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Herpetofaunal Species Abundance, Richness, and Diversity in a Dry Tropical Forest and Agricultural Matrix at the Sakaerat Biosphere Reserve, Thailand

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Abstract.—**With ongoing rapid anthropogenic deforestation and habitat degradation, it is critical that we understand the role of human-disturbed areas in conserving the biodiversity of the world. Despite intensive deforestation in Southeast Asia, there are few studies investigating faunal communities in human-modified landscapes there. We assessed herpetofauna in dry dipterocarp forest, disturbed forest, and *Eucalyptus* plantations in the Sakaerat Biosphere Reserve of Thailand. In May and June of 2015, we conducted surveys using 12 passive trapping arrays with funnel and pitfall traps. We captured 436 individuals representing 38 species, with 266 amphibians (13 species) and 170 reptiles (25 species). The *Eucalyptus* plantations and highly disturbed forest sites hosted higher amphibian abundance and richness than protected areas of dry dipterocarp forest. Reptile species richness did not differ between habitat types, but capture rates in the *Eucalyptus* plantations were significantly higher than in the dry dipterocarp forest. Funnel traps yielded significantly higher reptile capture rates than pitfall traps. For reptiles, the results support other studies, which have concluded that reptiles are less sensitive than amphibians to disturbance and possibly positively affected by human disturbance in some cases. The terrestrial amphibian abundance and species richness documented here indicate that disturbed habitats may provide suitable areas for these species.**

Key Words.—degraded forest; dipterocarp forest; herpetofaunal survey; human disturbance; passive trapping; Southeast Asia

Introduction

Loss of species in species-rich tropical regions is one of the greatest threats to global biodiversity loss as these areas undergo extensive deforestation, losing large swaths of forested land each year (Brooks et al. 2002; Bradshaw et al. 2009). Human-dominated land use types, such as plantation forests and agricultural developments, are rapidly replacing natural forests (Rudel et al. 2005; Wright 2005). The conversion of forests to agrarian lands has left mosaics of small forest patches surrounded by a mixture of anthropogenic land uses (Saunders et al. 1991). These changes are important because land use composition significantly influences the structure of tropical species assemblages found within disturbed habitat patches (Urbina-Cardona et al. 2006).

Southeast Asia has among the least remaining forest cover of all of the historically forested tropical regions (Achard et al. 2002; Laurance 2007) yet continues to experience high rates of deforestation (Sodhi et al. 2004; Stibig et al. 2014). Replacing forests with other land use types is detrimental for many taxa (Kwok and Corlett 2000; Sodhi et al. 2004). Conservationists, however, are only beginning to understand the underlying mechanisms for this in a few well-studied taxa, such as birds (Hsu et al. 2010), mammals, and invertebrates (Trimble and van Aarde 2012). Herpetofauna are in need of intensive conservation effort as these taxa are facing global declines resulting from a variety of factors, such as habitat loss, spread of invasive species, overcollection of naïve fauna (Gibbons et al. 2000), climate change (Reading et al. 2010; Araújo et al. 2006), and pandemic disease (Alford 2011; Lips et al. 2005). Despite the severity of threats faced by many herpetofaunal taxa, Böhm et al. (2013) described one in five reptilian species as Data Deficient using IUCN Red List categories, with tropical species being among the most deficient.

Research on the impacts of replacing forests with plantations or other land uses is plentiful for temperate regions (Trimble and van Aarde 2012). The studies

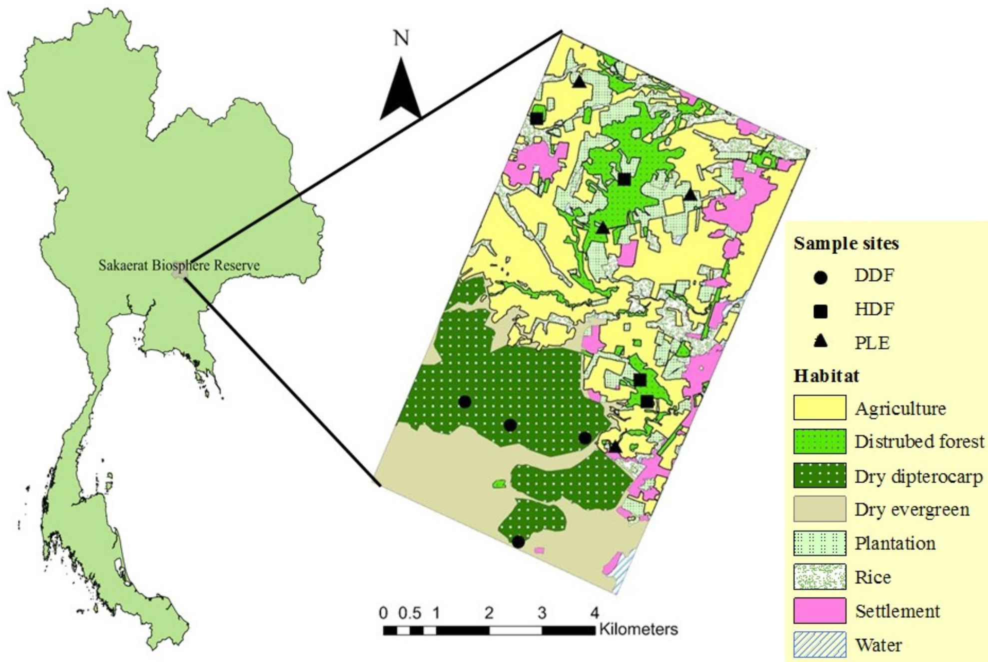

Figure 1. Map of Thailand with the location of the Sakaerat Biosphere Reserve (left) and the land use provided by the Thai Land Development Department (2004) for the study area with sampling sites (right). (Map created by Mathew S. Crane)

have produced varied results. While some studies argue that there are detrimental impacts upon herpetofaunal diversity (Faria et al. 2007; Gardner et al. 2007; Wanger et al. 2010; Kurz et al. 2014), others suggest that plantations are capable of housing herpetofaunal communities similar in composition to those in primary forests (Germano et al. 2003; Fredericksen and Fredericksen 2004; Folt and Reider 2013). Others suggest that these changes may have a positive influence on amphibian diversity by maintaining forest cover through the conversion to plantation forests (Vonesh 2001; Fredericksen and Fredericksen 2002).

The lack of consensus in the literature highlights the need for more studies to help determine the causes for the wide range of herpetofaunal community responses to land use change. There have been relatively few studies in Southeast Asia concerning herpetofaunal community composition and land use changes (Wanger et al. 2010; Sung et al. 2012). It is critical to assess the conservation value of plantation and remnant forests for herpetofaunal communities in Southeast Asia to construct a broad set of policy guidelines targeting land use management.

In Thailand, several studies have investigated herpetofaunal diversity along elevation (Kongjaroen and Nabhitabhata 2007; Phochayavanich et al. 2010) and disturbance gradients (Konlek and Lauhachinda 2008; Kaensa et al. 2014), as well as between habitats (Inger and Colwell 1977; Suttanon and Lauhachinda 2008). The studies showed contrasting results with one study showing a negative impact on amphibian diversity from disturbance (Kaensa et al. 2014) and another finding higher amphibian abundances in highly disturbed agricultural habitats (Phochayavanich et al. 2008). We aimed to further elucidate the impacts of forest degradation in Thailand by comparing reptile and amphibian assemblages between protected and degraded forest habitats in the northeastern part of the country. We also aimed to further address the knowledge gap concerning the diverse reptile and amphibian communities in Thailand using standardized passive trapping techniques across both protected and degraded habitats. We hypothesized that forest degradation affects species diversity and we predict that more degraded forest habitats would have relatively lower amphibian abundance and species richness with little to marginal influence of the habitat degradation on reptiles compared to non-degraded forests.

Materials and Methods

Study site.—We conducted the study within the Sakaerat Biosphere Reserve (SBR) located in Nakhon Ratchasima Province, Thailand (14.44–14.55°N, 101.88–101.95°E). The reserve has an 80 km^2 protected Core Area, surrounded by a 360 km^2 belt, comprised of the Buffer and Transitional Zones (Fig. 1). The forested areas occur predominately in the Core and Buffer Zones, which consist of primary-growth dry evergreen forest, dry dipterocarp forest and secondary reforestation (Trisurat 2010). The Transition Zone comprises nearly

Figure 2. Passive trapping array showing the line (A), wings (B), and traps (C and D). (Photographed by Mathew S. Crane)

82% of the Reserves total area and is characterized by isolated forest fragments in a patchwork of agricultural fields, small plantation forests, and human settlements.

Site selection and herpetofaunal sampling.—We assessed herpetofaunal community assemblages across a gradient of human disturbance, specifically in remnant dry dipterocarp forests (DDF), highly disturbed forests (HDF), and eucalyptus plantation forests (PLE). The HDF consisted of patches of dry dipterocarp forest embedded within the agricultural matrix of the SBR Transition Zone. The HDF forest patches are characterized by high levels of anthropogenic disturbance, but still retain similar understory vegetation composition to remnant forests with endemic cycads and bamboo grasses. We identified eucalyptus plantations as forest stands planted with *Eucalyptus camaldulensis* for economic production. Eucalyptus trees are planted in linear stands, separated by sparsely vegetated gaps, and are harvested on 3–5 y intervals.

To identify the three forest types, we used a combination of 2004 land use maps provided by the Thai Land Development Department and more recent satellite imagery of 2014 accessed through Google Earth (https://www.google.com/earth/ [Accessed 24 April 2014]). We digitized satellite imagery from Google Earth to create polygons for each identifiable plantation and secondary forest within the study area. Using ArcMap, we randomly selected four plot sites in each forest type with a minimum distance of 450 m apart in an attempt to control for spatial autocorrelation.

To effectively sample the forest areas, we built Y-shaped drift-fence arrays with double-chambered funnel traps and 40 L pitfall traps. We attached two double funnel traps measuring 2 × 0.5 × 0.3 m at the end of each fence line in the array. We placed a 40 L pitfall trap, extending 36 cm deep into the soil from the surface at the midpoint of each line for a total of three pitfalls per array. Additionally, we attached six double-chambered funnel traps to the center of each array for a total of 12 funnel traps and three pitfall traps (Fig. 2). To avoid flooding during rains, we drilled holes into the base of each pitfall trap. We fitted pitfall bases with a sheet of screen wire to prevent blockages and animals from escaping through the drainage holes. To reduce capture mortalities from exposure, we constructed shade covers over the traps from plastic mesh tarps affixed to bamboo stakes. We placed sponges in the second compartment of each funnel trap and in the pitfall traps to avoid desiccation for sensitive taxa. During sampling periods, we checked traps early in the morning to reduce the exposure of a captive to extreme temperatures.

We sampled in May and June of 2015 to assess herpetofauna. May had slightly higher average temperatures (26.3° C) and less rain (8.55 mm) compared to June (25.6° C; 79.25 mm). Each month, we opened a group of six plots (two from each habitat type) for 3 d before switching to the second set of six plots for a total of 6 d/mo. However, one of the PLE sites was destroyed after the first sampling period in May, which created an uneven sampling effort between habitats.

We identified all herpetofauna captures to their species, except for snakes, following Chanard et al. (2015). To identify snakes, we used Cox et al. (2012) and the American Museum of Natural History electronic database (American Museum of Natural History. 2017. Amphibian Species of the World: Online Reference. Version 6.0. Available from http://research.amnh.org/herpetology/amphibia/index.html. [Accessed 12 April 2017]) for amphibians. We cross-referenced all species with the Reptile Database (Reptile Database. Available from http://www.reptile-database.org [Accessed 12 April 2017]) to adjust for current taxonomic name changes.

Environmental variables.—We recorded landscape factors for each plot, including distance to water, number of water types, patch size, and elevation. We defined the number of water types as the number of different water source types (ponds, streams, and rice paddy/wetlands) within 450 m of each plot. Additionally, at each site we visually assessed ground story vegetation density at six 1 m^3 quadrats. Within each 1 m^3 quadrat, we categorized ground story vegetation density on an ordinal scale from one to six as: None (0%), Very light (1–15%), Light (16–25%), Medium (25–65%), Heavy (66–80%), and Very Heavy (81–99%). At each of the six points, we also visually estimated canopy cover using a simple homemade densitometer following the same categories

Table 1. Comparison of habitats by environmental variables collected at each sampling site in northeastern Thailand. An asterisk (*) indicates that the variable was collected on an ordinal scale with median and upper and lower quantile reported rather than mean and SD.

Environmental variable	Dry dipterocarp forest Mean ± SD	Highly disturbed forest Mean ± SD	*Eucalyptus* plantation Mean ± SD	Statistical test	*F* or X^2	df	*P*
Distance to water (m)	512 ± 324	226 ± 107	113 ± 72	ANOVA	4.163	2	0.052
Water types	0.5 ± 0.58	1.25 ± 0.50	2 ± 0.82	Kruskal-Wallis	6.182	2	0.045
Patch size (ha)	440.6 ± 174.3	32.9 ± 20.7	3.7 ± 3.4	Kruskal-Wallis	9.434	2	0.009
Elevation (m)	361 ± 46.1	260.5 ± 9.7	259.8 ± 12.6	Kruskal-Wallis	7.385	2	0.025
Canopy cover*	5 (5 - 4)	2 (5 - 1)	1.5 (2 - 1)	Kruskal-Wallis	18.195	2	< 0.001
Ground story vegetation*	3 (4 - 2)	3 (4 - 2)	2 (4.75 - 2)	Kruskal-Wallis	0.311	2	0.856

used for ground story vegetation density. Due to the limited sampling effort at each site (n = 6), we did not directly compare the habitat features to species richness, abundance, or diversity to avoid drawing overly broad conclusions from our results.. Instead, we looked at each variable independently to see which factors showed significant differences between habitat types to identify general characteristics for each habitat type. We calculated all distances using ArcGIS 10.1.

Data analysis.—We tested all environmental parameters for normality and homoscedasticity using Shapiro-Wilks test and Levene's test, respectively. We applied a one-way ANOVA or a Kruskal-Wallis test as appropriate. We analyzed amphibian and reptile communities separately for all methods, as life-history traits and response to human disturbance can vary dramatically between the two groups.

To account for the potential bias in trapping method evident in reptiles, we combined captures from pitfall and funnel traps for a more accurate estimate of abundance and species richness. To compare abundance between habitat types, we ran Generalized Linear Mixed Models (GLMMs) with a Poisson log link function. We used habitat type and month as fixed effect and plot ID as a random effect. We selected the top model based on Akaike Information Criteria corrected for small sample sizes (AICc) values.

To analyze species richness, we first created sample-based rarefaction curves using each day that a plot was open as a sampling unit. To standardize for unequal sampling effort, we conducted sample-based rarefaction analysis for each habitat type using Hill numbers to extrapolate species richness and create 95% confidence intervals with package iNEXT (Hsieh et al. 2016). We visually assessed the 95% confidence interval overlap of the rarefaction curves to compare between habitat types. Due to limited sample size (n = 6), we were not able to calculate species richness for each site and thus only compared richness between habitat types.

We calculated both reptile and amphibian diversity for each site using the Shannon-Wiener index, which incorporates species richness and evenness to calculate diversity. We tested the data for normality and heterogeneity using Shapiro's test and Levene's test respectively, and then compared reptile diversity between habitats using one-way ANOVA, and amphibian diversity using the Wilcoxon rank-sum test. To visually assess species evenness, we compared rank abundance plots for each habitat type. We performed all analyses in R Studio using packages lme4, vegan, and iNEXT (Kindt and Coe 2005; R Core Team 2014; Oksanen et al. 2016). For all tests, $\alpha = 0.05$ unless specified otherwise.

Results

Habitat variation.—The DDF, HDF, and PLE habitat types showed significant differences amongst some environmental characteristics (Table 1). Additionally, plantation forests were composed of significantly smaller patches than the dry dipterocarp forest (W = 16.0, *P* = 0.026). The PLE and HDF habitats did not show a significant difference in patch size (W = 1.0 , *P* = 0.059). Distance to water did not show a significant difference at the 95% confidence interval, but did at the 90% (Table 1). The pairwise comparison showed that the PLE sites were closer to water than the DDF (Tukey HSD; df = 2, adjusted *P* = 0.046), but there was no significant difference between the HDF and DDF (Tukey HSD; df = 2, adjusted *P* = 0.150) or PLE (Tukey HSD; df = 2, adjusted *P* = 0.710). Additionally, the PLE sites had a greater variety of water types compared to the DDF sites (W = 0.5, *P* = 0.036), but not compared with the HDF sites (W = 12.5, *P* = 0.210). The difference in water types between HDF and DDF sites did not reach significance at the 95% confidence interval (W = 1.0, *P* = 0.052).

Capture rates.—We captured 436 individuals (266 amphibians and 170 reptiles) from 38 species (13 amphibians and 25 reptiles; Table 2). We captured 21 species in funnel traps, 16 reptiles and five amphibians, which we did not observe in the pitfall traps, while only two reptile species and two amphibian species were

Table 2. Number of individuals of amphibian and reptile species captured in each habitat type throughout the study in northeastern Thailand. An asterisk (*) indicates that the species was captured only in a funnel trap, while + indicates the species was unique to pitfall traps. Habitat types are DDF = dry, dipterocarp forest, HDF = highly disturbed forest, and PLE = *Eucalyptus* plantation.

		Habitat type			
Family	Species	DDF	HDF	PLE	Total
Bufonidae	*Duttaphrynus melanostictus*		7	5	12
Dicroglossidae	*Fejervarya limnocharis*		64	19	83
	Occidozyga lima+			1	1
Microhylidae	*Glyphoglossus molossus*+		1		1
	Kaloula mediolineata		12	7	19
	Kaloula pulchra	1	3	2	6
	*Microhyla butleri**		3	1	4
	*Microhyla heymonsi**		15	1	16
	*Micryletta inornata**			1	1
	Microhyla fissipes		63	10	73
	Microhyla pulchra		40	5	45
Ranidae	*Hylarana erythraea**			1	1
	*Hylarana macrodactyla**		2	2	4
Agamidae	*Calotes versicolor*	3		1	4
	Leiolepis reevesii		4	7	11
Gekkonidae	*Dixonius siamensis*	11	9	17	37
	Gehyra lacerata	3	2	2	7
	Hemidactylus frenatus		4	9	13
Scincidae	*Eutropis macularia*	7	16	22	45
	Lygosoma bowringii	3	7	3	13
Colubridae	*Boiga multimaculata**	2	1		3
	*Boiga siamensis**	1			1
	*Chrysopelea ornata**	1		1	2
	*Coelognathus radiatus**	1			1
	*Dendrelaphis subocularis**			1	1
	*Hypsiscopus plumbea**			1	1
	*Lycodon capucinus**	7	4	3	14
	*Lycodon laoensis**		1		1
	*Oligodon fasciolatus**	1		2	3
	*Oligodon pseudotaeniatus**	2	1		3
	*Oligodon taeniatus**			1	1
	*Rhabdophis chrysargos**			1	1
Elapidae	*Bungarus candidus**		1	1	2
	*Calliophis maculiceps**	1			1
	*Naja siamensis**	1	1		2
Typhlopidae	*Indotyphlops albiceps*+		1		1
	Indotyphlops braminus+		1		1
Viperidae	*Calloselasma rhodostoma**			1	1
Total		45	263	128	436

unique to pitfall traps. Both amphibian and reptile abundance showed variation between habitat types and by the month sampled (Fig. 3) and the best model for amphibian abundance included both habit type and month (Table 3). The month of May (estimate = -0.989, Z = -7.166, $P < 0.001$) and both HDF (estimate = 5.266,

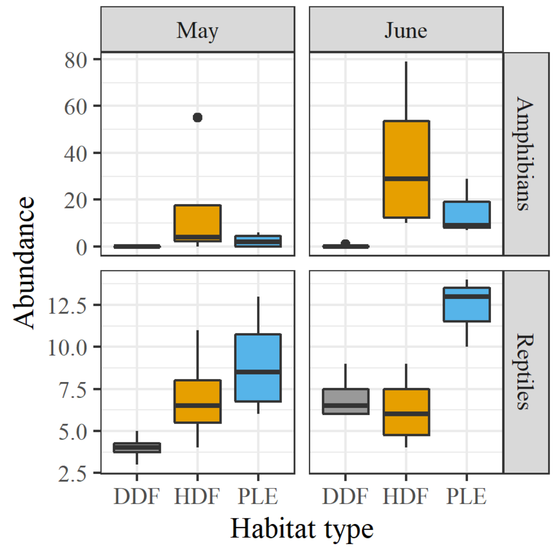

Figure 3. Boxplots of amphibian and reptile abundance for the dry dipterocarp forest (DDF), highly disturbed forest (HDF), and *Eucalyptus* plantations (PLE) in northeastern Thailand, divided by each sampling month.

Z = 4.261, $P < 0.001$) and PLE (estimate = 4.092, Z = 3.295, $P < 0.001$) showed significant coefficient estimates, with May predicting lower abundance and both the HDF and PLE predicting higher when compared to the DDF in June which was the intercept (estimate = -2.076, Z = -1.813, P = 0.070).

The PLE showed a larger effect size on amphibian abundance than the HDF, but both still overlapped in the 95% confidence interval (Fig. 4). Additionally, many exhibited slightly higher amphibian abundance, but the effect size was relatively small (Fig. 4). For reptile abundance the two top performing models included habitat type and month and only habitat type (Table 3). When only including forest type in the model, the PLE (estimate = 0.640, Z = 3.352, $P < 0.001$) showed a significant effect for increased abundance compared to the DDF (estimate = 1.70, Z = 11.308, $P < 0.001$) while the HDF (estimate = 0.186, Z = 0.912, $P = 0.362$) did not show a significant effect. However, the effect size for the predicted abundance in the PLE was small as the coefficient estimate was less than 1 (Fig. 5).

Rank abundances.—Amphibian rank abundance curves had similar slopes when comparing between the HDF and the PLE (Fig. 6). Two species, the Common Pond Frog (*Fejervarya limnocharis*) and the Ornate Narrowmouth Frog (*Microhyla fissipes*), ranked as the top two most abundant species in both the HDF and PLE. The HDF had a higher relative abundance of mircohylid species than the PLE (Table 2).

Reptile rank abundance curves had similar slopes across forest types (Fig. 7). Snakes contributed to the highest number of reptile species, but we captured them

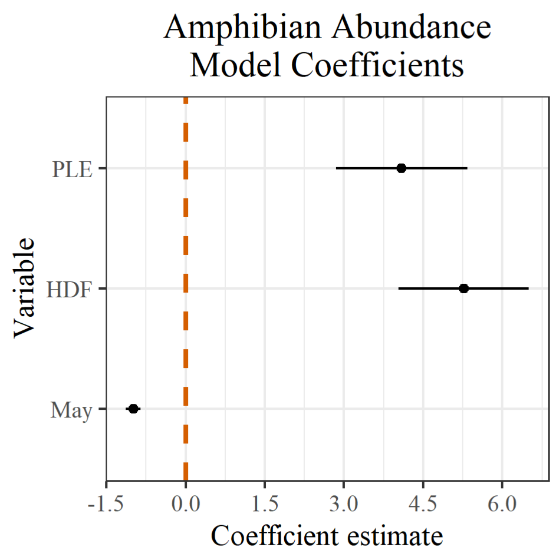

Figure 4. Effect size estimates for amphibian abundance from the top-performing GLMM model.

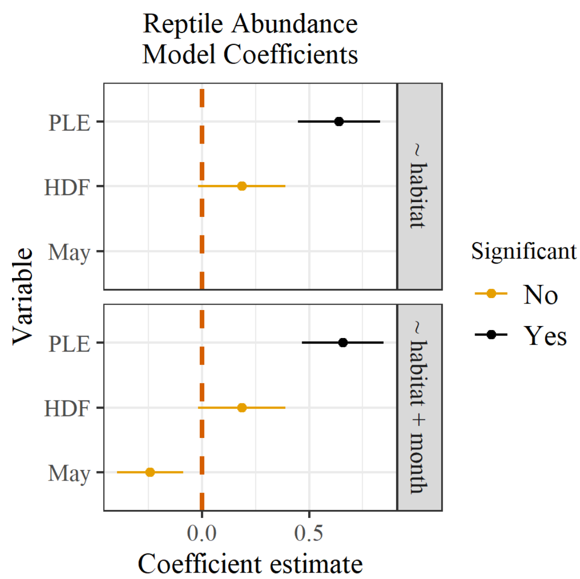

Figure 5. Effect size estimates for reptile abundance from the two top-performing GLMM models.

in very low abundances. Comparing between forest types showed that one species, *Hemidactlyus frenatus* was abundant in both the HDF and PLE but was not present in the DDF.

Species richness and diversity.—We captured 12 amphibian species in the PLE and 10 species in the HDF, but only a single species in the DDF sites. Because we captured only a single amphibian individual from the DDF, we did not include the DDF sites in species richness comparisons (Table 2). Visual inspection of the 95% confidence intervals revealed that the HDF and PLE did not show any significant difference in amphibian species richness (Fig. 8). Reptile species richness did not differ between forest types with 14 observed species in the HDF and DDF and 16 in the PLE. The accumulation curves all fell within the 95% confidence interval (Fig. 8). Similarly, reptile diversity showed no significant differences between habitat types ($F_{2,9}$ = 1.656, *P* = 0.777). Amphibian diversity did not differ significantly between the HDF and PLE sites (W = 9.0, *P* = 0.886; Table 4).

Discussion

Our results contribute to the growing literature on herpetofaunal abundance, species richness and diversity in fragmented forests. The results support similar studies indicating that reptiles show low sensitivity to disturbance (Wanger et al. 2010), with no difference in species richness across habitats and only a difference in reptile abundance between plantation sites and conserved forests. Our results suggest that some amphibian species can thrive in disturbed habitats, which contributes to the conflicting literature showing both negative (Pearman 1997; Suazo-Ortuño et al. 2008; Wanger et al. 2010) and positive impacts (Vonesh 2001; Fredericksen and Fredericksen 2002) on amphibian communities. Despite this, our results should not be overstated as the sampling was limited to just two months. Additionally, our surveying methodology was targeted at fossorial species. While we did capture species that are at least semi-arboreal, we did not capture any amphibians from the family Rhacophoridae. This indicates that our

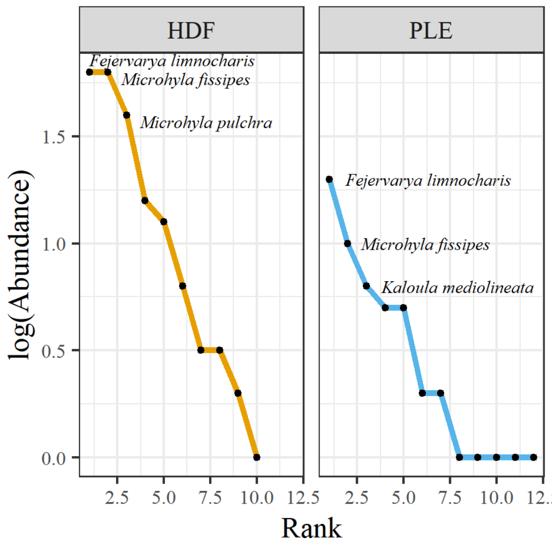

Figure 6. Amphibian rank abundance curves for the highly disturbed forest sites (HDF) and the *Eucalyptus* plantation sites (PLE) in northeastern Thailand.

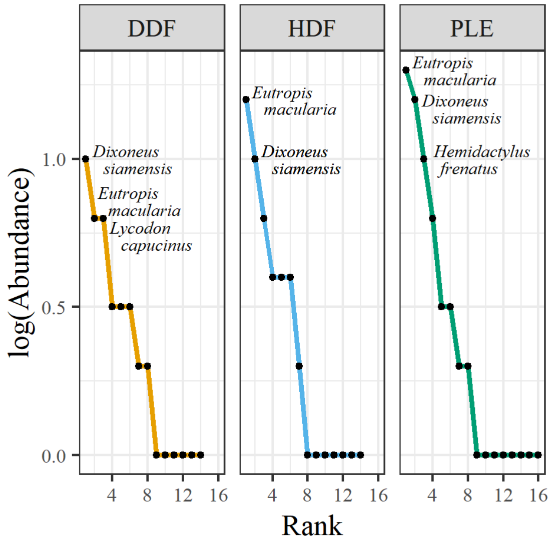

Figure 7. Reptile rank abundance curves for the dry dipterocarp forest (DDF), highly disturbed forest (HDF) and the *Eucalyptus* plantation sites (PLE) in northeastern Thailand.

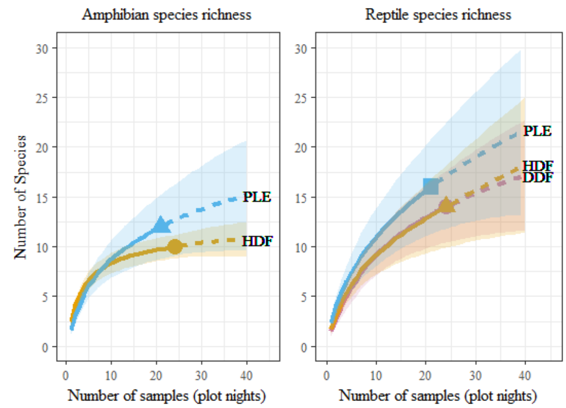

Figure 8. Species richness curves for amphibian and reptiles captured in each habitat type in northeastern Thailand.

sampling was biased against arboreal species. Inger and Colwell (1977) captured a high abundance of several arboreal species in the dry dipterocarp forest and/or agricultural areas, such as the *Polypedates leucomystax* and *Chiromantis nongkhorensis*, indicating that we did miss a number of species known to be present.

The limited number of sampling sites restricted our ability to identify significant environmental differences between habitat types, but our results suggest several trends, especially when comparing between the disturbed habitats (HDF and PLE) and the protected habitat (DDF). Specifically, the disturbed sites showed a trend towards higher availability of water associated habitats such as rice paddies, streambeds, and ponds. While we did not attempt to identify habitat characteristics that correlated with higher herpetofaunal abundance and species richness due to sample size, other studies have identified several microhabitat correlates such as canopy cover, leaf litter cover, temperature, and, distance to streams (Urbina-Cardona et al. 2006; Wanger et al. 2010).

Three species, *Fejervarya limnocharis*, *Microhyla fissipes* and *Microhyla pulchra*, constituted over 75% of all amphibian captures. Each of these three species show tolerance to habitat disturbance and can breed in temporary pools in disturbed areas (Heyer 1973; Inger and Colwell 1977). As we only sampled for two months, it is likely that rare species are underrepresented or missing in our study. The disturbed habitats showed a trend towards higher availability of water associated habitats such as rice paddies, streambeds, and ponds. Another study from Thailand reported similar results with higher amphibian abundance in agricultural areas (Phochayavanich et al. 2008), but the researchers only sampled along stream habitats and did not look at sites further away from the permanent water source. Our results compliment these previous findings, by indicating that fossorial amphibian communities can also persist in small forest patches embedded in agricultural landscapes along with riparian areas.

Some of the water availability in the disturbed habitats can be attributed to human disturbance through agricultural practices such as the maintenance of man-

Table 3. Model comparison of amphibian and reptile abundances using GLMM with plot as a random effect. The variables compared are the number of parameters (K), the Aikaike Information Criterion corrected for small samples (AICc), the difference in AICc from the top performing model (ΔAICc), the weight for support of the model (ω), the cumulative weight (cumulative ω), and the log likelihood.

Taxonomic group	Model set	K	AICc	ΔAICc	ω	Cumulative ω	Log likelihood
Amphibians	~ month + habitat type	5	145.75	0	1	1	-66.11
	~ month	3	157	11.24	0	1	-74.87
	~ habitat type	4	200.17	54.42	0	1	-94.98
	(Null)	2	210.96	65.21	0	1	-103.18
Reptiles	~ habitat type	4	113.68	0	0.54	0.54	-51.73
	~ month + habitat type	5	114.52	0.84	0.36	0.9	-50.5
	(Null)	2	117.95	4.26	0.06	0.96	-56.67
	~ month	3	118.9	5.22	0.04	1	-55.82

Table 4. Comparison of Shannon-Wiener diversity index of reptiles and amphibians in each sampled habitat type in northeastern Thailand. Abbreviations are ANSO = average number of species observed and ASDI = avearge Shannon Diversity Index.

	Amphibians		Reptiles	
Forest Type	ANSO	ASDI	ANSO	ASDI
Dry dipterocarp forest	0.25 (0–1)	N/A	5.25 (4–6)	1.47 ± 0.23
Highly disturbed forest	5.75 (4–9)	1.36 ± 0.16	4.75 (3–6)	1.51 ± 0.58
Eucalypus plantation	4.25 (0–7)	1.08 ± 0.75	7 (4–9)	1.66 ± 0.23

made reservoirs and irrigation canals. The protected dry dipterocarp forest also occurs at a higher elevation suggesting that the disturbed sites benefit from increased water accumulation. Bickford et al. (2010) found that breeding site heterogeneity provided the strongest predictor for amphibian abundance and species richness in Singaporean forest fragments. These findings support our inference that the difference in water availability between the disturbed and protected sites may account for the stark contrast in amphibian captures.

The higher availability of water in the disturbed habitats, may have also inflated capture rates. Marsh et al. (2000) determined that inter-pond distance contributed to male site fidelity in tungara frogs, with lower site fidelity when breeding ponds were closer together. As passive trapping relies on animal activity, higher capture rates in the disturbed habitats may be attributed to high water resource density. We observed a single site in the HDF that captured more amphibians than all other sites combined. The outlier site differed from the others because it was surrounded by rice paddies rather than the abundantly cultivated drier crops such as cassava and sugar cane, supporting the claim that resource availability may influence amphibian capture rates.

While water is available at different sources (i.e., agricultural canals, rice paddies, and reservoirs) within the disturbed habitats, the protected forest is limited to streams and seasonally available temporary pools. Kaensa et al. (2014) investigated the difference between unprotected and protected forest habitats in upper Northeast Thailand, revealing that unprotected forest habitats had lower abundance compared to similar protected forest habitats. The results directly conflict with our findings, but they also identified that woodland habitats sites in protected areas had the lowest amphibian captures. A previous study within the Sakaerat Biosphere Reserve found that within the undisturbed forest areas amphibians were limited to riparian galleys (Inger and Colwell 1977). When examining similar dry tropical forests in the Western Ghats of India, Vijayakumar et al. (2006) found evidence suggesting amphibian distributions are clustered around riparian habitats. Our results may be skewed as amphibians cluster around streambeds in protected forests and because we did not adequately sample for arboreal species. More intensive sampling within various micro-habitats of the dry dipterocarp forest may reveal different amphibian abundance and species richness than we observed.

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 can thrive in partially disturbed habitats (Wanger et al. 2010); however, our results also show that the slopes of the sample-based rarefaction for reptiles did not reach an asymptote and further samples may provide a more comprehensive comparison between forest types as the accumulation curve levels off. Because our study was confined to the dry season, these results are not surprising. Within the Sakaerat Biosphere Reserve, Suttanon and Lauhachinda (2008) found 45 reptile species in the DDF, over double the amount we observed, but found much higher species richness during the rainy season (61 species) than the dry season (38 species). In contrast, one study from the nearby Khao Yai National Park reported higher amphibian abundance in the dry season (Kongjaroen and Nabhitabhata 2007). Sampling across multiple seasons could provide information on seasonal shifts and a more accurate picture of the species richness of an area. Additionally, we captured a relatively high number of snake species, but few individuals compared to other reptiles, which may also account for the steep rarefaction curve. Snakes in particular are difficult to obtain adequate captures to analyze with current statistical methods (Steen, 2010; Durso et al. 2011). Although the proportion of captures is low, snakes are still an important taxon to include in the analyses, especially when estimating richness and diversity.

The higher capture rates in the *Eucalyptus* plantations compared to the natural forest does suggest that at least some species can use these habitats), but the biggest difference in reptile abundance between habitat types derives from the two most abundant species found across all habitats, *Dixonius siamensis* and *Eutropis macularia*. Both species are widely distributed throughout Thailand (Chan-ard et al. 2015) and appear to be tolerant of anthropogenic disturbances). Their taxonomy has yet to be satisfactorily resolved, with the possibility that they may constitute species complexes (Ota et al. 2001). The yet undescribed diversity within these cryptic species may result in different responses to anthropogenic habitats (Bickford et al. 2007) and so hinders our ability to extrapolate our conclusions.

Sampling methodology can lead to significantly different community composition results, and studies comparing methods often fail to assess the underlying bias in all methods (Rodda et al. 2001). Results from passive trapping methods depend upon activity patterns for species in each habitat type, which likely influences the observed patterns. Accounting for habitat specific factors, and how they may influence capture rates, may prove beneficial for generating more accurate representations of herpetofaunal communities. Within *Eucalyptus* plantations many shelter sites, such as large rocks and fallen logs, are removed as part of the planting and clearing process. Zani et al. (2009) documented that Side-blotched Lizards (*Uta stansburiana*) exhibited behavioral changes in response to distance from refuge sites. While we did not record shelter site density at each site, we suggest future studies include this aspect as lower density of shelters sites may increase reptile activity leading to inflated capture rates.

Our results provide important baseline data for herpetofaunal species including recording a species previously unknown to be present in agricultural habitats, *Kaloula mediolineata*. The results represent preliminary trends and raise further questions concerning the assemblage of Southeast Asian herpetofauna and their responses to anthropogenic activity. Within our study areas, we suggest focusing on what habitat and landscape factors, such as breeding site availability and patch isolation, which influence habitat suitability in an agricultural mosaic. Additionally, we suggest looking at seasonal variation in both amphibian and reptile abundance and diversity within these habitats. Understanding how seasonality effects herpetofauna in human modified habitats could provide important information on how natural patterns shift in a changing environment.

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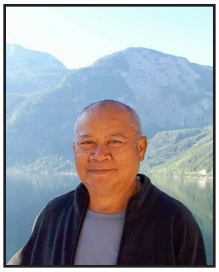

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