	T	3	2	1	1	-
19	*Vitex peduncularis*	Labitae	T	2	1	3	1	4
20	*Croton longgissimus*	Euphorbiaceae	S	52	159	162	117	143
21	*Berrya cordiforlia*	Tiliaceae	T	1	3	9	4	4
22	*Harrisonia perforate*	Simaroubaceae	S	1	1	24	16	112
23	*Xylia xylocarpa* var. *kerrii*	Mimosoideae	T	1	2	1	6	-
24	*Morinda coreia*	Rubiaceae	T	2	3	2	5	11
25	*Sterculia guttata*	Sterculiaceae	T	1	9	2	15	31
26	*Phyllanthus orientalis*	Euphorbiaceae	S	18	9	11	17	4
27	*D. cultrate*	Paplionoideae	T	13	30	35	11	43
28	*Diospyros ehretioides*	Ebenaceae	T	1	1	3	1	1
29	*Bombax anceps*	Bombaceae	T	3	1	2	2	2
30	*Artocarpus lacucha*	Moraceae	T	-	2	-	2	-
31	*Albizia odoratissima*	Mimosoideae	T	1	3	1	5	5
32	*Aporusa villosa*	Euphorbiaceae	T	8	6	30	3	3
33	*Holarrhena antidysenterica*	Apocynaceae	T	2	9	19	4	-
34	*Garuga pinnata*	Berseraceae	T	-	1	1	-	-
35	*Ficus hispida*	Moraceae	T	1	2	1	1	1
36	*Canarium subulatum*	Burseraceae	T	-	1	1	1	-
37	*Phyllanthus emblica*	Euphorbiaceae	T	1	1	5	-	1
38	*Chukrasia tabularis*	Meliaceae	T	1	1	1	1	1
39	*Gmelina arborea*	Labiatae	T	-	1	10	1	-
40	*Millettia pendula*	Paplionoideae	T	-	1	-	-	10
41	*Hymenodictyon orixense*	Rubiaceae	T	1	1	-	1	1
42	*Terminalia alata*	Combretaceae	T	-	-	1	-	1
43	*Cassia fistula*	Caesalpinioideae	T	1	-	2	1	-
44	*Nephelium hypoleucum*	Sapidaceae	T	3	-	8	1	2
45	*Memecylon scutellacum*	Melastomataceae	S	-	-	4	-	-
46	*Spondias pinnata*	Anacardiaceae	T	-	-	1	1	1
47	*Berrya mollis*	Tiliaceae	T	1	-	1	-	1
48	*Terminalia mucronata*	Combretaceae	T	3	-	2	1	2

Appendix A.4 (Continued)

49	*Irvingia malayana*	Irvingiaceae	T	1	-	1	-	-
50	*Siphonodon eclastrineus*	Celastraceae	T	-	-	8	-	-
51	*Oroxylum indicum*	Bignoniaceae	T	-	-	1	-	-
52	*Catunaregam tpmentosa*	Rubiaceae	S	2	-	1	2	1
53	*Croton roxburghii*	Euphorbiceae	S	1	-	2	1	-
54	Unidentified sp 13	-		-	-	1	-	3
55	*Vitex canescens*	Labiatae	T	1	-	1	-	2
56	*Capparis tenara*	Capparceae	S	-	-	1	-	1
57	*Microcos paniculata*	Tiliaceae	T	1	-	1	-	3
58	*Terminalia triptera*	Combretaceae	T	2	-	-	-	-
59	*Terminalia calamansanai*	Combretaceae	T	1	-	-	-	-
60	*D. negrescens*	Paplionoideae	T	-	-	-	1	2
61	*Millingtonia hortensis*	Bignoniaceae	T	-	-	-	1	-
62	*Careya sphaerica*	Lecythidaceae	T	-	-	-	1	-
63	Diospyros sp.	Ebenaceae	T	-	-	-	1	1
64	*Miliusa velutina*	Annonaceae	T	-	-	-	1	1
65	Bhauhinia sp.	-		-	-	-	1	1
66	*Garuga pinnata*	Burseraceae	T	-	-	-	1	1
67	Unidentified sp. 14	-	S	-	-	-	1	1
68	*Diospyros mollis*	Ebenaceae	T	-	-	-	1	11
69	*Scheleichera oleosa*	Sapindaceae	T	-	-	-	1	1
70	*Acacia rugata*	Mimosoideae	S	-	-	-	1	1
71	*Bauhinia scandens*	Caesalpinioideae	C	-	-	-	1	1
72	*D. oliveri*	Paplionoideae	T	6	1	-	10	7
73	*Millettia sp.*	Paplionoideae	S	32	12	29	19	17

Appendix A.5: Density of non-woody species in five mixed tree species plantations in Lampang province, northern Thailand.

No	Scientific name	Family	Plant habit	Average density (Individuals/125 m^2)				
				T	TT	TG	TTG	TTA
1	*Imperata cylindrica*	Gramineae	H	1494.67	558.67	914.00	497.67	319.00
2	*Chromolaena odorata*	Compositae	H	19.33	16.00	15.67	18.00	12.67
3	*Gigantochloa albociliata*	Gramineae	Bamboo	29.00	25.00	18.00	25.67	21.33
4	Commelina sp.	Comelinaceae	H	10.33	23.33	3.00	34.33	27.67
5	*Getonia floribunda*	Combretaceae	C	11.00	4.33	12.67	2.00	1.00
6	Unidentified sp 15	-	H	2.67	4.00	1.00	0.00	1.00
7	Curcuma spp.	Zingiberraceae	H	1.67	2.00	0.00	3.33	12.67
8	Lygodium sp.	Scizaeaceae	H	0.00	2.67	0.67	1.67	1.33
9	Unidentified sp 16	-	H	0.00	2.00	2.67	0.00	1.33
10	*Curculigo orchidoides*	Hypocidaceae	H	0.00	1.67	0.00	0.00	0.00
11	*Congea tomentosa*	Verbenaceae	C	0.00	1.00	2.00	6.00	0.00
12	Carex sp.	Cyperaceae	C	0.00	0.67	1.67	0.33	4.00
13	*Dalbergia stipulacea*	Papilionoideae	C	2.33	1.00	0.00	2.33	1.67
14	Unidentified sp 9	-	C	3.33	0.33	0.00	5.00	0.00
15	*Dalbergia volubilis*	Papilionoideae	C	0.00	1.00	0.00	0.67	4.00
16	Unidentified sp 17	-	H	0.33	0.00	3.00	0.00	0.33
17	Neyraudia sp.	Gramineae	H	0.00	0.00	0.67	0.00	0.00
18	Uvaria sp.	Annonaceae	C	0.00	0.00	2.00	0.00	0.00
19	Unidentified sp 18	-	H	0.33	0.00	1.33	0.00	2.33
20	Cleistanthus sp.	Euphorbiaceae	H	0.00	0.00	0.00	0.00	2.67
21	*Costus speciosus*	Costaceae	H	0.00	0.00	0.00	1.33	1.67
22	Unidentified sp 3	-	C	0.00	0.00	0.00	0.00	3.00
23	Unidentified sp 19	-	H	0.00	0.00	0.00	0.00	0.33
24	Unidentified sp 20	-	H	0.00	0.00	0.00	0.67	0.00
25	Pseudodracontium spp	Araceae	H	0.00	0.00	0.00	0.67	0.00
26	Dioscorea sp.	Dioscoreaceae	H	0.00	0.00	0.00	0.33	0.00
27	*Cephalostachyum pergracile*	Graminae	Bamboo	1.00	1.67	2.00	0.00	2.33
28	Unidentified sp 21	-	H	0.00	1.00	0.00	0.00	0.00

Appendix A.6: Density of woody seeds in natural mixed deciduous forest with teak (MDF), ecotone (ECO) and mixed tree species plantation (MP) in Lampang province, northern Thailand.

Scientific name	Abbreviations	Average density of seed dispersal (seeds/ha)			Average density of seed bank (seeds/ha)		
		MDF	ECO	MP	MDF	ECO	MP
Tectona grandis	T	7,127.53	14,255.06	37,986.11	-	15386.98	28263.70
Lannea coromandelica	KUK	10,799.69	242,105.26	-	-	2297.13	-
Vitex peduncularlis	KSP	12,791.01	462,368.42	-	2394.17	2604.15	-
Grewia eriocarpa	PKT	10,059.52	303,000.00	-	1190.46	1770.82	-
Antidesma ghaesembilla	MYAI	3,344.48	157,500.00	-	668.89	2280.69	-
Phyllanthus cf. orientalis	YHI 1	-	56,578.95	1,805.56	-	3152.40	-
Spondias pinnata	MKO	-	40,000.00	-	119.04	374.99	-
Gmelina arborea	SO	-	7,543.86	14,027.78	-	-	-
Erythrina subumbrans	TL	2,962.96	12,500.00	-	-	3333.32	-
Lagerstroemia duperreana	TB	8,750.00	1,333.33	-	3571.38	263.15	4166.64
Nephelium hypoleucum	KL	8,228.62	2,017.54	-	1635.39	1315.78	-
Unidentified sp 24	MKAE	9,259.26	-	-	3888.89	-	-
Chukrasia tabularis	YHI 2	8,273.81	614.04	-	-	-	-
Pterospermum acerifolium	PHO	6,147.65	-	-	1614.45	-	-
Pterocarpus macrocarpus	P	1,717.03	263.16	3,888.89	1918.28	1884.85	2152.76
Getonia floribunda	KTT	5,555.56	-	-	5555.55	-	-
Anacardium occidentale	CNUT	-	-	4,513.89	-	-	3680.53
Morinda coreia	YPA	-	2,500.00	2,222.22	-	-	347.22
Croton roxburghii	PYAI	-	87.72	3,888.89	-	1080.03	-
Unidentified sp 9	KHAN	2,531.28	789.47	-	803.67	1085.52	-
Xylia xylocarpa var. kerrii	D	2,777.78	-	-	-		-
Canarium subulatum	ML	1,960.24	-	-	2196.81		-
Afzelia xylocarpa	MKA	1,538.46	-	-	1739.11		-
Albizia lebbeck	TUT	952.38	-	416.67	833.32	166.66	-
Buchanania latifolia	MHMW	1,070.23	-	-	1471.56		-
Croton longissimus	PNOI	-	-	902.78	401.33	87.71	833.32
Irvingia malayana	KB	892.86	-	-	1011.89		-
Albizia odoratissima	KKM	802.68	-	-	-	87.71	-
Harrisonia perforata	KOTA	-	614.04	-	-	2289.46	-
Oroxylum indicum	LP	535.71	-	-	-	-	
Unidentified sp 22	KNH	401.34	-	-	-	-	-
Colona siamica	PYAB	66.89	-	-	-	-	-
Cassia fistula		-	-	-	773.79	520.83	-
Bombax anceps	NGIU	-	-	-	-	-	277.77
Miliusa velutina	KHM	-	-	-	357.13	-	-
Schleichera oleosa	TKO	-	-	-	1481.48	-	-
Catunaregam tomentosa	NTANG	-	-	-	1070.22	-	-
Terminalia calamansanai	HAAN	-	-	-	-	-	555.55
Bhauhinia spp.	KDLING	-	-	-	-	166.66	-
Unidentified sp 23	MM	-	-	-	-	-	138.88

Appendix A.7: Productivity of enrichment planting speecies in five mixed tree species plantation in Lampang province, northern Thailand.(*TM: Total biomasss*)

No	Species	T1		T2		T3		TT1		TT2		TT3		TG1		TG2		TG3		TTG1		TTG2		TTG3		TTA1		TTA2		TTA3	
		TM0	TM12	TM0	TM12	TM0	TM12	TM0	TM12	TM0	TM12	TM0	TM12	TM0	TM12	TM0	TM12	TM0	TM12	TM0	TM12	TM0	TM12	TM0	TM12	TM0	TM12	TM0	TM12	TM0	TM12
1	*Pterocarpus macrocarpus*	6.46	7.07	11.35	9.92	8.94	22.47	6.46	7.07	11.35	9.92	8.94	22.47	10.46	9.99	8.22	9.85	8.13	4.53	13.11	7.07	14.40	23.76	10.46	11.93	8.94	5.59	10.69	9.16	7.84	5.08
2	*Pterocarpus macrocarpus*	8.41	4.98	5.09	12.25	6.81	3.85	8.41	4.98	5.09	12.25	6.81	3.85	11.13		8.68	8.23	10.91	13.03	10.88	17.01	9.51	6.29	12.23	12.98	9.20	7.55	12.37	11.25	8.59	5.59
3	*Pterocarpus macrocarpus*	6.94	4.87	8.03	12.83	10.91	19.28	6.94	4.87	8.03	12.83	10.91	19.28	9.51	11.77	10.12	11.64	12.78	3.69	8.50		14.06	4.53	12.37	11.77	8.13	7.43	16.03	16.12	8.68	5.08
4	*Pterocarpus macrocarpus*	6.46	8.12	7.73	6.82	11.55	16.28	6.46	8.12	7.73	6.82	11.55	16.28	12.62	12.98	11.04	8.57	8.03	7.07	17.38	20.21	13.88		15.23	12.25	8.32	7.55	12.92	12.54	8.94	7.55
5	*Pterocarpus macrocarpus*	7.84	9.02	10.12	9.99	10.58	14.78	7.84	9.02	10.12	9.99	10.58	14.78	13.19	14.63	7.63	9.02	11.44		11.50	6.94	8.03		13.71	10.62	8.32	4.53	20.11	15.31	9.03	5.78
6	*Pterocarpus macrocarpus*	8.22	4.03	11.50	13.58	6.88	8.14	8.22	4.03	11.50	13.58	6.88	8.14	8.94	10.38	8.68	10.96	8.41		14.90	16.28	9.88	4.28	13.19	9.44	11.23		9.88	8.11	9.20	8.76
7	*Pterocarpus macrocarpus*	8.41	9.16	12.23	13.58	13.84	21.51	8.41	9.16	12.23	13.58	13.84	21.51	8.86	14.64	10.00	11.77	7.38	4.98	10.46	8.57	13.53	6.33	11.02	10.12	14.09	14.63	10.00	9.72	9.20	9.72
8	*Pterocarpus macrocarpus*	11.65	12.24	9.39	9.58	9.11	11.13	11.65	12.24	9.39	9.58	9.11	11.13	11.50	9.99	10.69	10.76	13.97		8.03	8.14	10.23		14.57	18.31	8.59	5.39	9.88	7.67	7.63	6.06
9	*Pterocarpus macrocarpus*	9.00	8.11	9.88	14.78	8.94	13.83	9.00	8.11	9.88	14.78	8.94	13.83	8.41	10.79	8.94	8.34	11.65	11.77	10.23	8.14	8.22	3.62	10.00	11.29	10.80	12.39	13.71	12.39	8.50	4.76
10	*Pterocarpus macrocarpus*	11.85	8.45	10.12	11.01	14.74	13.03	11.85	8.45	10.12	11.01	14.74	13.03	8.94		9.88	7.07	11.95	28.12	11.94	13.58	9.51	7.43	12.92	15.31	9.52	8.23	14.90	5.88	8.03	4.53
11	*Pterocarpus macrocarpus*	6.88	4.53	7.84	6.69	9.28	8.57	6.88	4.53	7.84	6.69	9.28	8.57	10.00	18.13	9.88	11.93	9.36	7.90	8.59	13.27	9.51	6.69	9.88	6.69	11.23	10.88	8.94	12.45	8.13	
12	*Pterocarpus macrocarpus*	6.60	5.08	8.59	3.61	8.77	11.77	6.60	5.08	8.59	3.61	8.77	11.77	8.50	15.31	9.51	13.27	9.03	10.38	15.23	17.96	9.39	6.01	9.88	10.25	14.90	14.37	13.19	17.35	11.02	16.59
13	*Pterocarpus macrocarpus*	9.76	7.43	8.50	18.84	9.51	20.32	9.76	7.43	8.50	18.84	9.51	20.32	8.50		8.50		10.69	13.69	11.04	11.64	10.23	11.29	8.03	10.38	10.23	10.88	8.32	8.01	13.84	12.69
14	*Pterocarpus macrocarpus*	11.55	6.15	6.67	17.39	10.69	15.31	11.55	6.15	6.67	17.39	10.69	15.31	8.86	9.99	8.03	11.29	9.11	9.16	12.51	27.93	10.69	9.85	10.69	14.98	9.68	12.40	9.44	2.90	11.79	16.82
15	*Pterocarpus macrocarpus*	9.11	10.63	8.22	17.76	9.11	5.59	9.11	10.63	8.22	17.76	9.11	5.59	8.94	9.58	8.94	16.28	9.39	7.62	12.99	8.57	9.26	7.95	8.94	7.07	8.94	5.88	14.40	8.23	10.46	8.76
16	*Pterocarpus macrocarpus*	8.94	1.85	8.32	11.45	9.36	12.84	8.94	1.85	8.32	11.45	9.36	12.84	10.80	19.19	8.03	5.08	12.78	25.52	13.71	4.87	9.39	7.07	8.50	12.69	9.60	6.76	12.23	14.30	6.81	5.59
17	*Pterocarpus macrocarpus*	7.56	7.55	10.00	10.51	8.94	12.54	7.56	7.55	10.00	10.51	8.94	12.54	14.57	19.19	8.77	12.69	8.59	4.53	13.88	12.08	10.58	5.71	8.32	9.44	10.69	2.21	11.79	13.96	9.88	17.01
18	*Pterocarpus macrocarpus*	7.93	8.12	11.02	20.11	8.22	17.01	7.93	8.12	11.02	20.11	8.22	17.01	8.50		10.46		12.64	20.04	11.50	8.57	12.37		9.88	8.57	6.88	6.33	12.51	9.99	9.20	
19	*Pterocarpus macrocarpus*	8.50	7.78	8.22	6.69	14.23	15.31	8.50	7.78	8.22	6.69	14.23	15.31	10.35	11.77	9.51	9.16	17.41	18.66	10.69	4.53	12.51	6.29	10.58	10.51	9.28	8.12	11.79	8.76	8.32	5.97
20	*Pterocarpus macrocarpus*	13.58	11.77	10.00	12.54	8.22	20.72	13.58	11.77	10.00	12.54	8.22	20.72	12.08	16.43	8.13	9.02	6.60	38.27	13.19	11.61	10.58	12.25	26.34	17.76	8.94	6.24	11.23	14.30	10.91	8.23
21	*Pterocarpus macrocarpus*	10.51	10.51	8.32	11.13	8.03	10.34	10.51	10.51	8.32	11.13	8.03	10.34	10.58		8.50	7.31	10.80	8.57	13.19	1.78	12.51	8.01	11.44	17.79	11.13		11.04	9.44	8.22	14.81
22	*Pterocarpus macrocarpus*	10.69	11.93	10.00	9.16	12.99	16.56	10.69	11.93	10.00	9.16	12.99	16.56	10.58	11.45	7.93	7.07	13.71	17.27	9.36	6.15	7.53	6.29	17.38	17.01	8.94	5.59	13.32	16.59	8.03	5.41
23	*Pterocarpus macrocarpus*	10.46	5.29	9.51	20.72	11.04	13.96	10.46	5.29	9.51	20.72	11.04	13.96	12.51	17.27	7.20	6.92	11.65	14.98	8.77	5.97	12.37	9.99	8.94	9.55	8.86	11.84	8.59		10.58	13.58
24	*Pterocarpus macrocarpus*	11.55	3.09	8.32	8.01	9.28	17.35	11.55	3.09	8.32	8.01	9.28	17.35	10.00	9.30	7.20	7.43	9.03	11.01	10.69	9.73	11.02	5.71	11.65		8.86	6.69	8.68	5.88	8.77	5.08
25	*Pterocarpus macrocarpus*	11.75	10.51	16.98	25.52	8.03	7.78	11.75	10.51	16.98	25.52	8.03	7.78	15.07	26.96	6.31	10.10	8.50	5.71	12.81	7.95	10.69	13.96	12.92	13.69	8.32	7.43	9.51	9.58	9.51	11.25
26	*Pterocarpus macrocarpus*	11.02	15.80	9.76	8.87	7.26	5.08	11.02	15.80	9.76	8.87	7.26	5.08	18.46	21.61	11.04	14.35	9.76	22.28	14.40	20.16	8.32	4.76	11.50	10.96	9.20	5.19	10.58	11.49	7.13	4.53
27	*Pterocarpus macrocarpus*	7.13	7.92	7.20	14.30	11.02	11.77	7.13	7.92	7.20	14.30	11.02	11.77	12.08	18.13	6.60	7.67	8.94	15.15	10.80	17.27	12.64	9.06	17.77	20.04	11.34	4.28	10.23	11.01	8.94	
28	*Pterocarpus macrocarpus*	8.03	8.11	6.94	8.01	12.08	10.79	8.03	8.11	6.94	8.01	12.08	10.79	8.32	9.16	8.03	6.92	6.46	10.10	10.69	10.25	14.23	13.69	16.03	16.28	12.25	9.26	13.53	17.20	7.93	7.19
29	*Pterocarpus macrocarpus*	6.74	5.71	9.51	8.97	9.11	10.12	6.74	5.71	9.51	8.97	9.11	10.12	9.51	9.02	8.32	6.92	15.87	16.43	8.59	7.07	10.58	11.01	13.88	16.43	10.07	5.97	7.93	7.43	9.76	14.30

Appendix A.7 (Continued)

30	*Pterocarpus macrocarpus*	8.59	1.56	9.64	17.76	5.31	5.59	8.59	1.56	9.64	17.76	5.31	5.59	7.63	3.61	8.22	10.28	8.03	6.42	9.39	7.45	9.88	5.86	10.69	8.01	16.50	15.31	14.40	7.31	8.13	11.29
31	*Xylia xylocarpa*	6.94	8.21	2.82		3.04		6.94	8.21	2.82		3.04		5.35	8.41	3.04		4.56	5.80	4.06	23.78	2.36	3.04	3.14	11.24	4.40		5.35		5.50	8.41
32	*Xylia xylocarpa*	3.04		5.19	3.04	3.45		3.04		5.19	3.04	3.45		1.54	5.65	3.35	10.16	4.23	6.94	1.41		1.66	8.62	3.04	2.71	2.60	2.71	2.29		7.59	8.56
33	*Xylia xylocarpa*	3.14		3.25		5.35	14.54	3.14		3.25		5.35	14.54	1.60		1.90	6.81	2.82		2.60		2.93		3.04		2.93		3.75	5.95	5.95	8.82
34	*Xylia xylocarpa*	2.82	13.55	3.04	7.59	3.35	6.10	2.82	13.55	3.04	7.59	3.35	6.10	2.60	12.29	1.72		3.45	8.41	2.36	17.02	2.36		3.04		3.14	10.97	3.35		3.85	6.39
35	*Xylia xylocarpa*	2.71	3.45	7.37	5.35	3.25	6.10	2.71	3.45	7.37	5.35	3.25	6.10	2.82	5.59	5.04	6.10	1.54		1.60		2.60	4.04	3.04	4.72	2.93	5.50	4.06		5.95	
36	*Xylia xylocarpa*	2.82		8.17	15.74	5.50	9.02	2.82		8.17	15.74	5.50	9.02	3.04	5.35	2.01		4.23		2.01	3.95	2.48	6.28	2.93		2.60		3.35	7.16	3.25	14.78
37	*Xylia xylocarpa*	4.40	17.64	3.75		3.95	13.55	4.40	17.64	3.75		3.95	13.55	4.40		3.14	6.10	2.82	3.04	2.71	4.40	2.60	9.32	5.04		5.35	8.41	6.94		3.45	
38	*Xylia xylocarpa*	2.71		4.72	13.55	4.88	10.37	2.71		4.72	13.55	4.88	10.37	3.25		3.25		2.36		1.48		3.14		5.50		4.72	5.65	3.45	1.72	4.56	
39	*Xylia xylocarpa*	4.88		5.50	8.41	4.88	17.02	4.88		5.50	8.41	4.88	17.02	3.04		4.72	5.19	2.82	8.82	4.06		2.60		3.04		1.66	7.23	3.35		3.04	1.66
40	*Xylia xylocarpa*	2.82	8.56	3.14	9.61	4.56		2.82	8.56	3.14	9.61	4.56		9.41	14.29	1.90		2.71	19.78	1.41		2.36	1.48	4.56		2.48	2.93	1.66	8.41	3.25	1.60
41	*Xylia xylocarpa*	3.04		7.09	15.09	2.82		3.04		7.09	15.09	2.82		9.80		4.56		2.93	6.10	4.56	5.04	9.21		10.97		2.71		11.77	16.91	3.04	6.10
42	*Xylia xylocarpa*	13.55		13.55	27.47	2.36		13.55		13.55	27.47	2.36		9.61	22.28	3.35		9.21	308.00	9.02	0.30	11.77	9.21	9.80	8.00	5.04		10.71	32.19	9.99	15.09
43	*Xylia xylocarpa*	9.21		2.71		8.41	15.74	9.21		2.71		8.41	15.74	3.35		9.41	10.97	15.42	41.98	12.54		2.82		9.41	40.35	8.41	9.61	6.81	8.41	2.48	5.19
44	*Largerstroemia duperreana*	6.99	17.87	15.30	27.13	26.23		6.99	17.87	15.30	27.13	26.23		5.46	7.69	10.17	19.43	11.44	23.38	5.09	5.93	15.80	42.50	9.08		7.83	6.47	6.64	8.85	6.47	
45	*Largerstroemia duperreana*	10.17	12.69	17.97	36.73	9.74		10.17	12.69	17.97	36.73	9.74		6.99		14.28	21.65	7.83	21.65	5.21	8.62	8.32	9.52	9.30	20.54	7.83	19.41	8.00	7.69	6.29	6.96
46	*Largerstroemia duperreana*	7.67	14.07	22.17	19.03	16.69	14.62	7.67	14.07	22.17	19.03	16.69	14.62	4.71	4.58	17.32	29.78	8.16	12.05	5.70	9.08	15.80	16.69	19.76	16.05	10.39	11.98	5.21	5.93	4.96	6.29
47	*Largerstroemia duperreana*	5.33	11.61	15.80	27.74	4.71	10.18	5.33	11.61	15.80	27.74	4.71	10.18	4.83	4.37	7.69	21.27	9.30	15.46	10.60	13.23	8.64	0.15	3.22	5.36	5.58	9.74	7.16	14.07	6.29	11.62
48	*Largerstroemia duperreana*	7.16	13.04	10.39	16.68	12.70	15.40	7.16	13.04	10.39	16.68	12.70	15.40	4.96	16.68	11.44	16.68	12.70	28.98	10.17	11.61	9.30	24.15	5.93	11.04	13.23	23.94	8.00	11.88	5.21	6.99
49	*Largerstroemia duperreana*	7.16	25.23	14.28	18.95	9.30		7.16	25.23	14.28	18.95	9.30		6.99	10.51	6.11	16.68	12.45	22.67	27.49	20.11	7.16	8.39	4.18	5.36	7.16	16.37	9.30	6.82	5.46	7.33
50	*Largerstroemia duperreana*	5.70	32.93	11.44	15.04	13.50	19.16	5.70	32.93	11.44	15.04	13.50	19.16	6.99	8.95	9.08	8.39	20.36	19.03	6.20	18.26	8.48	4.18	5.21	4.58	9.08	12.97	6.29	12.43	5.21	7.69
51	*Largerstroemia duperreana*	9.96	14.07	35.60	17.02	11.44	22.62	9.96	14.07	35.60	17.02	11.44	22.62	3.91	5.93	11.33	14.07	11.61	15.52	16.04	24.95	9.96	11.64	11.61	27.91	11.02	13.04	4.58	4.98	9.30	4.16
52	*Largerstroemia duperreana*	9.08	40.06	7.67	11.04	13.50	28.15	9.08	40.06	7.67	11.04	13.50	28.15	6.99	13.76	5.09	9.30	3.63	6.11	16.04	28.98	7.16	14.28	5.21	9.89	4.58	13.50	5.21	12.34	6.64	11.98
53	*Largerstroemia duperreana*	7.45	11.33	7.83	5.93	11.02	5.93	7.45	11.33	7.83	5.93	11.02	5.93	7.93		14.28	16.05	5.82	1.09	8.48	10.17	7.16	16.68	6.64	4.37	13.23	22.62	8.00	9.11	7.16	
54	*Largerstroemia duperreana*	6.99	7.69	11.02	3.91	13.76	17.02	6.99	7.69	11.02	3.91	13.76	17.02	4.78	8.67	5.70	3.91	9.96	12.70	10.39	9.11	13.76		6.47	8.67	10.17		5.21	3.63	13.23	6.47
55	*Largerstroemia duperreana*	3.91	16.05	14.28	23.50	30.58	16.63	3.91	16.05	14.28	23.50	30.58	16.63	6.29		12.43	15.04	9.30	10.47	6.47	6.96	12.65	27.49	7.69	16.69	14.28	2.42	12.25		3.91	0.40
56	*Largerstroemia duperreana*	10.17	9.59	8.16	22.31	11.44	48.04	10.17	9.59	8.16	22.31	11.44	48.04	4.31	9.59	14.02	16.37	4.58	7.93	6.82	24.37	19.46	9.35	7.16	13.73	10.81	10.18	12.25	22.73	10.47	16.68
57	*Schleichera oleosa*	3.60	4.55	7.64		6.00	10.74	3.60	4.55	7.64		6.00	10.74	6.00		3.50	4.18	6.00		1.72		3.60	5.64	2.66		3.50	3.60	1.47	1.42	4.73	
58	*Schleichera oleosa*	3.50		8.48		6.00		3.50		8.48		6.00		3.50	4.91	3.40		7.64		2.87		1.47		2.76	3.50	3.60	6.18	3.40	6.18	5.46	
59	*Schleichera oleosa*	1.31		3.40		2.98	3.44	1.31		3.40		2.98	3.44	1.72		2.87	0.19	5.46		1.67		2.76		1.67	4.91	2.87	6.54	4.36	3.60	5.82	
60	*Diospyros mollis*	9.09	15.35	13.51		15.39	20.25	9.09	15.35	13.51		15.39	20.25	19.91	59.83	13.76	32.36	30.20	38.22	15.86	0.17	12.57	22.33	18.57	45.20	14.00	26.50	21.15	48.92	22.63	29.52
61	*Diospyros mollis*	14.00	29.83	24.09		21.15	51.91	14.00	29.83	24.09		21.15	51.91	15.86	9.62	12.35	36.88	24.62	9.44	33.82	49.97	14.93	42.92	24.09	40.44	16.08	31.66	27.62	24.40	16.54	17.83

Appendix A.7 (Continued)

62	*Toona ciliata*	7.26	12.85	12.79	27.29	5.88	15.98	7.26	12.85	12.79	27.29	5.88	15.98	7.43	19.83	5.75	12.71	6.73	24.32	5.34	6.88	6.15	15.34	6.28	13.58	5.88	11.65	5.88	17.37	6.91	8.85
63	*Toona ciliata*	6.28		6.91		6.91	11.54	6.28		6.91		6.91	11.54	6.28	27.73	5.48	31.81	5.88	2.97	7.60		6.15	3.09	6.15	16.67	6.41	33.65	9.76	15.63	5.34	15.05
64	*Millettia leucantha*	24.80	43.78	42.18	61.39	27.97	71.54	24.80	43.78	42.18	61.39	27.97	71.54	36.23	56.56	29.34	45.89	20.88	40.02	38.24	151.88	22.56		26.68	41.33	49.77	51.40	20.12	36.86	24.82	36.39
65	*Millettia leucantha*	22.33	40.46	34.90	65.35	19.85	25.06	22.33	40.46	34.90	65.35	19.85	25.06	32.83	65.46	22.56	49.79	28.53	62.57	40.21	86.21	23.08	48.26	17.85	25.92	24.80	66.98	20.12	44.82	47.17	55.79
66	*Afzelia xylocarpa*	9.59	17.03	8.91	18.33	18.99	24.09	9.59	17.03	8.91	18.33	18.99	24.09	8.91	11.08	9.59	7.55	15.13		12.61	24.09	11.46	21.66	11.46	24.84	12.41	20.98	5.16	13.60	8.91	14.60
67	*Afzelia xylocarpa*	8.68	9.23	6.63	11.64	14.10	15.44	8.68	9.23	6.63	11.64	14.10	15.44	9.05	5.35	10.52	26.21	7.55	30.98	8.36	9.99	9.59	10.14	17.43	8.68	8.22	11.08	10.52	15.36	16.91	24.05
68	*Chukrasia velutina*	8.86	7.79	7.96	3.67	7.74	6.78	8.86	7.79	7.96	3.67	7.74	6.78	11.59	7.18	8.09	5.76	6.83	9.45	9.45	1.86	8.36	4.01	6.93	3.54	8.49	10.63	7.03	7.74	8.43	8.09
69	*Chukrasia velutina*	8.80	10.39	10.41	10.02	8.23	8.53	8.80	10.39	10.41	10.02	8.23	8.53	9.27	14.54	8.80	8.91	6.93	6.30	9.86	5.52	10.81		8.68	14.16	8.68		8.62	10.91	8.43	10.02
70	*Spondias pinnata*	6.78	6.89	6.42	4.59	15.28	21.51	6.78	6.89	6.42	4.59	15.28	21.51	9.30	8.84	8.04	9.30	7.87	25.11	9.15	18.12	10.96	28.73	9.45	10.96	11.94		7.12	11.94	10.55	49.58
71	*Spondias pinnata*	15.58	54.94	7.12	24.94	7.87	21.34	15.58	54.94	7.12	24.94	7.87	21.34	11.16	20.04	11.75	15.72	5.93	9.00	7.67	23.41	10.76		14.09	49.74	6.83	16.17	13.11	23.72	13.36	35.08
72	*Dalbergia dongnaiensis*	7.96	33.22	7.27	14.84	6.51	28.26	7.96	33.22	7.27	14.84	6.51	28.26	6.89	15.52	5.50	33.92	8.21	16.19	4.80		4.19	14.35	5.10	38.44	3.44	7.44	6.89	17.51	4.65	31.07
73	*Dalbergia dongnaiensis*	8.70	24.72	7.08		8.45	15.85	8.70	24.72	7.08		8.45	15.85	4.80	11.27	3.55	46.67	3.76	19.12	7.27	22.93	4.80	6.11	6.31	30.27	3.34	34.81	4.95	4.35	7.71	42.47
Total		599.08	816.01	756.88	965.29	732.47	1036.08	664.14	872.19	663.57	878.01	726.84	928.75	655.08	843.62	614.17	891.7	697.39	1256.48	749.99	967.64	682.28	667.28	723.07	945.52	667.41	783.85	694.36	819.85	643.84	783.57

APPENDICES B

Appendix B.1 Ecological indices equations

1. The equation for Shannon-Wiener's index;

$$H' = -\sum pi\ ln\ pi, \qquad \text{(Equation 1)}$$

Where H' = Shanon-Wiener's index for species diversity
pi = Proportion of species i to total of number.

2. The equation for evenness index;

$$E = H' / H_{max} \qquad \text{(Equation 2)}$$

where E = Evenness index
H' = Shannon-Wiener's index
H_{max} = Maximum for Shannon-Wiener's index = $\log_{10} S$
S = Total number of species

3. The equation for Sorensen similarity index;

$$C = (2c/A+B) \times 100 \qquad \text{(Equation 3)}$$

Where C = Similarity index
c = The number of species shared by site i and j
A = Total number of species in site i
B = Total number of species in site j

Appendix B.2 Productivity of enrichment seedling calculation by used allometric method

The seedling biomass both above-ground and below-ground were calculated for each species in each plot by using biomass regressions that relate total and component-wise dry mass to stem basal diameter and height (Allometric method). After measure diameter above ground level and height, seedlings were selecting for 8-20 seedlings per species. Then seedling samples were dividing into 4 components; stem, leaves, branch and root. All of components were taken and oven-dried to constant weight at 105 ^{o} C to yield conversion factor used to calculate seedling biomass on dry-weight basis, and develop allometric regression. Allometry formulates the relation between D_0^2H and dry weight of stem, branch, leaf and root as $y = Ax^h$ or log y = h log x + log A, where y is the weight of stem, branch, leaf and root, and A and h are specific constants (Ogawa, 1967). Therefore, the equation for each component is;

Log Ws = log A+hLogD_0^2H	Ws; Stem dry weight
Log Wb = log A+hLogD_0^2H	Wb; Branch dry weight
Log Wl = log A+hLogD_0^2H	Ws; Leaf dry weight
Log Wr = log A+hLogD_0^2H	Ws; Root dry weight

The regression equations for the 11 species are as follows:

1. *Xylia xylocarpa* var. *kerrii*;

Log Ws = 0.863+0.738 log (D_0^2h);	r^2 = 0.832	(Equation 4)
Log Wb = 0.599+0.583 log (D_0^2h);	r^2 = 0.810	(Equation 5)
Log Wl = 1.119+0.593 log (D_0^2h);	r^2 = 0.918	(Equation 6)
Log Wr = 1.527+0.860 log (D_0^2h);	r^2 = 0.850	(Equation 7)

2. *Pterocarpus macrocarpus*;

Log Ws = 0.891+0.642 log (D_0^2h);	r^2 = 0.955	(Equation 8)
Log Wb = 0.411+0.672 log (D_0^2h);	r^2 = 0.838	(Equation 9)
Log Wl = 0.799+0.566 log (D_0^2h);	r^2 = 0.769	(Equation 10)
Log Wr = 1.071 +0.448 log (D_0^2h);	r^2 = 0.792	(Equation 11)

3. *Largerstroemia floribunda*;

Log Ws = 1.066+0.820 log (D_0^2h);	r^2 = 0.918	(Equation 12)
Log Wb = 0.653+0.727 log (D_0^2h);	r^2 = 0.821	(Equation 13)
Log Wl = 1.149+0.788 log (D_0^2h);	r^2 = 0.853	(Equation 14)
Log Wr = 1.217 +0.626 log (D_0^2h);	r^2 = 0.859	(Equation 15)

4. *Schleichera oleosa*;

Log Ws = 1.052+0.761 log (D_0^2h);	r^2 = 0.905	(Equation 16)
Log Wb = 0.370+0.544 log (D_0^2h);	r^2 = 0.927	(Equation 17)
Log Wl = 1.187+0.681 log (D_0^2h);	r^2 = 0.972	(Equation 18)
Log Wr = 1.990 +1.209 log (D_0^2h);	r^2 = 0.815	(Equation 19)

5. *Chukrasia tabularis*;

Log Ws = 0.660+0.416 log (D_0^2h);	r^2 = 0.923	(Equation 20)
Log Wb = 0.314+0.650 log (D_0^2h);	r^2 = 0.916	(Equation 21)
Log Wl = 0.907+0.610 log (D_0^2h);	r^2 = 0.841	(Equation 22)
Log Wr = 0.965+0.267 log (D_0^2h);	r^2 = 0.950	(Equation 23)

6. *Dalbergia oliveri*;

Log Ws = 1.046+0.900 log (D_0^2h);	r^2 = 0.944	(Equation 24)
Log Wb = 0.521+0.621 log (D_0^2h);	r^2 = 0.907	(Equation 25)
Log Wl = 0.662+0.595 log (D_0^2h);	r^2 = 0.851	(Equation 26)
Log Wr = 1.341+0.778 log (D_0^2h);	r^2 = 0.963	(Equation 27)

7. *Spondias pinnata*;

Log Ws = 0.838+0.726 log (D_0^2h);	r^2 = 0.920	(Equation 28)
Log Wb = 0.512+1.554 log (D_0^2h);	r^2 = 0.937	(Equation 29)
Log Wl = 0.897+0.575 log (D_0^2h);	r^2 = 0.912	(Equation 30)
Log Wr = 1.403+0.627 log (D_0^2h);	r^2 = 0.966	(Equation 31)

8. *Afzelia xylocarpa*;

Log Ws = 1.153 + 1.343 log (D_0^2h);	r^2 = 0.905	(Equation 32)
Log Wb = -0.463+1.819 log (D_0^2h);	r^2 = 0.749	(Equation 33)
Log Wl = 0.589 +1.206 log (D_0^2h);	r^2 = 0.965	(Equation 34)
Log Wr = 1.008 + 1.126 log (D_0^2h);	r^2 = 0.918	(Equation 35)

9. *Diospyros mollis*;

Log Ws = 1.073+0.931 log (D_0^2h);	r^2 = 0.973	(Equation 36)
Log Wb = 0.689+1.102 log (D_0^2h);	r^2 = 0.900	(Equation 37)
Log Wl = 0.918+0.692 log (D_0^2h);	r^2 = 0.892	(Equation 38)
Log Wr = 1.542 +0.834 log (D_0^2h);	r^2 = 0.804	(Equation 39)

10. *Toona ciliata*;

Log Ws = 0.947+0.647 log (D_0^2h);	r^2 = 0.872	(Equation 40)
Log Wb = 0.302+1.070 log (D_0^2h);	r^2 = 0.912	(Equation 41)
Log Wl = 1.032+1.292 log (D_0^2h);	r^2 = 0.929	(Equation 42)
Log Wr = 1.062 +0.671 log (D_0^2h);	r^2 = 0.955	(Equation 43)

11. *Millettia leucantha* var. *leucantha*;

Log Ws = 1.091+0.938 log (D_0^2h);	r^2 = 0.980	(Equation44)
Log Wb = 0.585+0.422 log (D_0^2h);	r^2 = 0.732	(Equation 45)
Log Wl = 0.953+0.406 log (D_0^2h);	r^2 = 0.757	(Equation 46)
Log Wr = 1.394 +0.482 log (D_0^2h);	r^2 = 0.808	(Equation 47)

[Ws, Wb, Wl and Wr]: g; [D_0]: cm; [h]: m

Appendix B.3 Absolute growth rate and relative growth rate of enrichment seedling

1. The equation for absolute growth rate is:

AGR yi = (Y2 –Y1)/t2 - t1; (Equation 48)

Where AGR yi = Absolute growth rate between t2 and t1
Y1, Y2 = Biomass at the first and second measurements
t1, t2 = Time of the first and second measurements

2. The equation of relative growth rate is:

$$RGR\ yi = (\ln y2 - \ln y1) / t2 - t1 \qquad \text{(Equation 49)}$$

Where RGR yi = Relative growth rate between t2 and t1
Y1, Y2 = Biomass at the first and second measurements
t1, t2 = Time of the first and second measurements

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BIOGRAPHY

Miss Puangpaka Kaewkrom was born on December 15, 1971 in Petchabun Province. Graduated with the degree of Bachelor of Agriculture (Animal Science) from King Mongkut's Institute of technology Ladkrabang in 1993. Graduated with Master degree of Science (Zoology) from Department of Biology, Chulalongkorn University in 1997.