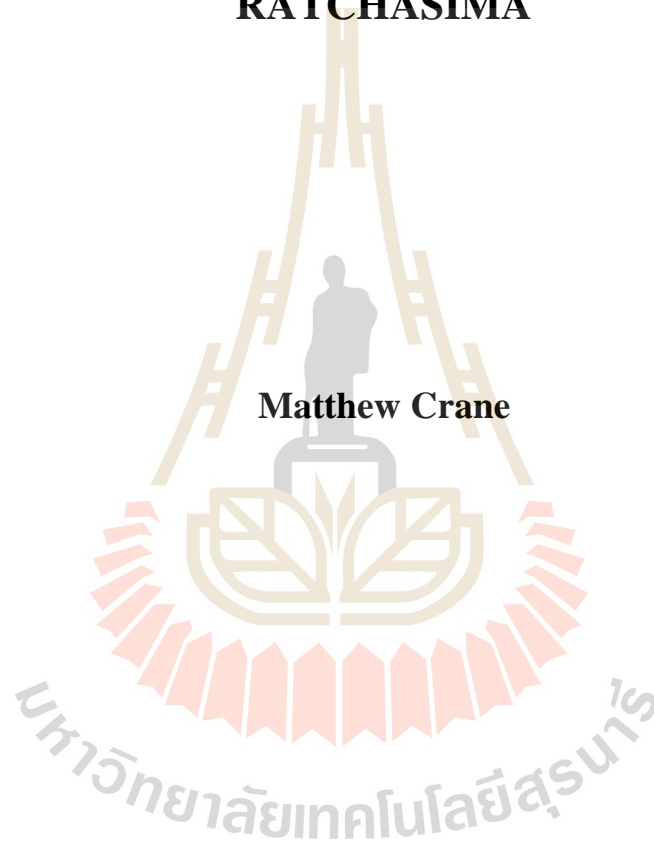


**EFFECT OF LAND-USE AND HUMAN DISTURBANCE  
ON HERPETOFAUNAL COMMUNITIES IN THE  
SAKAERAT BIOSPHERE RESERVE, NAKHON  
RATCHASIMA**



**A Thesis Submitted in Partial Fulfillment of the Requirements for the**

**Degree of Master of Science in Environmental Biology**

**Suranaree University of Technology**

**Academic Year 2015**

ผลกระทบจากการใช้ที่ดินและการรบกวนของมนุษย์ต่อชุมชนของ  
สัตว์เลื้อยคลานและสัตว์สะเทินน้ำสะเทินบก ในเขตสงวนชีวมณฑลสะแกกราช  
จังหวัดนครราชสีมา



วิทยานิพนธ์นี้เป็นส่วนหนึ่งของการศึกษาตามหลักสูตรปริญญาวิทยาศาสตรมหาบัณฑิต  
สาขาวิชาชีววิทยาลิ่งแวดล้อม  
มหาวิทยาลัยเทคโนโลยีสุรนารี  
ปีการศึกษา 2558

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Suranaree University of Technology has approved this thesis submitted in partial fulfillment of the requirements for a Master's Degree.

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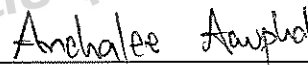
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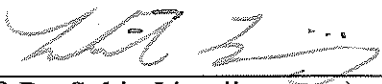


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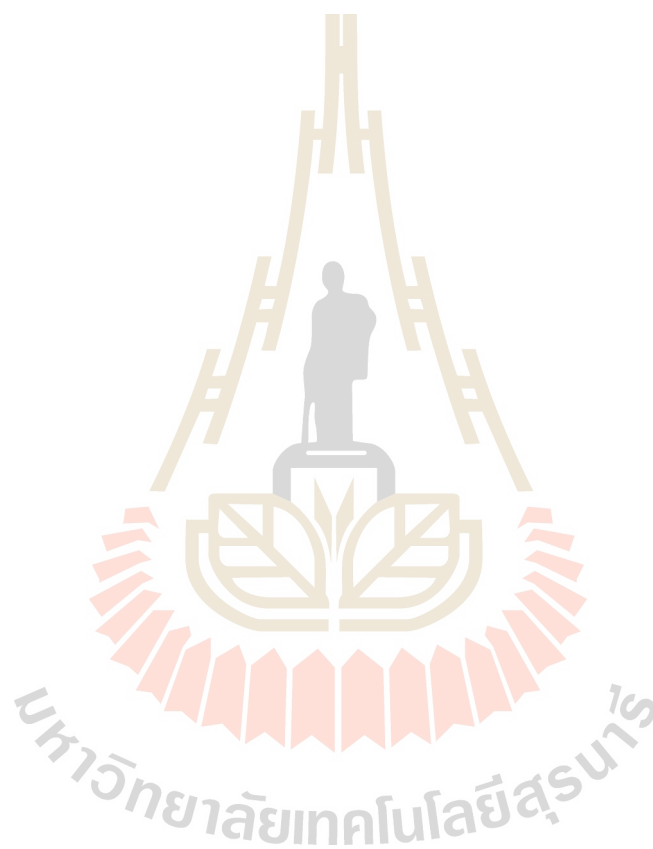
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พื้นที่อนุรักษ์เพียงอย่างเดียวไม่สามารถช่วยรักษาความหลากหลายทางชีวภาพไว้ได้อย่าง  
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จึงเป็นเรื่องที่จำเป็น ในแถบเอเชียตะวันออกเฉียงใต้ซึ่งป่าไม้ถูกทำลายไปอย่างมากมาแต่ยังขาด  
การวิจัยด้านสังคมของสัตว์เลื้อยคลานและสัตว์สะเทินน้ำสะเทินบกในพื้นที่ถูกมนุษย์รบกวน  
งานวิจัยนี้มุ่งศึกษาชุมชนของสัตว์เลื้อยคลานและสัตว์สะเทินน้ำสะเทินบกในป่าเต็งรัง ป่าที่ถูก  
รบกวน และสวนป่ายุคาลิปตัส การสำรวจดำเนินการในเดือนพฤษภาคม มิถุนายนและกันยายน  
พ.ศ. 2558 โดยการวางกับดักตาข่ายจำนวน 11 แห่ง แต่ละแห่งประกอบด้วยกับดักกรวยสองชั้น  
จำนวน 12 กับดักและหลุมกับดักขนาด 40 ลิตรจำนวน 3 หลุม และเก็บข้อมูลลักษณะแหล่งที่อยู่  
อาศัยแต่ละแห่งจากแปลงขนาด 1 x 1 ตารางเมตร จำนวน 6 แปลง ข้อมูลด้านภูมิประเทศเช่น  
ความสูงจากระดับน้ำทะเลและขนาดของพื้นที่ศึกษา ถูกกำหนดโดยโปรแกรม ArcGIS การ  
วิเคราะห์ความสอดคล้องเชิงพหุ (Multiple Correspondence Analysis: MCA) พบว่าพื้นที่ศึกษามี  
ความแตกต่างกันอย่างเป็นอิสระ

ผลการศึกษาพบสัตว์เลื้อยคลานและสัตว์สะเทินน้ำสะเทินบกจำนวน 861 ตัว จำแนกได้  
40 ชนิด โดยป่ายุคาลิปตัส (361 ตัว, 37 ชนิด) และป่าที่ถูกรบกวน (373 ตัว, 29 ชนิด) มีความชุกชุม  
และจำนวนชนิดมากกว่าป่าเต็งรัง (127 ตัว, 29 ชนิด) สำหรับสัตว์สะเทินน้ำสะเทินบกพบความชุก  
ชุมในป่ายุคาลิปตัสและป่าที่ถูกรบกวนมากกว่าในป่าเต็งรัง แต่ในสัตว์เลื้อยคลานกลับพบความชุก  
ชุมในป่ายุคาลิปตัสมากกว่าป่าทั้งสองประเภท

ความมากชนิดของสัตว์เลื้อยคลานและสัตว์สะเทินน้ำสะเทินบกไม่ได้ผันแปรตาม  
ประเภทของป่า แต่เมื่อเปรียบเทียบเฉพาะความมากชนิดของสัตว์สะเทินน้ำสะเทินบกเท่านั้นพบว่า  
ในป่าเต็งรังมีพบความหลากหลายทางจำนวนชนิดของสัตว์สะเทินน้ำสะเทินน้อยกว่าในป่าที่ถูก  
รบกวนและป่ายุคาลิปตัส

ผู้วิจัยใช้เทคนิค non-metric multidimensional scaling และทดสอบด้วย PERMANOVA เพื่อศึกษาหาความแตกต่างโครงสร้างของชุมชนสัตว์เลื้อยคานและสัตว์สะเทินน้ำสะเทินบกไม่พบความแตกต่างอย่างมีนัยสำคัญระหว่างประเภทของป่ากับโครงสร้างของชุมชน



สาขาวิชาชีววิทยา  
ปีการศึกษา 2558

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MATTHEW CRANE : EFFECT OF LAND-USE AND HUMAN  
DISTURBANCE ON HERPETOFAUNAL COMMUNITIES IN THE  
SAKAERAT BIOSPHERE RESERVE, NAKHON RATCHASIMA.  
THESIS ADVISOR : ASST. PROF. PONGTHEP SUWANWAREE, Ph.D  
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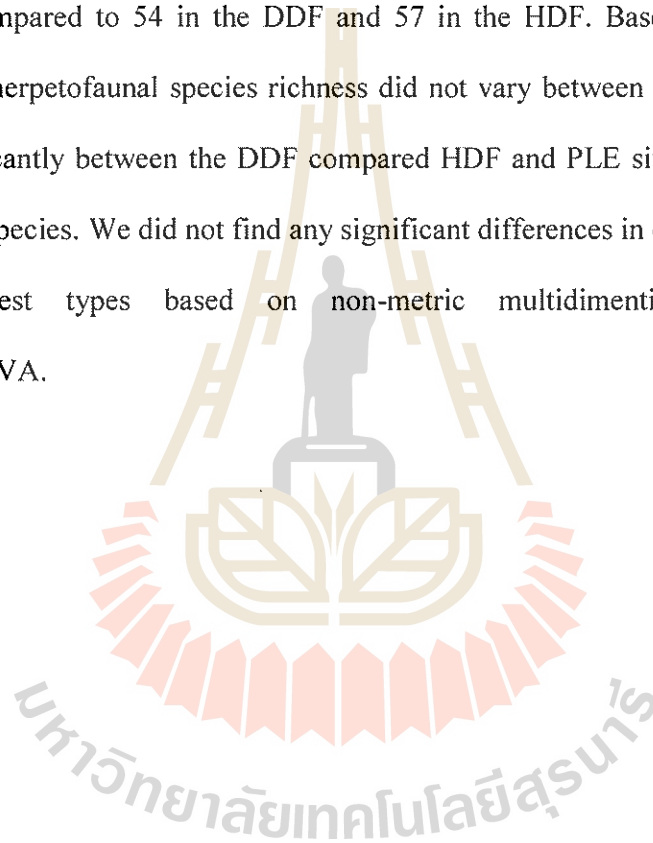
#### DIVERSITY/SPECIES RICHNESS/COMMUNITY STRUCTURE/HABITATS

Protected areas cannot completely sustain biodiversity, thus understanding the role of human-disturbed areas in conserving the world's diversity is critical. Despite intensive deforestation, Southeast Asia is underrepresented in studies investigating faunal communities in human-modified landscapes. This project assessed the herpetofaunal community in dry dipterocarp forest, secondary disturbed forest, and eucalyptus plantations in the Sakaerat Biosphere Reserve. In May, June, and September of 2015, we surveyed using 11 passive trapping arrays. Each array consisted of 12 double funnel traps and three 40 L pitfall traps. We assessed habitat characteristics at each site collected from six 1 m x 1 m quadrats at each site. To collect landscape variables such as elevation and patch size, we used ArcGIS mapping software. Analysis of the habitat characters using multiple correspondence analysis (MCA), did not reveal any clustering of the sites.


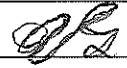
The results showed 861 individuals were captured representing 40 species. The eucalyptus plantations (361 individuals, 37 species) and highly disturbed forest

sites (373 individuals, 29 species) hosted higher herpetofauna abundance and richness than protected areas of dry dipterocarp forest (127 individuals, 29 species).

Both the PLE and HDF (224 and 275 individuals respectively) had higher amphibian abundance than the DDF (57 individuals). Reptiles in contrast showed the highest abundance in the PLE sites compared to the other two forest types, with 100 captures compared to 54 in the DDF and 57 in the HDF. Based on sample based rarefaction herpetofaunal species richness did not vary between forest types, but did vary significantly between the DDF compared HDF and PLE sites when comparing amphibian species. We did not find any significant differences in community structure among forest types based on non-metric multidimensional scaling and PERMANOVA.



School of Biology  
Academic Year 2015

Student's Signature   
Advisor's Signature P. Suran  
Co-advisor's Signature 

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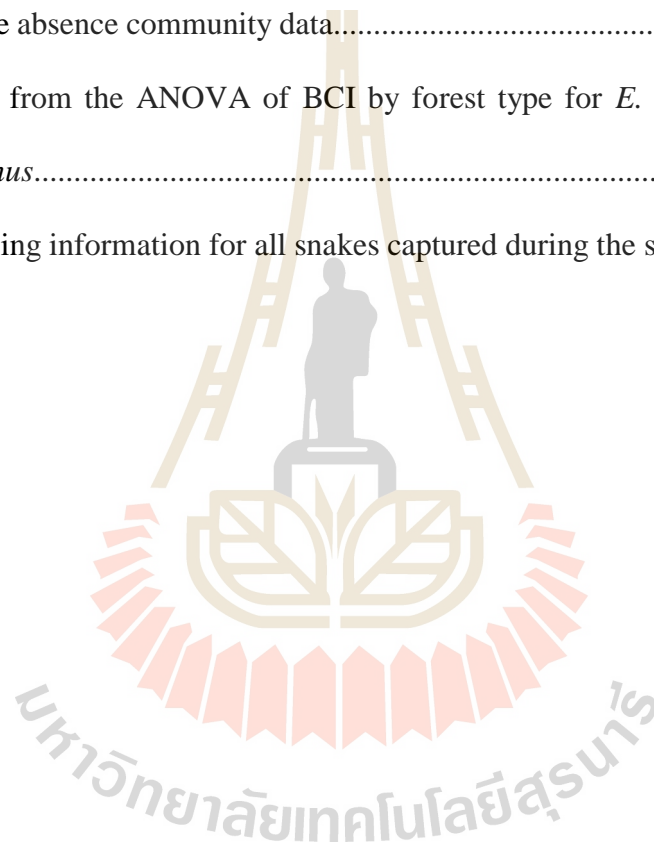
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# CHAPTER I

## INTRODUCTION

### 1.1 Introduction

Global biodiversity stands on the brink of a mass collapse. Human activities threaten species across all taxa and regions, but tropical regions in particular are a major battle ground for global biodiversity. The tropics hold over 60% percent of the Earth's species, yet human population growth has directly conflicted with biodiversity conservation causing widespread deforestation in many tropical regions (Dirzo and Raven, 2003). Conservation measures generally focus on developing and expanding the global protected area network as refuges for biodiversity.

The global protected area network protects 460 million ha (~12.5% of total forest area) (FAO, 2010) of forest cover from deforestation; however many reserves are becoming isolated from other large tracts of undisturbed landscapes (Sánchez-Azofeifa et al., 1999; DeFries and Hansen, 2005). Isolation could mean extinction for a multitude of species as global climate change puts additional pressure on populations by shifting suitable habitat ranges (Bickford et al., 2010). Additionally, many reserves may not effectively protect biodiversity as illegal poaching can lead to the large scale defaunation (Harrison, 2011). Rodrigues et al. (2004) calculated a conservative estimate of gap species, those not represented in protected reserves, by overlaying species distributions and protected area maps, identifying 1,424 species not protected under the current reserve system. The

limitations for the current reserve system for biodiversity conservation have led to an increase in research focusing on the role of human-disturbed areas in sustaining the world's diversity.

Deforestation and the accompanied loss of biodiversity is a particular challenge for sustainable development in Southeast Asia (Fox and Vogler, 2005). Southeast Asia has the highest rates of annual deforestation of all tropical regions, which has been projected to result in the loss of 13-42% of all regionally endemic species in the next century (Brooks et al., 2002). Thailand in particular has been victim to extensive loss of forest cover as land-use changes over time. From 1961 to 1998 estimates indicated that the forest cover declined from 53% to 25% in the 37 year period (Charupat, 2000). A landscape change assessment for the Northeast of Thailand showed an increase in the isolation for a protected forest along with an increase in agriculture and urban development in the area from 1980 to 2010 (Sutthivanich and Ongsomwang, 2015).

Deforestation and other anthropogenic impacts such as urbanization and poaching are causing declines across all taxa; however, amphibians are the most threatened of all terrestrial vertebrates (Sodhi et al., 2010). Despite very different morphological and life history traits reptile species are imperiled by many of the same anthropogenic influences (Gibbon et al., 2000). A recent assessment on the global conservation status of reptiles revealed that over 18% of the 9,084 described species are threatened (Böhm et al., 2013). The status of amphibians mirrors that of reptiles with 7.4% of the 5,743 known species listed as Critically Endangered on the IUCN red list (Stuart et al., 2004). Along with lizards and amphibians snake populations are thought to be in decline globally (Reading and Luiselli, 2010), highlighting the need for herpetofaunal

community and population studies to monitor the observed trends. Reptile populations face a wide array of extinction threats from anthropogenic activities. Species affected by anthropogenic habitat loss were particularly prevalent in Southeast Asia (Böhm et al., 2013). The conservation value of human-modified landscapes must be assessed to inform conservation action plans in the region.

Despite the high rate of deforestation, Southeast Asia is generally underrepresented in studies on faunal community response to habitat loss and response to human-modified landscapes (Trimble and Aarde, 2012). Additionally, herpetofauna is globally underrepresented in community studies and in their response to anthropogenic disturbance (Voris, 2006; Trimble and Aarde, 2012). Thailand is home to more than 142 species of amphibians and over 218 species of reptiles (IUCN, 2014), and has the smallest area of remnant forest cover in Southeast Asia (Sodhi et al., 2010). The herpetofaunal diversity and the level of human disruption make Thailand an ideal site to investigate the impacts of land-use change on tropical amphibian and reptile communities.

The United Nations Educational, Scientific, and Cultural Organization (UNESCO) Man and the Biosphere Programme was developed to establish sustainable landscapes; balancing biodiversity conservation and sustainable human development. The Sakaerat Biosphere Reserve is home to 41% of Thailand's reptile species and 22% of the amphibian species listed on the IUCN red list (IUCN, 2014; SERS, 2014). With a diverse landscape of protected forest and mixed agricultural land it is an ideal location to study herpetofauna and the impacts of land-use on communities. Recent studies in the area have led to a deeper understanding of specific amphibian and reptile species such as hatching plasticity in the frogs (Poo and Bickford, 2014), the

spatial ecology and behavior of king cobras (Strine et al., 2014), and the discovery of a new frog species (*Limnonectes megastomias*) (McLeod, 2008). However only one study from the area has assessed a herpetofaunal community, which was focused only on semi-aquatic snake species (Genus *Enhydris*) (Karns et al., 2010). The proposed project will be the first full herpetofaunal assessment of the disturbed landscapes in the Sakaerat Biosphere Reserve.

## 1.2 Research objectives

- 1) To determine abundance, richness, and diversity of reptile (excluding birds) and amphibian species in the (a) secondary forests, (b) plantation forests, and (c) protected areas of the Sakaerat Biosphere Reserve.
- 2) To evaluate the structure of amphibian and reptile communities in three different land use types
- 3) To determine the impact of land-use on body condition of highly abundant species

## 1.3 Scope and limitations

Starting in May 2015 herpetofaunal community sampling took place in the core area and the transition area of the Sakaerat Biosphere Reserve. The study area was confined to the Northeast including the majority of the dry dipterocarp forest in the core area and a patchwork of mixed agricultural and small plantation. We sampled three land use types: heterogeneous disturbed forest (HDF), eucalyptus plantation forest (PLE), and primary dry dipterocarp forest (DDF). Dry evergreen forest which is present within the Southwest of the Biosphere Reserve were not sampled, as the

disturbed areas were most likely dry dipterocarp forest before conversion by human activities. Plantations forests were considered monoculture forest stands of eucalyptus. Heterogeneous disturbed forests are fragments of natural forests with high levels of anthropogenic disturbance embedded in an agricultural matrix. The survey methods were limited to passive trapping arrays (12) using both double-funnel and pitfalls traps. We sampled once each month in May, June, and September.



## **CHAPTER II**

### **LITERATURE REVIEW**

#### **2.1 Herpetofaunal diversity in human-dominated landscapes**

Research projects investigating community response to human-disturbance are typically heavily biased towards specific taxa and regions. Most studies come out of the Neotropical forests and focus on mammals, plants, or invertebrates (Trimble and Aarde, 2012). However, several studies have assessed both the methods of studying herpetofauna in human-disturbed landscapes and the impact on community assemblages (Urbina-Cardona et al., 2006; Gardner et al., 2007b; Ribeiro-Júnior et al., 2008) (Table 2.1). The ability of plantation forests to maintain herpetofaunal diversity is often compared to pristine forests to elucidate similarities in community structure impacts from land-use change.

**Table 2.1** The impacts of human landscapes on herpetofaunal communities in the tropics.

Habitat types	Places	Amphibian		Reptile		References
		Species	Individuals	Species	Individuals	
Remnant forest bordered with palmetto		18	142	10	58	
Remnant forest bordered with pasture	Near La Selva Biological Station, Costa Rica	17	225	10	61	Kurz et al. (2014)
Palmetto plantation		11	68	2	2	
Pasture		9	46	12	66	
Mature primary rainforest	Northeastern Brazilian Amazonia, Brazil	22	515	25	681	Gardner et al. (2007b)
14-19 years secondary forest		14	510	15	87	
4-5 years <i>Eucalyptus</i> plantation		5	714	14	757	
40-400 years secondary forest	Hong Kong, China	6	52	7	237	Sung et al. (2012)
30-60 years <i>Lophostemon confertus</i> plantation		4	16	8	134	
Primary rainforest	Kibale National Park, Uganda	7	46	2	2	Vonesh (2001)
Selective logged rainforest		8	94	4	17	
Pine plantation		8	102	4	4	
Primary rainforest	La Selva Biological Station, Costa Rica	8	236	6	49	Folt and Reider (2013)
<i>Pentaclethra macroloba</i> plantation		5	95	4	13	
<i>Virola koschnyi</i> plantation		7	117	3	25	
<i>Vochysia guatemalensis</i> plantation		8	128	5	48	
Fragmented tropical rainforest	Los Tuxtlas Biosphere Reserve, Mexico	13	588	22	258	Urbina-Cardona et al. (2006)
Edge of fragmented tropical rainforest		14	570	25	208	
Pasture		12	98	11	157	
Tropical dry forest	Chamela Biosphere Reserve, Mexico	4-13	9-46	22-28	178-276	Suazo-Ortuño et al. (2008)
Disturbed tropical dry forest		6-8	23-40	19-29	238-304	

While some results did not agree, most studies found significant differences in the abundance, richness, and community structure responses for amphibians and reptiles (Gardner et al., 2007b; Wanger et al., 2010, Kurz et al., 2014;). Typically amphibian communities were more sensitive to human disturbance than reptiles, which makes ecological sense as amphibian morphology (for example semi-permeable skin) means that environmental factors can have a greater impact (Cushman, 2006). However, the literature does not support a unanimous conclusion about the response of amphibian communities to human disturbance gradients.

One study from La Selva Biological Station in Costa Rica tested the influence of land use along the forest edge on herpetofaunal communities sampling palmetto, pasture, and natural forests (Kurz et al., 2014). They used generalized linear mixed models and determined the best habitat classification model for describing the observed species richness and abundance. The best fit model for amphibians grouped both disturbance areas (open pasture and palmetto plantation) as a single type compared to forest. The results indicate that regardless of the land-use the amphibian communities were sensitive to human habitat disturbance. A second study from the same site aimed at answering a different question related to human disturbance impacts.

Folt and Reider (2013) examined plantation forests of three separate species (*Pentaclethra maculosa*, *Virola koschnyi*, and *Vochysia guatemalensis*) compared to natural growth forest. The study found a significant difference in amphibian species richness between only *one* plantation type (*P. maculosa*) and the reference forest at the 95% confidence interval. The two other plantation forests under review contained comparable species richness to the reference forest. However, the authors also note



that community assemblage differed significantly between both *P. macroloba* and *V. guatemalensis* when compared to the primary forest amphibian community structure. Frog abundances were also highest in one of the plantation forests (*V. koschnyi*) than any other forest type. The study indicates that based on species richness, native mono-culture plantation stands can support biodiversity conservation. The two studies from La Selva Biological Station illustrate the complexity of community response to human disturbance and the need for further research to isolate all of the factors in disturbance impacts.

Work from the Brazilian Amazonia tropical forests found similar results to both of the La Selva Station studies on the influence of human disturbance schemes on amphibian communities. Gardner et al. (2007a) sought to standardize the comparison of primary, secondary, and plantation forests in a Brazilian rainforest. Instead of investigating a reference primary forest in comparison to native species mono-culture plantations, the study sampled a gradient of disturbed forests: primary, secondary, and non-native *Eucalyptus* plantations. Results from the study found that amphibian abundances were similar across all forest types while the community structure varied between primary forest and secondary and plantation forests. The amphibian species found in the two disturbed forest types (secondary and plantation) were a subset of the primary forest amphibian community.

Similar to Folt and Reider (2013) who determined that plantation forests can support some species diversity, however Gardner et al. (2007b) did not find any disturbed forest that was comparable to the reference forest. Gardner et al. (2007b) additionally determined that secondary forests held significantly more amphibian species than plantation forests.

Additional evidence exists that secondary forests can contain relatively higher amphibian abundances than non-native plantation forests, even if species composition does not differ (Sung et al., 2012). Additionally, work in a Neotropical dry forest supported the conclusions that amphibian abundances are similar across disturbed and primary landscapes, while species richness and community structure decrease (Suazo-Ortuño et al., 2008). In Southeast Asia these results also are confirmed from the only study conducted in the region in Indonesia. Amphibian species richness decreased significantly along the human disturbance gradient (Wanger et al., 2010).

Despite many studies arriving at similar conclusions some notable studies found that disturbed areas had either a neutral impact or even a positive impact on amphibian communities. Vonesh (2001) investigated the differences between an undisturbed primary forest, a similar forest with historical selective logging, and a pine plantation forest in Kibale National Park, Uganda. He determined that the logged forest had higher herpetofaunal species richness and abundance than the undisturbed forest, but that the pine plantation forest had the highest overall species richness and abundance. Fredricksen and Fredricksen (2004) also arrived at very different conclusions in terms of amphibian species richness and abundance. The researchers also studied the impact of selective logging on amphibian communities, but found no significant differences between logged and unlogged habitats in Bolivia. The conflicting results demands more research to assess either the methodological shortcomings that caused the results or identify critical factors that can lead to such drastically different conclusions.

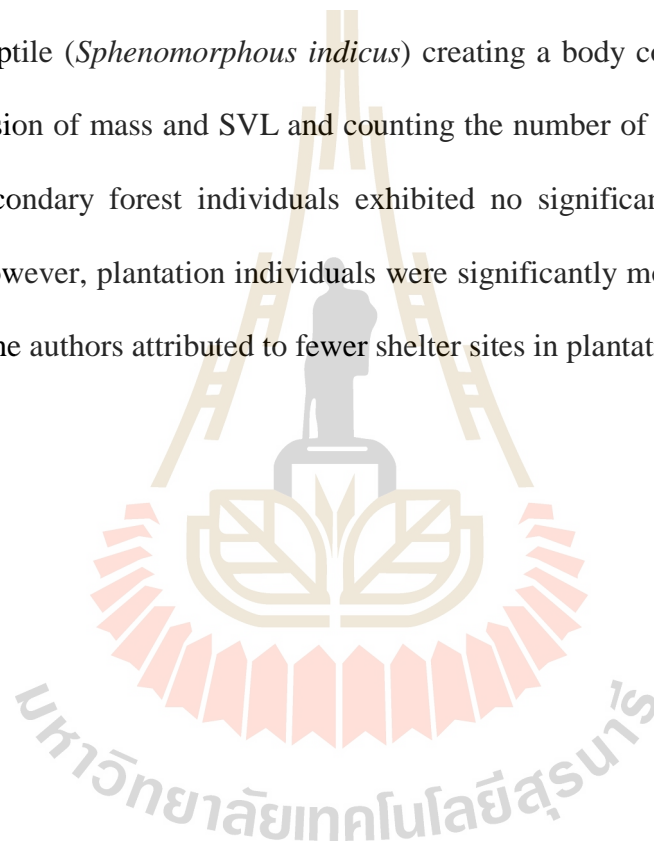
Reptile communities show different trends than amphibians in response to human habitat conversion. In general, reptile communities were more resilient to forest

change and human disturbance, and in some cases species richness and abundance increased along disturbance gradients. For example, the previously discussed Folt and Reider (2013) study determined that reptile community assemblages between the primary forest and two of the plantation forests were indistinguishable (*V. koschnyi* and *V. guatemalensis*) as opposed to a single plantation type for amphibians. Kurz et al. (2014) did not find that the studied disturbed forest type (palmetto) held higher reptile species richness or abundance compared to primary forest, but did determine that palmetto was less dissimilar to primary forest than heavily disturbed pasture land. Gardner et al. (2007b) found that primary forests held higher species diversity than secondary or non-native plantation forests, but that the species richness between secondary and plantation forests did not differ significantly for reptiles in direct contrast to amphibian species richness. Additionally, several unique reptile species were found only in plantation forests. While none of these studies found that disturbance increased reptile species richness or abundance, all of them illustrated that reptile communities are relatively less sensitive to change than amphibians.

Two studies on herpetofaunal communities did discover that reptile species richness increased. In an agricultural matrix around a Neotropical dry forest, lizard species richness, diversity, and abundance was higher than in the remnant comparison forest (Suazo-Ortuño et al., 2008). However, from the same study, turtles were highly sensitive to habitat conversion, and no snake species was sensitive to disturbance, indicating that treating all reptiles as a group may lead to misinterpreting the results. In the study by Wanger et al. (2010) out of Indonesia, reptile species richness and abundance peaked in a natural shade-cacao plantation forest. The results from all the

studies show positive implications for the availability of plantation and regrowth forest regimes in maintaining reptile diversity and abundance.

In addition to investigating the effect of human disturbance on reptile species richness, abundance, and community structure one study also attempted to assess the secondary impacts of disturbance on individual health. Sung et al. compared secondary forests and plantation forests in Hong Kong selecting the most abundant species of reptile (*Sphenomorphus indicus*) creating a body condition index from a linear regression of mass and SVL and counting the number of individuals exhibiting tail loss. Secondary forest individuals exhibited no significant difference in body condition; however, plantation individuals were significantly more likely to show tail loss, which the authors attributed to fewer shelter sites in plantation forests.



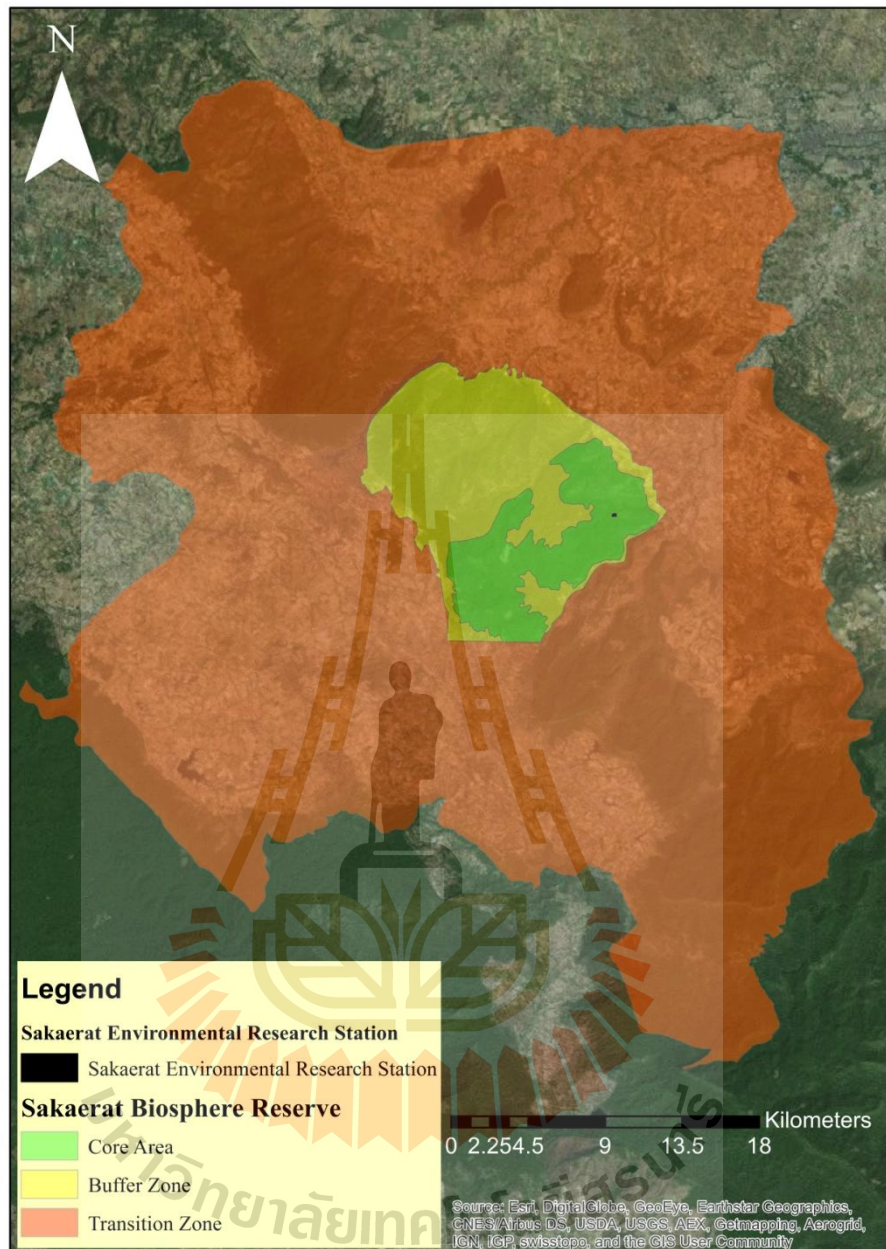
## CHAPTER III

### METHODS

#### 3.1 Study site

The study was conducted within the Sakaerat Biosphere Reserve located in Nakhon Ratchasima Province, Thailand (14.44–14.55°N, 101.88–101.95°E). The reserve has an 80 km<sup>2</sup> core area and a combined 360 km<sup>2</sup> making up the buffer and transitional zones, which consist mostly of agricultural and settlement areas (Figure 3.1). The core area predominately consists of primary growth dry evergreen forest (60%), dry dipterocarp forest (18%), and secondary plantation forest (<18%) (Tongyai, 1980). The dry dipterocarp forest is endemic to South East Asia and is characterized with thick *Vietnamosasa pusilla* ground cover and dipterocarp trees such as *Shorea siamensis* and *Shorea obtusa* (Lamotte et al., 1998). The transition zone of SBR comprises nearly 82% of the total area and is characterized by isolated forest fragments in a patchwork of agricultural fields, small plantation forests, and human settlements.

The Sakaerat Biosphere Reserve hosts 93 reptile and 29 sp amphibian species. Four reptiles and three amphibian species are categorized as Threatened or Near Threatened on the IUCN Red List (IUCN, 2014; Sakaerat Environmental Research Station, 2014). However, the exact number of species is still unknown as populations of known species in the area may in fact be separate species (Voris, 2006). The estimate is also low as several species are Data Deficient or not present on the list.



**Figure 3.1** Map of the Sakaerat Biosphere Reserve delineating core, buffer, and transition areas.

### 3.2 Assessment of forest types

I assessed the herpetofaunal community assemblages across a gradient of human disturbance, specifically in remnant dry dipterocarp forest (DDF), highly disturbed forest (HDF), and eucalyptus plantation forest (PLE). For the purpose of the study disturbed forests were considered as secondary forests embedded in an agricultural matrix, characterized by high levels of anthropogenic change, but still similar to remnant forests in dominant vegetative cover type. Cycads and bamboo grass are typically present, along with species of Dipterocarpaceae. We considered, eucalyptus plantations as forest stands planted with *Eucalyptus camaldulensis* for economic production. Within SBR eucalyptus stands are typically harvested every 2-5 years, at which point the trees are cut while the base is allowed to regrow. The eucalyptus trees are planted in straight lines with roughly two to five meters between rows.

Current land use maps provided by the Thai government were inaccurate with 7 out of 10 randomly selected points not matching the indicated classification. I used satellite imagery from Google Earth to create polygons by hand of each identifiable plantation and secondary forest within the study area. To test the accuracy of visual identification, I ground-truthed the newly created areas, which resulted in 9 out of 10 randomly selected sites falling in appropriate land-use type. This initial assessment indicated that the hand generated plantation and disturbed forest areas are sufficient to randomly select sites; however, further analyses of the area was required to accurately determine land use composition of the transition area.

We randomly selected twelve sites (4 in each forest type) using ArcMap 10.1. Plots were set no closer than 450 meters from each other to control for spatial autocorrelation. The total area of the disturbed forest patches (127.13 ha) and the

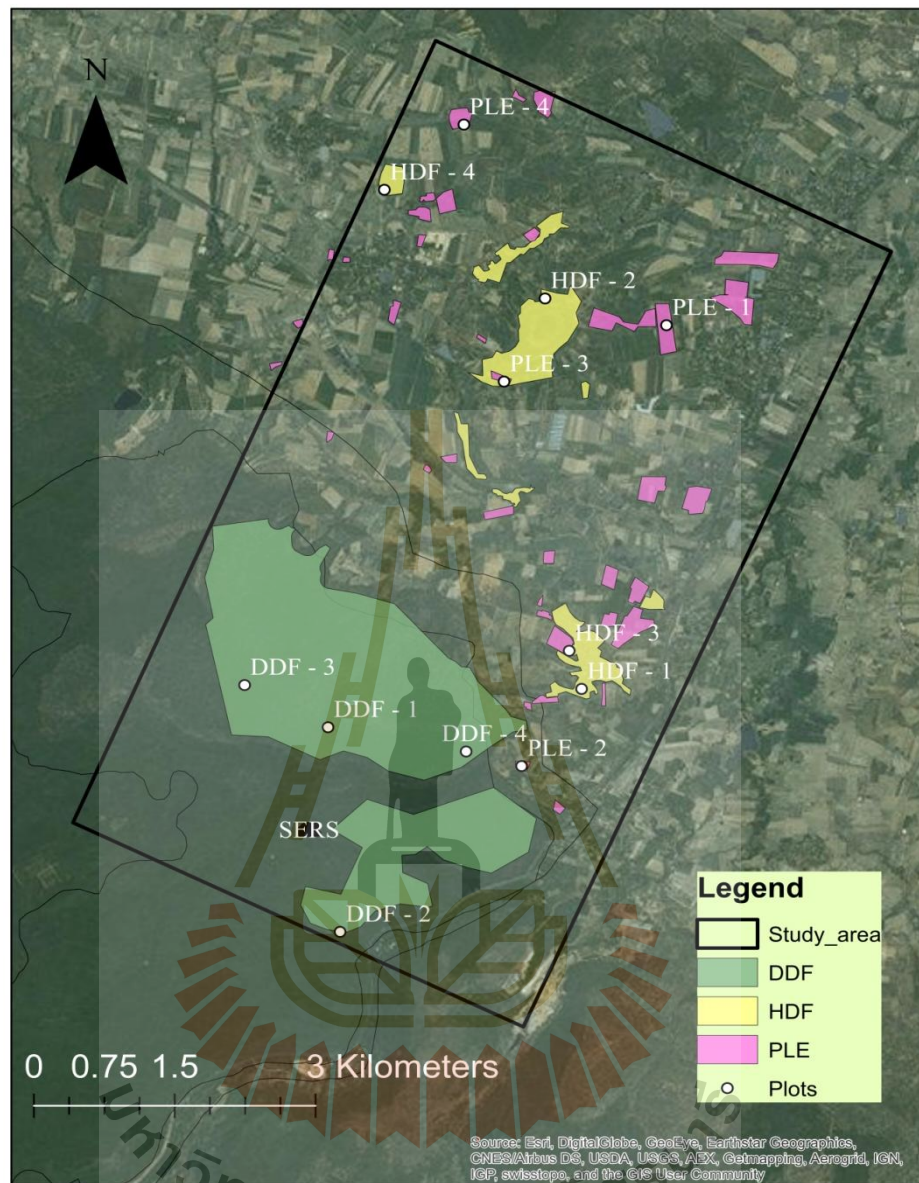
plantation forests (110.97 ha) were roughly equal; however, the plantation forests were much more fragmented and smaller (Figure 3.2).

**Table 3.1** Landscape characteristics for each plot site used in the study, negative values are used to indicate that site is located inside the SBR core area.

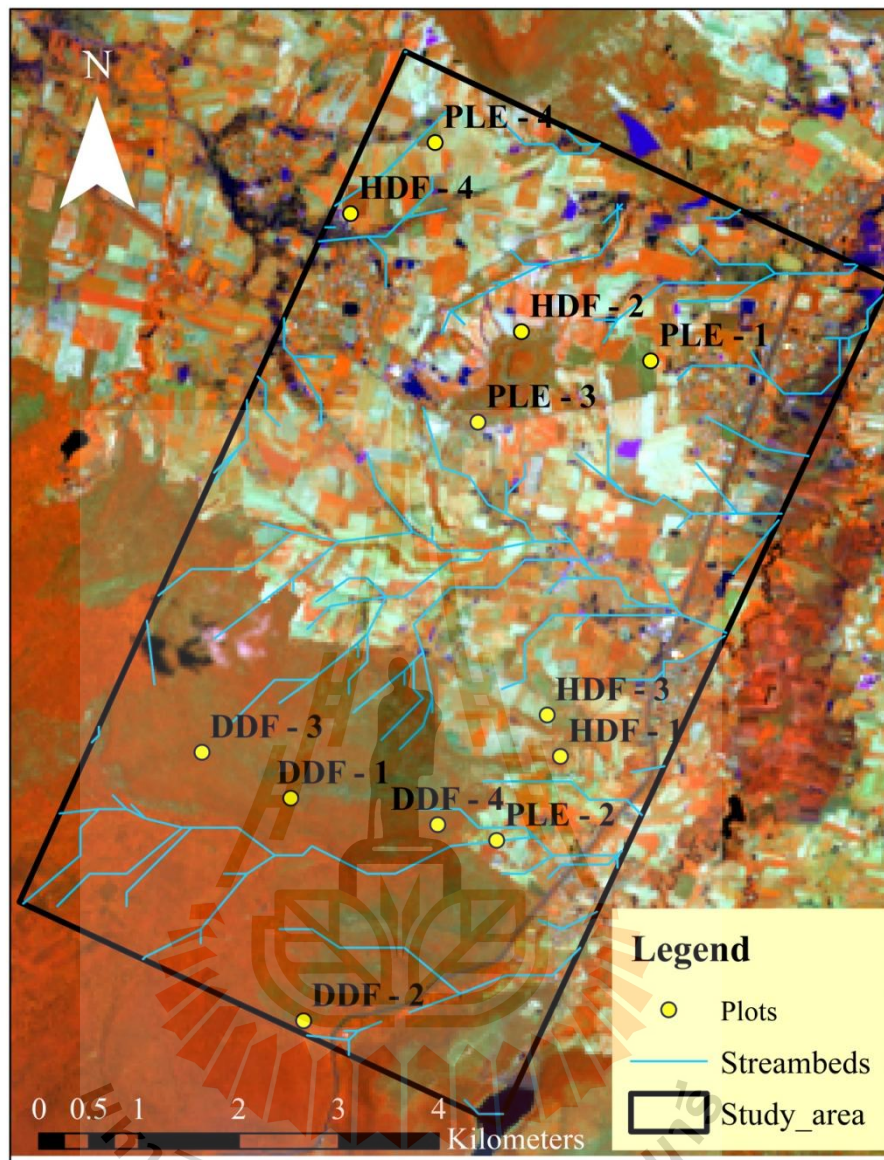
Site	Elevation (m)	Patch size (ha)	Slope (°)	Distance from edge of core area (m)	Distance to water (m)	Water type
DDF - 1	368	527.7	7.0	-1298	475	Stream
DDF - 2	386	179.2	24.2	-324	950	Stream
DDF - 3	396	527.7	7.5	-1125	680	Stream
DDF - 4	294	527.7	12.1	-218	185	Stream
HDF - 1	254	33.1	4.5	1081	150	Pond
HDF - 2	272	57.9	7.1	3252	150	Pond
HDF - 3	265	33.1	1.5	979	310	Pond
HDF - 4	251	7.3	2.4	3150	46	Pond
PLE - 1	246	8.3	2.5	3988	160	Pond
PLE - 2	253	1.1	3.1	328	12	Stream
PLE - 3	274	1.3	9.6	2430	123	Pond

I investigated landscape factors for each plot site, including distance to water, patch size, slope, elevation, and distance to the edge of the protected area (Table 3.1). To determine distance to water I used a 32-day composite of Landsat 8 data from April 7th to May 9th to identify sources of water within the study area. Additionally I used a digital elevation model of the study area to determine stream beds using the Hydrology toolbox in ArcGIS 10.1 (Figure 3.3).





**Figure 3.2** All three forest types that were sampled in the Sakaerat Biosphere Reserve (Highly disturbed forest, dry dipterocarp forest, and plantation forest).



**Figure 3.3** Map of the study area showing streambeds and composite of Landsat 8 images to clearly determine water sources.

To effectively sample the forest areas, we built Y-shaped drift-fence arrays with double funnel traps and 40 L pitfall traps. We attached two double funnel traps measuring 2 m x 0.5m x 0.3 m at the end of each line. A 40 L pitfall trap was placed at the midpoint of each line for a total of three pitfalls per array. Six double funnel traps were affixed to the center of each array for a total of 12 funnel traps and 3 pitfall traps (Figure 3.4). Each arm of the plot was 15 m in length with ground vegetation cleared half a meter on both sides. Each plot was oriented with 1 line running North-South with the two other lines angled 120° from the North-South line.

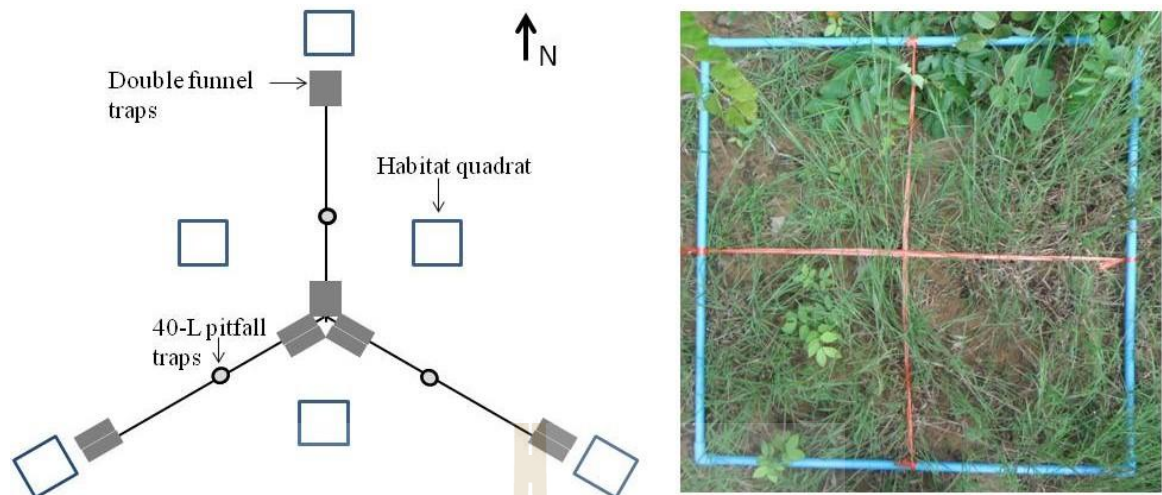


**Figure 3.4** Passive trapping array showing the line (A), wings (B), and traps (C and D).

Each site was sampled for three days in May, June, and September. We sampled half of the plots, representing an equal number for each forest type, for 3 days and then switched to the second set. Prior to sampling each site was visited and repairs made to ensure that all plots were equivalent.

To account for habitat variations between sites, measured several environmental variables at each plot including: percent canopy cover, percent groundcover, leaf litter depth, leaf litter ground coverage, and vegetative density at the groundstory (<1 m), understory (1-3), midstory (3-5 m), and abovestory (> 5 m). Each variable was assessed from a 1 m x 1 m quadrat at 6 sites per plot, three set 7.5 m from the center away from each line, and three set 3 m away from the end of each line (Figure 3.5). To measure canopy density, I used hand-made densiometer created from a PVC pipe with fishing line creating four equal areas. Additionally, I assessed ground cover within each quadrat including percent coverage of leaf litter, rocks, grass, dead vegetation, fallen logs, trees/saplings, and bare ground. When leaf litter was present, I used a rigid ruler to measure depth to the closest 1-mm. Ground cover and vegetation density were divided into 7 classifications (None, very light, light, medium, heavy, very heavy, complete).

Both May (8.55 mm) and June (79.25 mm), had lower precipitation than September (259.85 mm). Despite the differences in precipitation in 2015, the average rainfall in May (107.8 mm) and June (90.8 mm) over the past 4 years are comparable. Due to differences in average temperature, relative humidity, and rain fall May and June were categorized as dry season samples and September represented a single rainy season month (see Appendix 3 and 7).



**Figure 3.5** Plot layout showing the locations used to assess habitats (Left) and an example quadrat used to estimate ground cover in the study (Right).

### 3.3 Captures and biometrics

All snakes captured through the study were brought to the lab for processing to increase measurement precision. Individuals were marked with a field cauterizing unit on the ventral scales for mark and recapture analysis (Winne et al., 2006). Isoflorane was used to anesthetize captured snakes to determine accurate biometrics, which is proven to be a more accurate method for measuring individuals (Blouin-Demers, 2003; Setser, 2007).

We collected biometrics for an additional five species: *Kaloula pulchra*, *Kaloula mediolineata*, *Dixonious siamensis*, *Eutropis macularia*, and *Leiolepis reevesii rubritaeniata*. Captured individuals were processed in the field, at the time and site of capture. We collected snout-to-vent lengths (SVL) as well as mass, using a digital caliper and digital scale respectively. Mass and SVL were used to construct a body condition index for abundant species (Schulte-Hostedde et al., 2005; Peig and Green, 2009). Lizards were marked with xylene free permanent markers on the left and right

posterior jaw. While not a long term marking method, the technique allowed us to determine recaptures within sampling sessions.

### 3.4 Data analysis

#### 3.4.1 Abundance, richness, and diversity

**Table 3.2** Each category that was assessed in the study and which data and statistical methods were used.

Category	Data used	Methods
Environment	Habitat from site	Multiple correspondence analysis
Abundance	Captures	ANOVA, Kruskal-Wallis
Species richness	Captures	Sample based rarefaction
Diversity	Captures	Kruskal-Wallis
Community structure	Captures; landscape data; habitat data	NMDS, PERMANOVA, hierarchic clustering
Body condition	Biometric data	ANOVA, general least squares regression

We analyzed amphibian and reptile communities separately for all methods as life history traits and response to human disturbance can vary drastically between these two groups. Additionally we conducted the same analyses for all herpetofauna captures to determine whether any observed trends were consistent for both groups as a whole. Additionally, we compared differences in abundance, richness, and diversity for each month that was sampled.

Abundance was plotted against patch size, elevation, and distance to water to determine whether any potential correlations may exist. However, since the sample

size was limited to 11 sites and we select sites based on forest type and not environmental variables further statistical tests to verify differences were not possible. I used the categorical variables in a multiple correspondence analysis to investigate which environmental variables if any contributed to separation of the three forest types. Multiple correspondence analysis is similar to principle component analysis, but incorporates categorical rather than numeric inputs (Abdi and Valentin, 2007).

We compared both reptile and amphibian abundances between forest types using the accumulated total for each plot. Additionally to control for unequal sampling between seasons, we also tested the difference in abundance between the forest types for each month. For any that was non-parametric, we conducted a Kruskal-Wallis signed rank test followed by a pairwise comparison using a Wilcoxon signed rank test for any significant results at the 95% confidence interval. For data that fit the assumptions of parametric testing we used ANOVA with Tukey's post-hoc testing to determine significant differences at the 95% confidence interval. Additionally, we created Whittaker plots for each forest type to display the species dominance patterns for the entire landscape and for each individual forest type.

To analyze species richness we first created sample-based rarefaction curves using each day that a plot was open as a sampling unit. Rarefaction curves were created for each forest type and then assessed for significance at the 95% confidence interval. To assess sampling completeness we divided the observed species richness at each site by the estimated richness calculated using the Chao 1 estimate. Chao1 is a non-parametric estimator that provides a lower bound for species richness (Chao, 1984).

We compared diversity between forest types using two different indices. Firstly we used the standard Shannon-Wiener index which incorporates species richness and evenness to calculate diversity.

$$H' = - \sum_{i=1}^s p_i \ln (p_i)$$

Additionally we investigated differences between diversity using the Simpson index which incorporates species dominance to define diversity.

$$\lambda = \sum_{i=1}^R p_i^2$$

### 3.4.2 Community structure

Prior to any testing I transformed the community data for each group. I applied a total relativization to amphibian captures as several sites had a much higher number of individuals. Both reptiles and total herpetofauna were transformed using a  $\log(x+1)$  to reduce the impact of highly abundant species (McCune and Grace, 2002). First to explore community similarity between sample sites I applied hierarchic clustering analysis based on the Bray-Curtis dissimilarity matrix for all plot sites. We used the complete linkage method to create the clusters, as our aim was to visualize the most compact groups with minimal within group spread (Manning et al., 2008). Next we tested for spatial auto-correlation between sites using the Mantel test. To remove any spatial correlations from later analysis we also created groups based on geographic distances using hierarchic clustering and then used that as an additional grouping variable.

We used non-metric multidimensional scaling (NMDS) as an ordination technique to visually represent the dissimilarity between sites. NMDS is a non-



parametric ordination technique that requires less assumptions from the data, and was more appropriate for our dataset which contained a high number of zeros (Zuur et al., 2007). We applied environmental fitting to the NMDS to determine which variables provided significant influence on site separation. Any variables that were in the 90% confidence were used in the formula for the follow up hypothesis testing. To test the hypothesis that community structure varied between forest types, we ran a PERMANOVA test with 999 permutations (Anderson and Walsh, 2013). We also used indicator species analysis to determine which species contributed to differences between forest types. All analyses were done in R Studio using packages "vegan", "MASS", and "BiodiversityR" (Venable and Ripley, 2002; Kindt and Coe, 2005; Husson et al., 2016; Oksanen et al., 2016; R Core Development Team, 2016).

### 3.4.3 Body condition

I created a body condition index using the residuals from a ordinary least squares regression of SVL to mass. Only species with measurements of at least 2 individuals from each forest type were select to create a body condition index. Two species, *D. siamensis* and *E. macularia*, fit the sample size requirements, but only *E. macularia* had data that was normally distributed and homogenous allowing for a linear regression.

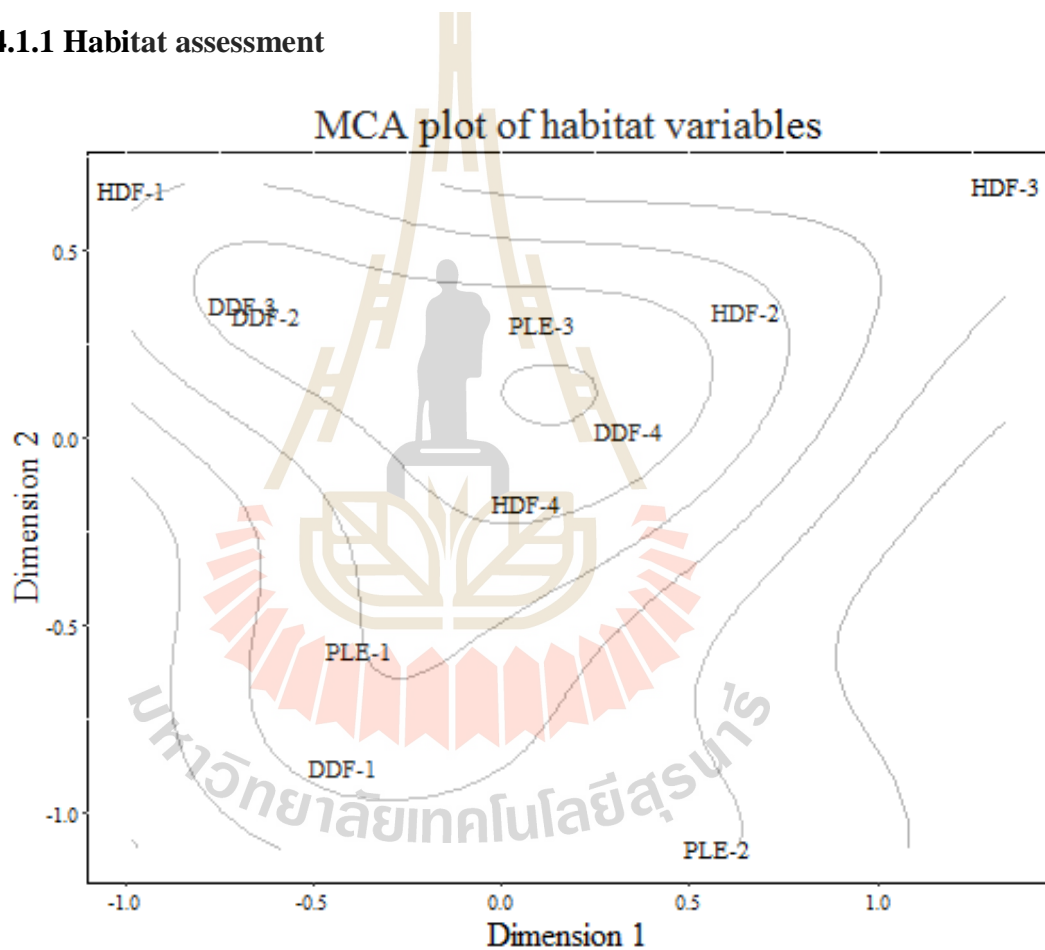
We also analyzed whether forest type influenced whether reptile individuals had lost their tails as an indicator for fitness. Tail loss indicates potential predation pressure, and requires energy reserves to re-grow the dropped appendage. We used a chi-square test for independence, to determine whether forest type influenced tail loss for all captured reptiles and for the two most abundant species: *D. siamensis* and *E. macularia*

# CHAPTER IV

## RESULTS AND DISCUSSION

### 4.1 Results

#### 4.1.1 Habitat assessment



**Figure 4.1** Plotting of sites based on multiple correspondence analysis of habitat variables.

The results of the multiple correspondence analysis did not show clear clustering of forest types (Figure 4.1). Additionally, the MCA results showed that no measured variables contributed strongly to the variance between each site (Table 4.1).

**Table 4.1** Contribution to the variance between sites from eigenvalues resulting from multiple correspondence analysis of site habitats.

Dimension	1	2	3	4	5	6	7	8	9	10
Variance %	0.41	0.34	0.32	0.25	0.22	0.17	0.14	0.12	0.07	0.04
% Variance explained	19.4	16.4	15.3	12.0	10.6	8.32	6.67	5.57	3.57	1.93
Cumulative % variance explained	19.4	35.9	51.2	63.3	73.9	82.2	88.9	94.5	98.0	100

#### 4.1.2 Plot captures

We sampled for a total of 1,977 trap nights over the course of the study. Two out of the 12 plots were stolen during the course of the study (1 in eucalyptus plantation and 1 in heterogeneous disturbed forest) resulting in uneven sampling between forest types (Table 4.2).

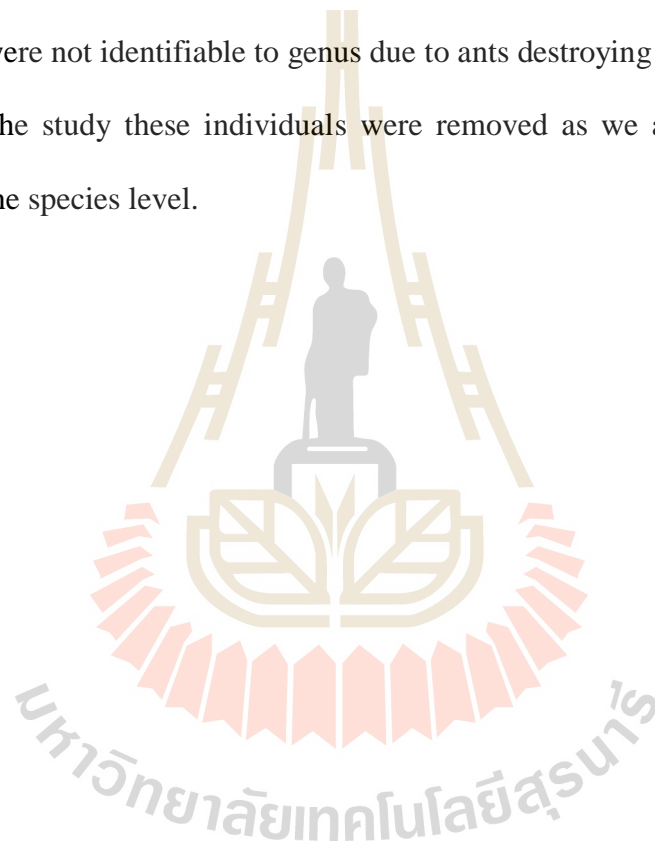
**Table 4.2** Comparison of sampling effort as the number of trap nights for each forest type.

Forest type	Funnel	Pitfall	Total
DDF	576	144	720
HDF	540	135	675
PLE	468	114	582
Grand Total	1584	393	1977

Throughout the study we recorded a total of 861 individuals comprising 40 recognized species. From the 861 captures 110 individuals (12.8% of total) were not completely identified to species, including 48 reptiles and 62 amphibians (Table 4.3

and Table 4.4). Geckos contributed to the majority of the unidentified reptiles (38 individuals), primarily due to the close similarity between the species *Hemidactylus frenatus* and *Hemidactylus garnottii*. For analysis we considered both *Hemidactylus* spp. as a single species.

Over half of the unidentified amphibians (35 individuals) came from the Microhylidae family and specifically the genus *Microhyla*. Additionally, several individuals were not identifiable to genus due to ants destroying the specimen. For the analyses in the study these individuals were removed as we assessed richness and diversity at the species level.



**Table 4.3** Amphibians captures by species from three forest types DDF, HDF, and PLE.

Family	Species	DDF	HDF	PLE	Total
Bufonidae	<i>Duttaphrynus melanostictus</i>		7	5	12
Microhylidae	<i>Calluella guttulata</i>		1	10	11
	<i>Glyphoglossus molossus</i>	18	3	28	49
	<i>Kaloula mediolineata</i>	20	27	11	58
	<i>Kaloula pulchra</i>	13	6	13	32
	<i>Microhyla butleri</i>	1	13	19	33
	<i>Microhyla heymonsi</i>	1	20	28	49
	<i>Microhyla fissipes</i>	1	78	51	130
	<i>Microhyla pulchra</i>	1	53	8	62
	<i>Microhyla</i> sp.		12	23	35
	<i>Micryletta inornata</i>			11	11
	Ranidae	<i>Hylarana erythraea</i>			1
<i>Hylarana macrodactyla</i>			2	3	5
Dicroglossidae	<i>Fejervarya limnocharis</i>	2	66	31	99
	<i>Occidozyga lima</i>			5	5
Unknown	Unknown		23	4	27
Species		8	11	14	14
Grand total		57	311	251	619

**Table 4.4** Reptile captures by species from three forest types DDF, HDF, and PLE.

Family	Species	DDF	HDF	PLE	Total
Agamidae	<i>Calotes</i> sp.			1	1
	<i>Calotes versicolor</i>	3		1	4
	<i>Leiolepis reevesii</i>	1	5	8	14
Colubridae	<i>Boiga multimaculata</i>	2			2
	<i>Boiga siamensis</i>	1			1
	<i>Chrysopelea ornata</i>	1		1	2
	<i>Coelognathus radiatus</i>	1			1
	<i>Dendrelaphis suborcularis</i>			1	1
	<i>Enhydris plumbea</i>			1	1
	<i>Lycodon capucinus</i>	7	4	4	15
	<i>Lycodon laoensis</i>		1		1
	<i>Oligodon fasciolatus</i>	1		2	3
	<i>Oligodon pseudotaeniatus</i>	2	1		3
	<i>Oligodon taeniatus</i>			1	1
	<i>Psammodynates pulverulentus</i>	1			1
	<i>Rhabdophis chrysargus</i>			1	1
	Gekkonidae	<i>Boiga multimaculata</i>		1	
<i>Dixoneus siamensis</i>		14	9	17	40
<i>Gehyra lacerata</i>		4	3	2	9
<i>Hemidactylus</i> sp.			6	14	20
Elapidae	<i>Bungarus candidus</i>		1	3	4
	<i>Calliophis maculiceps</i>	1			1
	<i>Naja siamensis</i>	1	1		2
Scincidae	<i>Eutropis macularia</i>	10	16	39	65
	<i>Lygosoma bowringii</i>	3	7	4	14
Typhlopidae	<i>Ramphotyphlops albiceps</i>		1		1
	<i>Ramphotyphlops braminus</i>	1	1		2
Viperidae	<i>Calloselasma rhodostoma</i>			1	1
Unknown	Unknown	16	5	9	30
Species		17	14	17	26
Grand total		70	62	110	242

### 4.1.3 Mortalities

Over the course of the study 12.9% (111 individuals) died either in the trap or from processing. However, morality rates fluctuated between forest type, trap type, and sample month (Table 4.5).

**Table 4.5** Reptile and amphibian trap mortalities from each forest type and for each month sampled.

Forest type	Month	Reptiles	Amphibians
DDF	May	1	0
	June	1	0
	September	0	0
	Total	2	0
HDF	May	3	23
	June	2	22
	September	0	3
	Total	5	48
PLE	May	2	2
	June	8	39
	September	1	4
	Total	11	45
Grand Total		18	93

HDF and PLE site had higher mortalities rates (14.2% and 15.5% respectively), when compared to the DDF sites (1.6%). Amphibians were more sensitive to trap related mortalities losing 93 individuals (15.0% of total), compared to just 18 reptiles (7.4% of total). Trap type also effected mortality rates with funnel traps proving more dangerous to animal safety with 57 mortalities compared to just eight.

### 4.1.4 Comparison of funnel and pitfall traps

Due to the plot design funnel traps accounted for a higher amount of trapping effort, and as a result caught a higher number of individuals. When

comparing trap efficiency (captured individuals/trap night), pitfall and funnel traps were similar for all herpetofauna; however when dividing reptile and amphibian captures, the results suggest that pitfall traps did not perform as well for reptiles than for amphibians (Table 4.6). Despite similar efficiencies in capturing individuals, funnel traps captured more amphibian and reptiles species, 14 and 24 species respectively, than pitfall traps, 6 and 12 species respectively. However, both trap types captured at least 1 unique species. I analyzed abundance, species richness, and diversity using the combined captures from both trap types to cover any biases in either method.

**Table 4.6** The number of amphibians and reptiles captured in the SBR from each forest type and the capture efficiency of each trap night.

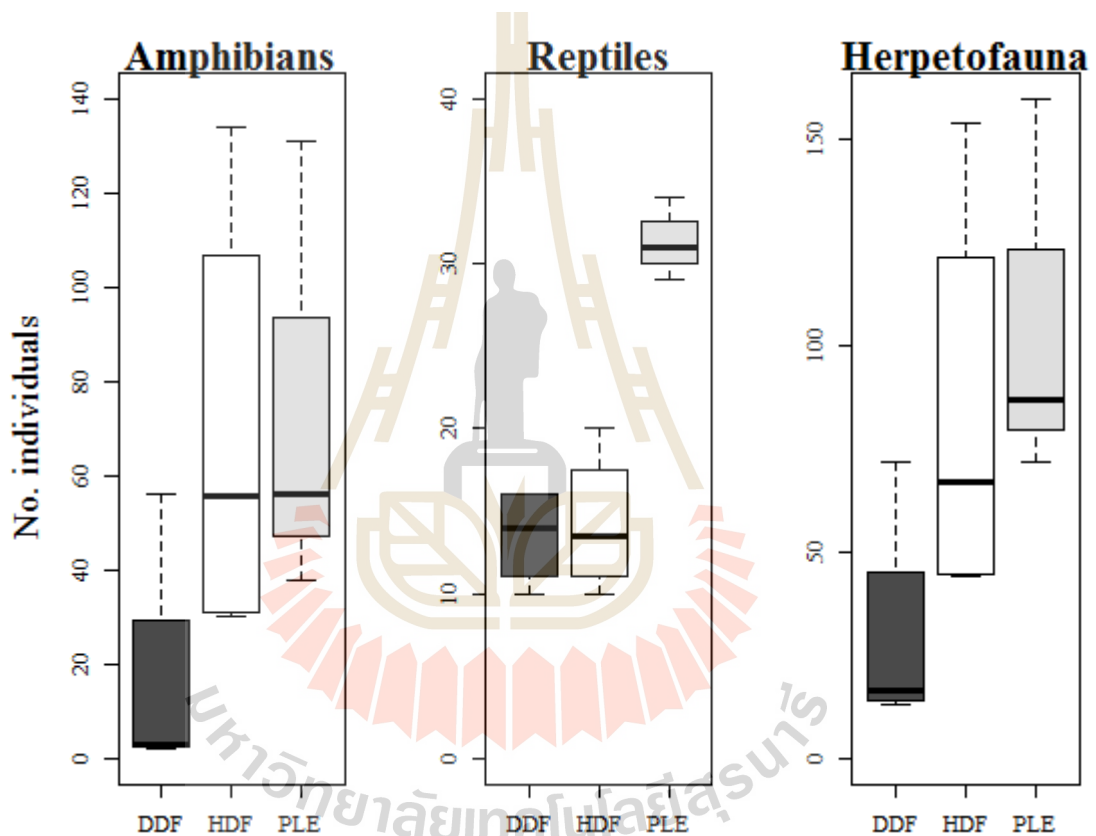
Trap Type	Forest Type	Amphibians		Reptiles		Total	
		Count	Efficiency (%)	Count	Efficiency (%)	Count	Efficiency (%)
Funnel	DDF	46	8.0	64	11.1	110	19.1
	HDF	242	44.8	55	10.2	297	55.0
	PLE	202	43.2	99	21.2	301	64.3
	Total	490	30.9	218	13.8	708	44.7
Pitfall	DDF	11	7.6	6	4.2	17	11.8
	HDF	69	51.1	7	5.2	76	56.3
	PLE	49	43.0	11	9.6	60	52.6
	Total	129	32.8%	24	6.1%	153	38.9%
Grand Total		619	31.3	242	12.2	861	43.6

#### 4.1.5 Abundances

Observed abundances varied across the three forest types under investigation for both amphibians and reptiles (Figure 4.2). Total herpetofaunal



abundances did not significantly vary by forest type (F-value = 3.146, df = 2,  $p$ -value = 0.0982). Additionally, complete amphibian abundances did not differ between forest types (F - value = 2.352, df = 2,  $p$ -value = 0.157), while reptile abundances did show a significant difference ( $\chi^2 = 6.0829$ , df = 2,  $p$ -value = 0.0478). Pairwise comparison results found that only HDF and PLE reptile abundances were significantly different ( $W = 0$ ,  $p$ -value = 0.04975). 2.352 0.157



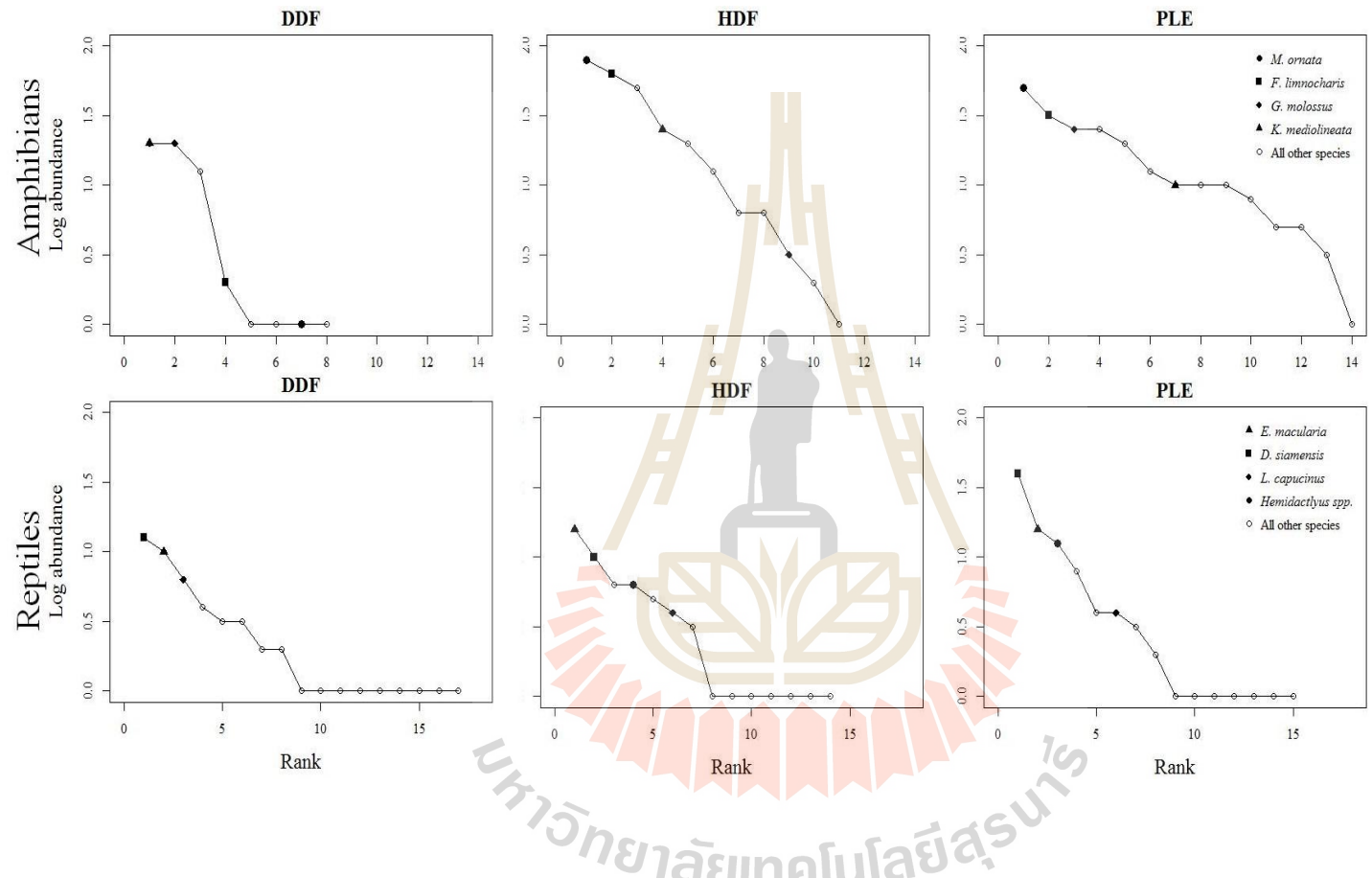
**Figure 4.2** A comparison of the observed abundances in each forest type for amphibians, reptiles, and all herpetofauna.

Reptile abundances differed significantly between forest types in the dry season ( $\chi^2 = 6.385$ , df = 2,  $p$ -value = 0.0411) and in the wet season ( $\chi^2 = 6.409$ , df = 2,  $p$ -value = 0.0406). Pairwise post hoc analysis revealed that over the dry season the dry dipterocarp did not differ significantly from the heterogeneous disturbed forest

( $W = 6$ ,  $p$ -value = 0.653), but both DDF and HDF significantly differed from eucalyptus plantation ( $W = 0$ ,  $p$ -value = 0.0477 and  $W = 12$ ,  $p$ -value = 0.0498 respectively). The pairwise comparison for reptile abundances in the wet season revealed that only DDF and PLE were significantly different ( $W = 0$ ,  $p$ -value = 0.0477), as HDF abundances did not differ significantly from DDF ( $W = 8$ ,  $p$ -value = 0.559) nor PLE ( $W = 0$ ,  $p$ -value = 0.0722).

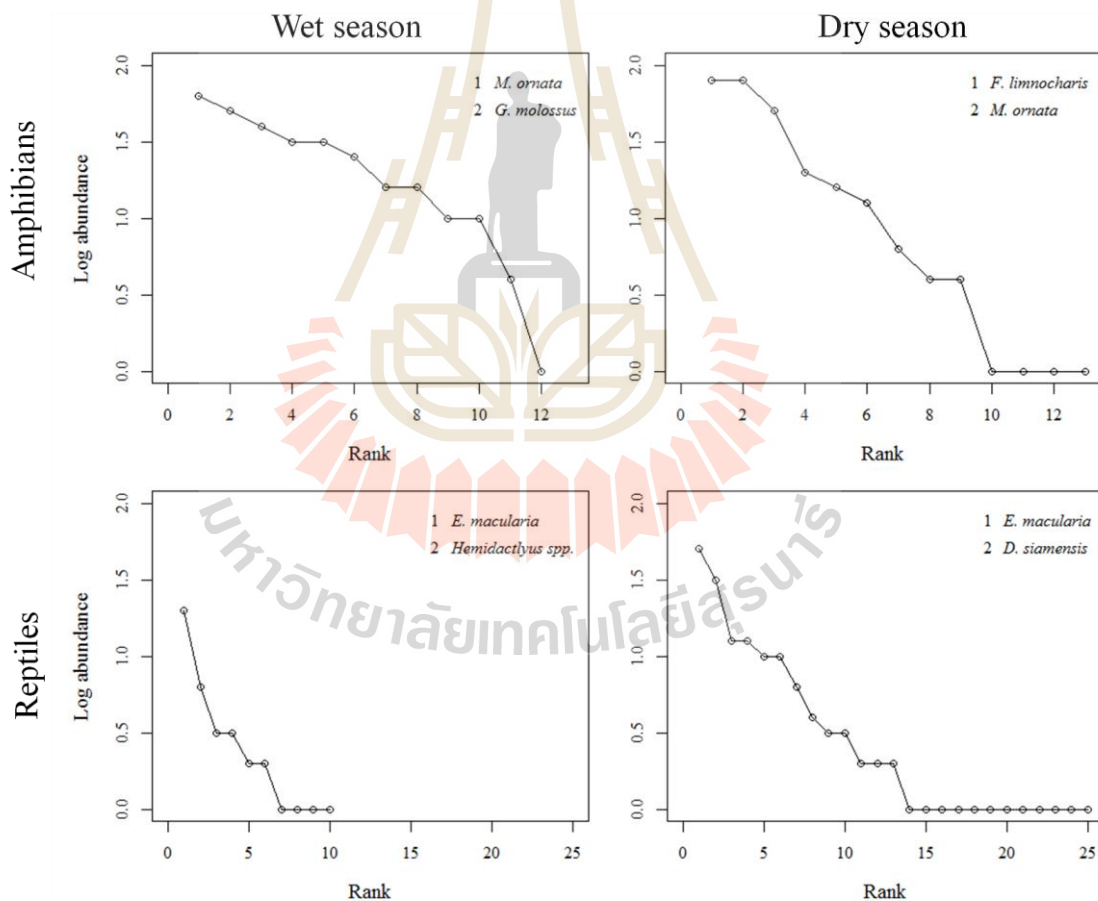
Amphibian abundances in the dry season varied significantly between forest types ( $\chi^2 = 7.616$ ,  $df = 2$ ,  $p$ -value = 0.0222), but did not differ significantly in the wet season ( $\chi^2 = 2.917$ ,  $df = 2$ ,  $p$ -value = 0.233). Dry season amphibian abundances were different between the DDF and both HDF and PLE ( $W = 0$ ,  $p$ -value = 0.0265 and  $W = 0$ ,  $p$ -value = 0.0436 respectively); however abundances were not different between HDF sites and PLE sites ( $W = 9$ ,  $p$ -value = 0.4).

The Whittaker plots generated for each forest type indicated that reptiles show high unevenness, with little difference in evenness between forest types (Figure 4.3). The two most abundant reptile species, *D. siamensis* and *E. macularia*, remained the same between forest types; however while in the HDF and PLE the *E. macularia* was the most common species the *D. siamensis* was more abundant in the DDF. The third most common species (or species complex) in both the PLE and HDF, *Hemidactylus* spp., did not occur at all within the DDF.



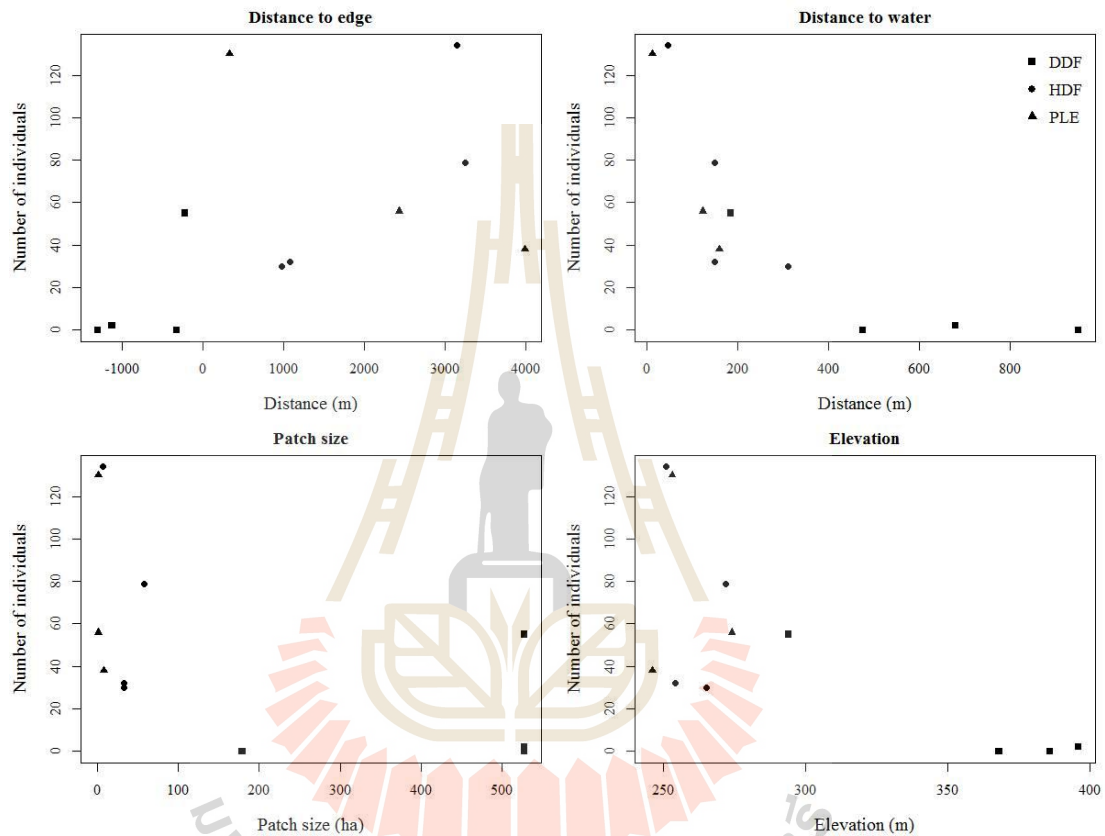
**Figure 4.3** Whittaker plots for complete amphibian captures (top row) and reptiles (bottom row) for DDF, PLE, and HDF

Amphibians in contrast show variation in the between DDF and both HDF and PLE. The two most abundant amphibian species, *M. fissipes* and *F. limnocharis*, in the PLE and HDF sites were identical, but two completely different species, *K. mediolineata* and *G. molossus*, dominated the DDF. Amphibians rank abundance changed seasonally, with the *F. limnocharis* moving from the top spot during the dry season to 7th in the wet season, while reptile rank abundances did not drastically vary between the two observed seasons (Figure 4.4).



**Figure 4.4** Seasonal rank abundance curves from reptile and amphibians species from all forest types combined.

Plotting amphibian abundance by different landscape factors did not reveal any potential correlations with the possible exception of distance to water (Figure 4.5). Reptile abundance did not show any potential trends when plotted by the same factors (Appendix 9)

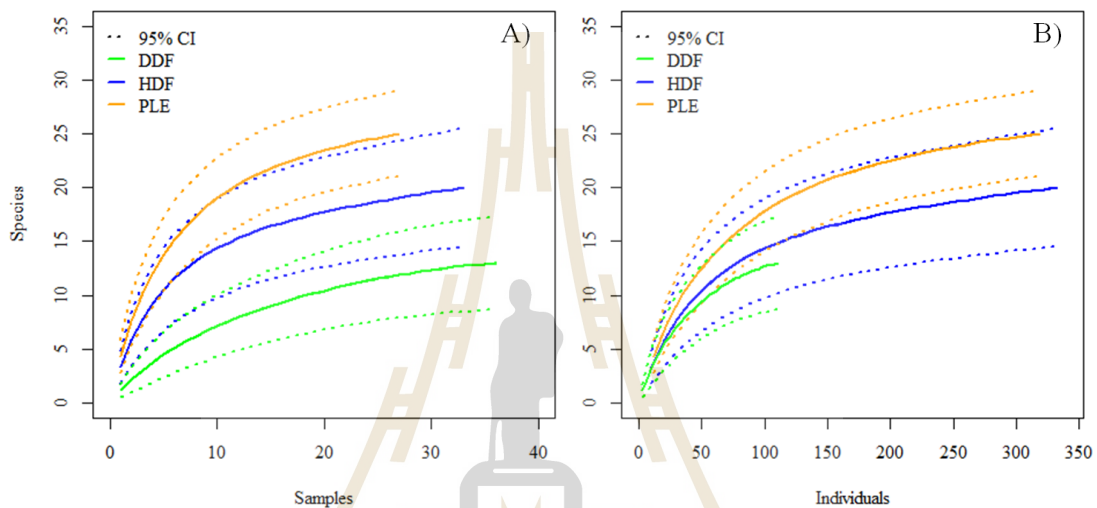


**Figure 4.5** Plot of amphibian abundance by distance to the edge of SBR, distance to water, patch size, and elevation.

#### 4.1.6 Species richness

Sampling completeness varied widely between sites for both amphibians and reptiles meaning that within forest types there was high variation between sites (Appendix 4). As DDF amphibian captures were primarily limited to a single site (DDF-4), the estimated richness was actually lower than the observed richness. The lower estimated richness occurred because we used sampling with replacement to

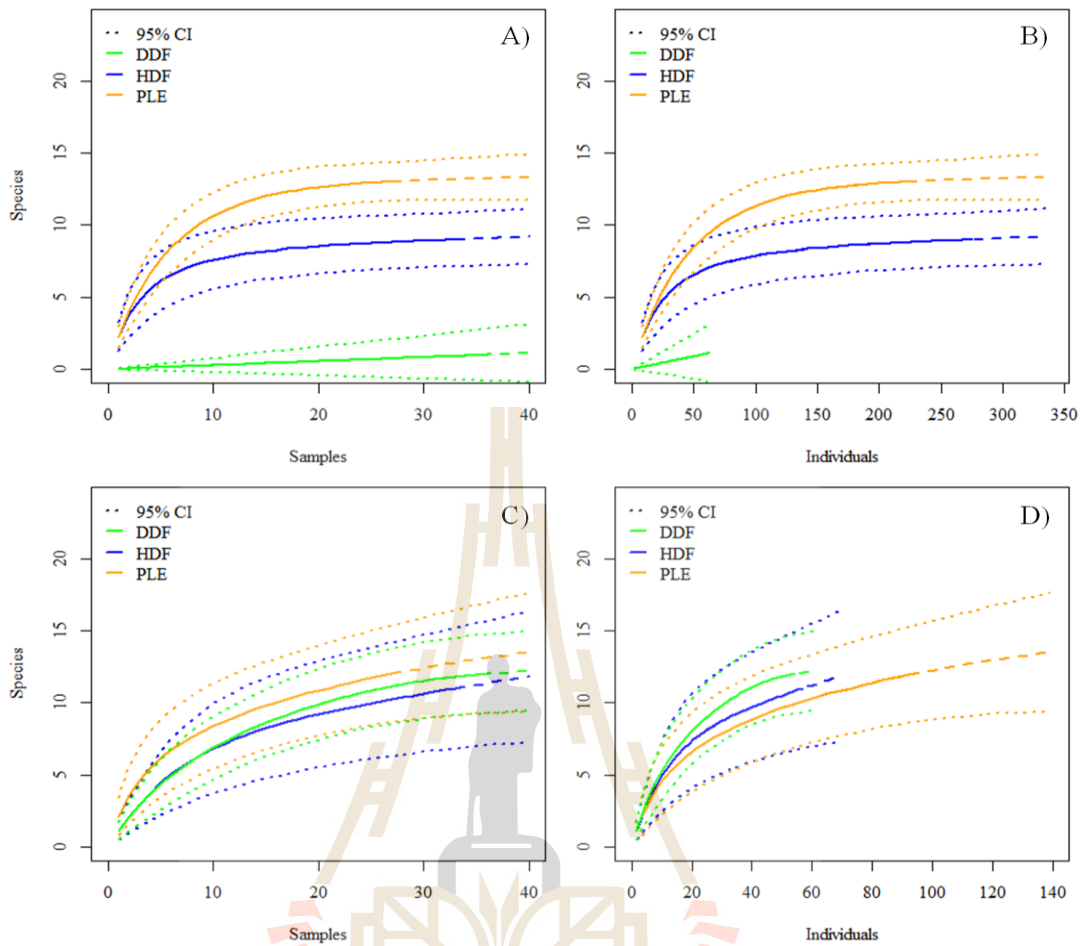
accurately determine confidence intervals. Additionally, sampling completeness also varied by season with more reptile species observed in May and June than in September. The trend shifted for amphibians in the DDF, with almost all species only observed in September.



**Figure 4.6** Sample and individual based rarefaction for amphibians (A and B) and for reptiles (C and D).

The extrapolated rarefaction curves for each forest type showed a significant difference in total herpetofauna species between PLE and DDF; however, HDF fell within the 95% confidence interval for both of the other two forest types (Figure 4.6). However, when correcting for the number of individuals using individual based rarefaction the significant differences were no longer apparent.

When comparing reptile species richness between forest types, no significant differences appeared; however the curves never reached an asymptote. For amphibian species richness the PLE showed significantly higher richness than the HDF, even when controlling for differences in the number of captured individuals (Figure 4.7).



**Figure 4.7** Sample and individual based rarefaction for amphibians (A and B) and for reptiles (C and D).

#### 4.1.7 Diversity

Diversity based on the Shannon-Weiner index differed significantly between forest types when considering all herpetofauna ( $\chi^2 = 6.4091$ ,  $df = 2$ ,  $p$ -value = 0.04058) and amphibians ( $\chi^2 = 8.2648$ ,  $df = 2$ ,  $p$ -value = 0.01604); however reptile diversity did not significantly vary between forest types ( $\chi^2 = 1.5455$ ,  $df = 2$ ,  $p$ -value = 0.4618) (Table 4.7). Pairwise comparison using Wilcoxon's rank sum test for all herpetofauna revealed that no significant differences existed between forests at the 95% confidence interval. When comparing amphibian diversity only DDF and PLE

were significantly different ( $W = 0$ ,  $p$ -value = 0.04975). Comparing the Simpson index between forest types showed only a significant difference when using all herpetofauna ( $\chi^2 = 6.1818$ ,  $df = 2$ , and  $p$ -value = 0.04546).

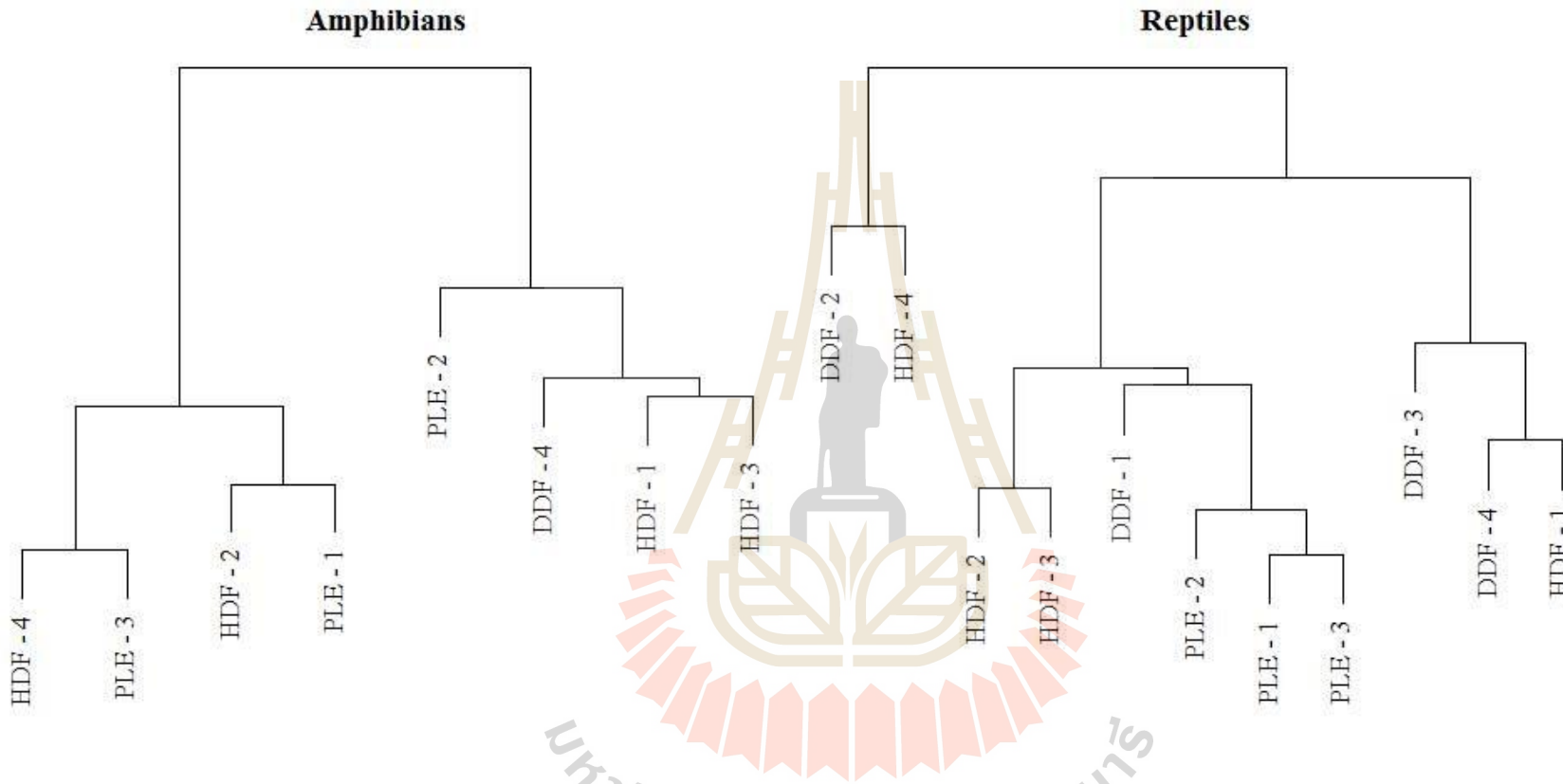
**Table 4.7** The calculated Shannon-Weiner and Simpson diversity indexes for each plot.

Plot	Amphibians		Reptiles		Total	
	Shannon-Weiner	Simpson	Shannon-Weiner	Simpson	Shannon-Weiner	Simpson
DDF - 1	0.00	1.00	1.72	0.78	1.72	0.78
DDF - 2	0.00	1.00	1.82	0.82	1.82	0.82
DDF - 3	0.69	0.50	1.42	0.72	1.75	0.79
DDF - 4	1.43	0.71	1.57	0.73	2.00	0.81
HDF - 1	1.66	0.80	1.63	0.78	2.24	0.88
HDF - 2	1.34	0.70	1.50	0.76	1.71	0.76
HDF - 3	1.47	0.67	2.08	0.86	2.31	0.84
HDF - 4	1.59	0.75	0.75	0.45	1.87	0.80
PLE - 1	1.89	0.79	1.79	0.78	2.53	0.89
PLE - 2	2.18	0.86	1.73	0.74	2.57	0.90
PLE - 3	1.91	0.82	1.74	0.76	2.50	0.89

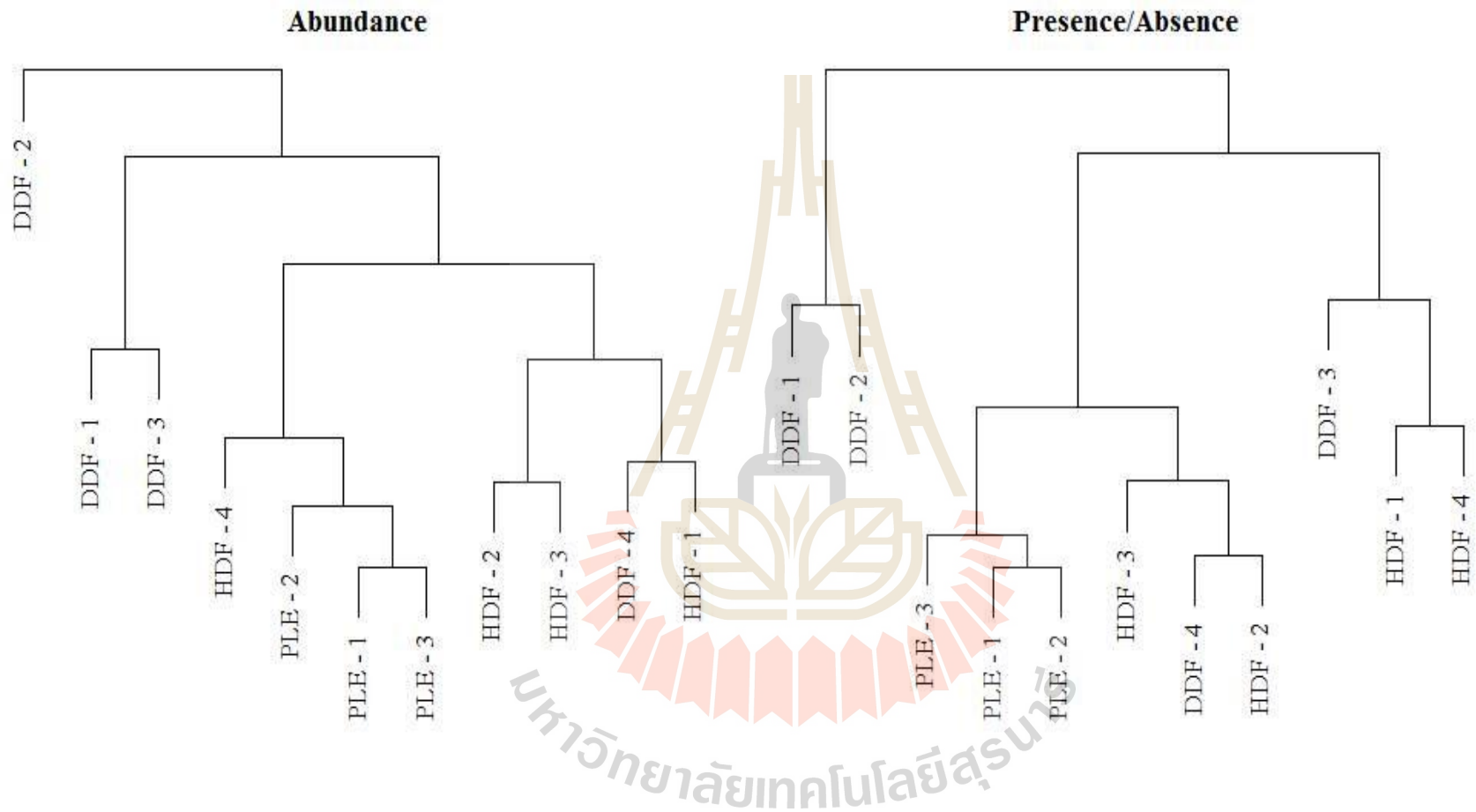
#### 4.1.8 Community structure

The Mantel test results showed that amphibian and total herpetofauna assemblages were spatially auto-correlated by plot ( $R = 0.5307$ ,  $p$ -value = 0.008 and  $R = 0.4728$ ,  $p$ -value = 0.002 respectively). Reptile communities however were not spatially auto-correlated ( $R = 0.1744$ ,  $p$ -value = 0.093). Spatial autocorrelation of amphibian communities could either indicate that the sampling sites were set to close together. To account for spatial autocorrelation, I created a new categorical variable called distance group. Distance groups were based on geographic distance between sites divided into 3 groups: far, edge, and forest.





**Figure 4.8** Comparison of hierarchic clustering of samples sites based on the recorded abundances of reptiles (right) and amphibians (left).

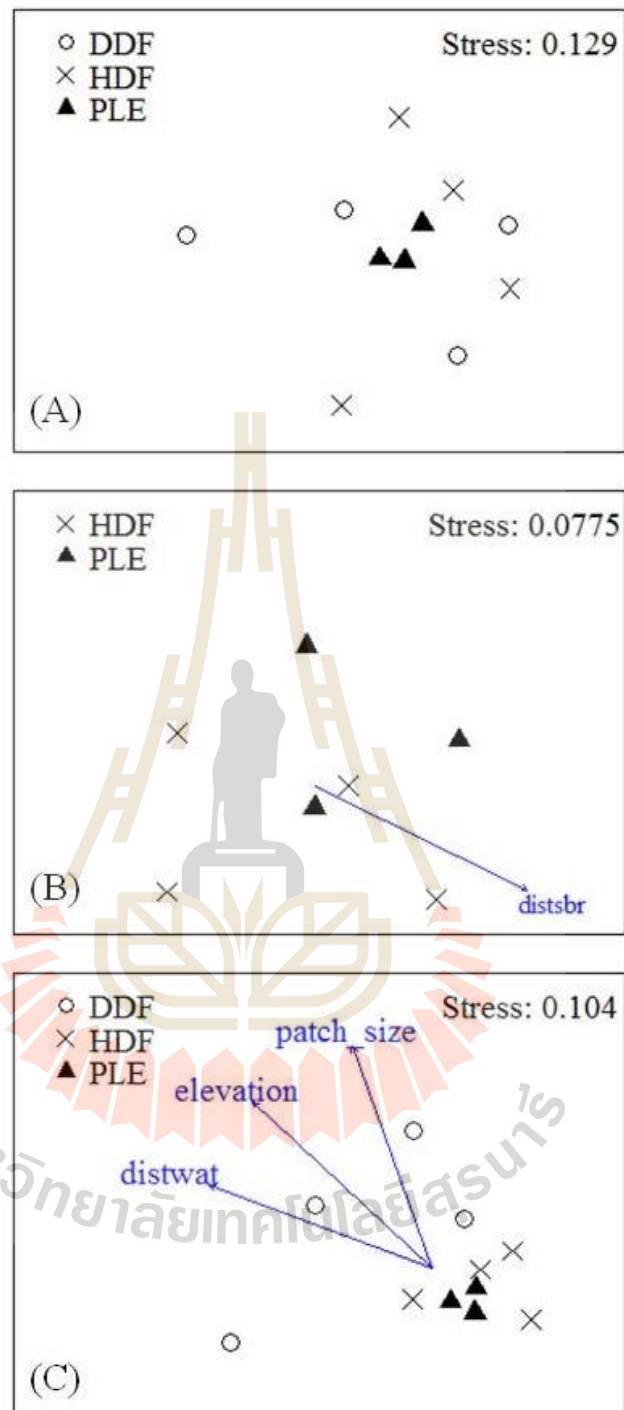


**Figure 4.9** Comparison of hierarchic clustering between abundance and presence absence data for all captured herpetofauna

Hierarchic clustering based on complete linkages showed that sites did not cluster based solely by forest type, and different patterns emerged when considering amphibians and reptiles separately (Figure 4.8). Additionally, clusters changed when using the abundance data and presence-absence data (Figure 4.9), which means that species abundances varied between sites and not just presence.

The results of NMDS on the abundance data did not show clustering of sites based on forest type. For both total herpetofaunal and reptile assemblages only the PLE sites showed a pattern of clustering on the NMDS. Since several sites had no recorded amphibian captures, only the PLE and HDF sites were used in the NMDS. Sites showed no clustering based on amphibian captures. The stress values for each NMDS plot all fell within the generally accepted range for good representation, meaning that the NMDS plots accurately displayed the resulting ordination in two dimensions.

The environmental fitting for each NMDS found that elevation, distance to water, and patch size were significant variables when analyzing all herpetofauna. Distance to SBR was the only significant variable for amphibians, while no variables fit for reptiles (Appendix 5). Only the environmental fitting for all herpetofauna captures showed that forest type was a significant factor in the NMDS, in addition to canopy cover (Appendix 6).



**Figure 4.10** Non-metric multidimensional scaling (N-MDS) for sites based on observed A) amphibians, B) reptiles, and C) herpetofauna with significant environmental variables.

Analysis of the beta dispersion for total, reptile, and amphibian captures, showed that only the total captures were significant different between forest types ( $F = 5.5083$  and  $p\text{-value} = 0.03132$ ). The results from the PERMANOVA found no significant difference between forest types, and no significance for any of the environmental variables used in the model with the exception of amphibians and distance grouping (Table 4.8). Additionally, the PERMANOVA for the presence absence data did not find any significant differences (Appendix 8).



**Table 4.8** PERMANOVA results modeling reptile, amphibian, and total captures based on forest type and other identified environmental factors.

	Variable	Df	Sum of Squares	Mean Squares	F-Model	R	<i>p</i> -value
Total	Distance group	2	0.99	0.50	2.27	0.44	0.11
	Forest type	2	0.20	0.10	0.46	0.09	0.91
	Distance to water	1	0.22	0.22	1.02	0.10	0.46
	Distance to SBR	1	0.06	0.06	0.25	0.02	0.96
	Canopy cover	1	0.28	0.28	1.26	0.12	0.37
	Elevation	1	0.21	0.21	0.96	0.09	0.55
	Patch size	1	0.08	0.08	0.36	0.04	0.92
	Residuals	1	0.22	0.22	0.10		
	Total	10	2.26	1.00			
	Reptile	Distance group	2	0.38	0.19	0.82	0.19
Forest type		2	0.20	0.10	0.43	0.10	0.95
Residuals		6	1.40	0.23	0.71		
Total		10	1.99	1.00			
Amphibian	Distance group	1	0.36	0.36	4.91	0.40	0.02
	Forest type	1	0.15	0.15	2.12	0.17	0.17
	Distance to water	1	0.13	0.13	1.80	0.15	0.22
	Distance to SBR	1	0.07	0.07	0.92	0.08	0.55
	Litter depth	1	0.10	0.10	1.43	0.12	0.32
	Residuals	1	0.07	0.07	0.08		
	Total	6	0.89	1.00			

The indicator species analysis based on abundance data identified five species as significantly contributing to forest type differences (Table 4.9). When using

only species presence absence, we found that only two species, *Bungarus candidus* and *Callouela guttalata*, were significant indicators. For all indicator species using both methods the test found them as indicators for the PLE forests.

**Table 4.9** Significant results from indicator species analysis using abundance and presence absence community data.

Data	Species	Forest type	Indicator value	<i>p</i> -value
Abundance	<i>Callouela guttalata</i>	PLE	0.93	0.029
	<i>Bungarus candidus</i>	PLE	0.80	0.041
	<i>Hemidactylus</i> spp.	PLE	0.73	0.035
	<i>Micryletta inornata</i>	PLE	0.67	0.039
	<i>Eutropis macularia</i>	PLE	0.67	0.042
Presence/Absence	<i>Callouela guttalata</i>	PLE	0.80	0.027
	<i>Bungarus candidus</i>	PLE	0.80	0.028
	<i>Micryletta inornata</i>	PLE	0.67	0.050

#### 4.1.9 Analysis of biometrics

From the 861 total captures we collected biometrics on 86 individuals, including 40 snakes and 48 individuals from the five species *D. siamensis*, *E. macularia*, *K. pulchra*, *K. mediolineata*, and *L. reevesii rubritaeniata*. We selected the three most abundant reptiles to create a BCI from the residuals of an ordinary least squares regression, *D. siamensis* (n = 15), *E. macularia* (n = 14) and *Lycodon capucinus* (n = 10). The SVL measurements for the *D. siamensis* was not normal ( $W = 0.5984$ ,  $p$ -value =  $2.463e-05$ ) and thus violated the assumptions for an ordinary least squares regression; however *E. macularia* and *L. capucinus* measurements fit the assumptions and were used to create a BCI. The results of an ANOVA on BCI to forest type found no significant difference for either *E. macularia* or *L. capucinus* (Table 4.10).

**Table 4.10** Results from the ANOVA of BCI by forest type for *E. macularia* and *L. capucinus*.

Species		Df	Sum Sq	Mean Sq	F-value	p-value
<i>E. macularia</i>	forest	2	0.6715	0.3358	1.344	0.301
	Residuals	11	2.7487	0.2499		
<i>L. capucinus</i>	forest	2	53.18	26.59	0.868	0.461
	Residuals	7	214.49	30.64		

We also analyzed whether forest type influenced fitness by assessing whether reptile individuals had lost their tails. The chi-square analysis for all reptiles showed that forest type did not influence tail loss ( $\chi^2 = 1.5353$ ,  $df = 2$ ,  $p$ -value = 0.4641). Looking at the two most abundant species, *D. siamensis* and *E. macularia*, also show that forest type and tail loss were independent ( $\chi^2 = 5.2724$ ,  $df = 2$ ,  $p$ -value = 0.07163 and  $\chi^2 = 0.2171$ ,  $df = 2$ ,  $p$ -value = 0.8971 respectively).

As snout-to-vent length and mass can vary widely between species, we did not conduct any tests on these metrics (Table 4.11 and Appendix C). From the 40 snakes processed, 30 were male compared to only 6 females, with 4 individuals too young to safely determine sex. The high male to female capture ratio pattern was observed across all forest types.

**Table 4.11** Processing information for all snakes captured during the study.

Forest type	Individuals	Species	Male:Female	Max SVL	Min SVL	Avg SVL
DDF	18	11	14:3	1266	84	445.5
HDF	8	4	7:1	1036	232	433.0
PLE	14	10	9:2	1070	144	415.2



## 4.2 Discussion

### 4.2.1 Abundance, species richness, and diversity

A single plot was responsible for over 90% of the amphibian captures in the DDF, which may be confounding the results for both the total abundance comparison and when comparing abundances specifically in the wet season. The low abundance of amphibians in the DDF, agree with similar results from several other studies such as Vonesh (2001) in which amphibian abundance was higher in disturbed forests and pine plantations than in primary forest. Additionally, Gardner et al. (2007b) found that amphibian abundances were actually highest in 4-5 year Eucalyptus plantations compared to primary and secondary forests in the Neotropics. However, several other studies arrived at different conclusions showing that plantations do not necessarily house higher numbers of amphibians (Folt and Reider, 2013; Kurz et al., 2014). However, open habitats, i.e. pastures, appear to show the greatest negative impact on amphibian abundance and diversity, which were not investigate in this study (Urbina-Cardona et al., 2006; Kurz et al., 2014). The species that were found in the DDF were also primarily burrowing frogs (*G. molossus* and *K. mediolineata*). As burrowing frogs, they spend a large portion of their time underground and potentially face less risk of desiccation in dry habitats such as the DDF (Suazo-Ortuño et al., 2008). All three of the species *G. molossus*, *K. pulchra*, and *K. mediolineata* abundances showed that they peaked in sites at the edge of the protected area and dropped in sites farther away from the edge. In contrast the most abundant amphibian species found in the HDF and PLE sites, *F. limnocharis* and *M. fissipes*, displayed the opposite trend.

The similarity in reptile species richness across all forest types supports the hypothesis that reptiles are not as sensitive to fragmentation as other taxa, and that

some species even thrive in partially disturbed habitats (Wanger et al., 2010). Canopy heterogeneity provides reptiles with a plethora of basking sites, and some studies suggests that at least insect prey communities are positive impacted in disturbed habitats (Heliölä et al., 2001; Suazo-Ortuño et al., 2008). However, our results also show that the slopes of the individual based rarefaction for reptiles did not reach an asymptote and that both the DDF and HDF had higher species richness when accounting for the number of individuals. More samples could provide a better comparison between forest types as the accumulation curve levels off.

The lack of significant difference in pairwise comparison of diversity between forest types potentially derives from an issue in sample size. The variation between sites in the same forest type was high which could be overcome with a larger sample size; however passive trapping arrays are time intensive and expensive to create.

The results from the MCA of habitat variables does not necessarily reflect that all forest types comprised similar habitats. One potential issue was the correlation between variables which violates the assumptions of MCA. Additionally since we collected the data as categorical values, small variations between forest types may have gone unnoticed.

#### **4.2.2 Community structure**

The drastic difference in amphibians between the majority of the DDF sites and the other sites created separation when comparing the total herpetofaunal community. The lack of any significant environmental variables for reptiles suggests that other factors are contributing to the difference or that no true differences exist on the limited spatial and/or temporal scale. However, the NMDS shows that the PLE

sites formed a tight grouping, but may overlap with the other forest types in a way that obscures the clustering.

Clustering for amphibians appears to occur based on the spatial distance between sites. As observed by Kurz et al. (2014) the habitat type along the edge of a forest did not influence herpetofaunal communities with the exception of forest to pasture. The sites near the edge, may house more species as individuals can still benefit from the primary forest. Further study in the area is needed to isolate the potential edge effect and the effective distance that supports amphibian communities.

#### **4.2.3 Biometrics**

While tail loss did not show a significant result at the 95% it did at the 90% indicating that a larger sample size may reveal a true significant difference. Sung et al. results support that reptiles in disturbed habitats may face increased predation pressure as evidenced through tail loss. Additionally comparing a BCI for the most abundant species does not necessarily reflect how habitat influences body condition as highly abundant species are typically generalists and are not as sensitive to habitat degradation. An assessment of rare species would require a longer time scale to achieve an acceptable sample size, but could provide contrasting information.

## CHAPTER V

### CONCLUSION

As a preliminary study on the effects disturbance on herpetofauna, this study provides useful results for establishing continued study in the Sakaerat Biosphere Reserve. The high mortality rates throughout the dry season suggest that sampling with passive trapping may not be ethical in all seasons. Disturbed habitats that experience high temperatures and low humidity may require different sampling procedures, such as only opening traps at night.

Our results also confirmed the assertion that reptiles and amphibians do not show the same patterns with regard to disturbance. Abundance, species richness, and diversity comparisons clearly showed that these two groups should not be combined when considering their response to an environmental gradient. Reptiles appeared less sensitive to habitat differences, while amphibians showed a high degree of difference.

The study also documented that amphibian abundance and richness was very low in the protected forest, which contrasts with findings in other studies. More research is required to determine what environmental factors may influence the low number of amphibians in the DDF, and whether seasonal shifts occur.

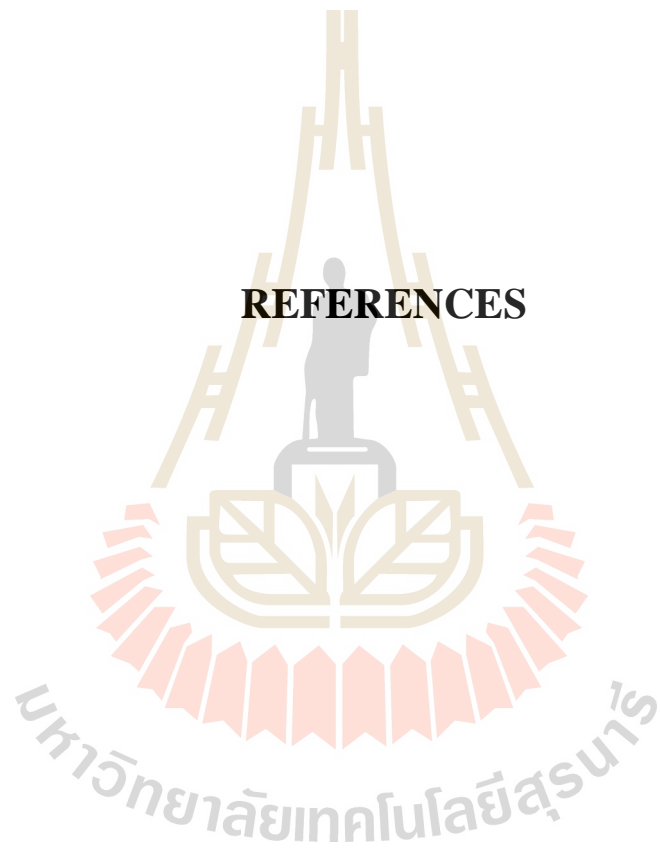
The biometrics of several abundant species did not reveal any significant differences between forest types. However, studying the most abundant species does not necessarily provide a clear indicator for habitat quality. Species that are abundant across all forest types, are generalists and should not show any evidence in reduced

body condition along a disturbance gradient. Future studies should investigate body condition for species that are present, but not necessarily common in all forest types to assess the impact of disturbance.

Our study did not find any significant differences in community structure between forest types using ordination (non-metric multidimensional scaling) nor with PERMANOVA. Additionally, continued sampling over several years and season would improve detection of rare species. Addressing both the temporal and spatial aspects will aid in more accurate ordination and multivariate hypothesis testing.

While some of the results suggest that there are significant differences between the studied forest types, the study did not provide enough conclusive evidence to completely explain how herpetofauna communities are effected by habitat disturbance in the SBR. One major issue that we encountered was that the forest type categories we created do not necessarily work as a disturbance gradient. We suggest that future studies focus on a single forest type and identify different factors, such as patch size, management activities, and distance to water, that influence herpetofauna communities.

## REFERENCES



## REFERENCES

- Abdi, H., and Valentin, D. (2007). Multiple correspondence analysis (pp 651-657). In . Salkind, N. J. (ed). Encyclopedia of Measurement and Statistics. Sage.
- Anderson, M., and Walsh, D. (2013). PERMANOVA, ANOSIM, and the Mantel test in the face of heterogeneous dispersions: What null hypothesis are you testing? Ecological Monographs. 83(4): 557–574.
- Bickford, D., Howard, S. D., Ng, D. J. J., and Sheridan, J. (2010). Impacts of climate change on the amphibians and reptiles of Southeast Asia. Biodiversity and Conservation. 19(4): 1043–1062.
- Blouin-Demers, G. (2003). Precision and accuracy of body-size measurements in a constricting, large-bodied snake (*Elaphe obsoleta* ). Herpetological Review. 34(4): 2–5.
- Böhm, M., Collen, B., Baillie, J. E. M., Bowles, P., Chanson, J., Cox, N., Hammerson, G., Hoffmann, M., Livingstone, S., Ram, M., et al.. (2013). The conservation status of the world's reptiles. Biological Conservation. 157: 372–385.
- Brooks, T., Mittermeier, R., Mittermeier, C., Da Fonseca, G., Rylands, A., Konstant, W., Flick, P., Pilgrim, J., Oldfield, S., Magin, G., and Hilton-Taylor, C. (2002). Habitat loss and extinction in the hotspots of biodiversity. Conservation Biology. 16(4): 909–923.

- Chao, A. (1984). Nonparametric estimation of the number of classes in a population. *Scandinavian Journal of Statistics*. 265–270.
- Charuphat T (2000) Remote sensing and GIS for tropical forest management. In: GIS Application Center (ed) Proceedings of the Ninth Regional Seminar on Earth Observation for Tropical Ecosystem Management, Khao Yai, Thailand, 20–24 November 2000. National Space Development Agency of Japan, Remote Sensing Technology Center of Japan, Royal Forest Department, and GIS Application Center/AIT, Khao Yai National Park, Thailand, 42–49.
- Cushman, S. (2006). Effects of habitat loss and fragmentation on amphibians: A review and prospectus. *Biological Conservation*. 128: 231–240.
- DeFries, R., and Hansen, A. (2005). Increasing isolation of protected areas in tropical forests over the past twenty years. *Ecological Applications*. 15(1): 19–26.
- Dirzo, R., and Raven, P. H. (2003). Global state of biodiversity and loss. *Annual Review of Environment and Resources*. 28(1): 137–167.
- Folt, B., and Reider, K. E. (2013). Leaf-litter herpetofaunal richness, abundance, and community assembly in mono-dominant plantations and primary forest of northeastern Costa Rica. *Biodiversity and Conservation*. 22(9): 2057–2070.
- Food and Agriculture Organization of the United Nations (FAO). (2010). *Global Forest Resources Assessment 2010*. Food and Agriculture Organization.
- Fox, J., and Vogler, J. (2005). Land-use and land-cover change in montane mainland Southeast Asia. *Environmental Management*. 36(3): 394–403.
- Fredericksen, N. J., and Fredericksen, T. S. (2004). Impacts of selective logging on amphibians in a Bolivian tropical humid forest. *Forest Ecology and Management*. 191(1-3): 275–282.



- Gardner, T., Barlow, J., and Peres, C. (2007a). Paradox, presumption and pitfalls in conservation biology: The importance of habitat change for amphibians and reptiles. *Biological Conservation*. 138: 166–179.
- Gardner, T., Ribeiro-Júnior, M. A., Barlow, J., Avila-Pires, T. C. S., Hoogmoed, M. S., and Peres, C. (2007b). The value of primary, secondary, and plantation forests for a neotropical herpetofauna. *Conservation Biology*. 21(3): 775–787.
- Gibbon, J. W., Scott, D. E., Ryan, T. J., Buhlmann, K. A., Tuberville, T. D., Metts, B. S., Greene, J., Mills, T., Leiden, Y., Poppy, S., and Winne, C. T. (2000). The global decline of reptiles, déjà vu amphibians. *BioScience*. 50(8): 653–666.
- Harrison, R. D. (2011). Emptying the forest: hunting and the extirpation of wildlife from tropical nature reserves. *BioScience*. 61(11): 919–924.
- Heliölä, J., Koivula, M., and Niemelä, J. (2001). Distribution of carabid beetles (Coleoptera, Carabidae) across a boreal forest–clearcut ecotone. *Conservation Biology*. 15(2): 370–377.
- Husson, F., Josse, J., Le, S., and Mazet, J.. (2016). *FactoMineR: Multivariate Exploratory Data Analysis and Data Mining*. R package version 1.31.5. <https://CRAN.R-project.org/package=FactoMineR>.
- International Union for Conservation of Nature. (2014). IUCN Red List of Threatened Species. Retrieved February 19, 2014, (Online). Available: [www.iucnredlist.org](http://www.iucnredlist.org)
- Karns, D., Murphy, J., and Voris, H. (2010). Semi-aquatic snake communities of the central plain region of Thailand. *Tropical Natural History*. 10(April): 1–25.
- Kindt, R. and Coe, R. (2005) *Tree diversity analysis. A manual and software for common statistical methods for ecological and biodiversity studies*. World Agroforestry Centre (ICRAF), Nairobi.

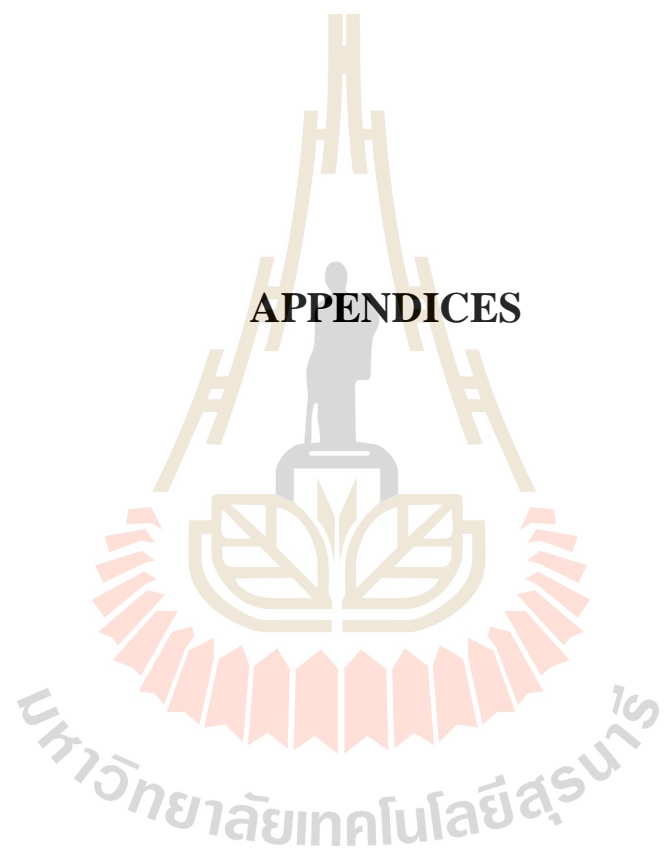
- Kurz, D. J., Nowakowski, A. J., Tingley, M. W., Donnelly, M., and Wilcove, D. S. (2014). Forest-land use complementarity modifies community structure of a tropical herpetofauna. *Biological Conservation*. 170: 246–255.
- Lamotte, S., Gajasen, J., and Malaisse, F. (1998). Structure diversity in three forest types of north-eastern Thailand (Sakaerat Reserve, Pak Tong Chai). *Biotechnology Agronomy Society and Environment*. 2(3): 192–202.
- Manning, C. D., Raghavan, P., and Schutze, H. (2008). *Introduction to Information Retrieval*. Cambridge University Press, Cambridge.
- McCune, B., Grace, J.B. and Urban, D.L., (2002). *Analysis of Ecological Communities*. Vol. 28: MjM software design.
- McLeod, D. (2008). A new species of big-headed, fanged dicroglossine frog (Genus *Limnonectes*) from Thailand. *Zootaxa*. 1807: 26–46.
- Oksanen, J. (2013). Multivariate analysis of ecological communities in R: vegan tutorial. In *R Documentation* (p. 43).
- Oksanen J., Blanchet, F. G., Kindt. R., Legendre P., Minchin, P. R., O'Hara R.B., Simpson, G., Solymos, P., Stevens H., and Wagner, H. (2016). *vegan: Community Ecology Package*. R package version 2.3-0. <http://CRAN.R-project.org/package=vegan>.
- Peig, J., and Green, A. J. (2009). New perspectives for estimating body condition from mass/length data: The scaled mass index as an alternative method. *Oikos*. 118(12): 1883–1891.
- Poo, S., and Bickford, D. P. (2014). Hatching plasticity in a Southeast Asian tree frog. *Behavioral Ecology and Sociobiology*. 68(11): 1733–1740.

- R Core Team (2014). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL <http://www.R-project.org/>.
- Reading, C., and Luiselli, L. (2010). Are snake populations in widespread decline? *Biology Letters*. 6: 777–780.
- Ribeiro-Júnior, M., Gardner, T. A., and Ávila-pires, T. C. (2008). Evaluating the effectiveness of herpetofaunal sampling techniques across a gradient of habitat change in a tropical forest landscape. *Journal of Herpetology* 42(4): 733–749.
- Rodrigues, A. S., Andelman, S. J., Bakarr, M. I., Boitani, L., Brooks, T. M., Cowling, R. M., Fishpool, L. C., Da Fonseca, G. B., Gaston, K. J., Hoffmann, M., et al. (2004). Effectiveness of the global protected area network in representing species diversity. *Nature*. 428(6983): 640 – 643.
- Sakaerat Environmental Research Station. (2015). Retrieved March 15, 2015, (Online): Available: <http://www.tistr.or.th/sakaerat/SakaeratE/index.php>.
- Sánchez-Azofeifa, G. A., Quesada-Mateo, C., Gonzalez-Quesada, P., Dayanandan, S., and Bawa, K. S. (1999). Protected areas and conservation of biodiversity in the tropics. *Conservation Biology*. 13(2): 407–411.
- Schulte-Hostedde, A., Zinner, B., Millar, J., and Hickling, G. (2005). Restitution of mass-size residuals: validating body condition indices. *Ecology*. 86(1): 155–163.
- Setser, K. (2007). Use of anesthesia increases precision of snake length measurements. *Herpetological Review*. 38(4): 2005–2007.
- Sodhi, N. S., Posa, M. R. C., Lee, T. M., Bickford, D., Koh, L. P., and Brook, B. W. (2010). The state and conservation of Southeast Asian biodiversity. *Biodiversity and Conservation*. 19(2): 317–328.

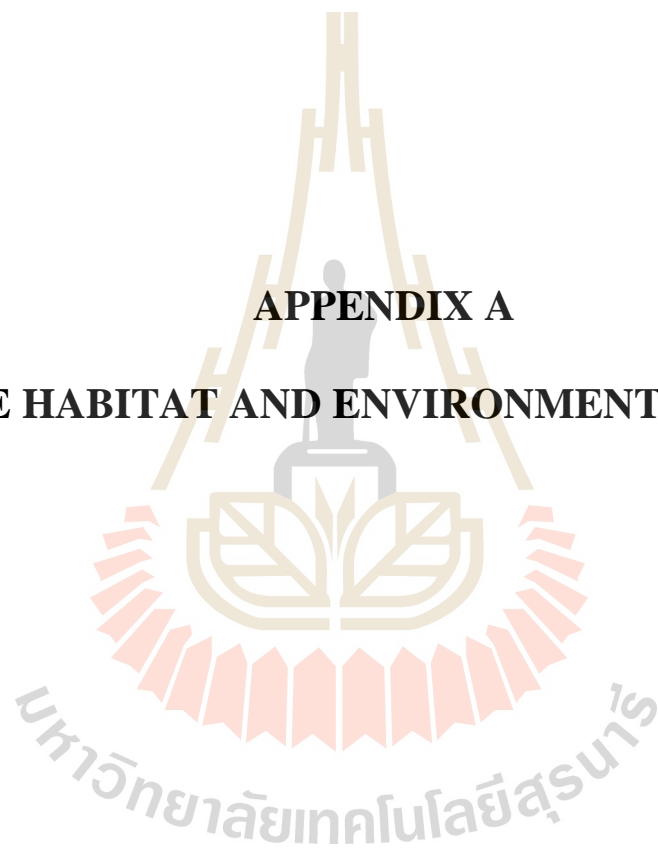
- Strine, C. T., Silva, I. M. S., Crane, M. S., Bartosz, N., Artchawakom, T., Goode, M., and Suwanwaree, P. (2014). Mortality of a wild king cobra, *Ophiophagus hannah* Cantor, 1836 (Serpentes: Elapidae) from Northeast Thailand after ingesting a plastic bag. *Asian Herpetological Research*. 5(4): 284–286.
- Stuart, S., Chanson, J., and Cox, N. (2004). Status and trends of amphibian declines and extinctions worldwide. *Science*. 306(5702): 1783–1786.
- Suazo-Ortuño, I., Alvarado-Díaz, J., and Martínez-Ramos, M. (2008). Effects of conversion of dry tropical forest to agricultural mosaic on herpetofaunal assemblages. *Conservation Biology*. 22(2): 362–74.
- Sung, Y., Karraker, N. E., and Hau, B. C. H. (2012). Terrestrial herpetofaunal assemblages in secondary forests and exotic *Lophostemon confertus* plantations in South China. *Forest Ecology and Management*. 270: 71–77.
- Sutthivanich, I., and Ongsomwang, S. (2015). Evaluation on landscape change using remote sensing and landscape metrics: A case study of Sakaerat Biosphere Reserve (SBR), Thailand. *International Journal of Environmental Science and Development*. 6(3): 182.
- Tongyai, P. (1980). The Sakaerat environmental research station. Thailand MAB Committee, Bangkok.
- Trimble, M., and Aarde, R. van. (2012). Geographical and taxonomic biases in research on biodiversity in human-modified landscapes. *Ecosphere*. 3(12): 1–16.
- Urbina-Cardona, J. N., Olivares-Pérez, M., and Reynoso, V. H. (2006). Herpetofauna diversity and microenvironment correlates across a pasture–edge–interior ecotone in tropical rainforest fragments in the Los Tuxtlas Biosphere Reserve of Veracruz, Mexico. *Biological Conservation*. 132(1): 61–75.

- Venable, W. N. and Ripley, B. D. (2002) *Modern Applied Statistics with S*. Fourth Edition. Springer, New York.
- Vonesh, J. R. (2001). Patterns of richness and abundance in a tropical African leaf-litter herpetofauna. *Biotropica*. 33(3): 502–510.
- Voris, H. (2006). Assessment of biodiversity among Southeast Asian amphibians and reptiles. *The Natural History Journal of Chulalongkorn University*. 6(May): 1–10.
- Wanger, T. C., Iskandar, D. T., Motzke, I., Brook, B. W., Sodhi, N. S., Clough, Y., and Tscharrntke, T. (2010). Effects of land-use change on community composition of tropical amphibians and reptiles in Sulawesi, Indonesia. *Conservation Biology*. 24(3): 795–802.
- Winne, C. T., Willson, J. D., and Andrews, K. M. (2006). Efficacy of marking snakes with disposable medical cautery units. *Herpetological Review*. 37(1): 52–54.
- Zuur, A. F., Ieno, E. N., and Smith, G. M. (2007). *Analyzing Ecological Data*. Springer.

**APPENDICES**



**APPENDIX A**  
**SITE HABITAT AND ENVIRONMENTAL DATA**





**Appendix A-1** Photographs of each dry dipterocarp forest site.







**Appendix A-2** Photographs of each highly disturbed forest site.





**Appendix A-3** Photographs of each eucalyptus plantation site.



**Appendix A-4** Daily weather conditions for the Sakaerat Biosphere Reserve in May, June, and September of 2015 for the all days sampled including the day traps were opened.

Date	Max temperature (°C)	Min temperature (°C)	Average temperature (°C)	Relative Humidity (%)	Rain (mm)
15/05/2015	35.25	24.75	25.35	75	0
16/05/2015	36	25.35	25.95	74.5	0
17/05/2015	35.6	26.05	26.65	73	0
18/05/2015	36.55	25.3	25.9	74	0
25/05/2015	35.75	26.4	27	72	0
26/05/2015	35.75	25.95	26.55	72.5	0
27/05/2015	35.55	25.8	26.4	75	0
28/05/2015	36.8	25.8	26.4	73.5	0
17/06/2015	31.35	23.5	24.1	83	0
18/06/2015	31.4	23.85	24.45	80	0
19/06/2015	33.75	24.1	24.7	77	0
20/06/2015	34.65	24.05	24.65	74.5	0
23/06/2015	30.8	25.05	25.65	76.5	2.35
24/06/2015	31.8	24.8	25.4	78.5	0
25/06/2015	32.05	25.35	25.95	75	0
26/06/2015	29.45	25.05	25.65	74	0
17/09/2015	25.8	23.2	23.8	92.5	38.45
18/09/2015	26.65	22.65	23.25	94	1.65
19/09/2015	31.2	22.25	22.85	93	0
20/09/2015	32.75	22.45	23.05	91	0
21/09/2015	32.8	23.6	24.2	88	28.55
22/09/2015	30.4	22.7	23.3	91.5	52.4
23/09/2015	30.15	22.6	23.2	91.5	0
24/09/2015	30.8	23.65	24.25	86	0

**Appendix A-4** Monthly weather conditions for May, June, and September at the Sakaerat Environmental Research Station.

Month	Temperature (°C)			Relative Humidity (%)	Rain (mm)
	Max	Min	Average		
May	36.0	25.8	26.4	74.8	8.6
June	34.2	25.0	25.6	75.7	79.3
September	30.9	23.4	24.0	84.3	259.9

**Appendix A-5** Ground cover assessment for each site.

Plot	Vegetative ground cover		
	Vegetative ground cover	Bare ground coverage	Leaf litter coverage
DDF-1	Light (16-25%)	Light (16-25%)	Medium (25-65%)
DDF-2	Medium (25-65%)	Medium (25-65%)	Very light (1-15%)
DDF-3	Medium (25-65%)	Very light (1-15%)	Medium (25-65%)
DDF-4	Light (16-25%)	Medium (25-65%)	Very light (1-15%)
HDF-1	Medium (25-65%)	Very light (1-15%)	Very light (1-15%)
HDF-2	Medium (25-65%)	Medium (25-65%)	Very light (1-15%)
HDF-3	Medium (25-65%)	Medium (25-65%)	None (0%)
HDF-4	Medium (25-65%)	Light (16-25%)	Very light (1-15%)
PLE-1	Medium (25-65%)	Light (16-25%)	Very light (1-15%)
PLE-2	Light (16-25%)	Medium (25-65%)	Medium (25-65%)
PLE-3	Medium (25-65%)	Medium (25-65%)	Very light (1-15%)

**Appendix 10** Canopy and vegetation factors for each plot site.

Plot	Canopy cover	Groundstory density	Understory density	Midstory density	Abovestory density
DDF-1	Heavy (66-80%)	Medium (25-65%)	Very light (1-15%)	Very light (1-15%)	Medium (25-65%)
DDF-2	Heavy (66-80%)	Heavy (66-80%)	None (0%)	Very light (1-15%)	Medium (25-65%)
DDF-3	Medium (25-65%)	Heavy (66-80%)	Very light (1-15%)	Very light (1-15%)	Light (16-25%)
DDF-4	Medium (25-65%)	Light (16-25%)	Very light (1-15%)	Very light (1-15%)	Very light (1-15%)
HDF-1	Medium (25-65%)	Heavy (66-80%)	Very light (1-15%)	Very light (1-15%)	Very light (1-15%)
HDF-2	Medium (25-65%)	Light (16-25%)	Light (16-25%)	Medium (25-65%)	Light (16-25%)
HDF-3	Very light (1-15%)	Light (16-25%)	Very light (1-15%)	None (0%)	None (0%)
HDF-4	Medium (25-65%)	Medium (25-65%)	Very light (1-15%)	Medium (25-65%)	Very light (1-15%)
PLE-1	Very light (1-15%)	Medium (25-65%)	Very light (1-15%)	Very light (1-15%)	Very light (1-15%)
PLE-2	Very light (1-15%)	Medium (25-65%)	Light (16-25%)	Very light (1-15%)	None (0%)
PLE-3	Light (16-25%)	Very light (1-15%)	Very light (1-15%)	Very light (1-15%)	Very light (1-15%)



**APPENDIX B**

**RICHNESS, ABUNDANCE, AND COMMUNITY DATA**

**Appendix B-1** Sampling completeness for trapping site including observed richness and estimated richness using Chao1 method.

Family	Forest type	Observed richness	Chao1 richness	Completeness	
Amphibians	DDF - 1	0	7.73	0.0%	
	DDF - 2	0	7.73	0.0%	
	DDF - 3	2	7.73	25.9%	
	DDF - 4	8	7.73	103.5%	
	HDF - 1	6	10.09	59.5%	
	HDF - 2	5	10.09	49.6%	
	HDF - 3	7	10.09	69.4%	
	HDF - 4	9	10.09	89.2%	
	PLE - 1	10	13.49	74.1%	
	PLE - 2	13	13.49	96.4%	
	PLE - 3	9	13.49	66.7%	
	Reptiles	DDF - 1	7	16.74	41.8%
		DDF - 2	7	16.74	41.8%
DDF - 3		5	16.74	29.9%	
DDF - 4		7	16.74	41.8%	
HDF - 1		6	12.63	47.5%	
HDF - 2		5	12.63	39.6%	
HDF - 3		9	12.63	71.3%	
HDF - 4		3	12.63	23.8%	
PLE - 1		9	14.3	62.9%	
PLE - 2		9	14.3	62.9%	
PLE - 3		9	14.3	62.9%	

**Appendix B-2** Monthly abundance, species, and reptile tail loss at each sample site.

Month	Plot	Amphibians		Reptiles		Reptile tail loss	
		Abundance	Species	Abundance	Species		
May	DDF - 1	0	0	4	4	0	
	DDF - 2	0	1	4	3	0	
	DDF - 3	0	0	3	2	0	
	DDF - 4	0	0	5	3	0	
	HDF - 1	5	3	7	4	0	
	HDF - 2	3	3	4	2	0	
	HDF - 3	0	0	6	4	0	
	HDF - 4	55	7	11	3	1	
	PLE - 1	6	5	13	6	0	
	PLE - 2	0	1	7	4	0	
	PLE - 3	4	3	10	5	2	
	PLE - 4	0	0	6	4	0	
	Total		73	11	80	21	3
June	DDF - 1	0	0	9	4	1	
	DDF - 2	0	0	7	6	0	
	DDF - 3	1	1	6	4	0	
	DDF - 4	0	1	6	2	0	
	HDF - 1	13	3	4	4	0	
	HDF - 2	45	5	5	4	0	
	HDF - 3	10	4	7	5	0	
	HDF - 4	79	8	9	2	0	
	PLE - 1	9	4	10	5	0	
	PLE - 2	7	4	14	6	0	
	PLE - 3	29	6	13	6	0	
	Total		193	11	90	17	1
	September	DDF - 1	0	0	3	3	2
DDF - 2		0	0	1	1	0	
DDF - 3		1	1	1	1	1	
DDF - 4		55	8	5	4	0	
HDF - 1		14	4	1	1	0	
HDF - 2		31	4	1	1	0	
HDF - 3		20	6	2	2	0	
PLE - 1		23	7	11	4	0	
PLE - 2		123	11	8	3	2	
PLE - 3		23	6	8	3	1	
Total			290	12	41	10	6
Grand Total			556	14	211	27	10



**Appendix B-3** Rank abundance for amphibian species captured in each forest type.

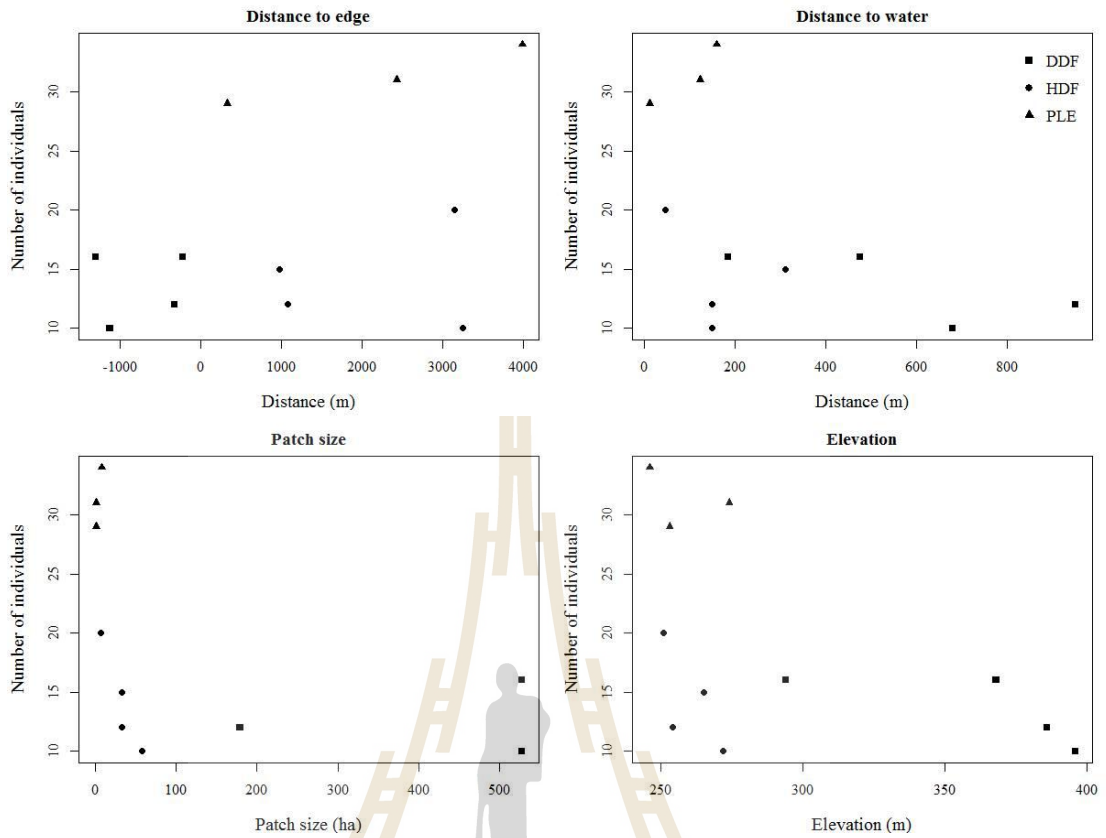
Forest type	Species	Rank	Abundance	Proportion	Log(Abund)
DDF	<i>Kaloula mediolineata</i>	1	20	35.1	1.3
	<i>Glyphoglossus molossus</i>	2	18	31.6	1.3
	<i>Kaloula pulchra</i>	3	13	22.8	1.1
	<i>Fejervarya limnocharis</i>	4	2	3.5	0.3
	<i>Microhyla butleri</i>	5	1	1.8	0
	<i>Microhyla heymonsi</i>	6	1	1.8	0
	<i>Microhyla fissipes</i>	7	1	1.8	0
	<i>Microhyla pulchra</i>	8	1	1.8	0
HDF	<i>Microhyla fissipes</i>	1	78	28.4	1.9
	<i>Fejervarya limnocharis</i>	2	66	24	1.8
	<i>Microhyla pulchra</i>	3	52	18.9	1.7
	<i>Kaloula mediolineata</i>	4	27	9.8	1.4
	<i>Microhyla heymonsi</i>	5	20	7.3	1.3
	<i>Microhyla butleri</i>	6	13	4.7	1.1
	<i>Duttaphrynus melanostictus</i>	7	7	2.5	0.8
	<i>Kaloula pulchra</i>	8	6	2.2	0.8
	<i>Glyphoglossus molossus</i>	9	3	1.1	0.5
	<i>Hylarana macrodactyla</i>	10	2	0.7	0.3
	<i>Calluella guttulata</i>	11	1	0.4	0
PLE	<i>Microhyla fissipes</i>	1	51	22.8	1.7
	<i>Fejervarya limnocharis</i>	2	31	13.8	1.5
	<i>Glyphoglossus molossus</i>	3	28	12.5	1.4
	<i>Microhyla heymonsi</i>	4	28	12.5	1.4
	<i>Microhyla butleri</i>	5	19	8.5	1.3
	<i>Kaloula pulchra</i>	6	13	5.8	1.1
	<i>Kaloula mediolineata</i>	7	11	4.9	1
	<i>Micryletta inornata</i>	8	11	4.9	1
	<i>Calluella guttulata</i>	9	10	4.5	1
	<i>Microhyla pulchra</i>	10	8	3.6	0.9
	<i>Duttaphrynus melanostictus</i>	11	5	2.2	0.7
	<i>Occidozyga lima</i>	12	5	2.2	0.7
	<i>Hylarana macrodactyla</i>	13	3	1.3	0.5
	<i>Hylarana erythraea</i>	14	1	0.4	0

**AppendixB-4** Rank abundance for reptile species captured in each forest type.

Forest type	Species	Rank	Abundance	Proportion	Log(Abun)
DDF	<i>Dixoneus siamensis</i>	1	14	25.9	1.1
	<i>Eutropis macularia</i>	2	10	18.5	1
	<i>Lycodon capucinus</i>	3	7	13	0.8
	<i>Gehyra lacerata</i>	4	4	7.4	0.6
	<i>Calotes versicolor</i>	5	3	5.6	0.5
	<i>Lygosoma bowringii</i>	6	3	5.6	0.5
	<i>Boiga multomaculata</i>	7	2	3.7	0.3
	<i>Oligodon pseudotaeniatus</i>	8	2	3.7	0.3
	<i>Boiga siamensis</i>	9	1	1.9	0
	<i>Calliophis maculiceps</i>	10	1	1.9	0
	<i>Chrysopelea ornata</i>	11	1	1.9	0
	<i>Coelognatus radiatus</i>	12	1	1.9	0
	<i>Leiolepis reevesii</i>	13	1	1.9	0
	<i>Naja siamensis</i>	14	1	1.9	0
	<i>Oligodon faciولاتus</i>	15	1	1.9	0
	<i>Psammodynastes pulverulentus</i>	16	1	1.9	0
	<i>Ramphotyphlops braminus</i>	17	1	1.9	0
HDF	<i>Eutropis macularia</i>	1	16	28.1	1.2
	<i>Dixoneus siamensis</i>	2	9	15.8	1
	<i>Lygosoma bowringii</i>	3	7	12.3	0.8
	<i>Hemidactylus spp</i>	4	6	10.5	0.8
	<i>Leiolepis reevesii</i>	5	5	8.8	0.7
	<i>Lycodon capucinus</i>	6	4	7	0.6
	<i>Gehyra lacerata</i>	7	3	5.3	0.5
	<i>Boiga multomaculata</i>	8	1	1.8	0
	<i>Bungarus candidus</i>	9	1	1.8	0
	<i>Lycodon laoensis</i>	10	1	1.8	0
	<i>Naja siamensis</i>	11	1	1.8	0
	<i>Oligodon pseudotaeniatus</i>	12	1	1.8	0
	<i>Ramphotyphlops albiceps</i>	13	1	1.8	0
	<i>Ramphotyphlops braminus</i>	14	1	1.8	0

## Appendix B-4 (Continued)

Forest type	Species	Rank	Abundance	Proportion	Log(Abun)
PLE	<i>Eutropis macularia</i>	1	39	41.5	1.6
	<i>Dixoneus siamensis</i>	2	15	16	1.2
	<i>Hemidactylus spp.</i>	3	12	12.8	1.1
	<i>Leiolepis reevesii</i>	4	8	8.5	0.9
	<i>Lygosoma bowringii</i>	5	4	4.3	0.6
	<i>Lycodon capucinus</i>	6	4	4.3	0.6
	<i>Bungarus candidus</i>	7	3	3.2	0.5
	<i>Gehyra lacerata</i>	8	2	2.1	0.3
	<i>Caloselasma rhodostoma</i>	9	1	1.1	0
	<i>Calotes versicolor</i>	10	1	1.1	0
	<i>Dendrelaphis subocularis</i>	11	1	1.1	0
	<i>Enhydryis plumbea</i>	12	1	1.1	0
	<i>Oligodon faciolutus</i>	13	1	1.1	0
	<i>Oligodon taeniatus</i>	14	1	1.1	0
	<i>Rhabdophis chrysargus</i>	15	1	1.1	0



**Appendix B-5** Plot of reptile abundance by distance to the edge of SBR, distance to water, patch size, and elevation.

**Appendix B-6** Results from environmental fitting on NMDS of abundance community data for continuous variables.

Community analyze	Environmental variable	NMDS1	NMDS2	R <sup>2</sup>	p-value
Reptiles	Groundstory	-0.294	-0.956	0.240	0.334
	Canopy cover	-0.794	-0.608	0.362	0.160
	Grass	0.201	-0.980	0.100	0.676
	Litter depth	-0.349	-0.937	0.117	0.628
	Elevation	-0.975	-0.220	0.159	0.503
	Distance to water	-0.998	0.065	0.310	0.223
	Distance to SBR edge	0.613	-0.790	0.032	0.889
	Patch size	0.911	-0.412	0.016	0.959
	Slope	-0.973	0.231	0.269	0.305
	Aspect	1.000	-0.022	0.013	0.946
Amphibians	Groundstory	0.721	0.693	0.401	0.350
	Canopy cover	0.702	-0.712	0.153	0.714
	Grass	0.515	-0.857	0.334	0.460
	Litter depth	0.432	0.902	0.753	0.064
	Elevation	-0.109	-0.994	0.494	0.291
	Distance to water	-0.401	-0.916	0.606	0.145
	Distance to SBR edge	0.900	-0.437	0.759	0.043
	Patch size	-0.089	-0.996	0.447	0.313
	Slope	0.555	-0.832	0.100	0.820
	Aspect	0.468	-0.884	0.526	0.225
Total	Groundstory	-0.801	-0.599	0.009	0.968
	Canopy cover	-0.969	0.246	0.488	0.078
	Grass	0.694	-0.720	0.197	0.458
	Litter depth	-0.273	0.962	0.105	0.669
	Elevation	-0.837	0.548	0.861	0.003
	Distance to water	-0.984	0.179	0.851	0.005
	Distance to SBR edge	0.631	-0.776	0.512	0.051
	Patch size	-0.428	0.904	0.768	0.007
	Slope	-0.936	-0.353	0.492	0.110
	Aspect	0.480	-0.877	0.431	0.123

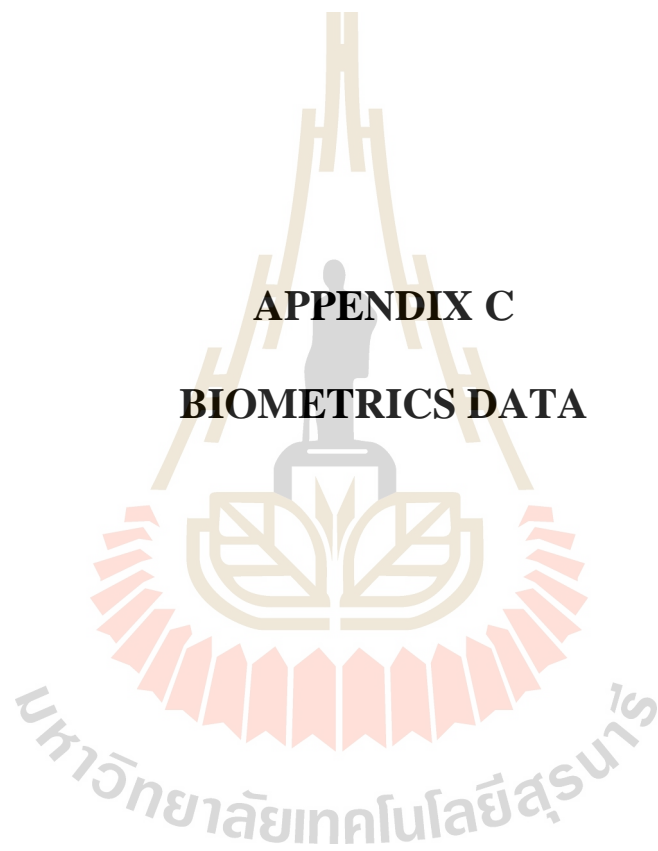
**Appendix B-7** Fitting of environmental factors on NMDS of abundance community data.

Community analyzed	Environmental variable	R <sup>2</sup>	p-value
Reptiles	Forest type	0.038	0.960
	Distance group	0.209	0.453
	Ground vegetation	0.222	0.705
	Leaf litter coverage	0.201	0.748
Amphibians	Forest type	0.206	0.387
	Distance group	0.441	0.014
	Ground vegetation	0.473	0.516
	Leaf litter coverage	0.611	0.254
Total	Forest type	0.448	0.039
	Distance group	0.591	0.003
	Ground vegetation	0.053	0.997
	Leaf litter coverage	0.195	0.748

**Appendix B-8** PERMANOVA results for all herpetofauna captures, reptile captures, and amphibian captures using presence/absence transformation.

Community analyzed	Variable	Df	Sum of Squares	Mean Squares	F-Model	R <sup>2</sup>	p-value
Total	Distance group	2	0.69	0.35	1.82	0.37	0.18
	Forest type	2	0.11	0.06	0.30	0.06	0.97
	Distance to water	1	0.14	0.14	0.75	0.08	0.60
	Elevation	1	0.18	0.18	0.93	0.10	0.52
	Canopy cover	1	0.17	0.17	0.90	0.09	0.54
	Patch size	1	0.17	0.17	0.90	0.09	0.54
	Residuals	2	0.38	0.19	0.21		
	Total	10	1.85	1.00			
Reptiles	Forest type	2	0.32	0.16	0.84	0.17	0.62
	Canopy cover	1	0.27	0.27	1.40	0.14	0.26
	Residuals	7	1.34	0.19	0.69		
	Total	10	1.93	1.00			
Amphibians	Distance group	1	0.02	0.02	0.34	0.07	0.79
	Forest type	1	0.08	0.08	1.09	0.21	0.45
	Litter depth	1	0.05	0.05	0.71	0.14	0.62
	Residuals	3	0.21	0.07	0.58		
	Total	6	0.36	1.00			

**APPENDIX C**  
**BIOMETRICS DATA**



**Appendix C-1** Captured snake biometrics for each species by forest type.

Forest type	Species	Count	Average SVL (mm)	Average Mass (g)
DDF	<i>Boiga multomaculata</i>	1	598	22.8
	<i>Boiga siamensis</i>	1	93.5	74.4
	<i>Calliophis maculiceps</i>	1	265	4.9
	<i>Chrysopelea ornata</i>	1	570	36.8
	<i>Coelognathus radiatus</i>	1	827	165.8
	<i>Lycodon capucinus</i>	5	360.8	25.4
	<i>Lycodon laoensis</i>	1	328	14.1
	<i>Naja siamensis</i>	1	1266	758.5
	<i>Oligodon fasciolatus</i>	1	616	86.7
	<i>Oligodon pseudotaeniatus</i>	1	269	8.7
	<i>Rhabdophis subminiatus</i>	1	248	5.7
HDF	<i>Boiga multomaculata</i>	1	518	16.8
	<i>Lycodon capucinus</i>	5	371.2	16.12
	<i>Lycodon laoensis</i>	1	321	12.5
PLE	<i>Bungarus candidus</i>	2	700.5	138.6
	<i>Calloselasma rhodostoma</i>	1	552	36.7
	<i>Chrysopelea ornata</i>	1	724	726
	<i>Dendrelaphis subocularis</i>	1	450	22.7
	<i>Enhydris plumbea</i>	1	182	1.5
	<i>Lycodon capucinus</i>	3	337.7	14.1
	<i>Oligodon fasciolatus</i>	2	376	46.6
	<i>Oligodon pseudotaeniatus</i>	1	288	13.4
	<i>Oligodon taeniatus</i>	1	307	15.3
<i>Rhabdophis chrysargos</i>	1	144	10.9	



**Appendix C-2** Biometric information for species by each forest type examined.

Species	Forest type	Individuals	SVL		Mass	
			Avg	StdDev	Avg	StdDev
<i>Dixoneus siamensis</i>	DDF	1	50.00	-	2.25	-
	HDF	8	49.88	1.89	3.24	0.31
	PLE	6	43.06	11.53	2.14	1.30
<i>Eutropis macularia</i>	DDF	4	51.50	7.55	4.13	2.22
	HDF	5	57.80	2.95	5.07	0.85
	PLE	5	49.51	3.77	3.47	0.97
<i>Kaloula mediolineata</i>	HDF	10	45.99	11.38	11.62	6.03
	PLE	1	22.48	-	1.28	-
<i>Kaloula pulchra</i>	HDF	2	61.00	1.41	17.34	0.44
<i>Leiolepis reevesii</i>	HDF	2	55.67	35.82	11.01	8.68
	PLE	4	73.85	5.66	10.52	2.76

## CURRICULUM VITAE

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### Publications

- Strine, C. T., Silva, I. M. S., **Crane, M. S.**, Bartosz, N., Artchawakom, T., Goode, M., and Suwanwaree, P. (2014). Mortality of a wild king cobra, *Ophiophagus hannah* Cantor, 1836 (Serpentes: Elapidae) from Northeast Thailand after ingesting a plastic bag. *Asian Herpetological Research*. 5(4): 284–286.
- Karraker, N. E., Strine, C. T., **Crane, M.**, and Devan-Song, A. (2015). *Dryocalamus subannulatus* (Malayan bridle snake) behavior. *Herpetological Review*, 46(2): 272–273.
- Strine, C., Silva, I., Nadolski, B., **Crane, M.**, Barnes, C., Artchawakom, T., Hill, J. and Suwanwaree, P.. (2015). Sexual dimorphism of tropical green pit viper *Trimeresurus (Cryptelytrops) macrops* in Northeast Thailand. **Amphibia-Reptilia**, 36(4): 327-338.