FIRE RESILIENCE OF NATIVE TREE SPECIES IN MONTANE FOREST ECOSYSTEM

PHUTTHIDA NIPPANON

**MASTER OF SCIENCE
IN ENVIRONMENTAL SCIENCE**

**GRADUATE SCHOOL
CHIANG MAI UNIVERSITY
JANUARY 2018**

FIRE RESILIENCE OF NTIVE TREE SPECIES IN MONTANE FOREST ECOSYSTEM

PHUTTHIDA NIPPANON

A THISIS SUBMITTED TO CHIANG MAI UNIVERSITY IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF MASTER OF SCIENCE IN ENVIRONMENTAL SCIENCE

GRADUATE SCHOOL, CHIANG MAI UNIVERSITY
JANUARY 2018

FIRE RESILIENCE OF NATIVE TREE SPECIES IN MONTANE FOREST ECOSYSTEM

PHUTTHIDA NIPPANON

THIS THESIS HAS BEEN APPROVED TO BE A PARTIAL FULFULLMENT OF THE REQUIREMENTS FOR THE DEGREE OF
MASTER OF SCIENCE
IN ENVIRONMENTAL SCIENCE

Examination Committee:

Pimonrat Tiansawat
.... Chairman
(Dr. Pimonrat Tiansawat)

.... Member
(Dr. Dia Panitnard Shannon)

.... Member
(Dr. Stephen Elliot)

.... Member
(Assistant Professor Dr. Lamthai Asanok)

Advisor:

....
(Dr. Dia Panitnard Shannon)

24 January 2018

To

the people across time

who supported and inspired

Acknowledgements

Books and nature are what I love. Also, getting a job that can create a positive impact to this world is what I expect. However, I have no idea how to get all of it. Therefore, I decided to study a Master's degree. The first reason is to search for what I really want to do, and the second reason is to travel deeply into the environmental world on academic side. And now it becomes clearer for what I really want to do and how to make a good impact to this world.

I thank FORRU-CMU for bringing me into the conservation world, lighting up my dream and being my role model. Moreover, this organization also taught me useful skills, such as adjustment to unfamiliar situations, management, problem solving, and English skills. From lack of confidence girl, I became stronger, more confident, and respected myself.

I thank Dr. Dia Panitnard Shannon for being my thesis advisor and university senior. Her office always welcomes me whenever I ran into a trouble and steered me to the right direction.

I acknowledge Dr. Pimonrat Tiansawat on helping in data analysis, data interpretation. She was the key person to help me understand that a biology paper is far easier than statistic paper. I am gratefully indebted to her for her very valuable comments on this thesis.

Thank you Aj. Steve for being my greatest role model! Your passion on tropical forest always inspires other people. If I can, I would like to be as clever, active, high inspirational and fun as you.

I thank Mr. Jatupoom Meesena and friends who involved in the validation survey for this research project. Without their passionate participation and input, the validation survey could not have been successfully conducted.

Last but not least, I would like to thank my family for their patience on waiting for my graduation, cheering me up, supporting and always standing by me in every part of my life.

Finally, I thank every lesson and experience which have influenced my growing. Master's degree might be already done, but my learning world is just starting, and I will take all of the lessons from university to create a positive impact to this world as I expected.

Phutthida Nippanon

หัวข้อวิทยานิพนธ์ ความสามารถในการฟื้นตัวหลังถูกไฟไหม้ของพรรณไม้ท้องถิ่นในระบบนิเวศป่าดิบเขา

ผู้เขียน นางสาว พุทธิดา นิพพานนท์

ปริญญา วิทยาศาสตรมหาบัณฑิต (วิทยาศาสตร์สิ่งแวดล้อม)

อาจารย์ที่ปรึกษา อ.ดร. เดีย พนิตนาถ แชนนอน

บทคัดย่อ

ระบบนิเวศป่าดิบเขาทำหน้าที่เป็นแหล่งต้นน้ำและที่อยู่อาศัยของสิ่งมีชีวิตหลากหลายชนิด ได้ถูกเปลี่ยนไปเนื่องจากความต้องการใช้ประโยชน์พื้นที่ในหลายวัตถุประสงค์ การฟื้นฟูระบบนิเวศดังกล่าวเป็นสิ่งจำเป็นเร่งด่วนระดับประเทศ เนื่องจากอุปสรรคสำคัญต่อความสำเร็จของการฟื้นฟูป่าในภาคเหนือคือการรุกรานของไฟในฤดูแล้ง การเลือกปลูกกล้าไม้ที่สามารถฟื้นตัวได้หลังไฟไหม้ จะช่วยเพิ่มความสำเร็จของการฟื้นฟู การศึกษานี้มีวัตถุประสงค์เพื่อแสดงความสัมพันธ์ระหว่างขนาดลำต้นต่อความสามารถในการรอดชีวิตและการแตกหน่อใหม่หลังถูกไฟรบกวน และเพื่อค้นหากล้าไม้ชนิดที่มีศักยภาพในการฟื้นฟูระบบนิเวศป่าดิบเขาที่เสี่ยงต่อการรุกรานของไฟ

ภายหลังเหตุการณ์ไฟไหม้แปลงฟื้นฟูป่าอายุ 1 2 14 และ 17 ปี (ปลูกเมื่อ พ.ศ. 2557 2556 2544 และ 2541 ตามลำดับ) ในช่วงฤดูแล้ง (เมษายน-พฤษภาคม) ของปี พ.ศ. 2558 มีการบันทึกข้อมูลจำนวนทั้งหมด 3 ครั้ง ที่ 2 18 และ 30 สัปดาห์ หลังจากเกิดไฟไหม้ บันทึกขนาดเส้นผ่านศูนย์กลางลำต้น ความสูง และจำนวนหน่อแตกใหม่ของต้นไม้ทุกต้นที่รอดชีวิตในแปลงตัวอย่างขนาด 40 x 40 เมตรต่อแปลง ต้นไม้ในแปลงฟื้นฟูอายุมากมีร้อยละการรอดชีวิตหลังถูกไหม้สูง (ร้อยละ 95.3 และ 98.6 ในแปลงอายุ 14 และ 17 ปี ตามลำดับ) ขณะที่การรอดชีวิตของต้นไม้ในแปลงอายุน้อยมีค่าน้อยกว่ามาก (ร้อยละ 42.9 และ 39.8 ในแปลงอายุ 1 และ 2 ปี ตามลำดับ) นอกจากนี้ยังพบว่าร้อยละการรอดชีวิตระหว่างการเก็บข้อมูลหลังไฟไหม้ทั้ง 3 ครั้ง ในแปลงอายุน้อยลดลงอย่างรวดเร็ว

เมื่อวิเคราะห์หาความสัมพันธ์ระหว่างขนาดลำต้นและการรอดชีวิตหลังถูกเผาโดยใช้ไค-สแควร์ การถดถอยเชิงเส้นอย่างง่าย และตัวแบบผสมเชิงเส้นวางในทั่วไป พบว่าขนาดของต้นมีผลต่อการรอดชีวิตอย่างมีนัยสำคัญ ต้นไม้ขนาดใหญ่จะมีโอกาสในการรอดชีวิตมากกว่าต้นไม้ขนาดเล็ก อาจเป็น

เพราะมีเปลือกหนาปกป้องเนื้อเยื่อเจริญด้านข้างส่วนวาสคิวลาร์แคมเบียมจากไฟ ซึ่งอาจทำให้ต้นไม้ตายจากการตายหรือการเสียรูปของเนื้อเยื่อส่วนลำต้น จากการทดลองนี้พบว่าต้นไม้ในแปลงฟื้นฟูที่มีขนาดลำต้นมากกว่า 40 มิลลิเมตร มีโอกาสในการรอดชีวิตเกือบร้อยละ 100

โดยภาพรวมขนาดลำต้นส่งผลทางลบต่อจำนวนหน่อที่แตกใหม่หลังถูกไฟไหม้ ต้นไม้ที่มีขนาดใหญ่กว่าจะสร้างหน่อใหม่จำนวนน้อยกว่าอย่างมีนัยสำคัญ เปลือกที่หนากว่าสามารถป้องกันตาพืชจากการรบกวนของไฟแต่ก็ยับยั้งการแตกหน่อใหม่ด้วยเช่นกัน หรืออาจเป็นเพราะตาของต้นไม้ขนาดใหญ่ที่อายุมากได้เสื่อมสภาพไปแล้ว เมื่อพิจารณาเฉพาะต้นไม้ขนาดเล็ก (ลำต้น <60 มิลลิเมตร) พบว่าจำนวนของหน่อแตกใหม่หลังถูกไฟไหม้เพิ่มขึ้นตามขนาดของลำต้น น่าจะสัมพันธ์กับความสามารถในการเก็บอาหารที่จำเป็นสำหรับกระบวนการสร้างหน่อใหม่ ความสามารถในการแตกหน่อใหม่หลังไฟไหม้จะเพิ่มขึ้นตามขนาดของลำต้น จนกระทั่งต้นไม้โตถึงขนาดหนึ่งจะสูญเสียความสามารถดังกล่าวไป

จากดัชนีความเหมาะสม (คำนวณจากร้อยละการรอดชีวิต จำนวนหน่อที่แตกใหม่ต่อต้นและอัตราการเติบโตสัมพัทธ์) การศึกษานี้แนะนำพืชท้องถิ่น 12 ชนิด สำหรับการฟื้นฟูป่าดิบเขาในพื้นที่เสี่ยงต่อไฟ พืช 3 ชนิดได้คะแนนในช่วงดีมาก (> ร้อยละ 75) ได้แก่ มณฑาแดง (*Magnolia garrettii*) เติม (*Bischofia javanica*) มะเดื่อใบใหญ่ (*Ficus auriculata*) พืช 7 ชนิดได้คะแนนอยู่ในช่วงยอมรับได้ (ร้อยละ 60 - 74) ได้แก่ มะเดื่อปล้อง (*Ficus hispida*) ตาเสือทุ่ง (*Heynea trijuga*) นางพญาเสือโคร่ง (*Prunus cerasoides*) ก่อใบเลื่อม (*Castanopsis tribuloides*) มะแฟน (*Protium serratum*) มะยาง (*Sarcosperma arboreum*) และหมอนหิน (*Hovenia dulcis*) และพืชจำนวน 2 ชนิด ถูกจัดอยู่ในกลุ่มสำรอง (<ร้อยละ 60) ได้แก่ มะเดื่อกวาง (*Ficus callosa*) และพืชตระกูลอบเชย (*Cinnamomum longipetiolatum*)

พืช 3 ใน 4 ของรายชื่อที่แนะนำข้างต้นถูกแนะนำเป็นพรรณไม้โครงสร้างเพื่อการฟื้นฟูป่าในภาคเหนือของประเทศไทย ยกเว้น มะแฟน มะเดื่อกวาง และพืชตระกูลอบเชย แม้ว่าโครงการฟื้นฟูสามารถจัดหากล้าไม้ท้องถิ่นที่มีความสามารถในการฟื้นตัวหลังโดนไฟที่แนะนำได้ อย่างไรก็ตามควรมีการวางแผนป้องกันไฟที่มีประสิทธิภาพร่วมกันในพื้นที่ที่มีความเสี่ยง อย่างน้อยเป็นเวลา 2 ปีหลังปลูก เพื่อเพิ่มโอกาสให้ระบบนิเวศฟื้นตัวไปในทิศทางตามเป้าที่ตั้งไว้

Thesis Title Fire Resilience of Native Tree Species in Montane Forest Ecosystem

Author Miss Phutthida Nippanon

Degree Master of Science (Environmental Science)

Advisor Dr. Dia Panitnard Shannon

Abstract

Montane forest ecosystems are important for watershed and as habitats for diverse organisms. They have been greatly modified by various protection land-uses. Therefore, restoring such ecosystems is a national priority. Fire disturbance is a major barrier for the success of forest restoration in northern Thailand, particularly during the dry season. Planting fire resilient saplings could increase restoration success. The objectives of this study were to quantify the effects of stem diameter on survival and resprouting ability after fire, as well as to identify potential candidate tree species for restoring montane forest ecosystem in fire-prone areas.

After the accidental burning of 1-, 2-, 14- and 17-year-old restoration plots (planted in 2014, 2013, 2001 and 1998 respectively) during the summer of 2015 (April-May), data were collected out 3 times (2, 18 and 30 weeks after the fire). Stem diameter, height and the number of resprouts of surviving trees were measured in 40 x 40 m samplings per plot. Survival after fire in the older plots (14- and 17-year-old plots) was high (95.3% and 98.6% respectively), whereas, in the younger plots (1- and 2-year-old plots) it was much lower (42.9% and 39.8%, respectively) Moreover, survival, throughout the monitoring, of younger trees declined more rapidly than that of older trees.

A chi square test, simple linear regression and generalized linear mixed models (GLMMs) all showed that the survival after the fire increased with increasing stem diameter. Bigger trees had thicker bark, which protected the cambium from fire, thus reducing mortality stem necrosis and deformation. Trees with DBH or RCD > 40 mm had almost 100% survival after fire.

In general, bigger trees produced significantly fewer reprouting shoots than smaller trees did. Although thick bark on larger stems protected buds from burning, it also inhibited resprouting by obstructing epicormic bud emergence. This might be associated with bud senescence, as the trees mature. Focusing on small trees (DBH or RCD <60 mm) the number of resprouting shoots significantly increased with the stem size. This relationship is possibly associated with resource storage and therefore helps to support resprouting. Resprouting ability increases with stem size, until the trees reach their adult stage. It then determinates.

A suitability index was calculated from survival, number of resprouting shoots per tree, and stem relative growth rates. Twelve native tree species were recommended for restoring areas with high fire risk. Three species were categorized as excellent group (>75%) (*Magnolia garrettii*, *Bischofia javanica* and *Ficus auriculate*). Seven species were acceptable (60 – 74%) (*Ficus hispida*, *Heynea* trijuga, *Prunus cerasoides*, *Castanopsis tribuloides*, *Protium serratum*, *Sarcosperma arboreum* and *Hovenia dulcis*) and 2 species were marginal (<60%) (*Ficus callosa* and *Cinnamomum longipetiolatum*).

Three quarters of the recommended species have been identified as framework trees for forest restoration in northern Thailand. The exceptions were *P. serratum*, *F. callosa* and *C. longipetiolatum*. Although seedlings of high fire resilient species are available for restoration projects, stakeholders should still implement fire prevention measures, at least during 2 consecutive years after planting, to keep successional pathway on track towards the ultimate goals of restoration.

CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1

Introduction

1.1 Background

Montane forest ecosystems are usually found at elevations above 1,000 m above mean sea level (Santisuk, 2006) and covers Thailand's most important water catchments. Such ecosystem types have been greatly modified by various types of land-use, particularly and shifting cultivation (Delang, 2002). Since the last century, the non-Thai hill people have illegally settled in mountain chains of the Northern Thailand and consequently altered the natural montane vegetation (Santisuk, 2006). Although most mountainous areas in northern Thailand are national parks (36% of all national parks), wildlife sanctuaries (47% of all wildlife sanctuaries), or other types of protected area (Department of National Parks, Wildlife and Plant Conservation, 2016). Most of the natural vegetation cover has been converted into grasslands or open secondary forests, due to logging, forest fires and social and economic factors (Schmidt-Vogt, 1999; Luukanen, 2001; Laurance, 2007; Frois *et al*., 2008).

Therefore, forest restoration is urgently required for this most important watersheds in the Thailand, to prevent run-off and soil erosion (Santisuk, 2006), and to recreate an ecosystem that can function effectively (Stanturf *et al*., 2014). In previous forest restoration and reforestation projects, launched by the government sector, *Pinus kesiya* was commonly planted on abandoned areas in northern Thailand (Santisuk, 2006) due to its fast-growing characteristic (Kiianmaa, 2005). Another species that has been planted widely is *Leucaena leucocephala*, because it helps to improve soil fertility and survives well during droughts (Frois *et al*., 2008). The use of pioneer species, either exotic or native species in reforestation projects, is mainly to reduce the growth of weeds and prevent soil erosion.

A technique with higher diversity is called the 'framework species method' which has been tested for restoring montane forests in northern Thailand over the past 20 years.

This technique employs a mixture of 20-30 native tree species, to accelerate natural regeneration. It results in a high level of biodiversity recovery in less than 10 years after planting (Elliott *et al.*, 2013). Although practitioners and local communities can implement restoration appropriately, fires in the dry season are a major problem. Fire destroys young seedlings and reduces growth. A single fire can eliminate most plants in a restoration plot, thus deflecting the successional pathway.

Fires are infrequent in montane forest ecosystems. Consequently, tree species in such ecosystems lack adaptations to fires (Cerda and Robichaud, 2009). Fire disturbance affects ecosystem regeneration: it destroys the soil seed bank (Nieuwstadt *et al.*, 2001; Lentile *et al.*, 2007), and encourages non-forest species to establish after the fire (Setterfield, 2002; Lentile *et al.*, 2007). Furthermore, in forest restoration projects, fire obstructs an ecosystems recovery, by burning the planted saplings (Lawes *et al.*, 2011a). Although, some species grow back after fire (resilience), repeated fires kill small trees and thus prevent recruitment of adult trees (Lawes *et al.*, 2011a).

In fire-prone ecosystems, trees have resilient characteristics that include thick bark (Pinard *et al.*, 1999; Hoffmann *et al.*, 2003), the ability to resprout (Kauffman, 1991) and/or a high growth rate (Hoffmann *et al.*, 2003). To restore montane forest ecosystems that are less adapted to fire, native tree species with fire resilient characteristics must be selected.

Climate change is increasing the risk of more frequent and more intense fires, especially where increasing temperature is accompanied by lower precipitation or longer dry seasons (Burton *et al.*, 2010). To prepare for this unpredictable situation, native tree species that survive after a fire should be identified and promoted in forest restoration. These native tree species not only restore a forest but also resilient ecosystem, to cope with unpredictable future fires, increase an ability to absorb changes, and to hold stability and return to the equilibrium state after disturbance (Holling, 1973). There are only a few publications on the effects of fire on native tree species in the tropics (Elliott *et al.*, 2003, Marod *et al.*, 2002). No report has been found on the minimum size of native trees that are likely to survive after a fire in the montane forest ecosystem in northern Thailand.

This study addressed 3 main research questions:

1) How much does size of native trees affect their survival after a fire?

2) How much does size of native trees affect their resprouting ability after a fire?

3) What are the most suitable fire-resilient native tree species to plant for restoring montane forest ecosystems in fire-prone areas?

1.2 Hypotheses

1) Post fire survival increases with increasing tree size.

2) Post fire resprouting ability increases with increasing tree size.

1.3 Research Objectives

1) To quantity the effects of tree size on their survival after fire.

2) To quantity the effects of tree size on reprouting ability after fire.

3) To identify potential tree species candidates to be used for restoring montane forest ecosystems in fire-prone areas.

1.4 Usefulness of the research

1) This study suggests a minimum size of trees which will likely survive better after burning. Therefore, this knowledge can be used to determine optimum size of seedlings when no further plot maintenance is needed, also optimum length of time to implement fire prevention.

2) This study provides a better understanding of relationship between tree size and resprouting ability after fire disturbance (fire resilience).

3) This study provides a list of potential tree species for restoring montane forest ecosystem in fire-prone area.

CHAPTER 2

Literature review

2.1 Montane forest ecosystem

Montane forest ecosystems are commonly found above 1,000 m elevation (Santisuk, 2006). It is predominant across most of Thailand's critical water catchments. Such ecosystems play an important role in the hydrologic cycle, due to its ability to prevent run-off after heavy downpours, ensuring continuous stream flow to lower-lying areas and protection against soil erosion (Ateroffa and Rada, 2000; Khamying et al., 2003). Montane forest ecosystems contain high biodiversity, some families that are commonly found e.g. Magnoliaceae, Theaceae, Lauraceae, Moraceae and Fagaceae (Santisuk, 2006; FORRU, 2006).

During the past few decades of rapid economic development, Thailand's natural resources have been severely degraded (ICEM, 2003). Montane forest ecosystems have been modified by various types of land-use, particularly shifting cultivation (Delang, 2002). Since the last century, people have illegally settled in the mountain chains of the North, altering the natural montane vegetation (Santisuk, 2006), and increasing the intensity of land use which has been driven by economic development and population growth (Santisuk, 2006; ICEM, 2003). Deforestation in northern Thailand caused by land encroachment, agricultural expansion, intensive shifting cultivation and fire disturbance (ICEM, 2003; Ongprasert, 2011). Although patches of trees can be found on northern mountains land degradation might have caused heavy soil erosion, siltation, flash floods etc. Continuingly, these events have become serious problems of the country (Maxwell and Elliott, 2001).

Another major cause of montane forest degradation is fire (Ongprasert, 2011). Due to high annual humidity in montane forest ecosystems, forest fire is naturally less frequent compared to other ecosystems in lower elevation (Santisuk, 2006; Hoffman, 2003; FORRU, 2006; Bruijnzeel et al., 2011). However, using fire to control weeds in highland agricultural areas is a major cause of fire disturbance in this ecosystem (Bruijnzeel et al., 2011). Furthermore, climate change could cause some areas to become drier, and therefore increase fire severity in the montane forests (Cochrane, 2003, Bruijnzeel et al., 2011).

2.2 Forest restoration in high elevation

SER (2004) proposed that ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed. The term restoration could be translated into different meanings or levels of practices in the context of Thailand. Forest restoration is a specialized form of reforestation. Elliott et al. (2013) defined forest restoration as re-instatement an ecological process, which accelerates recovery of a forest's structure, ecological functioning and biodiversity levels, towards those typical of a climax forest.

Reforestation in northern Thailand began in 1906 in province of Phrae. Its goal was to replant teak trees in logging concession areas, and was later expanded to include planting more tree species in the watershed areas (Royal Forestry Department, 2009). This same period of time, the Royal Forestry Department focused on planting few species of high value timber, such as teak (Tectona grandis) and rose wood (Xylia xylocarpa) (Royal Forestry Department, 2009). Later in 1941, they changed their strategy from planting high value timber species to fast growing species (Kiianmaa, 2005), such as eucalyptus (Eucalyptus oblique) and pine (Pinus kesiya). Most reforestation programs focused on montane forest ecosystem, such as in Doi Inthanon National Park (Werner and Santisuk, 1993). Another species that had been planted widely was Leucaena leucocephala. It is used in many plantations because it helps to improve soil fertility and survives well in drought condition (Frois et al., 2008). The use of pioneer species, either exotic or native species in reforestation projects, is mainly to reduce the incidence of weeds and prevent soil erosion.

Planting few tree species can bring back green cover and some ecological functions but does little for biodiversity. A restoration technique called framework species method was first coined in Queensland, Australia, and later adopted in Thailand by the Forest Restoration Research Unit, Chiang Mai University (FORRU-CMU). It has been scarcely tested in northern Thailand, since 1997 (FORRU, 2006). This technique calls for planting a mixture of 20-30 indigenous tree species, to encourage the re-establishment more balance in the ecosystems (FORRU, 2006) (Figure 2.1). This technique encourages seed dispersers from nearby intact forests such as birds, squirrels and civets by providing resources such as fruits and suitable habitats. Its results showed a high level of biodiversity recovery in less than 10 years after planting (Elliott et al., 2013).

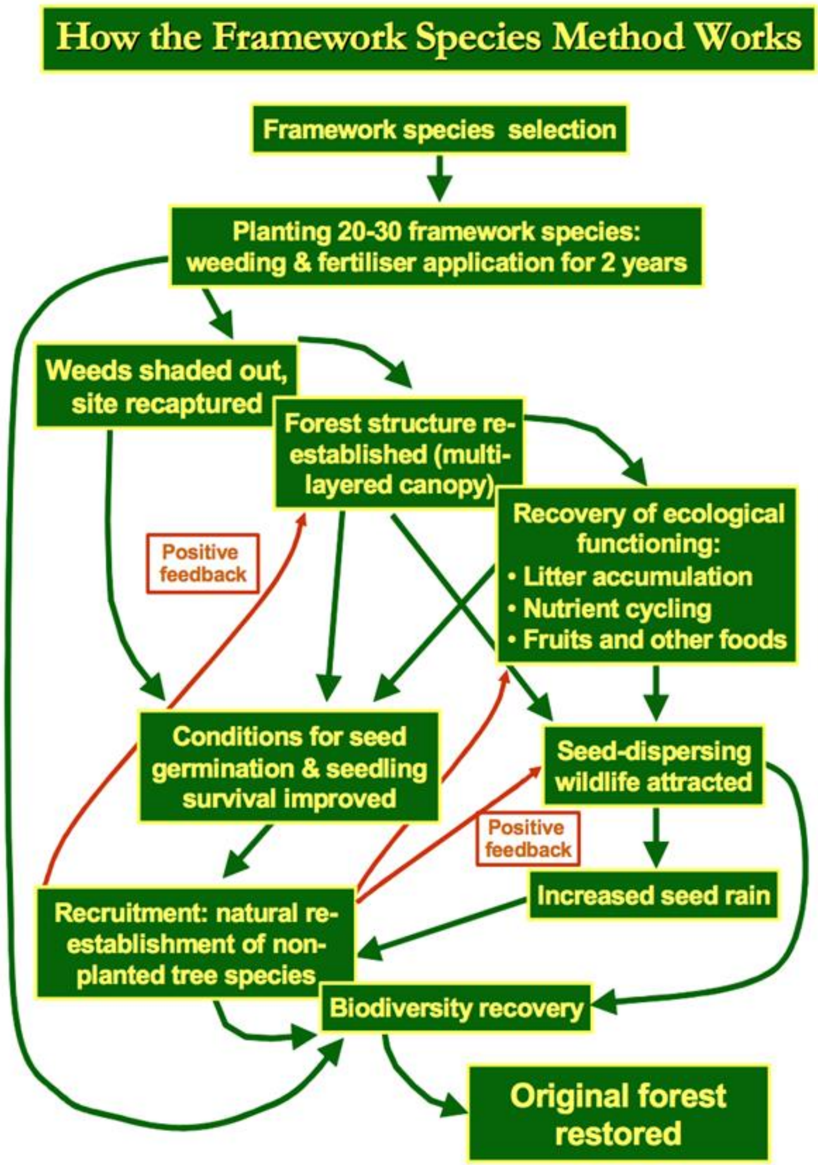


Figure 2.1: Concept of framework species method (FORRU, 2006)

2.3 Fire disturbance in montane forest ecosystem

For successful restoration in northern Thailand, fire prevention has been widely emphasized. An ecosystem, affected by fire, has decreased in ecosystems dynamics (Hoffman, 1996; Schmoldt et al., 1999), ecosystem functionality (Attiwill, 1994), plant species diversity and plant abundance (Attiwill, 1994; Vaidhayakarn and Maxwell, 2010).

In many tropical forest ecosystems, fires damage mature vegetation (Maxwell and Elliott, 2001), affect ecosystem regeneration, and destroy the soil seed bank (Maxwell and Elliott, 2001; Nieuwstadt et al., 2001; Lentile et al., 2007). Ground flora dominated by fire resistant-grasses are proliferated and therefore can obstruct forest recovery (Maxwell and Elliott, 2001; Setterfield, 2002; Lentile et al., 2007).

In Thailand, fire disturbance is common in deciduous forests, and sparsely occurs in evergreen forests which contains high moisture biomass content (Junpen et al., 2013). Fire occurs during dry season (December to May), which peak in March (Junpen et al., 2013). The Forest Fire Control Division of the Royal Forestry Department (FFCD) (2011) reported that all fires are human-caused with various reasons (% of incident) including gathering of forest non-timber products (39%) such as mushroom, hunting (24%), agriculture residue burning for land clearing before the next crop (19%), incendiary fire (10%), illegal logging (2%) and other (6%).

Junpen et al. (2013) reported fire hotspot mainly occurs in upper northern, western, upper northern eastern, and east side of the northeastern regions in Thailand. The study of fire hot spots in Thailand during 2005 - 2009 found the intensity of fire hotspot in 2007, which El Nino occurred in this year (Junpen et al., 2013) (Figure 2.2). Furthermore, Cochrane (2001) reported that climate change is one of major causes of drier and higher temperatures in the dry season, which can contribute to higher intensity and frequency of fire disturbance in terrestrial ecosystems. Climate change may increase the likelihood of fire disturbance in montane forest ecosystem where fires are less frequent (Hoffman, 2003; FORRU, 2006; Bruijnzeel et al., 2011). Therefore, plants in this ecosystem that barely evolved along with frequent fires, are more likely to disappear after a fire (Hoffman, 2003; Conchrone, 2003; ICEM, 2003; Marrinan et al., 2005).

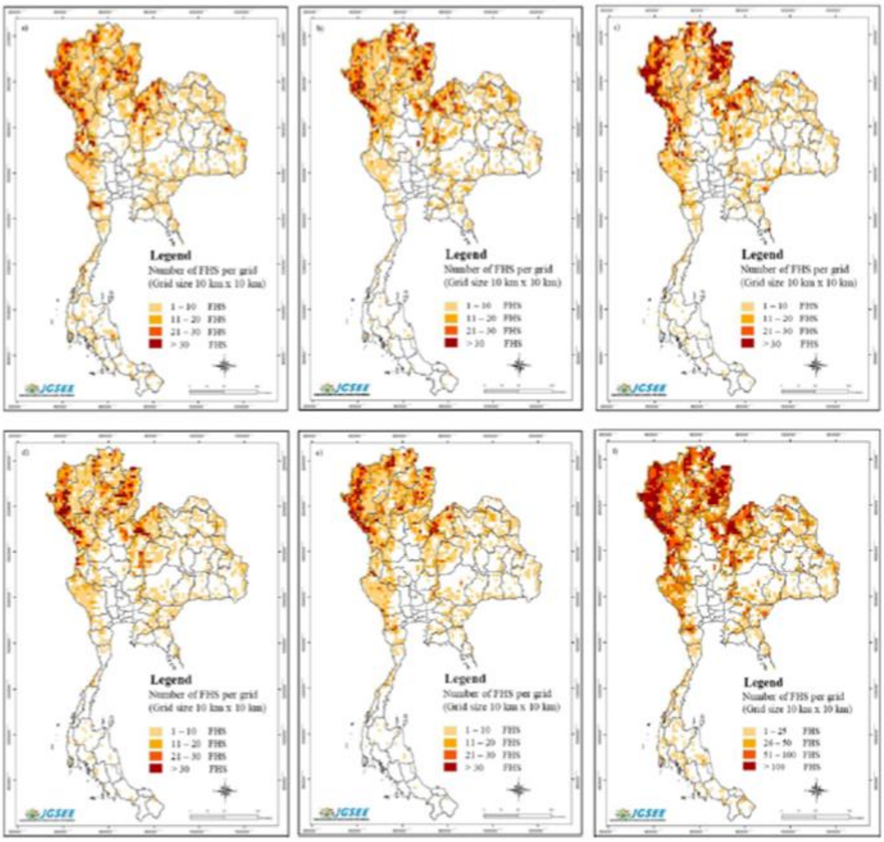


Figure 2.2: Forest fire hotspots density distribution on yearly scale for (a) 2005, (b) 2006, (c) 2007, (d) 2008, (e) 2009, and (f) cumulative forest fire hotspots density distribution during 2005 - 2009 (Junpen et al., 2013)

2.4 Factors affecting survival after fire disturbance

Fire kills trees by destroying the cambium during burning. However, post-fire mortality is a consequence of stem necrosis (Dickinson et al., 2004; Bova and Dickinson, 2005) and stem deformation (Michaletz et al., 2011). Stem necrosis is a cause of preventing downward translocation of photosynthate, trees will survive until root starvation (Nobel, 2005). Stem deformation is another cause of post-fire mortality. In normal stages, lignin, hemicellulose and cellulose polymer in the conduit wall are hard and glassy, but soft in high temperature. Heating softens cell walls and sap surface tension. These components become permanently deformation in cooler stages after a fire, and results in permanent disruption of xylem flow and reduce hydraulic conductivity (Michaletz et al., 2011).

Many studies reported that post-fire survival is correlated with bark thickness (Pinard et al., 1999; Hoffmann et al., 2003; Midgley et al., 2010; Lawes et al., 2011a; Lawes et al., 2011b; Xaub et al., 2013), that protect epicormic buds, cambium, hydraulic system and stem injury. Bark slows the heat transfer to cambium, that prevents stem necrosis (Bova and Dickinson, 2005), deformation (Michaletz et al., 2011) and failure of the hydraulic system (Midgley et al., 2010). Several studies in tropical forests showed positive relationship between bark thickness and stem diameter in tropical species (Bova and Dickinson, 2005; Midgley et al., 2010; Lawes et al., 2011a). Therefore, bigger trees have thicker bark and higher survival ability after fire. Forest species require considerably greater stem diameter to ensure stem survival during a burn (Hoffmann et al., 2003).

2.5 Factors affecting resprouting ability after fire disturbance

Resprouting is the response of initial growth from buds follow disturbance; this response implies the potential for repeated vegetative regeneration from a source of protected buds and meristem (Clarke et al., 2012). This trait is important for achieving persistence at the species level and can survive diverse disturbances (Marrinam et al., 2005; Vesk and Westoby, 2004; Clarke et al., 2012). Resprouting is also resilient to severe disturbances at the community level, which might have caused consequences on vegetation dynamics, community composition, and species coexistence (Poorter et al., 2010; Clarke et al., 2012).

The influences and consequences of resprouting from individuals to communities are discussed in several publications (Figure 2.3). Clarke et al. (2012) proposed a new conceptual framework for resprouting theory, the buds-protection-resources (BPR) scheme. This concept considered resprouting as a plant functional trait based on bud location, their protection, and resourcing of regrowth, in response to disturbance (Clarke et al., 2010; Lawes and Clarke, 2011; Hoffmann et al., 2012) (Figure 2.3).

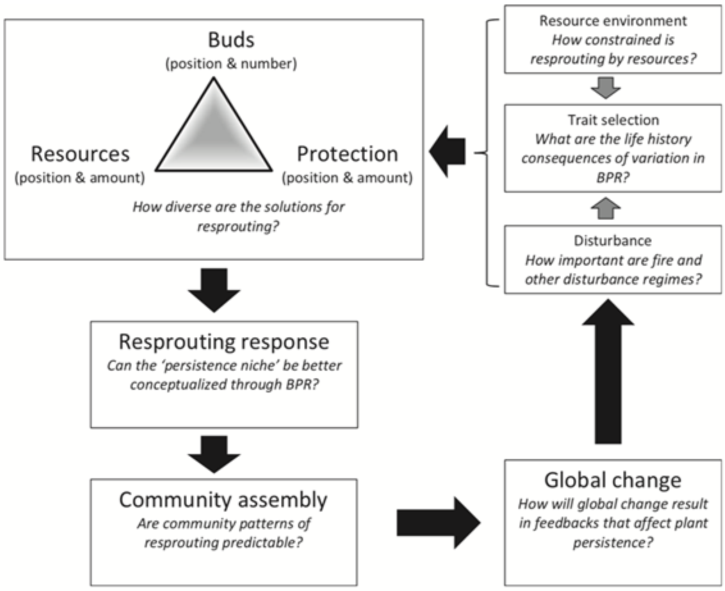


Figure 2.3: The influences and consequences of resprouting from individuals to communities (Clarke et al., 2012)

Protecting buds from scorching flames is an important factor for resprouting (Kauffman, 1991; Marrinam et al., 2005). Clarke et al. (2012) summarized that this factor contains 4 mechanisms; there are growing rapidly, a thicker stem, thick bark and special protection. First, rapid growth helps plants to reach an escape height which allows buds in the crown to escape being scorched. Secondly, thicker stem helps to buffer the xylem against hydraulic failure. Thirdly, thick bark protects xylem and phloem from fire damage; however in a forest ecosystem where fire is less selective, bark thickness inhibits resprouting by hindering epicormic bud emergence. Special protection is the last mechanism that depends on species, such as deeply embed meristems protection of Eucalyptus.

When a plant loses above ground biomass from a disturbance, resources allocation to stem and root is a potential source to support resprouting (Clarke et al., 2012). Carbohydrate reserve was reported as the important resource (Bond and Midgley, 2001; Shibata et al., 2016), which correlated with the root size (Shibata et al., 2016). Clarke et al. (2012) also reported that in more fire-prone system, allocation of resources to roots is increased.

Furthermore, bud senescence also affects resprouting ability, but decreases with age. Bud senescence involves with genetics, physiological and anatomical change (Clarke et al., 2012). Bond and Midgley (2001) found that resprouting ability is more common in juveniles than adults. The study in a tropical forest in Malaysia by Kauffman (1991) found the significant inverse relationship between resprouting and stem diameter, bigger trees have lower resprouting ability. Similarly, failure rate of resprouting was observed in older oak tree with thicker bark (Johnson et al., 2002).

2.6 Fire resilience of indigenous tree species in montane forest ecosystem

Ecological resilience is defined as an amount of disturbance that an ecosystem could withstand without changing self-organizing processes and structures (Gunderson, 2000). Each ecosystem has a certain ability to absorb changes, hold, maintain stability and return to the equilibrium state after a disturbance (Holling, 1973). In any ecosystem, there is a disturbance threshold, a critical point at which an ecosystem could switch to another state (Standish et al., 2014). Small disturbances might cause ecosystem shift, if resilience is lacking. On the other hand, an ecosystem with high resilience can withstand a high intensity disturbance. The stability of any ecosystem depends on two factors; level of disturbance and ecosystem resilience (Standish et al., 2014).

Nieuwstadt et al. (2001) who studied burned forest of east Kalimantan in Indonesia, suggested 4 main processes to recover after burnt; they were tree survival, resprouting of damaged trees, germination of seeds from seed bank and the seed rain. Resprouting after fire disturbance is an effective persistence mechanism after a fire disturbances (Marrinam et al., 2005) by a shorten time to recovery with a rapid regrowth (Kauffman, 1991; Marrinam et al., 2005; Lawes et al., 2011a) and a greater capacity for exploitation of limited resources in tropical forest (Kauffman, 1991). Four factors affecting on resprouting ability including intensity of disturbance, size, resources allocation and post neighboring vegetation (Bond and Midgley, 2001; Vesk and Westoby, 2004).

Knowledge of fire resilience of indigenous tree species in montane forest ecosystem is essential for restoration. Forest restoration which is aiming to recover this forest type ought to be concerned about planting native tree species with fire resilient characteristics (FORRU, 2006). Only one study on fire resilience of native tree species from montane forest ecosystem exist; Elliott et al. (2003) reported that by three growing seasons after planting (trees were 33 months old), most framework tree species are large enough to recover well after fire, whereas younger trees (21 months old) showed greater variability in their response. However, no statistical analysis was performed to explain this interesting finding.

2.7 Restoring a resilient ecosystem in changing climate

Fire severity may increase with dryness due to climate change (Cochrane, 2003, Bruijnzeel et al., 2011). The highest intensity of fire hotspots reported in 2007 when El Nino occurred (Junpen et al., 2013). Climate change has become more obvious and caused serious consequences currently. Therefore, restoring biodiversity and ecological processes might not be enough to withstand this change. Restoring ecosystem resilience may be an effective way to build adaptive capacity to climate change (Padgham, 2014).

Bernhardt and Leslie (2013) reported mechanisms that enhancing the resilience: diversity, connectivity and adaptive capacity. Biodiversity is increasing the likelihood of some species and/or functional groups that they are resistant to disturbance, to compensate some species within community and facilitate ecological processes. Multiple forms of connectivity can stabilize ecosystems and enhance recovery following a severe disturbance. Adaptation to climate change is an important trait to survive new condition.

In ecosystems with resistance species or species that are able to recover quickly following disturbance, these are able to maintain ecosystem processes that sustain function and lack of loss in productivity (Padgham, 2014). In the planning process of forest restoration, diversity of native tree species, connectivity between species and /or family, and species with fire adaptation, is needed to create fire resilient systems: this should be integrated holistically.

CHAPTER 3

Methodology

3.1 Site Description

Four different forest restoration plots with varying ages were selected for this study. The youngest plot (1-year-old) was in Mon Long (ML) area and the second plots (2-, 14- and 17-year-old plots) were near Ban Mae Sa Mai and Ban Mae Sa Noi (BMSs). All plots were in the Mae Rim district, in the north of Chiang Mai province, in northern Thailand (Figure 3.1 and Table 3.1). Mon Long (ML) is the highest mountain in the Mae Rim district at about 1,450 m a.s.l. The ML plot was located slightly below the highest point between 1,350 - 1,380 m a.s.l. Ban Mae Sa Mai/Ban Mae Sa Noi (BMS) are Hmong villages located at about 1,200 m a.s.l. BMSs plots are situated about 3 km further from the villages between 1,260 - 1,320 m a.s.l. Annual mean temperature during this study (2015) was between 22 - 23 degree Celsius, average rainfall was between 1,350 - 2,500 mm/year and relative humidity was 70 - 80% (Mae SA Mai Royal project, 2016 - unpublished information).

Before the 1960s the land in northern Thailand was covered with montane forest. Dominant tree families in this forest ecosystem sites were Magnoliaceae, Theaceae, Lauraceae and Fagaceae (Werner and Santisuk, 1993). Such forests have been extensively cleared and replaced with cash crops e.g. cabbages, maize, carrots, lettuces, etc. Intensive maintenance and heavy chemical inputs are required continuingly for these crops. Before restoration activity began in 1997, the abandoned fields were previously dominated by herbaceous weeds such as Pteridium *aquilinum* (L.) Kuhn (Dennstaedtiaceae), *Bidens pilosa* L., *Ageratum conyzoides* (L.) L., *Eupatorium odoratum* L. (all Compositae), *Commelina diffusa* Burm.f. (Commelinaceae) and grasses e.g. *Phragmites vallatoria* (Benth.) Mabille, *Imperata cylindrical* (L.) Raeusch and *Thysanolaena latifolia* (Roxb. Ex Hornem.) Honda (both Gramineae).

Forest fires are common during the dry season in northern Thailand. Although fire breaks were created around all restoration plots in both ML and BMSs sites, some plots were partially burned due to various factors; topography (e.g. fire can easily cross a fire break strip in an area with steep slopes), location in the landscape (e.g. the plots are surrounded with agricultural activities which usually use fire to clear away weeds), and lacking a holistic plan of fire management. Fire entered BMSs plots in mid-April 2015. An 80,000 m2 area was burned. Later in early May 2015, the same thing happened over an area of 960 m2 in ML plot.

Table 3.1 GPS coordinates of all studied restoration plots

Site	**Planting year**	**Plot age (year)**	**Planted area (m2)**	**GPS coordinates**	**Elevation (m)**
ML	2014	1	960	N 18° 55’ 22.50’’ E 98° 50’ 27.90’’	1350 - 1380
MSMs	2013	2	3,200	N 18° 51’ 19.80’’ E 98° 50’ 53.29’’	1260 - 1290
MSMs	2001	14	1,600	N 18° 51’ 25.26’’ E 98° 50’ 57.60’’	1280 - 1310
MSMs	1998	17	1,600	N 18° 51’ 27.24’’ E 98° 50’ 53.29’’	1280 - 1320

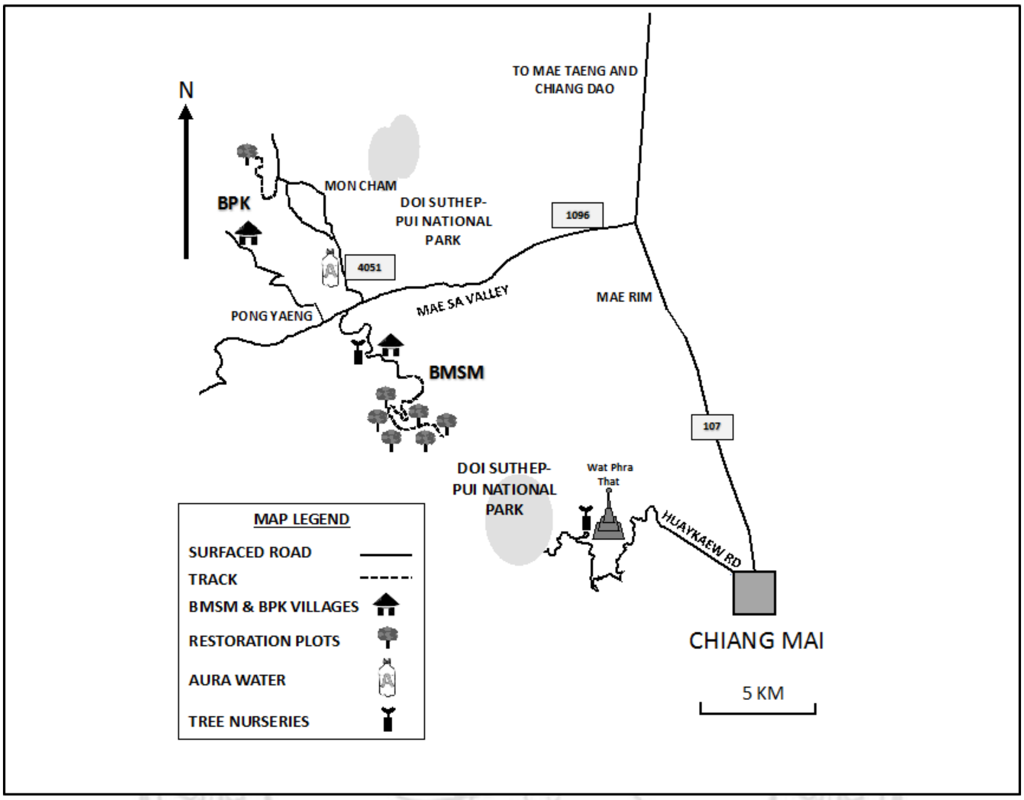


Figure 3.1: Map BMSs and ML restoration sites

3.2 Studied species

A total of 39 native tree species were studied because they had a minimum of five surviving trees after the fire in the summer of 2015. The list of all studied species is presented in Table 3.2, only Prunus cerasoides existed in all plots. There were 6, 23, 14 and 18 species presented in 1-, 2-, 14- and 17-year-old plot (planted in 2014, 2013, 2001 and 1998 respectively).

Table 3.2 The number of surviving trees in all studied plots

Scientific name	Family	Number of trees			
		1-y	2-y	14-y	17-y
Acrocarpus fraxinifolius Arn.	Leguminosae	5			
Alangium kurzii Craib	Cornaceae	6			
Artocarpus nitidus Trécul	Moraceae		18	18	
Bischofia javanica Blume	Phyllanthaceae		37	6	41
Castanopsis acuminatissima (Blume) A.DC.	Fagaceae		25	8	
Castanopsis calathiformis (Skan) Rehder & E.H.Wilson	Fagaceae				7
Castanopsis diversifolia (Kurz) King ex Hook.f.	Fagaceae			6	
Castanopsis tribuloides (Sm.) A.DC.	Fagaceae	8	16		
Choerospondias axillaris (Roxb.) B.L.Burtt & A.W.Hill	Anacardiaceae		18	7	20
Cinnamomum iners Reinw. ex Blume	Lauraceae		30		7
Cinnamomum longipetiolatum H.W.Li	Lauraceae	6			
Diospyros glandulosa Lace	Ebenaceae		27		8
Duabanga grandiflora (DC.) Walp.	Lythraceae		8		
Ficus altissima Blume	Moraceae				9
Ficus auriculata Lour.	Moraceae		40		
Ficus callosa Willd.	Moraceae		34		
Ficus hispida L.f.	Moraceae		29		

Table 3.2 The number of surviving trees in all studied plots (Continued)

Scientific name	**Family**	**Number of trees**			
		1-y	**2-y**	**14-y**	**17-y**
Ficus subulata Blume	Moraceae			6	
Garcinia mckeaniana Craib	Clusiaceae				6
Gmelina arborea Roxb.	Lamiaceae				6
Helicia nilagirica Bedd.	Proteaceae				13
Heynea trijuga Roxb. ex Sims	Meliaceae		30	6	
Hovenia dulcis Thunb.	Rhamnaceae		37		8
Magnolia baillonii Pierre	Magnoliaceae		27	15	
Magnolia garrettii (Craib) V.S.Kumar	Magnoliaceae		24		10
Markhamia stipulata (*Wall*.) Seem.	Bignoniaceae			5	
Melia azedarach L.	Meliaceae				11
Ocotea lancifolia (*Schott*) Mez	Lauraceae				7
Podocarpus neriifolius D. Don	Podocarpaceae		25		
Protium serratum (Wall. ex Colebr.) Engl.	Burseraceae		26		
Prunus cerasoides Buch. -Ham. ex D.Don	Rosaceae	6	22	56	14
Pterospermum grandiflorum Craib	Malvaceae		34		
Quercus semiserrata Roxb.	Fagaceae	7		5	10
Sapindus rarak DC.	Sapindaceae				16
Sarcosperma arboreum Hook.f.	Sapotaceae		23	10	19

Table 3.2 The number of surviving trees in all studied plots (Continued)

Scientific name	Family	Number of trees			
		1-y	2-y	14-y	17-y
Scleropyrum pentandrum (Dennst.) Mabb.	Santalaceae		14		
Styrax benzoides W. G. Craib	Styracaceae		10		
Syzygium albiflorum (Duthie ex Kurz) Bahadur & R.C.Gaur	Myrtaceae			6	8
Syzygium tetragonum (Wight) Wall. ex Walp.	Myrtaceae		21	14	
Total species		**6**	**23**	**14**	**18**

3.3 Data collection and data analysis

A 40 x 40 m sampling plot was established in each restoration plot that had been burnt in 2015 (planted in 1998, 2001 and 2013 at BMSs), except the plot planted in 2014 at ML where two sampling plots were laid out due to the size of burned area. The distance between each plot in BMSs was about 200 m, whereas only one plot was located in the ML site.

3.3.1 Tree size and survival after a fire disturbance

The plots were surveyed 3 times during this study; 2 weeks (end of dry season), 18 weeks (beginning of rainy season) and 30 weeks (end of rainy season) after the plots had been burned (during April and May 2015). Girth at breast height (GBH) of all trees that survived was measured unless they were smaller than 10 cm, then their root collar diameters (RCD) were measured instead. GBH was then converted to diameter at breast height (DBH) for data analysis.

According to DBH measured from the last monitoring (30 weeks after fire), the trees were grouped into 13 classes (40 mm interval in each class), then calculated their survival percentage. Trees sizes with stem diameter between 1 - 40 mm were analyzed by Chi-square test at 95% confidence interval. This test was conducted to assess the survival of burnt tree at 30 weeks after fire, response in difference size of tree. Trees with DBH ≤ 40 mm were grouped into 8 classes (5 mm interval), and grouped into bigger group if not significant for testing Chi-square.

Simple linear regression used to explore relations between stem diameter (only tree that DBH ≤ 40 mm) and survival after fire. The survival at 30 weeks after fire disturbance was response variable and stem diameter was explanatory variable. R2 was interesting in figure out the correlation of this relation.

Only small trees (DBH between 1 - 50 mm) were analyzed by a generalized linear mixed models (GLMMs) with a binomial family test in the R program version 3.3.1, to determine the effects of tree size on their survival after fire. Trees were categorized as dead (0) or alive (1) and then used as a dependent variable. Tree stem diameter (DBH or RCD) was used as an independent variable (fixed effect). For 1- and 2-year-old plot, the size monitored from before the fire in the summer 2015 (November 2014 and December 2014 respectively) was used. Unfortunately, this data is not available for the old plots, so the size of surviving trees monitored at 2 weeks after burning was used for the 14- and 17-year-plot. In addition, planting plots were carried out as another independent variable (random effect) to reduce the variance of size on survival ability in GLMMs analysis.

3.3.2 Tree size and resprouting ability

In addition to tree survival from 3.31, the number of resprouting shoots were recorded from all plots. For all trees that survived during the last monitoring (30 weeks after a fire event), they were grouped into 17 classes (30 mm range in each class) to represent the ability of resprouting.

Only trees with a stem diameters between 1 to 210 mm were subsequently analyzed using generalized linear mixed models (GLMMs) with a Poisson family test in the R program version 3.3.1. To determine the effects of tree size on its resprouting ability, stem diameter (same data used in 3.3.1) was used as an independent variable (fixed effect), while the planting plot was assigned as a random effect.

3.3.3 Suitability index

A suitability index was calculated from the data of the young plots (1- and 2-year-old plots). This index aims to select suitable native tree species for restoring forest ecosystem in fire-prone sites. Three different variables were selected; (i) survival percentage (ii) relative growth rate (RGR) of stem diameter and (iii) resprouting ability (number of resprouting shoots after fire) of each native tree species.

(i) Survival percentage of each native tree species was calculated as a proportion of the number of trees survived before fire disturbance and during the last monitoring (30 weeks after the fire).

$$\text{Percent survival } (\%) = \frac{Number\ of\ survived\ trees}{Total\ number\ of\ trees} \times 100$$

(ii) RGR was presented as a percentage of growth in one year (365 days).

$$\text{Relative Growth Rate (RGR)} = \frac{\ln(G2)-\ln(G1)}{T2-T1} \times 365 \times 100$$

Where; G1 = Stem diameter (mm) at 1st monitoring (2 weeks after fire)

G2 = Stem diameter (mm) at 3rd monitoring (30 weeks after fire)

T1 = Date of 1st monitoring (2 weeks after fire)

T2 = Date of 3rd monitoring (30 weeks after fire)

(iii) Resprouting ability was presented for each species as a proportion of the number of resprouting shoots and the number of all burned trees.

$$\text{Resprouting ability} = \frac{No.\ of\ resprouting\ shoot}{Every\ trees\ that\ burnt}$$

The highest value from all three parameters was then selected and then converted into a score, ranging from 0 to 100. The survival percentage was multiplied by two after converting due to their importance to fire resilience. All scores were summed for each species and they were again ranked from 0 to 100.

CHAPTER 4

Results and discussion

4.1 Percent survival after fire between old and young plot

Figure 4.1 presents survival following fire. By the 3rd monitoring (30 weeks after fire), the percent survival of trees in the 1-, 2-, 14- and 17-year-old plots was 42.8, 39.8, 95.3 and 98.6 respectively. Inspection of the trees in the 14- and 17-year old plots revealed no immediate mortality due to fire (so 1st monitoring survival was 100%), whereas pre-fire surveys in the younger plots was used to calculate immediate mortality as a result of the fire.

Trees in the older plots (14- and 17-year-old plots) survived well (mean survival across species 95.3% and 98.58% respectively). These remaining trees can be seed source for post-fire ecosystem recovery. Furthermore, forest structure remained largely intact, retaining shady conditions. This result agreed with those of Swaine (1992), who also found that standing trees remaining after fire provide seed sources and shady conditions in dry forest in western Africa.

In contrast, tree survival after burning in the 1- and 2-year-old plots was much lower (mean across species 42.9% and 39.8, respectively) which shows the greater susceptibility of small trees to fire mortality. Fire opened up the younger plots, reducing canopy cover and allowing grasses and herbs to re-invade the sites. This result was similar to that obtaining in Australian tropical forest by Setterfield (2002) who also reported that grass and forb ground cover increased after fire. Such invasion by grasses and vines after fire has also been reported in the Amazonia rain forest (Cochrane, 2003).
Invasion by grasses can increase flammability and increase the subsequent fire risk (Pinard *et al*., 1999; Bond and Keane, 2017).

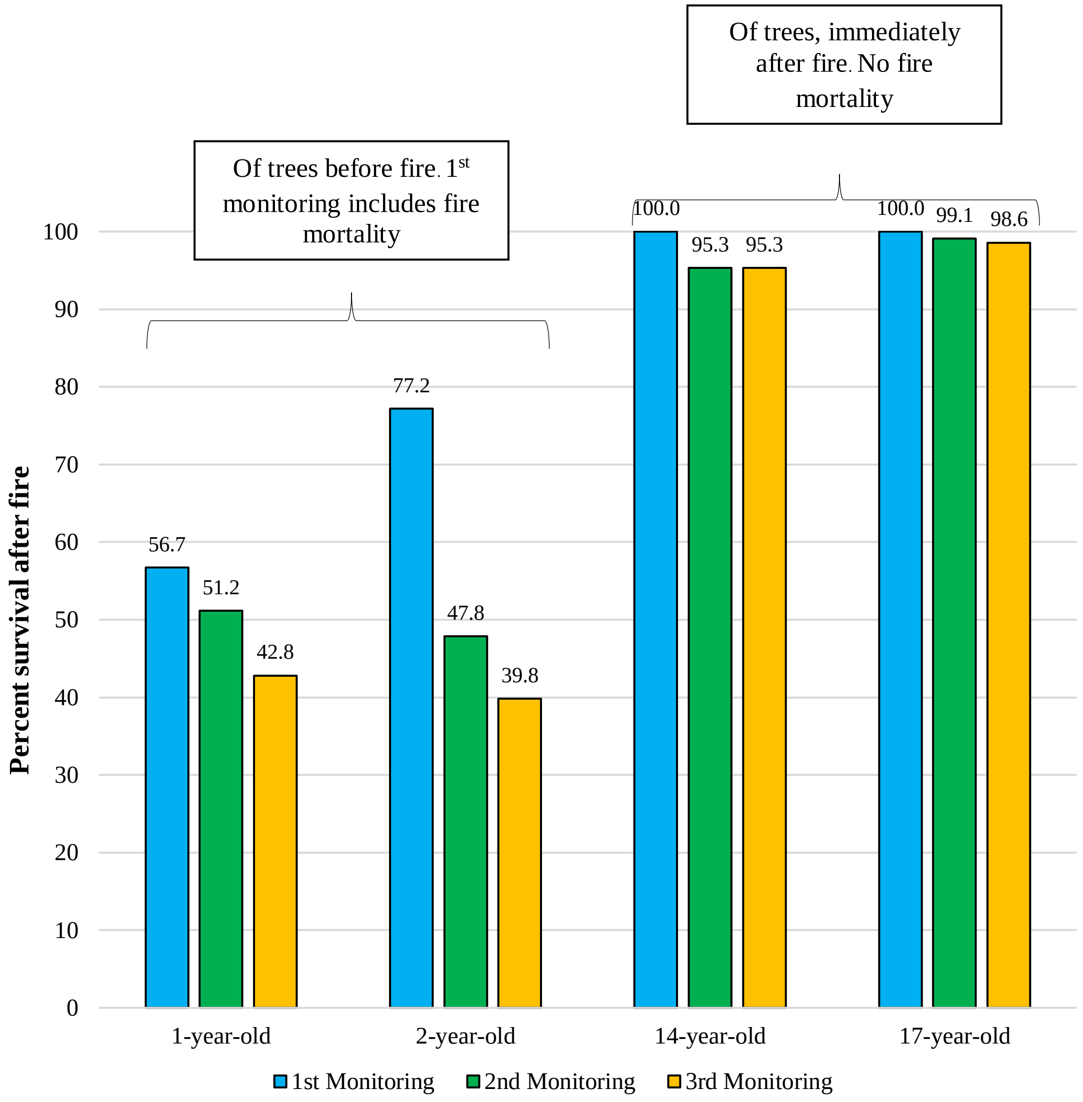


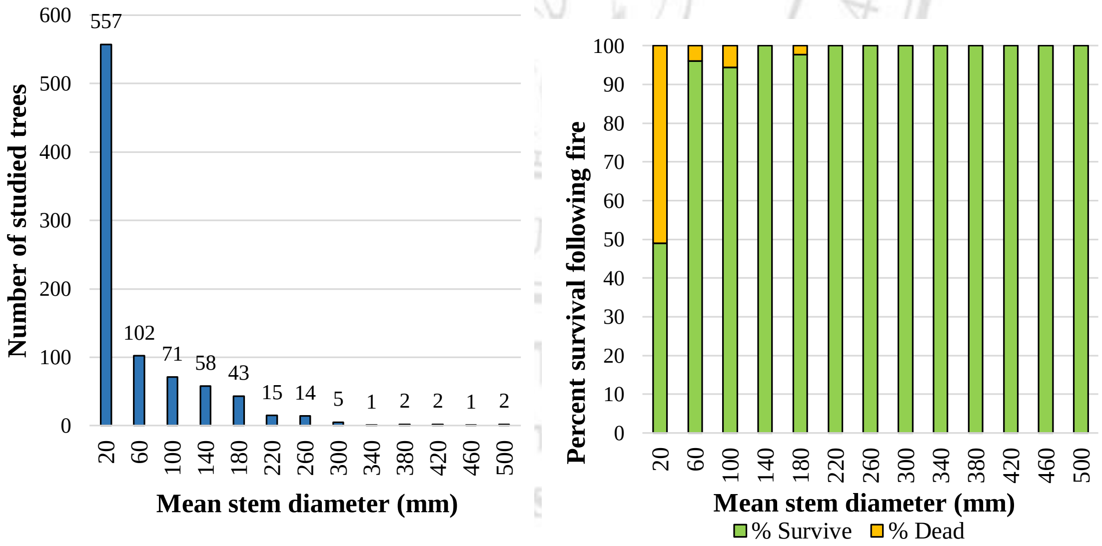
Figure 4.1: Survival percentage of native tree species at 1st, 2nd and 3rd monitoring after fire in each plot

4.2 Percent survival after fire between monitoring

In the younger plots, mortality continued to occur throughout the 30 weeks of the study whereas in the older plots very little additional mortality occurred after the 2nd monitoring (Figure 4.1).

Throughout monitoring, survival percentage of younger trees in this study was more rapidly decreased than older trees, assuming that younger trees faced cambium injury which could cause tree mortality within weeks (Michalet *et al.*, 2011). In Amazonian forest, fire damage continued to kill the trees during 1 and 3-years post fire due to death of the cambium (Barlow *et al.*, 2003). A similar phenomenon was reported in West African forest up to 2 – 4 years die back after fire (Swaine, 1992). Therefore, cambium die-back after fire is an important cause of tree mortality in montane forest ecosystems in northern Thailand, if fires become more frequent or more severe.

Apart from cambium die back, heat is also an important cause of xylem and phloem deformation. Heat reduces xylem conductivity by deform conduit wall, and later become permanent deformation at lower temperature (Michalet *et al.*, 2011). Unlike younger trees, several older trees might not die from cambium die back but facing conduit deformation. With tree conduit deformation, the trees can survive longer period until store carbon reserves are deplete (Michalet *et al.*, 2011).



(a) (b)

Figure 4.2: Number of studied trees (a) and percent survival after fire (b) of 13 classes, stem diameters were classified with 40 mm in each class

4.3 Effects of tree size on survival after fire

Thirteen classes of burned trees were categorized by stem diameter (40 mm interval). The size before burning was used for 1- and 2-year-old plot, whereas data from 1st monitoring was used for 14- and 17-year-old plot. In figure 4.2 (a), mean stem diameter represents each size class, the smallest size class (stem diameter ≤40 mm; mean = 20 mm) consists of the highest number of studied trees, while smaller numbers are in bigger size classes.

Figure 4.2 (b) shows percent survival after fire of trees in each size class, smaller trees were more vulnerable to fire than large ones were. This figure shows that trees larger than 40 mm stem diameter had a 94-100% chance of survival, where as those <40 mm stem diameter had a more or less equal chance of dying or surviving (51% chance of dying).

4.3.1 Chi-square test

Looking more closely at the smallest size class (stem diameter ≤40 mm) (Fig. 4.3), a steady increase in survival with size class can clearly be seen from 29% for the smallest trees to 100% for the largest. A chi-square test was performed to examine the relation between stem diameter and survival after fire. I separated trees into 8 groups by the stem diameter, 5 mm in each class. And I grouped trees that not significant between group, 4 groups were interesting; they were 1.0 – 9.9 mm, 10.0 – 24.9 mm, 25.0 – 34.9 mm and 35.0 – 40.0 mm. Chi-square tests revealed that increases in survival from DBH size classes 1.0 – 9.9 mm to 10.0 – 24.9 mm (X^2 (1, N = 487) = 3.841), from 10.0 – 24.9 to 25.0 – 34.9 mm (X^2 (1, N = 330) = 3.481) and from the latter to 35.0 – 40.0 mm (X^2 (1, N = 70) = 3.481) were highly significant (p = 0.05). Increasing stem diameter significantly increased survival.

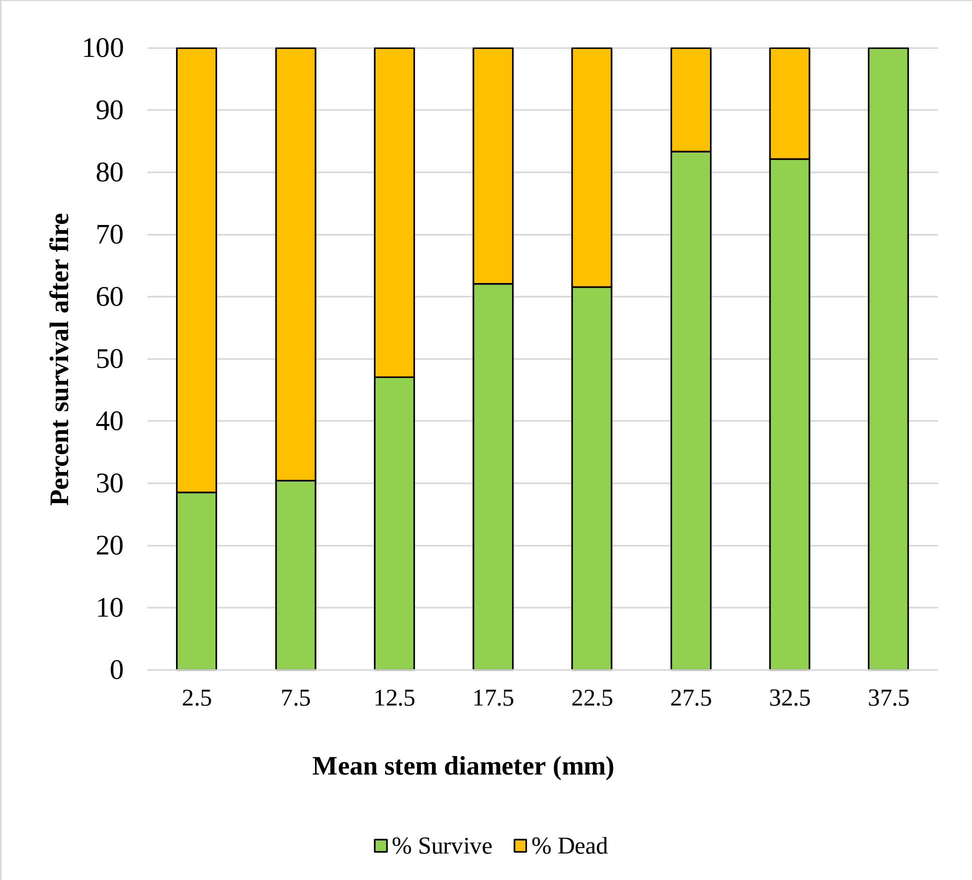


Figure 4.3: Survival and mortality percentage after fire of native tree species in class 1 (1.00 – 39.99 mm), 8 sub-class were classified with 5 mm in each class

4.3.2 Simple linear regression

Figure 4.4 shows a linear relationship between mean stem diameter and percent survival after fire. The results of the regression indicated a positive significant effect between stem diameter and survival (R^2 = 0.9634, p = 1.59 x 10^{-6}). It was found that significantly predicted tendency (β_1 = 2.064, p = 0.002), as did agreeableness (β_0 = 20.609, p = 1.59 x 10^{-6}).

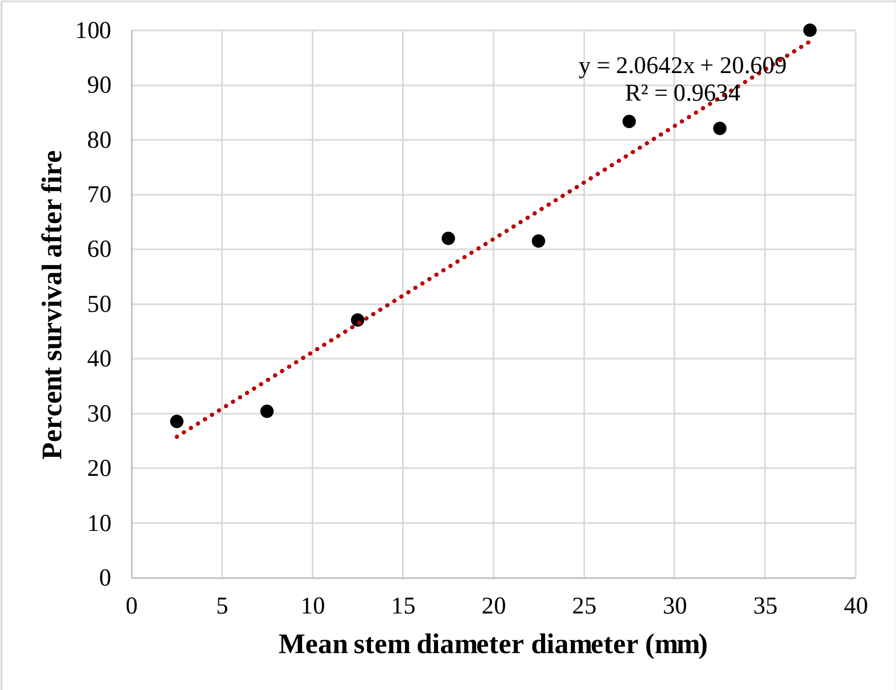


Figure 4.4: Linear relationship between mean stem diameter and percent survival after fire

4.3.3 GLMMs

GLMMs was carried out, to test the effects of stem diameter on survival, after fire. Only small trees (stem diameter ≤50 mm) were focused on this section. Figure 4.5 shows a significantly positive relationship between stem diameter and survival chance (co-efficient estimate ± SE = 0.10 ± 0.01, $z = 6.90$, $p = 5.3\text{x}10^{-12}$), increasing of stem diameter contributed to higher chance of survival after fire. In montane forest, the trees with stem diameter bigger than 40 mm tend to survive better (Figure 4.5).

This trend is similar to that reported in studies in the tropical forests in South America and Asia. In tropical Amazon forest, Xaub *et al*. (2013) reported that survival of trees (>10 cm DBH) after fire was between 79-92%, while similar survival percentage observed in bigger trees from sub-humid tropical forest in eastern Bolivia (84% from trees with DBH > 40 cm) (Pinard *et al*., 1999). In Eastern Kalimantan, Indonesia showed wide range of survival percentage after fire among species (20-95%) of the trees that have DBH > 30 cm (45% survival in average)

(Nieuwstadt *et al.*, 2001). Unsurprisingly, smaller trees with DBH <5-10 cm, showed lower survival percentage after fire, broader range (1-40%) had been reported from tropical Asian forest (Slik and Eichhorn, 2003; Nieuwstadt and Sheil, 2005) compared to 26% from tropical Amazon forest (Pinard et al., 1999).

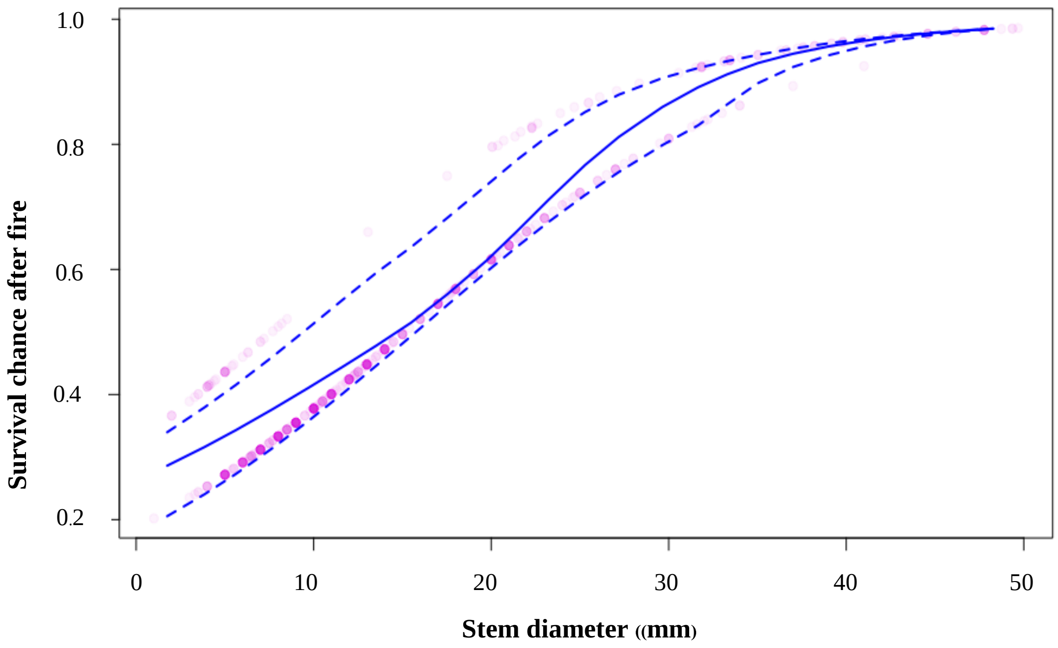


Figure 4.5: Relationship between stem diameter size and chance of tree survival after fire predicted by GLMMs, pink dot is observed value, blue line is predicted line and dash line is 95% interval for the regression line

The studies from Amazonian tropical forests reported high survival chance after fire of trees with a minimum of 10 cm DBH for (Xaub *et al*., 2013), especially trees with double size stem (DBH ≥ 20 cm) can protect cambium tissue from fire (Cochrane *et al.*, 1999). Fire transfers heat to tree stem that causes stem necrosis (Bova and Dickinson, 2005; Midgley *et al*., 2010) and stem deformation (Michalet *et al.*, 2011), both affect survival after fire. Stem necrosis from fire prevents the downward supply of photosynthates and causes of hydraulic system failure (Midgley *et al.*, 2010). In addition, fire reduces xylem conductivity because of vessel deformation and changing in sap surface tension (Michalet *et al.*, 2011). Both intensity of stem necrosis and deformation correlated to the bark thickness, which corresponds to a stem diameter (Bova and Dickinson, 2005; Midgley *et al.*, 2010). Thick bark prevents heat transfer to vascular cambium, so, thicker bark trees had higher percent survival after fire.

In an ecosystem with frequent fire, in resources are allocated to the bark as a defense component to fire, such as *Pinus* species which inhabit in frequently burned area (Jackson *et al.*, 1998). However, the studies of relationship between tree bark and stem size in tropical forests in south central Brazil (Hoffmann *et al.*, 2003) and Australia (Lawes *et al.*, 2011a) showed positive relationship between bark thickness and stem diameter in tropical tree species, bigger tree usually comes with thicker bark.

Bark thickness was considered as a strong strategy to prevent stem from fire (Pinard *et al.*, 1999; Hoffmann, 2003; Xaub *et al.*, 2013; Lawes *et al.*, 2011a; Lawes *et al.*, 2011b). In savanna ecosystems, with annual fires, bark thickness is an important component to prevent the cambium from burning (Hoffmann *et al.* 2003, 2003; Lawes *et al.*, 2011a). Even in tropical forest in Amazonian where fires are rare, the thicker bark tree showed higher ability to survive after fire than the thin bark (Cochrane *et al.*, 1999).

Therefore, bigger trees in this study assumed to have thicker bark that provided a fire defense component, which showed in the results of higher survival percentage in bigger trees in montane forest ecosystem, northern Thailand.

4.4 Percent survival of native tree species in each plot

From 4.3, bigger stems had increased chance of survival. However, different species might contain different traits to survive after fire. Therefore, looking into the species level could contribute to better understanding about this relationship between stem size and survival after fire. The size of the largest tree that died and the smallest tree that survived of each species can provide useful guidance to restoration practitioners.

4.4.1)1-year-old plot

Figure 4.6 shows the survival percentage of six species in 1-year-old plot after fire in May 2015. Three species showed good survival performance (>50%); they were *C. tribuloides*, *C. longiopetiolatum* and *P. cerasoides*. While three other species expressed low survival percentage (<40%); *A. kurzii*, *Q. semiserrata* and *A. fraxinifolius*. According to Elliott *et al*. (2003), percent survival after fire has been suggested as one of the criteria for selecting framework species in northern Thailand (Elliott *et al*., 2003). With this standard, *C. tribuloides* was the only one that could be categorized as "excellent" (75% survival), while *C. longiopetiolatum* and *P. cerasoides* were "acceptable" (50% survival), and the rest were rejected due to low survival percentage after fire.

When considering the largest tree of each species that died after fire, the size of the largest tree of *C. tribuloides* (only one species in the excellent group) was the smallest among all species (4.5 mm). Not much different with other 2 species in the acceptable group, the largest tree of *C. longiopetiolatum* and *P. cerasoides* was 5.4 and 5.0 mm respectively. For these 3 species, the average of smallest stem that survived and the largest stem that died were 3.0 and 5.0 mm respectively. Unclear relationship between stem size and survival ability was found in this plot.

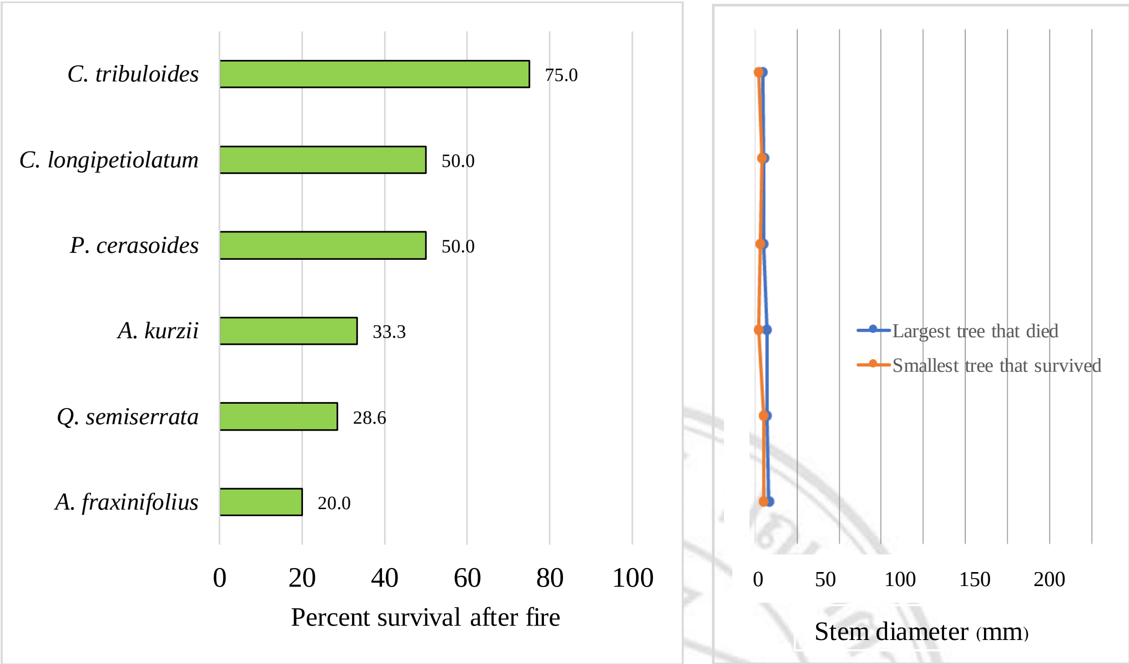


Figure 4.6: Survival percentage and stem size of the largest tree that died and the smallest tree that survived in 1-year-old plot (planted in 2014)

Jackson *et al.* (1998) found that oak species tend to have thicker bark in more frequent fire-prone habitat. Compare to other plots, fires were more frequent in 1-year-old plot as the evidences from burning were commonly found on the standing trees and rocks in this plot. Bark of *C. tribuloides* is thick (FORRU, 2006), even in young saplings this trait could help on protecting an injury from fire.

A trait of *C. longiopetiolatum* was never studied. However, *Cinnamomum caudatum*, which was the same genus of this species, was studied before. Survival and growth rate of *C. caudatum* at 17 months old was unacceptable (37.5% survival and height shorter than 1.25 m), but percent survival after fire at 21 months old was 60% (Elliott *et al.*, 2003). The results from Elliott *et al.* (2003) showed better survival after fire.

Percent survival of 1- year-old *P. cerasoides* was slightly lower (50%) than 21-month-old (60%), compared with Elliott *et al.* (2003)'s. Moreover, FORRU (2006) reported rapid growth of this species that could contribute to larger stem and thicker bark.

4.4.2) 2-year-old plot

Figure 4.7 shows the survival percentage after the plot was burned in April 2015. Using the minimum field performance standard for selecting framework tree species in evergreen forest sites in northern Thailand (Elliott *et al.*, 2003), four categories have been proposed according to percent survival after fire. Excellent performance (>70%) included *M. garretii* (83.3%), *B. javanica* (78.4%) and *F. auriculata* (70.0%). Acceptable performance (50 – 69%) included *P. serratum* (61.5%), *P. cerasoides* (59.1%) and *H. trijuga* (56.7%). Four species in marginal performance group (45 – 49%); they were *H. dulcis* (48.6%), *F. hispida* (48.3%), *S. arboreum* (47.8%) and *F. callosa* (47.1%). The remaining 13 species were in rejected group (< 45%).

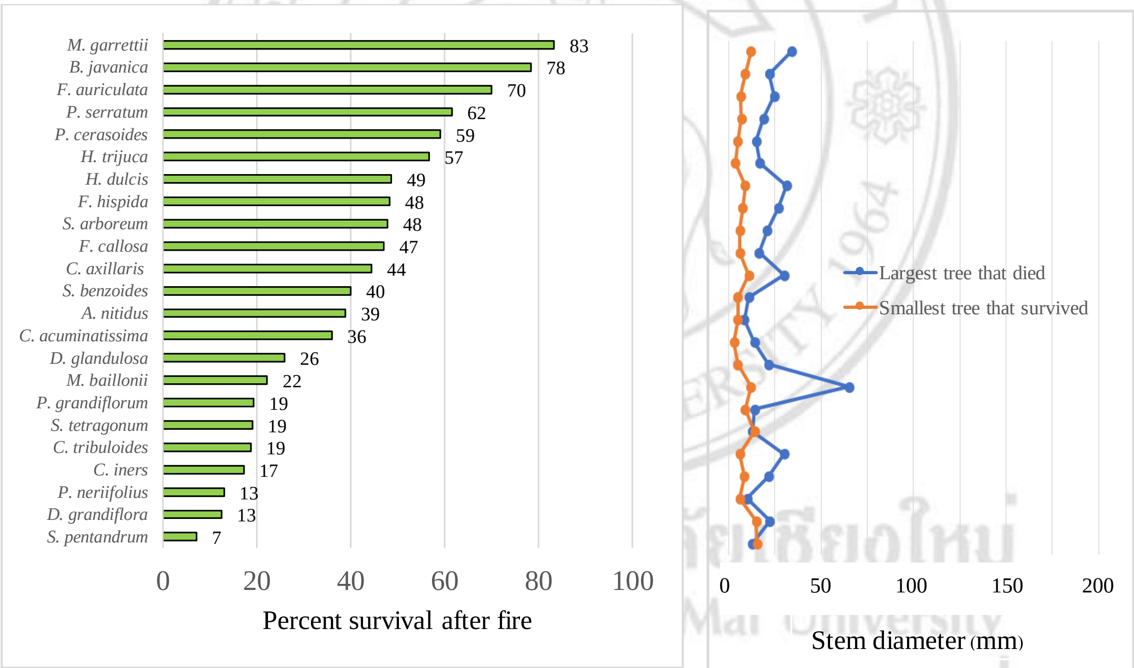


Figure 4.7: Survival percentage, the largest tree that died and the smallest tree that survived in 2-year-old plot (planted in 2013)

The species in excellent (*M. garretii*, *B. javanica* and *F. auriculata*) and acceptable (*P. serratum*, *P. cerasoides* and *H. trijuga*) groups are interesting to investigate further for future implementation. Seedlings of *M. garrettii* and *F. auriculata* grow well in the restoration sites (FORRU, 2006), this characteristic could contribute to larger stem and therefore thicker bark. This relationship has been commonly reported in tropical forests (Hoffmann *et al*., 2003; Lawes *et al.*, 2011a). In addition to rapid growth, *F. auriculata* can grow dense root system which able to help increasing the size of protection structure (e.g. bark) and containing high carbohydrate, which therefore increasing the ability to resprout (Shibata *et al.*, 2016) and survive after fire. The last species in this group, *B. javanica* showed positively fire resilient performance after fire. Higher percent survival found in older saplings, 87% at 33 months old in Elliott *et al*. (2003)'s study, and 78.4% at 24 months old in this study.

In the acceptable group, *P. serratum* had the highest percent survival after fire among other species (61.5%), whereas no existing data to compare elsewhere. Next on the list was *P. cerasoides*, their saplings showed similar survival percent after fire (about 59-60%) in this study and Elliott *et al.* (2003)'s. Similar to *B. javanica* in the previous group, *H. trijuca* showed higher percent survival when the saplings were older (67.0% at 33 months old) in Elliott *et al*. (2003)'s compared to this study (56.7% at 24 months old). It is still unclear why this species performed better among other studied species, however rapid growth has been emphasized to be an important characteristic of the species with fire resilience (e.g. *P. cerasoides)*.

The mean stem diameter of biggest tree that died of these 6 species was 22.0 mm. And the mean stem of smallest tree that survived was 7.2 mm. Among all six species in both excellent and acceptable groups, the biggest difference between the largest tree that died (30 mm) and the smallest tree that survived (6 mm) was found in *P. cerasoides*. Apart from the size of stem diameter, fire intensity (Slik and Eichorn, 2003) and duration of heating (Lawes *et al.*, 2011) were an important cause of survival. Therefore, the bigger trees with higher fire intensity and/or longer duration of heating might lower chance of survival than the smaller trees with lower fire intensity and shorter heating duration.

4.4.3) 14-year-old plot

Figure 4.8 shows percent survival in 14-year-old plot, the smallest trees that survived were presented for all species, whereas the largest trees that died were presented only for the species that had percent survival less than 100. According to

Elliott *et al.* (2003), all species in this plot were classified as excellent species based on their survival percent after fire. Eight species had 100% survival, they were *C. axillaris*, *Q. semiserrata*, *M. stipulata*, *F. subulata*, *S. tetragona*, *S. albiflorum*, *C. acuminatissima* and *B. javanica*. The rest of the species had more than 80% survival; they were *P. cerasoides* (96.4%), *A. nitidus* (94.4%), *S. arboretum* (90.0%), *M. baillonii* (86.7%), *H. trijuca* (83.3%) and *C. diversifolia* (83.3%).

For eight species that had 100% survival after fire, the range of their smallest trees that survived is between 20.1 – 141.7 mm. When considering all species in this plot, the smallest size of trees that survived also varies between 20.1 - 141.7 mm, while the largest tree that died is between 22.6 - 117.8 mm. Mean stem diameter among species of the largest tree that died was 68.4 mm, mean stem diameter of the smallest tree that survive was 44.7 mm. The largest tree that died in this plot had 117.8 mm of stem size (*C. diversifolia* - 83.33 % survival). The smallest tree that survived had 20.1 mm of stem size (*C. acuminatissima* and *M. stipulate* - both were 100% survival).

Similar to the 2-year-old plot, fire intensity and heating duration (Slik and Eichorn, 2003; Lawes *et al*., 2011a) are affected to stem injury, that can cause mortality after fire. Bigger trees might have a lower chance of survival in the higher fire intensity and/or longer heating duration.

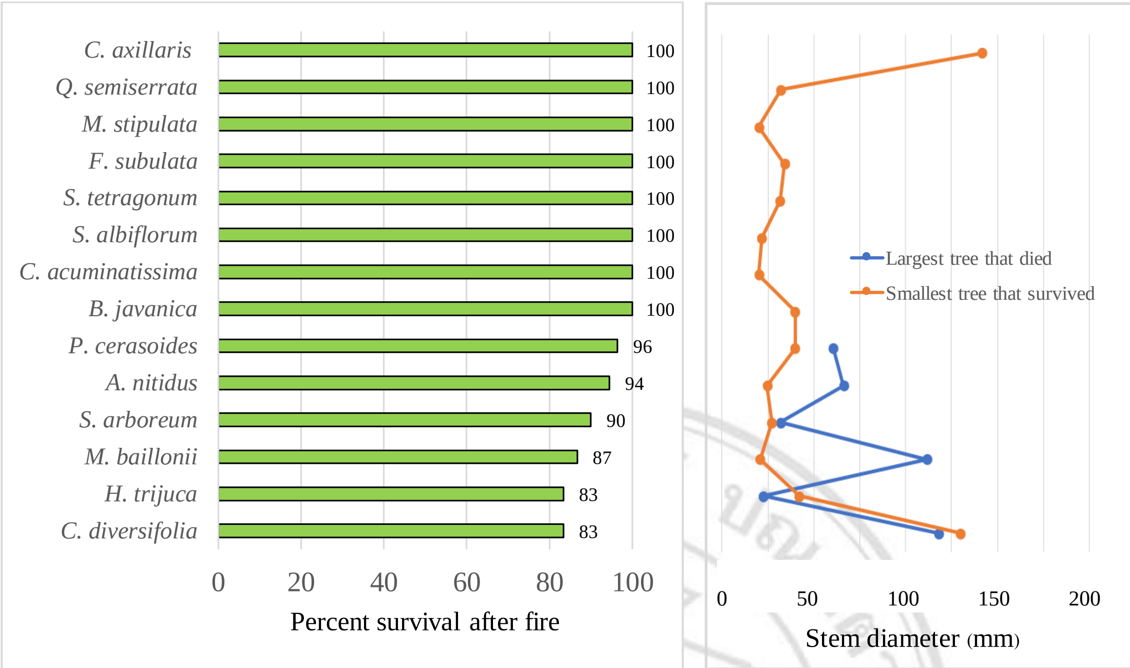


Figure 4.8: Survival percentage, the largest tree that died and the smallest tree that survived in 14-year-old plot (planted in 2001)

4.4.4) 17-year-old plot

Figure 4.9 shows percent survival, stem size of the largest trees that died and the smallest trees that survived in 17-year-old plot (planted in 1998). Most species at this age can survive fire disturbance. Only 3 species had survival percentage less than 100; they were *B. javanica* (97.6%), *C. axillaris* (95.0%) and *M. azedarach* (81.8%). All of them were classified into the excellent group of fire resilience performance (> 70% survival after fire) (Elliott *et al*., 2003). Mean stem diameter of the largest tree that died was 113.9 mm, while, mean stem diameter of the smallest tree that survived was 53.6 mm. The largest tree that died was *M. azedarach* (181.5 mm) with 81.8% survival. The smallest tree that survived after fire in this oldest plot was *G. mckeaniana* (13.1 mm) with 100% survival.

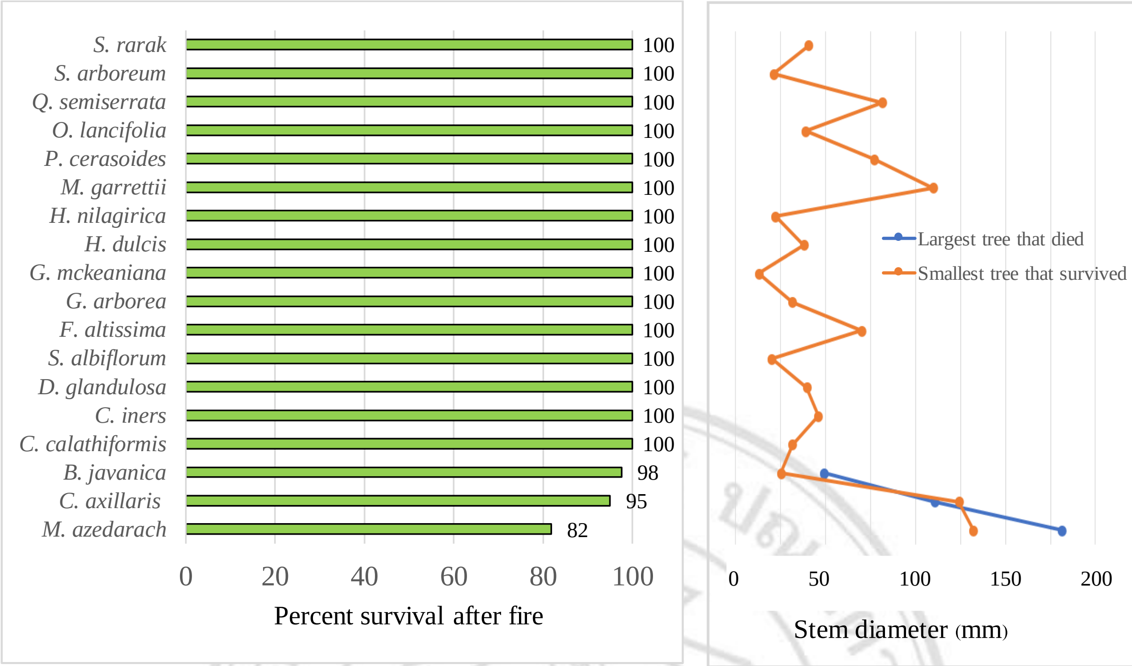


Figure 4.9: Survival percentage, the largest tree that died and the smallest tree that survived in 17-year-old plot (planted in 1998)

Planted trees 14 and 17-year-old plot were big enough to survive after fire. Nieuwstadt and Sheil (2005) studied in the tropical forest in Kalimantan, Indonesia, they found survival of burnt trees were decreased in stem size less than 10 cm and increased in stem size bigger than 70 cm. Another studied in tropical forest at eastern Bolivia (Pinard *et al*., 1999) was found percent survival of small tree (73% survival of 10 – 40 cm) was lower that large trees (84% survival of tree bigger than 40 cm). Survived of big trees were not only persist pre-fire ecosystem, but also valuable for source of seed and shade providing (Swaine, 1992) and decrease ability of un-wanted species (such as grass) to grow after burnt.

Mean stem diameter of the largest trees that died was increased in older plot (6.2, 22, 68.4 and 113.9 mm in 1-, 2-, 14-, 17-year-old plot). Mean stem diameter of the smallest trees that survived was also increased in older plot (6, 7.2, 44.7 and 53.7 mm in 1-, 2-, 14- and 17-year-old plot). Although bigger tree is contribute thicker bark and larger stem, fire intensity and duration of heating also affected to survive of burnt trees. Moreover, habit of tree was probably affected to the survival ability after fire. Three species that percent survival less than 100% in 17-year-old plot

(*M. azedarach*, *C. axillaris* and *B. javanica*) were categorized as pioneer species (Maxwell and Elliott, 2001). Pioneer species was mature early and die back in 15-20 years (Elliot *et al*., 2013), this senescence probably a cause of more sensitive to fire disturbance and decrease survival ability when get older.

4.5 Resprouting ability after fire disturbance

At 30 weeks after burning, the smallest size tree class (0-30 mm) produced the highest number of resprouting shoots. From this study, the number of shoots decreased when the stem size increased until it reached 90 mm, then the numbers fluctuated until the stems were bigger than 210 mm. Very few resprouting shoots were observed from the trees in big size classes (210 - 510 mm) (Figure 4.10).

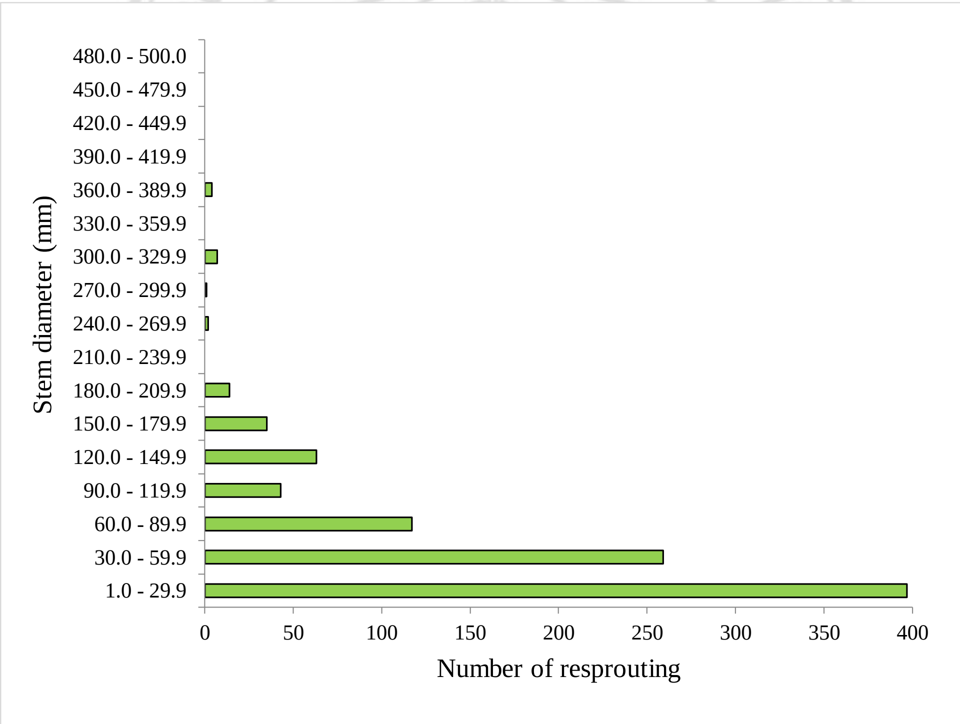


Figure 4.10: Number of resprouting shoots of the trees survived at 30 weeks after burning (30 mm interval)

To examine this closer, only the trees with DBH <210 mm were focused (10 mm range in each size class). The number of resprouting shoots was high (58-193) in smaller class sizes (0-60 mm). The trees that survived after burning tended to produce less number of resprouts (31 or less) if they were bigger than 90 mm. Only few resprouting shoots were observed from the trees of DBH > 190 mm (Figure 4.11).

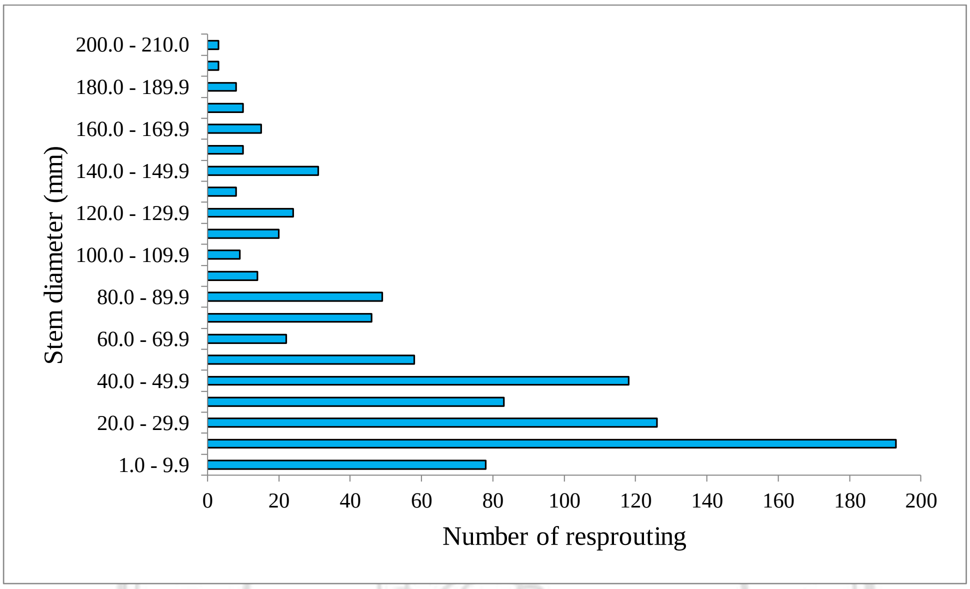


Figure 4.11: Number of resprouting shoots of the trees survived at 30 weeks after burning (10 mm interval)

4.6 Effects of tree size on resprouting ability after fire

Only the trees with a DBH <210 mm were included in GLMMs analysis in this section. Figure 3 shows negative significant between stem diameter and the number of resprouting shoots (co-efficient estimate ±SE = -0.006 ± 0.001, z = 6.01, p = 1.86x10^{-9}). Trees with larger stem diameters significantly resprouted less after fire in montane forest ecosystem of northern Thailand. Similar to studies in tropical forests in Malaysia by Kauffman (1991), who reported that resprouting ability was decreased in bigger tree.

According to the positive correlation of stem diameter and bark thickness (Lawes *et al.*, 2001a; Hoffmann *et al.*, 2003), bigger trees could possibly have thicker bark. Although bark thickness can prevent cambium necrosis from fire, but thick bark also inhibiting resprouting because it hinders epicormic bud emergence (Clarke *et al.*, 2012). The study in oak species confirmed a failure rate of resprouting in oaks with thick bark (Johnson *et al.*, 2002).

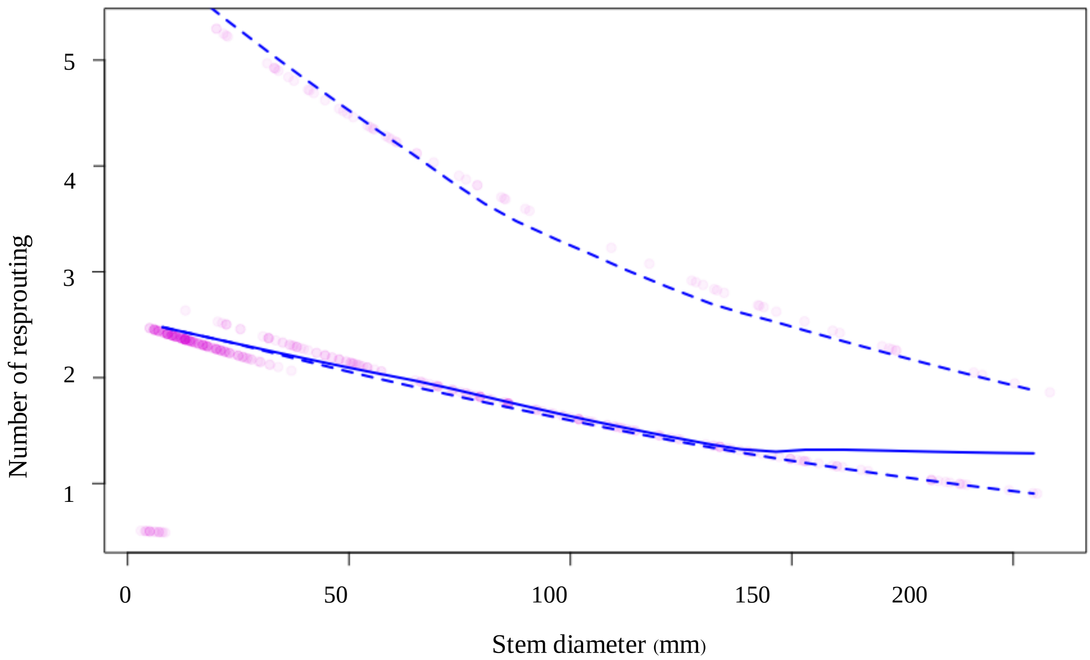


Figure 4.12: Predicted values for number of resprouting: an interaction between number of resprouting and stem diameter (DBH<210 mm), pink dot is observed value, blue line is predicted line and dash line is 95% interval for the regression line

From Figure 4.12, the number of resprouting shoots were high (58-193) in smaller class sizes (0-60 mm). Therefore, the trees in these small class sizes were included in GLMMs analysis to quantify the effects of stem size on resprouting ability after a fire. Differently, Figure 4.13 shows positive significance between stem diameter and resprouting ability of small trees with a DBH <60.0 mm (co-efficient estimate ± SE = -0.009 ± 0.004, z = 2.07, p = 0.04). Trees with a larger stem in these categories showed a significantly amount of resprouted shoots after they were burned. Pinard *et al*. (1999) confirmed that trees smaller than 100 mm DBH in a tropical Amazonian forest were relatively common on resprouting ability, but uncommon for bigger trees than 100 mm DBH. In addition, Shibata *et al*. (2016) found that resprouting ability increased with bigger stem of woody species in a temperate forest.

There was a positive relationship between stem diameter and resprouting ability of trees with DBH <60 mm. Inside this range, larger trees could have better bud protection and resource reserver. Clarke *et al*. (2012) reported that thicker stem helps to protect buds from burning, buffer xylem against hydraulic failure, and prevent phloem and xylem necrosis from the heat transfer through cambium. In addition, bigger stems are associated with resource storage (Shibata *et al*., 2016), which shows higher ability to resprout shoots than smaller stems.

Consistent with stem diameter and the age of the tree (mean stem diameter was 5.0, 13.2, 101.7, 124.1 mm in 1-, 2-, 14- and 17-year-old trees respectively), its resprouting ability decreases: this might be linked to bud senescence. The study of functional trait of resprouting by Clarke *et al*. (2012) found that this occurring was possibly arise from combination of genetic, physiological and related anatomical changes. Bond and Migdley (2001) confirmed that resprouting ability increases with size until the trees reach an adult stage and losing the capacity to resprout.

Resprouting is a tolerant trait to persist at the plant level, resilient to severe disturbance at community level (Clarke *et al*., 2012), and shortens the time to recovery (Lawes *et al*., 2011b). Therefore, smaller trees that were mostly killed by fire resprout after the fire disturbance to restore their photosynthesis capacity (Lawes *et al*., 2011b) and persist in a site (Vask and Westoby, 2004). Conversely, trees that were big enough to escape fire damage lost their capacity to resprout due to their remaining photosynthesis structure (Bond and Midgley, 2001).

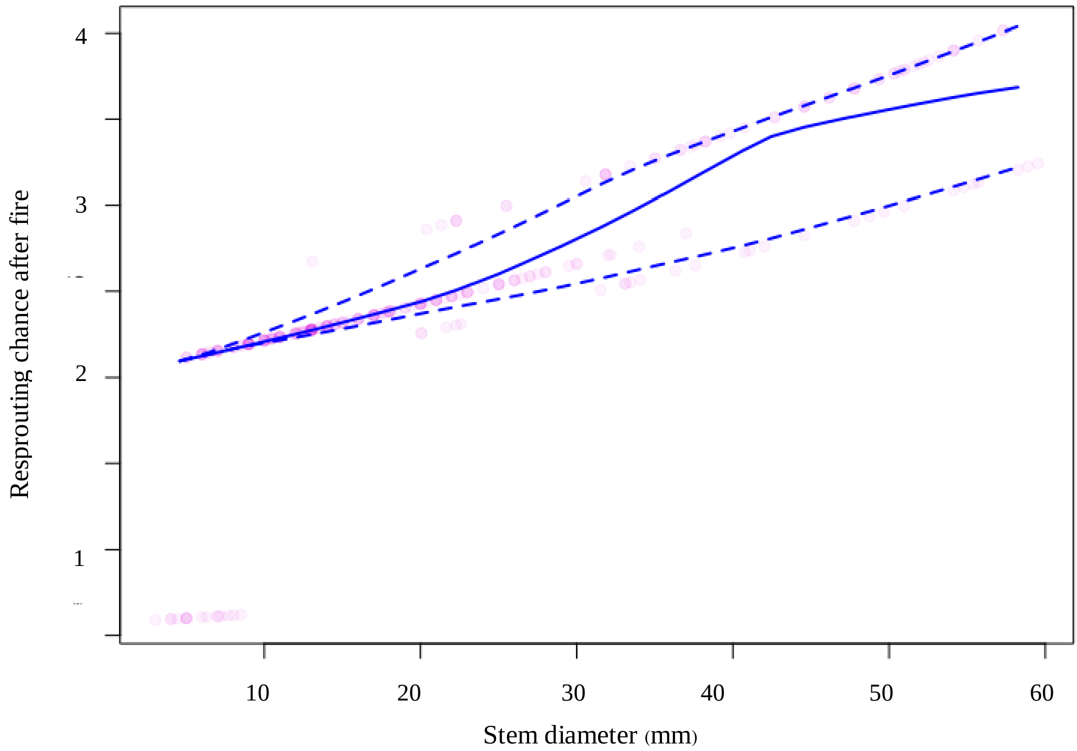


Figure 4.13: Predicted values for resprouting: an interaction between the number of resprouting and stem diameter (DBH<60 mm), a pink dot is the observed value, the blue line is the predicted line and the dash line is the 95% interval for the regression line.

4.7 Potential tree candidates for restoring fire-prone montane forest

Most trees in 14- and 17-year-old plot (planted in 2001 and 1998, respectively) were big enough to survive a fire disturbance in the summer of 2015 (April – May). Here I particularly focused on the trees that survived in the young plots; the 1- and 2-year-old plot (planted in 2014 and 2013, respectively), using survival data from the 3rd monitoring at 30 weeks after the fire. According to Elliott *et al.* (2003), tree species with a percent survival (after burning) that were lower than 45% were rejected, a total of 12 species were therefore selected to calculate the suitability index for this study. Only *P. cerasoides* has two values calculated from both 1- and 2-year-old plot (Table 4.1).

I propose 3 classes of suitability based on the scores calculated from 3 parameters (survival percentage, RGR and resprouting ability); excellent (>75), acceptable (60-75), and marginal (<60). High growth rate correlated with both survival and resprouting ability in juvenile trees (Hoffmann, 2003). From 12 species (10 families), there were three species (*F. auriculata*, *F. hispida* and *F. callosa*) from Moraceae that were available for this calculation. *Ficus* was selected because of its fire resilience, rapid growth after burning and its root system penetrates deep underground (FORRU, 2006), which can prevent it from being burned.

Three species were categorized into as “excellent” (rank suitability score >75); they were *M. garrettii*, *B. javanica* and *F. auriculata*. These three native tree species had been recommended as potential framework tree species (FORRU, 2006) for restoring forest ecosystems in northern Thailand where fires are prone. Elliott *et al*. (2003) confirmed that *B. javanica* was resilient to fires (87 % survival after a fire at 33-month-old), and resprouted well after fire (FORRU, 2006). Apart from fire resilient characteristics, *B. javanica* and *F. auriculata* have been reported by FORRU (2006) that these species can produce fruits within 6 years after planting. They have the ability of attracting wildlife in the early stage of restoration. Even though *M. garrettii* cannot produce fruit within 7 years, but seeds of this species attract birds and squirrels. Moreover, *M. garrettii* develops broad leaves and dense crown which effectively shade out weeds (FORRU, 2006). For restoring in montane forest where fire disturbance is common, these 3 species were highly recommended.

Seven species were grouped in the “acceptable” (rank suitability score 60-75); *F. hispida*, *H. trijuca*, *P. cerasoides* (2-year-old), *C. tribuloides*, *P. serratum*, *S. arboreum* and *H. dulcis*. Six out of 7 species were recommended as potential framework tree species for montane forest in northern Thailand (FORRU, 2006), except *P. serratum*. Among six potential framework tree species, *C. tribuloides* was reported to resprout rapidly, while *F. hispida* and *H. dulcis* were mentioned to have >70% survival after a fire. Although *S. arboreum* is a slow growing tree species but it showed high survival and rapidly resprouting characteristic. With similar percent survival after fire, *H. trijuca* and *P. cerasoides* were identified into acceptable category in both Elliott *et al*. (2003) and in this study.

During the 3rd monitoring, *C. tribuloides*’ seedlings were found in both 1- and 2-year-old plots, however only those in the 1-year-old plot (ML) survived >45%. In temperate forest, *Pinus* and *Quercus* that grow in an area with high frequency of fire tend to have thicker bark (Jackson *et al.*, 1998) and this trait could possibly be inherited to next generation. Local seed source could help to maximize possibility that seedlings will survive and adapt to local disturbance (e.g. fire). *C. tribuloides*' seedlings in the 1-year-old plot (ML) produced from local seed source, whereas those planted in the 2-year-old plot (BMSs) propagated from another location (FORRU, 2014 - unpublished paper).

Due to a rank suitability score of < 60%, 3 species were categorized as "marginal"; *F. callosa*, *P. cerasoides* (1-year-old) and *C. longipetiolatum*. Two out of 3, *F. callosa* and *P. cerasoides* were recommended as potential framework tree species (FORRU, 2006), whereas *C. longipetiolatum* had never been studied before. Interestingly, older seedlings of *P. cerasoides* (2- year-old) expressed better resilient performance compared to those younger seedlings in the 1-year-old plot. For those trees DBH <60 mm, trees with larger stems have better bud protection and resource reserve. *P. cerasoides*' trees in the 2- year-old plot produced 8 times more resprouting shoots than those in the 1- year-old plot. As mentioned in 4.6 that a bigger stem is associated with resource storage, it has more of the ability to produce resprouting shoots than smaller stems (Shibata *et al*., 2016).

All species in Table 1 should be considered during species selection process for restoring montane forest in northern Thailand. Nine out of the 12 species are potential framework tree species which can provide resources for wildlife and shade out weeds effectively within 3 - 5 years after planting. Planting these 9 framework tree species with 3 additional species (*P. serratum*, *F. callosa* and *C. longipetiolatum*) can help to increase species richness and also ecosystem resilience simultaneously

Table 4.1: Suitability index of native tree species in northern Thailand for restoring montane forest

No.	Scientific name	Plot age (year)	Survival (%)[a]	Largest tree that died (mm)	Smallest tree that survived (mm)	RGR (%/year)[b]	No. of resprouting shoot/tree[c]	Standardized suitability score[d]	Rank suitability score
1	*M. garrettii*	2	100.0	34.0	12.0	42.9	100.0	342.9	100.0
2	*B. javanica*	2	94.1	22.0	9.0	37.2	70.2	295.5	86.2
3	*F. auriculata*	2	84.0	24.7	6.4	24.4	69.8	262.2	76.5
4	*F. hispida*	2	57.9	27.0	7.5	100.0	35.5	251.3	73.3
5	*H. trijuca*	2	68.0	17.0	3.5	65.8	49.0	250.8	73.1
6	*P. cerasoides**	2	70.9	15.0	5.0	38.3	66.8	246.9	72.0
7	*C. tribuloides**	1	90.0	4.5	2.0	45.2	12.2	237.4	69.2
8	*P. serratum*	2	73.8	19.0	7.0	35.8	43.3	226.8	66.1
9	*S. arboreum*	2	57.4	21.0	6.1	29.8	76.7	221.3	64.5
10	*H. dulcis*	2	58.4	31.3	9.0	55.8	33.1	205.7	60.0
11	*F. callosa*	2	56.5	16.5	6.0	37.3	30.3	180.5	52.6
12	*P. cerasoides**	1	60.0	5.0	3.0	50.8	8.2	179.0	52.2
13	*C. longipetiolatum*	1	60.0	5.4	4.1	17.9	16.3	154.2	45.0

*Native tree species that were planted in both 1- and 2-year-old plot

[a,b,c] Standardized by converting the maximum value to 100, then adjusted other values proportionately

[d] Standardized suitability score = 2a + b + c

CHAPTER 5

Conclusion

5.1 Tree size and survival after a fire

Older trees survived better (more than 2 times) after a fire comparing to younger trees. Small differences of survival percentages were detected between each monitoring in the older plots, but there was a dramatic gap found in younger plots. It is assumed that younger trees faced with cambium injury after burning, and therefore died within weeks. More attention had been paid to small trees with DBH <40 mm because they were more vulnerable to a fire (about 50% chance of dying).

From all analyses done in this study, it has been confirmed that trees with bigger stems survived significantly better than small ones do. A higher chance of survival in bigger trees are probably due to the development of protection, such as bark and stem thickness that positively correspond with stem diameter. Larger trees tend to have thicker bark to protect vascular cambium tissue from fire which can cause tree mortality via stem necrosis and deformation.

When considering each plot separately, the relationship between stem size and survival after a fire has become less clear due to influences of other factors, such as the fire's intensity, duration of burning, and surrounding conditions which are not mainly focused in this study. Moreover, successional status or life history of different species could also contribute to the level of sensitivity to fire disturbance. For planted trees in a restoration program to survive a fire disturbance, fire prevention should be emphasized until the trees reach a minimum of 40 mm DBH.

5.2 Tree size and resprouting after a fire

Smaller trees produced a higher number of resprouting shoots after burning compared to larger trees. This trend was observed in trees with DBH < 90 mm, but the number was fluctuated in trees with DBH 90-210 mm. Very few resprouting shoots produced if the trees' DBH was bigger than 210 mm.

When considering all plots which contained a broad range of different tree sizes, the resprouting ability was found to decrease in bigger trees. It has been mentioned in the previous chapter that there was a positive correlation between stem diameter and bark thickness. Although thick bark can protect buds from burning; however, it also inhibits resprouting via obstructing epicormic bud emergence. Most trees found in the old plots are large and produced very few resprouting shoots after the fire. This might be correlated with bud senescence in the adult stage, which is a result from a combination of genetic, physiological and related anatomical changes.

In contrary, when considering small trees with a DBH < 60 mm, the trees with larger stems within this size class produced a higher number of resprouting shoots after a fire. Bigger stems are associated with resource storage and therefore help the trees to resprout efficiently. Resprouting ability is controlled by the interaction of the disturbance regime that harm buds and resource needed for resprouting, and the environment that affect growth and resource allocation.

5.3 Suitable tree species for restoring montane forest

From this study, all 12 species should be targeted for forest restoration in montane forest ecosystem in northern Thailand. They have a high potential to capture the site, increase species richness and also withstand fire disturbance. According to the suitability index, three species were categorized as excellent species; they were *M. garrettii*, *B. javanica* and *F. auriculata.* Seven species were grouped in the acceptable class; *F. hispida*, *H. trijuca*, *P. cerasoides*, *C. tribuloides*, *P. serratum*, *S. arboreum* and *H. dulcis.* The last three species were identified in the marginal group; they were *F. collosa*, *P. cerasoides,* and *C. longipetiolatum.*

Nine out of 12 species have been identified as potential framework tree species for restoring montane forests in northern Thailand, except *P. serratum*, *F. collosa*, and *C. longipetiolatum*. One big barrier for forest restoration is an understanding about native species particularly phenology, seed biology, and silviculture. The reason why these three species have not been included in the list of potential framework tree species is possibly because of not having seedling available. Their survival percentages are acceptable but not many reprouting shoots produced and growth was quite slow (only *C. longipetiolatum*).

Interestingly, *P. cerasoides* was classified into 2 groups, acceptable and marginal, because this species was tested for 2 consecutive years. Clearly, older *P. cerasoides*' seedlings (2 years old) performed better in selected criteria compared to those younger seedlings (one year old) due to better bud protection and resource storage.

Although restoration practitioners select target species with a high fire resilient ability; however, a fire prevention program should still be designed with stakeholders, to prevent unnecessary lost throughout the landscape, and direct successional pathway toward an ultimate restoration goal.

5.4 Suggested further research

Following the effect of stem diameter on survival and resprouting probability, stem size may not only be one factor that affects both survival and resprouting. Other stem traits recommended for further study included bark thickness, bark moisture content and density, bud location, structure of meristem and resources allocation.

Furthermore, fire intensity, heat duration, microclimate of ecosystem, plant life history and successional type of the plant would have an effect on the fire resilience characteristic. Therefore, these factors should be considered for further study.

REFERNCES

Ateroffa and Rada, 2000

Ataroff M. and Rada F., Deforestation Impact on Water Dynamics in a Venezuelan Andean Cloud Forest, A Journal of the Human Environemt, 29(7): 440-444, Royal Swedish Acedemic of Science, 2000

Attiwill, 1994 Attiwill P. M., The disturbance of forest ecosystem: the ecological basis for conservative management, Forest ecology and management, Vol. 63, 1994, 247 - 300

Balch *et al*., 2008 Balch J.K., Nepstad D.C., Brando P.M., Curran L.M., Portela O., Carvalho Jr. O. and Lefebvre P., Negative fire feedback in a transitional forest of sountheastern Amazonia, Global change biology (2008) 14, 2276 – 2287

Barlow et al., 2003 Barlow J. P., Peres C. A., Lagan B.O. and Haugaasen T., Large tree mortality and the decline of forest biomass following Amazonian wildfires, Ecol. Lett., Vol. 6, 2003, 6 -8

Bernhardt and Leslie, 2013

Bernhardt J. R. and Leslie H. M., Resilience to climate change in costal marine ecosystems, the annual review of marine science, Vol. 5, 2013, 371 - 392

Bond and Keane, 2017

Bond W. J. and Keane R. E., Fires, Ecological effects of, Elsevier Inc., 2017

Bond and Midgley, 2001

Bond W. J. and Midgley J.J., Ecology of sprouting in woody plants: the persistence niche, Trend in ecology and evolution, Vol. 16, No. 1, 2001, 45 - 51

Bova and Dickinson, 2005

Bova A.S. and Dickinson M. B., Linking surface-fire behavior, stem heating, and tissue necrosis, can. J. For. Res., Vol. 55, 2005, 814 - 822

Bruijnzeel *et al*., 2011

Bruijnzeel L. A., Scatena F. N. and Hamilton L. S., Factors controlling rates of forest recovery and management solution, Tropical montane cloud forest: science for conservation and management, 2011, 102 – 109

Burton *et al*., 2010 Burton O.J., Phillips B.L. and Travis J.M.J., Trade-offs and the evolution of life-histories during range expansion, Ecology letters, (2010) 13, 1210 – 1220

Cerda and Robichaud, 2009

Cerda A. and Robichaud P.R., Fire regime and resilience: the fire regime-resilience coupled model, Fire effects on soils and restoration strategies, 257 – 298.

Clarke *et al*., 2012 Clarke P. J., Lawes M. J., Midgley J. J., Lamont B. B., Ojeda F., Burrows G. E., Enright N. J. and Knox K. J. E., Resprouting as a key functional trait: how buds, protection and resources drive persistence after fire, New phytologist, Vol 197, 2013, 19-35

Cochrane *et al.*, 1999

Cochrane M.A., Alencar A., Schulze M.D., Souza Jr. C.M., Nepstad D.C., Lefebvre P and Devidson E.A., Positive feedbacks in the fire dynamics of closed canopy tropical forests, sciencemag, vol. 284, 1832 – 1835

Cochrane, 2001 Cochrane M.A., Spreading like wildfire - tropical forest fires in Latin American and the Caribbean: prevention, assessment and early warning (United Nations Environment Program (UNEP), 2001

Cochrone, 2003 Cochrane M., Fire science for rain forest, Nature publishing group, volume 421, 2003, 913 – 919

Delang, 2002 Delang C. O., Deforestation in Northern Thailand: The result of Hmong farming practices or Thai development strategies?, 2002, 483-501

Delang, 2002 Delang. C. O., Deforestation in Northern Thailand: The result of Hmong farming practices or Thai development strategies, Society and Natural resources, Vol 15, 2002, 483-501.

Duckinson *et al.*, 2004

Dickinson M.B., Jolliff J. and Bova A. S., Vascular cambium necrosis in forest fires using hyperbolic temperature regime to estimate parameters of a tissue-response model. Australia Journal of botany, Vol.52, 2004, 757 - 763

Elliott *et al.*, 2003 Elliott S. , Navakitbumrung P. , Kuarak C. , Zangkum S. , Anusarnsunthorn V. and Blakesley D., Selectinf framework tree species for restoring seasonally dry tropical forest in northern

Thailand base on field performance, Forest Ecology and management, Vol 184, 2003, 177 – 191

Elliott *et al*., 2013 Elliott S. D., BlakesleyD. And Hardwick K., Restoring tropical forest: a practical guide, Royal botanic garden, Kew, 2013

FORRU, 2006 Forest Restoration Research Unit, how to plant a forest: The principles and practice of restoring tropical forest, Biology department, Science faculty, Chiang Mai University, 2006.

Frois *et al*., 2008 Frois. N., Chan T. Y. W., Kaunisvesi L. K., Mitterer M., Omoro L. M. A., Tolonen A. A. J., Mulyani Y. A. M. and Songchan R., Forest ecosystem restoration in Northern Thailand, FORRSA RE_2 Field Course: Tropical forest landscape restoration in southern Asia, 2008, 1-41.

Gunderson, 2000 Gunderson L. H., Ecological resilience - in theory and application, Ann. Rev. Ecol. Syst., Vol. 31, 2000, 425-439

Gustafsson et al., 2016

Gustafsson M., Gustafsson L, Alloysius D., Falck J., Yap S., Karlsson and Ilstedt U., life history traits predict the response to increased light among 33 tropical rainforest tree species, Forest ecology and management 362 (2016), 20-28

Hoffman *et al*., 2003

Hoffman W. A., Orthen B., Kielse P. and Nascimento V. D., Comparative fire ecology of tropical savanna and forest trees, Functional ecology, 2003, 17, 720 – 726

Hoffmann *et al*., 2012

Hoffmann W. A., Geiger E. L., Gotsch S. G., Rossatto D. R., Silva L. C. R., Lau O. L., Haridasan M. and Franco A. C., Ecological thresholds at the savanna-forest boundary: how plant traits, resources and fire govern the distribution of tropical biomes, Ecology letter, Vol. 15, 2012, 759-768

Hoffmann, 1996 Hoffmann W. A., The effects of fire and cover on seedling establishment in Neotropical savanna, British ecological society, 1996, 383-393

Holling, 1973 Holling C. S., Resilience and stability of ecology system. Annu. Rev. Ecol. Syst, Vol 4, 1973, 1-23

Holling, 1973 Holling C.S., Resilience and stability of ecological systems, Institute of research ecology, 1973, 1-23

ICEM, 2003 ICEM., Protected areas and development: lessons from Thailand, Lessons leadership in Cambodia, Lao PDR, Thailand and Vietnam. Review of Protected Areas and Development in the Lower Mekong River Region. Indooroopilly, Queensland, Australia, 2003, 61 – 83

Jackson *et al*., 1999

Jackson F. J., Adams D. C. and Jackson U. B., Allometry of constitive defense: a model and a comparative test with tree bark and fire regime, the American naturist, Vol. 153 (6), 1999

Johnson *et al*., 2002

Johnson P. S., Shifley S. R. and Rogers R., The ecology and silviculture of oaks, CABI publishing, 2002

Junpen *et al.*, 2013 Junpen A. J., Garivait S. and Bonnet S., Estimating emissions from forest fires in Thailand using MODIS active fire product and country specific data, Asia-Pasific J. Atmos. Sci., Vol 49(3), 2013, 389 - 400

Kamying *et al.*, 2003

Kamyong S., Lykke A. M., Seramethakun D. and Barfod A. S., Species composition and vegetation structure of an upper montane forest at the summit of Mt. Doi Inthanon, Thailand., Nordic Journal of Botany, February, 2003

Kauffman, 1991 Kauffman J. B., Survival by sprouting following fire in tropical forests of the eastern Amazon, Biotropica 23(3), 219-224

Kiianmaa, 2005 Natural regeneration and ecological succession in *Pinus kesiya* watershed plantations in northern Thailand: implications for plantation management, University of Helsinki, Department of forest ecology

Laurance W. F., 2007

Forest destruction in tropical Asia, current science, Vol. 93, NO 11

Lawes and Clarke, 2011

Lawes M. J. and Clarke P. J., Ecology of plant resprouting: populations to community responses in fire-prone ecosystems, Plant ecology, Vol. 212, 2011, 1937 - 1943

Lawes *et al*., 2011a Lawes M. J., Adie H., Russell-Smith J., Murphy B. and Midgley J. J., How do small savanna trees avoid stem mortality by fire? The roles of stem diameter, height and bark thickness, Ecosphere, 2011, volume 2(4), 1-13

Lawes *et al*., 2011b Lawes M.J., Richards A., Dathe J. and MIdgley J.J., Bark thichness determines fires resistance of selected tree species from fire-prone tropical savanna in north Australia, Plant ecology: Sprouting ecology, 2011

Lentile *et al*., 2007 Lentile L. B., Morgan P., Hudak A. T., Bobbitt M. J., Lewis S. A., Smith A. M. S. and Robichaud P. R., Post-fire severity and vegetation response following eight large wildfires across the western united states, Fire ecology special issue, Vol. 3, No.1, 2007, 91-108

Lentile *et al*., 2007 Lentile L. B., Morgan P., Hudak A. T., Bobbitt M. J., Lewis S. A., Smith A. M. S. and Robichaud P. R., Post-fire severity and vegetation response following eight large wildfires across the western united states, Fire ecology special issue, Vol. 3, No.1, 2007, 91-108

Luukkane O., 2001 The vanishing and rappearing tropical forest: forest management and land use in Thailand. In: Myllyantaus T and Saikku M (ed) Encountering the past in nature: Essays in environmental history, pp. 74 – 93. Ohio University Press, Athens.

Marod *et al*., 2002 Marod D., Kutintara U., Tanaka H. and Nakashizuka T., The effects of drought and fire on seed and seedling dynamics in a tropical seasonal forests in Thailand, Plant ecology 161, 41-57, 2002

Marrinan *et al*., 2005

Marrinam M. J., Edwards W. and Landsberg J., Resprouting of saplings following tropical rainforest fire in north-east Queensland, Australia, Austral Ecology (2005) 30, 817 – 826

Maxwell and Elliott, 2001

Maxwell J. F. and Elliott S., Vegetation and vascular flora of Doi Sutep- Pui National park, northern Thailand, Thai studies in Biodiversity, No. 5, 2001, 1-205

Michaletz et al., 2011

Michaletz S. T., Johnson, E. A. and Tyree M. T., Moving beyond the cambium necrosis hypothesis of post-fire tree mortality: cavitation and deformation of xylem in forest fires, New phytologist, Vol. 194, 2011, 254 - 263

Midgley et al., 2010

Midgley J. J., Kruger L. M. and Skelton R., How do fire kill plants? The hydraulic death hypothesis and Cape Proteaceae “fire-resisters”, South African journal of botany, Vol. 77, 2010, 381-386

Nieuwstadt and Sheil,2005

Nieuwstadt M. G. L. and Sheil D., Drought, fire and tree survival in a Borneo rain forest, East Kalimantan, Indonesia, Journal of ecology, Vol. 93, 2005, 191-201.

Nieuwstadt *et al*., 2001

Nieuwstadt M. G. L., Sheil D., Kartawinata K., The ecological consequences of logging in the burned forests of East Kalimantan, Indonesia, Conservation biology, Vol 15 (4), 2001, 1183-1186.

Nobel, 2005 Nobel P. S., Physicochemical and environmental plant physiology, Elsevier academic press., 2005

Office of natural resources and environmental policy and planning, 2009

Office of natural resources and environmental policy and planning, Thailand: National report in the implementation of the convention on biological diversity, Ministry of natural resources and environment, 2009, 76

Ongpresert, 2011 Onprasert P., Forest Management in Thailand, International forestry cooperation office, 2011

Padgham, 2014 Padgham B. L. T., Examining spurces of ecological resilience to climate change for restoration planning, 2014

Pinard *et al*., 1999 Pinard M.A., Putz F.E and Licona J.C., Tree mortality and vine proliferation following a wildfire in a subhumid tropical forest in eastern Bolivia, Forest ecology and management 116 (1999), 247 – 252

Poorter *et al*., 2010 Poorter L., Kitajima K., Mercado P., Chubina J., Melgar I., Prins HHT., Resprouting as a persistence strategy of tropical forest trees: relations with carbohydrate storage and shade tolerance, Ecology, Vol 91, 2010, 2613-2627

Royal Forestry Department, 2009

Royal Forestry Department, Forest management in Thailand, International forestry cooperation office, 2009

Santisuk, 2006 Santisuk T., Forest in Thailand, The Forest Herbarium (BKF), National park, wildlife and plant conservation department of Thailand, 2006, 33-38.

Schmoldt *et al.*, 1999

Schmoldt D.L., Peterson D.L., Keane R.E., Lenihan J.M., McKenzie D., Weise D.R. and Sandberg D.V., Assessing the effects of fire disturbance on ecosystem: A scientific agenda for research and management, United states department of agriculture – Forest service, General technical report PNW-GTR-455, 1999

SER, 2004 Society for Ecological Restoration (SER), The SER international primer on ecological restoration, 2004, 1-13

Setterfield, 2002 Setterfield S. A., Seedling establishment in an Australian tropical savanna: effects of seed supply, soil disturbance and fire, Journal of applied ecology, Vol. 39, 2002, 949-959.

Setterfield, 2002 Stterfield S. A., Seedling establishment in an Australian tropical savanna: effects of seed supply, soil disturbance and fire, Journal of applied ecology, 2002, 949-959

Shibata *et al.*, 2016 Shibata R., Kurokawa H., Shibata M., Tanaka H., Lida S., Masaki T. and Nakashizuka T., Relationships between resprouting ability, species traits and resource allocation patterns in woody species in a temperate forest, Functional ecology, Vol. 30, 2016, 1205-1215.

Slik and Eichhorn, 2003

Slik J.W.F. and Eichhorn K.A.O., Fire survival of lowland tropical rain forest trees in restoration to stem diameter and topographic position, Oecologia (2003) 137, 446 – 455

Standish *et al.*, 2014

Standish R. J., Hobbs R. J., Mayfield M. M., Bestelmeyer B. T., Suding K. N., Battaglia L. L, Eviner V., Hawkes C. V., Temperton V. M., Cramer V. A., Harris J. A., Funk J. L. and Thomas P. A., Resilience in ecology: abstraction, distraction, or where the action is?, Biological Conservation, Vol. 177, 2014, 43-51.

Stanturf *et al*., 2014 Stanturf J.A., Palik B. J and Dumroese R. K., Contemporary forest restoration: A review emphasizing function, forest ecology and management 331 (2014), 292-323

Swain, 1992 Swaine M.D., Characteristics of dry forest in West Africa and the influence of fire, Journal of vegetation science 3, 1992, 365 – 374

Vaidhayakarn and Maxwell, 2010

Vaidhayakarn C. and Maxwell J. F., Ecological status of the lawland deciduous forest in Chang Kian vallay, Chiang Mai University, northern Thailand, Maejo International Journal of Science and technology, Vol 4(02), 2010, 268-317

Vesk and Westoby, 2004

Vesk P. A. and Westoby M., Sprouting ability across diverse disturbances and vegetation types worldwide, Journal of ecology 2004, 92, 310 – 320

Werner and Santisuk, 1993

Werner W. L. and Santisuk T., Conservation and restoration of montane fores communities in Thailand, Restoration of tropical forest ecosystem, Kluwer Academic Publishers., 1993, 193-202.

Xaub *et al.*, 2013 Xaud H. A. M., Marins F. S. R. and Santos, J. R., Tropical forest degradation by mega- fires in the northern Brazilian Amazon, Forest ecology and management 294 (2013), 97 – 106

CURRICULUM VITAE

Name Ms. Phutthida Nippanon

Date of birth September 6, 1990

Education Bachelor degree of Science (Biology), Chiang Mai University, 2014

Proceeding Nippanon, P., 2016. Fire Resilience of Framework Tree Species in Hill Ever Green Forest Ecosystem. Proceeding of the 3rd National Meeting on Biodiversity Management in Thailand. June 15th - 17th, 2016 at The Impress Nan Hotel, Nan, Thailand

Experience Field Research Officer, Forest Restoration Research Unit, Chiang Mai University 2013 – 2016

Local coordinator, the PUR Projet Thamma Raksa (Thailand) Co.,Ltd., 2016

Conservation Executive, Flight of the Gibbon, Chiang Mai, 2017 – Present