Madagascar Terrestrial Program

Nosy Be, North West Madagascar

MGF Phase 163 Science Report

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1. Introduction

Madagascar has been designated as a global biodiversity hotspot due to its exceptional concentrations of endemic species (Myers et al., 2000; Ganzhorn et al., 2001). The island has been heralded as 'the single highest biodiversity conservation priority in the world' (Myers et al., 2000). Roughly 90% of Madagascar's endemic species are forest dwelling and are highly susceptible to anthropogenic disturbance (Dufils, 2003). The island's unique biota is currently being threatened by high levels of deforestation and environmental degradation (Myers et al., 2000; Ganzhorn et al., 2001). There is much debate over the exact percentage of historical forest cover still remaining (Dufils, 2003; Quéméré et al., 2012), but it is believed that over 90% has been destroyed. Analysis of satellite imagery has shown that 40% of contemporary forest cover was lost between 1950 and 2000 (Harper et al., 2007). Despite increased environmental awareness in recent decades, deforestation has continued, with 0.9% of remaining forest lost annually from 1990 to 2000 (Scott et al., 2006; Irwin et al., 2010). Clearance for agriculture remains a major driver of forest destruction, as well as extraction of timber for fuel and building materials (Irwin et al., 2010).

Species extinctions have substantially reduced the native vertebrate community over the past 10,000 years (Dewar, 2003). Although it is difficult to pinpoint the mechanisms for these extinctions, it is likely that the rate has increased since human colonisation around 2,000 years ago (Dewar, 2003). Anthropogenic habitat destruction and fragmentation are considered to be the most important factors driving extinction in Madagascar (Andreone et al., 2005). Deforestation resulting in habitat loss, degradation and fragmentation has a detrimental effect on species diversity (Goodman and Rakotondravony, 2000; Ramanamanjato and Ganzhorn, 2001). The ability of a species to tolerate or exploit human modified landscapes will determine their persistence and future survival (Scott et al., 2006). In order to mitigate the deleterious effects of deforestation it is essential for conservation purposes to identify susceptible species. Such information will aid in determining communities for conservation priority and could assist in the development of effective land use and management practices. Madagascar National Parks (MNP, formerly ANGAP) currently manage 46 protected areas at 44 sites, covering an area of 170,000 km², with the aim of preserving a broad spectrum of Madagascar’s biota. However, there is a further 40,246 km² outside reserves (Randrianandianina et al., 2003), which contains substantial latent biodiversity (Ingram and Dawson, 2006).

The Sambirano domain is Madagascar’s youngest biome, distributed around the Sambirano River and the town of Ambanja, in northwest Madagascar. This humid forest habitat developed approximately eight million years ago, when a climatic shift caused an increase in rainfall (Wells, 2003). The sambirano forest domain is characterised by a climax community of lowland forest with a shared species composition relating to both Eastern and Western Madagascar domains. This is due to the close geographical proximity and similar climates of the regions, but an additional component of the taxa is endemic (Du Puy and Moat, 2003). The region experiences annual rainfall of 2,250 mm (Wells, 2003), with a short dry period of three to four months (Legris and Blasco, 1965) and mean annual temperatures of 28°C, ranging from 15°C to 35°C (Andreone et al., 2003). The Sambirano domain was historically a belt of unbroken forest running from North to South Madagascar; however it is now limited to the northern end of the central mountain range. Due to its restricted area and high levels of human disturbance, the Sambirano region is considered one of the most endangered habitats in Madagascar (Langrand, 1990).
Nosy Be, Madagascar’s largest offshore island, is situated in the northwest and falls within the Sambirano domain. It was connected to the mainland as recently as 8,000 years ago, when rising sea levels isolated the island and a number of smaller outcrops in the Nosy Vorona Bight (Goodman and Benstead, 2003). Analysis of the biodiversity of small islands, such as Nosy Be, is important for identifying areas of species endemism. These studies also increase our understanding of historical patterns of animal and plant dispersal and distribution, due to the fact that extinctions occur at an accelerated rate on small islands because of their limited size and reduced recruitment (Andreone et al., 2003). The island is an important centre for agriculture and tourism and large parts have been deforested for agriculture. Rice paddies, coffee, ylang-ylang and sugar cane plantations now form the largest component of vegetation, with only a small fragment of the original forest remaining (Andreone et al., 2003). This protected area represents one of the last remaining relatively undisturbed areas of Sambirano forest (Andreone et al., 2003). As a result, Nosy Be is the type locality of many of the floral and faunal species of the historic Sambirano domain. The area represents a high research priority with much of its unique biota in need of cataloguing.

2. Survey Sites

The Frontier Madagascar terrestrial research programme is situated in the village of Ambalahonko, Nosy Be (Figure 1), bordering Lokobe National Park. Twelve research sites were selected along a gradient of human disturbance, based on the time since they were last cleared and their current land usage. The sites range from primary forest that has never experienced clearing and has low levels of anthropogenic disruption, to those that are cleared annually for agricultural purposes. All research sites are located in close proximity to the village and are accessible via the beach, except at times when the tide is high. The primary survey sites are towards the west, near the Lokobe National Park, whilst the most disturbed sites are those nearest Ambalahonko and neighbouring Antafondro. A distance of about 5 km separates the survey sites. To account for the effect of human disturbance and habitat degradation on vertebrate communities, sites were selected carefully and chosen for their contrasting histories.
There are nine sites that are currently used for reptile surveying, with three being designated as primary, three as secondary, and three as degrade (Figure 2). The transects within these sites are all 200m in length and follow existing trails through the forest. The primary transects are all located close to the border with Lokobe National Park. There is one secondary forest transect which begins along the beach. The other two secondary forest transects are located within an area designated as a communal protected park by the Fokotany of Ampasipohy. The degraded transects are all located close to the village of Ambalahonko, in forested areas that undergo regular clear felling and disturbance. There are three transects that are presently used for bird and butterfly surveys. Transects within these sites are all 400m in length and utilise existing pathways. Two of the transects are within forested areas representing both primary and secondary habitats. The degraded transect is located in open habitat, primarily consisting of plantations. A map showing the locations of the sites is given in figure 2 and a summary of each site in table 1.

![Figure 1: Location of Nosy Be in relation to mainland Madagascar (top left and top right) and the position of the MGF base camp within the village of Ambalahonko (red dot), next to the Lokobe Integral Reserve.](image)

![Figure 2: Location of survey sites in relation to Ambalahonko and Lokobe National Park. Red indicates reptile transects, and yellow bird/butterfly transects. Image taken from Google Earth (2014).](image)
Table 1: An overview of each of the survey sites, describing their exact locations and their present habitat use.

<table>
<thead>
<tr>
<th>Site Code</th>
<th>GPS Location</th>
<th>Survey Type</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>BP</td>
<td>24°48’’ 20°2’’</td>
<td>Reptile</td>
<td>Primary mature forest, running along the border of Lokobe National Park. The trail is kept free of vegetation to mark the boundary. Steep in places.</td>
</tr>
<tr>
<td>GG</td>
<td>24°40’’ 20°8’’</td>
<td>Reptile</td>
<td>Predominantly primary mature forest, with many Ravenala palms. Steep in places.</td>
</tr>
<tr>
<td>LKL</td>
<td>23°52’’ 20°28’’</td>
<td>Reptile</td>
<td>Secondary closed canopy forest, located in Ampasipohy. Dense ground vegetation. Part of transect runs along border of Lokobe National Park.</td>
</tr>
<tr>
<td>SS</td>
<td>24°12’’ 20°30’’</td>
<td>Reptile</td>
<td>Young secondary regrowth, primarily used for timber extraction. Located on hillside</td>
</tr>
<tr>
<td>PP</td>
<td>24°7’’ 20°57’’</td>
<td>Reptile</td>
<td>Located along a main pathway. Recently, clear felled recently for charcoal production and agriculture.</td>
</tr>
<tr>
<td>LL</td>
<td>24°26’’ 20°59’’</td>
<td>Reptile</td>
<td>Located along a main pathway. Recently, clear felled recently for charcoal production and agriculture.</td>
</tr>
<tr>
<td>ZB</td>
<td>24°31’’ 20°18’’</td>
<td>Bird and Butterfly</td>
<td>Closed canopy forest, which has not been heavily cleared for many years. Accessed via beach. Some individual trees have been recently removed for timber</td>
</tr>
<tr>
<td>TS</td>
<td>24°20’’ 21°0’’</td>
<td>Bird and Butterfly</td>
<td>Pathway through mixed plantations</td>
</tr>
</tbody>
</table>
3. The effect of habitat degradation on reptile communities

Introduction

Reptiles are in a state of dramatic decline worldwide (Gibbons et al., 2000). In Madagascar the primary cause of this decline is through the destruction of habitats (Andreone et al., 2005). Much of island’s forests are still under threat, with deforestation reducing and fragmenting available habitat. The reptiles of Madagascar are highly diverse, with 91% of the species being endemic (Raxworthy and Nussbaum, 1994). Despite the importance of studying and conserving this taxa, relatively little is known and new species are routinely being described. There are around 60 species of reptile recorded from Nosy Be, although some of these may be ambiguous or now extirpated. Roughly 80% are considered to be found in Lokobe National Park (LNP), whose 740ha only covers around 3% of the total area of the island (Andreone et al., 2003).

There are several species which are also thought to exist only within the boundaries of LNP itself (Andreone et al., 2003). For this reason, the importance of maintaining this small protected area and its surrounding buffer zone of vegetation is high. Although there has been previous research work conducted on the herpetofauna of Nosy Be (e.g. Glaw and Vences, 2007; Andreone et al., 2001; Andreone et al., 2003; Andreone et al., 2005), this has so far been primarily cataloguing of the species present. There is still much work to be done from an ecological, conservation and genetic standpoint. Irwin et al., (2010) identified the need for a greater understanding of the patterns of anthropogenic change on Madagascan species. The paper highlighted the importance of being able to predict the outcomes of continued deforestation and disturbance on species richness and abundance. It is expected that diversity will be reduced by anthropogenic disturbance, with more generalist and tolerant (and potentially invasive non-endemic) species able to expand their populations in response to habitat modification (Glaw and Vences, 2007).

The high level of diversity and endemism of animals and plants in Madagascar is well documented (Myers et al., 2000; Ganzhorn et al., 2001). The comprehension that understanding of biogeographic patterns is crucial to assess conservation priorities has led to an increased interest in floral and faunal inventories. However, most studies have been conducted in protected areas on the mainland (Raxworthy and Nussbaum, 1995; Raxworthy et al., 1998; Raselimanana et al., 2000), while relatively few data are available for non-protected and secondary habitats in Madagascar (Andreone et al., 2000). Even less is known about the herpetofauna of the Malagasy offshore islands. These may be important centres of endemism and studies could aid in the understanding of historical patterns of reptile distribution and dispersal (Andreone et al., 2003). Moreover, islands represent a particular conservation priority as their limited size may accelerate extinction. This has previously been described for lemurs and birds by Goodman (1993), where human disturbance and intensive hunting has caused the extinction of the snail-eating coua bird and the disappearance of native lemurs at Sainte Marie. Only limited data exists in regards to the effect of habitat degradation on reptile abundance and species richness on Malagasy offshore islands (Andreone et al., 2003). A lack of information on how reptilian responses to habitat disturbance is a major impediment to effective conservation in Madagascar.
The study area surrounding the village of Ambalahonko and the Lokobe buffer zone is well suited to conducting studies of this nature as it comprises a variety of habitats with varying levels of disturbance within close proximity to one another. A portion of this area close to the border is composed of primary and secondary vegetation, which is contiguous with the park itself, therefore it is important to study and protect. Small-scale deforestation is still ongoing in the area (pers. obs. 2016), and further protective measures are required in order to safeguard this small fragment of intact forest. This study aims to collect data on the abundance of reptiles, assessing species richness and diversity between forests which have experienced various levels of anthropogenic disturbance, in order to determine the effects of habitat loss on reptile assemblages.

Aims and Objectives

Aim: To monitor the effects of habitat disturbance on the reptile diversity and abundance between primary, secondary and degraded habitats.

Objectives:
1. To compare reptile abundance between different habitat types from visual encounter surveys.
2. To compare reptile species richness between different habitat types from visual encounter surveys.
3. To examine the relationship between environmental factors (rainfall and temperature) and reptile abundance.

Methodology

Reptile Community

An assessment of reptile abundance and species richness was conducted along 200m transects using visual encounter surveys (VES) in line with previous assessments made for reptile surveying in the tropics (Doan, 2003; Jestrzemski et al., 2013). Nine transects were selected for reptile surveying. As forest areas are owned by local citizens, pre-existing trails were utilised to avoid demarcating private property. Transects were designated as either primary, secondary or degraded habitat based on the level of human disturbance. Two transects were devoted to each habitat type. The VES searching method is not a time-constrained technique (Doan, 2003). Researchers walked each transect using as much time as was needed to thoroughly examine each area (Jestrzemski et al., 2013). Observers kept noise and other disturbances to a minimum, in order to prevent scaring animals away before they had been encountered. During surveys reptiles were identified at the species level. In circumstances where species identification was not possible, differentiation was conducted at the genus level.
Statistical Analysis

Reptile species diversity among habitat types (primary, secondary and degraded) was assessed via the Shannon diversity index ($H'$).

$$H' = - \sum_{i=1}^{n} p_i \ln p_i$$

Where: $p_i$ is the proportion of the sample that belongs to the $i$th species (Stirling, 1999). It generally varies from 1 (low diversity) to 5 (high diversity) (Gering et al., 2002). The index takes into account both the number and evenness of species. The diversity value is increased through a greater number of species combined with a more even distribution (Stirling, 1999).

Analysis of reptile species richness was carried out in EstimateS, using rarefaction curves to produce smoothed curves, and the Chao II estimations of species richness were used to predict the likely number of species if curves reach asymptote.

The Shapiro-Wilk Test has been labelled as the preferred test for normal distribution due to its good power properties (Medes and Pala, 2003). It is recognised as the first statistical procedure to detect departures from normality due to skewness (Razali and Wah, 2011). The Shapiro-Wilk Test was selected to assess if variables had normal distributions as the procedure is regarded as optimal for sample sizes less than 2000 (Razali and Wah, 2011).

Kruskal-Wallis $H$ tests were used to determine whether a significant difference exists among the habitat types (primary, secondary and degraded). The Kruskal-Wallis $H$ test is a non-parametric test used to determine whether three or more independent groups are the same or different on some variable of interest. It is used in situations where an ordinal level of data or an interval or ratio level of data is available (Chan and Walmsley, 1997).

To assess the relationship between fragmentation characteristics and relative reptile abundance, the study used Spearman’s rank correlation coefficient. Spearman’s rank correlation coefficient is a non-parametric test and does not require variables to be normally distributed (Kranzler, 2003c). The correlation coefficient was used to determine if a relationship existed between reptile abundance and environmental factors such as rainfall and temperature.

Shapiro-Wilk, Spearman’s rank correlation coefficient and Kruskal-Wallis Test was conducted using SPSS (IBM v17.0, 2016). All tests were two-tailed, and statistical significance was set at $p < 0.05$.

Results

A total of 95 visual encounter surveys were completed across the course of the phase (July-September), encountering 440 individuals across 22 unique species. Across all habitats, chameleons accounted for 43.9% of all encountered species, ground lizards - 19.1%, geckos – 33.9% and snakes only 3.2% of species during this survey period.
Kruskal-Wallis analysis between habitat groups showed no significant difference between mean primary and secondary habitat species abundance (primary, mean – 5.23 ± 1.74 SE; secondary, mean – 4.72 ± 2.24 SE; p>0.05). Degraded habitat showed a statistically significant difference in mean abundance between both primary and secondary habitats (mean – 2.39 ± 0.91 SE ;p<0.05).

Figure 3. Mean abundance of reptiles in three habitat types (n= 440).

Figure 4. Rarefactions curves highlighting species richness for primary, secondary and degraded habitats. Chao II richness estimates have also been included to show the potential projected species richness increase over time to reach their asymptote.
Analysis of rarefaction curves (fig 4.) showed statistically different richness estimates for each of the habitats, primary, secondary and degraded. At respective equivalence points (maximum sampling intensity) degraded was lowest (6 spp.), primary (8 spp.) with secondary the highest (11 spp.)

Chao II estimates of species richness showed potential species richness of each habitat as degraded (6.48), primary (8.59) and secondary (18.78), where estimates are larger than identified species richness suggest that curves have not reached their asymptote.

Table 2. Number of species observed within each habitat with associated $H'$ index (Shannon diversity index). Species unique to either primary, secondary or degraded are also listed.

<table>
<thead>
<tr>
<th>Species Observed</th>
<th>Primary</th>
<th>Secondary</th>
<th>Degraded</th>
</tr>
</thead>
<tbody>
<tr>
<td>$H'$</td>
<td>1.95</td>
<td>2.17</td>
<td>1.89</td>
</tr>
<tr>
<td>Unique Species</td>
<td>Zonosaurus subunicolor</td>
<td>Brookesia minima Callula nasutum Paraedura ovicaps Mimophis mahfalensis Stenophis granuliceps</td>
<td>Trachylepis gravenhorsti</td>
</tr>
</tbody>
</table>

Primary (n=15, $H' = 1.95$) and degraded (n= 13, $H' = 1.89$) habitats showed similar numbers of species observed, also with similar $H'$ indices. Secondary habitat (n=20, $H' =2.17$) contained more species and consequently, a greater diversity index than its compared habitats.

**Discussion**

Despite visual encounter surveying being a widely used and accepted method of surveying (Andreone et al., 2003; D'Cruze et al 2008), not all species will be surveyed with equal efficiency as their appearance or behaviour will differ. Species living in high canopy or amongst the leaf litter will be particularly under-represented. Those which are sensitive to the presence of researchers may be more likely to flee or hide, despite best efforts to minimise disturbance, and is presumably more likely to occur in more pristine habitats (D'Cruze et al., 2008). Structural complexity also tends to mean species are more difficult to spot in less open habitats. The aim of not restricting surveys to a time limit was intended to try to account for this, as a correspondingly larger times needs to be spent searching dense vegetation.

Reptile abundance across all three habitats showed a decreased abundance in the mean number of individuals found per transect, likely due to increased levels of environmental degradation (Primary, Secondary, Degraded). Primary and secondary forest represented the highest mean abundances (5.23± 1.74 SE; 4.72 ± 2.24 SE) with degraded habitat significantly lower (2.39± 0.91 SE). Degraded habitats were found to be significantly different to both primary and secondary forest, suggesting that the degradation taking place in this habitat...
through deforestation and agriculture is having a significantly greater effect on abundance than in primary and secondary forests. This difference however is not supported by results from previous phases (Lorenti, 2015; Lorenti, 2016), which have shown degraded to have a higher mean abundance or no statistically significant differences between them. This may be explained by the increased degree of deforestation taking place through surveyed areas which has progressively increased, removing suitable refuges from degraded habitats which is known to reduce reptile abundance and richness (Irwin et al., 2010; Gibbons et al., 2000). Weather conditions during this phase have been characterised by cooler temperatures, shorter days and very little rainfall, which are likely to have impacted periods spent by reptiles in brumation (Abe, 1983), a winter dormancy state similar to mammalian hibernation (McEachern et al., 2015). Increased onset on brumation brought on by this weather cycle is likely to have impacted the abundance of reptiles being observed on surveys.

Primary and degraded habitat were each found to have one unique species (Zonosaurus subunicolor) and (Trachylepis gravenhorsti) respectively. Secondary however was found to have 5 unique species (Brookesia minima, Calluma nasutum, Paradoxura oviceps, Mimophis mahfalensis, Stenophis granulicpes). This increased richness is also supported by habitat shannon-diversity indices (Table 2.) with secondary showing the highest levels of species richness. Although these results are representative of surveys conducted during the phase, many species were observed in different habitats outside of survey periods during this phase suggesting longer surveys periods or more comprehensive transect coverage of habitats is required to fully reflect the diversity in each habitat. Low detectability of certain species, such as snakes, allows for the misinterpretation that certain species cannot tolerate or are specialised to habitats which may not be the case. While secondary forest showed the highest species diversity index, all three habitats showed relatively low diversity (1.89-2.17) overall, with the index considering score less than 2, as low (Gering et al., 2002). These results are supported by previous research showing that areas outside of Lokobe national park are heavily influenced by anthropogenic factors and have a lower overall diversity (Andreone et al., 2013).

Species were compared across habitats using rarefaction curves to predict species richness estimates using Chao II. Both primary and degraded habitat were shown to be approaching asymptote, suggesting that this survey period detected most species within available habitats. Secondary however, deviated significantly from the predicted species richness curve suggesting greater survey effort is required to observe the full diversity of the habitat. However, simply comparing sites by species richness, or other diversity measures, does not account for differences in community composition which are likely to occur between habitat types (Andreone et al., 2000). Different habitat types may contain a similar number of species, but they are not necessarily the same ones. This is an important point when considering the effects of disturbance upon ecological communities. It is likely that species more sensitive to disturbance, or with more specialised niches, are unable to compete with more generalist species which essentially replace their role within the ecosystem (Andreone et al., 2003). It is therefore important for studies to examine the presence or absence of each species for each research site.
4. The effect of habitat degradation on bird communities

Introduction

The avifauna of Madagascar is often considered species-poor, hosting just 258 species (204 breeding species), when compared to other landmasses of equivalent size (e.g. Borneo hosts 688 species) (Goodman et al., 2003). Despite its low bird diversity, nearly half of the species found in Madagascar are endemic (115 species) (Morris and Hawkins, 1998), meaning the fauna are still significant and have high ecological value. Another striking feature of Madagascan avifauna is the high degree of specialisation found within endemic lineages, most notably their dependence on forest environments, with 80 of the 115 endemic species (representing 37 endemic genera) being restricted to forest habitats (Morris and Hawkins, 1998).

There has been little previous research conducted on the avifauna of Nosy Be. As the island is close to the mainland, it is relatively easy for birds to disperse to and from the island, and as such there are no species currently recognised as being endemic to the region. Virtually all the species known to be present on the island have very large ranges and are of less conservation concern unless their abundances are low across the range (IUCN, 2014). The small area of protected forest in Lokobe National Park and its surrounding buffer zone harbours many species of bird. These are species not found in secondary or degraded habitat and their association with forest habitat increases the likelihood that they are of high functional value. Such information can be used to determine which species are more likely to be locally extirpated, should deforestation of the buffer zone continue.

Aims and objectives

Aim: To examine the effect of habitat degradation on bird species richness and abundance.

Objectives:
1. To compare bird abundance between different habitat types from point count surveys.
2. To compare bird species richness between different habitat types from point count surveys.

Methodology

Survey Methodology

Bird surveying was carried out using point counts along a 400m transect, with a point count station positioned every 200m along this transect (Gregory et al., 2004; Hutto et al., 1986). Point counts are used to explore bird habitat relationships and to determine species richness and abundance (Ralph et al., 1995). Bird species were identified by species based on both auditory and visual confirmations. Surveys in heavily vegetated habitats relied almost exclusively on auditory detections (Faanes and Bystrak 1981; Scott et al., 1981). A sampling period of five minutes was used at each point count station. All bird species within an estimated radius of 25m from the observers were recorded (Hutto et al., 1986; O'Dea et al., 2004). Birds which were heard and then subsequently seen were recorded as seen, and those
located outside of the 25m radius and subsequently entered it were recorded as being within the point count radius. Surveys were conducted in the morning, between 0630 and 0730 in order to record bird abundance at peak activity times.

**Statistical Analysis**

Bird species diversity among habitat types (open and closed forest) was assessed via the Shannon diversity index ($H'$).

$$H' = - \sum_{i=1}^{n} p_i \ln p_i$$

Where: $p_i$ is the proportion of the sample that belongs to the $i$th species (Stirling, 1999). It generally varies from 1 (low diversity) to 5 (high diversity) (Gering et al., 2002). The index takes into account both the number and evenness of species. The diversity value is increased through a greater number of species combined with a more even distribution (Stirling, 1999).

Analysis of bird species richness was carried out in * EstimateS*, using rarefaction curves to produce smoothed curves, and the ACE estimations of species richness were used to predict the likely number of species if curves reach asymptote.

The Shapiro-Wilk Test has been labelled as the preferred test for normal distribution due to its good power properties (Medes and Pala, 2003). It is recognised as the first statistical procedure to detect departures from normality due to skewness (Razali and Wah, 2011). The Shapiro-Wilk Test was selected to assess if variables had normal distributions as the procedure is regarded as optimal for sample sizes less than 2000 (Razali and Wah, 2011).

Kruskal-Wallis $H$ tests were used to determine whether a significant difference exists among the habitat types (primary, secondary and degraded). The Kruskal-Wallis $H$ test is a non-parametric test used to determine whether three or more independent groups are the same or different on some variable of interest. It is used in situations where an ordinal level of data or an interval or ratio level of data is available (Chan and Walmsley, 1997).

Both the Shapiro-Wilk Test Kruskal-Wallis conducted using SPSS (IBM, v.17, 2016). All tests were two-tailed, and statistical significance was set at $p < 0.05$

**Results**

Variation in the mean number of bird species found per transect is shown in figure 5.
Figure 5. Mean number of individuals found in two differing habitat types (n=921) per transect.

Kruskal-Wallis analysis between closed forest and open habitat reveal a significant difference (Kruskall-Wallis H= 8.88, df=1, p <0.05). The mean number of birds per transect was highest in open habitat at 10.02 ± 4.79 (mean ±SE). Closed forest showed a mean abundance of 4.45 ± 1.55 (mean ± SE).

Figure 6. Rarefaction curves showing species richness for closed forest and open habitats. Dashed lines indicate species richness estimates, showing potential species richness and projected time until their asymptote is reached.
Analysis of species richness using rarefaction between habitat types (Fig.6) showed significant differences between the sites. At their respective equivalence points, closed forest (16) had a significantly lower species richness than open habitat (20.58). Projected species richness estimates place comparable equivalence points at closed forest (20) and open habitat (21.82) suggesting considerably more comparable asymptotes between habitats.

Species observed in each habitat, along with associated shannon-diversity indices are shown in table 3. Unique species encountered in each habitat are also shown. Species encounters were only used if the species was encountered on the transect and does not include casual encounter observations.

Table 3: Total number of bird species recorded in each habitat type. Shannon diversity index values are denoted by $H'$. List of species unique to either primary, secondary or degraded habitat.

<table>
<thead>
<tr>
<th></th>
<th>Closed Forest</th>
<th>Open Habitat</th>
</tr>
</thead>
<tbody>
<tr>
<td>Species Observed</td>
<td>18</td>
<td>20</td>
</tr>
<tr>
<td>$H'$</td>
<td>2.23</td>
<td>1.89</td>
</tr>
<tr>
<td>Unique Species</td>
<td>Ashy cuckoo shrike Chaberts vanga Madagascar bee-eater Madagascar white-eye</td>
<td>African palm swift Alpine swift Cattle egret Long billed green sunbird Madagascar hoopoe Madagascar kestrel</td>
</tr>
</tbody>
</table>

Discussion
Quantifying the species diversity of bird communities has gained increasing importance in environmental impact assessments, conservation planning (Bibby et al., 1992; Stotz et al., 1996), and ecological research (Huston, 1994). The precise mechanisms for the creation and maintenance of avian species diversity are still being debated, although there is a general consensus that variables related to habitat heterogeneity play a prominent role (Kissling et al., 2008). Different groups of organisms that show similar direct or indirect responses to environmental factors are expected to display species richness patterns that are spatially congruent (Kissling et al., 2008). Such patterns could have profound implications for biodiversity conservation and for assessing the effects of habitat modification on birds.

Degraded habitat shows an increased species richness over primary closed forest. This is contradictory to previous research indicating that anthropogenic disturbance results in a loss of diversity and abundance (Irwin et al., 2010; Watson et al., 2004). Despite increased species richness within the degraded habitat, a comparison of estimated species richness curves between the two habitats suggests more comparable species richness across the habitats. The observed increase in richness in degraded habitats is therefore difficult to explain. As the majority of species found within the Ambalahanoko region are generalists and exist within a wide variety of habitats, the observed increase in species richness may be a function of bias within the sampling between the two habitats (Sinclair and Langrand, 2013).
Biases are unavoidable when using the point count methods, there will be differences in
detectability in closed forest as opposed to open habitat. Birds may remain undetected by
researchers in dense vegetation and there is the potential for vocalisations to be too faint, as
sound attenuation in closed forest is reduced (Schiek, 1997). It may be the case that true
forest specialists, such as the Vasa Parrots (*Coracopsis spp.*) (Morris *et al.*, 1998), existed
historically on the island of Nosy Be, but have been extirpated, leaving only more tolerant
species. Surveying more pristine core forest areas on the edge of Lokobe National Park would
be desirable for further examination of the effect of habitat degradation on avian
communities.

Despite a higher shannon-diversity index, closed forest habitat had only 4 species which were
unique in appearing in this habitat type, the Ashy cuckoo shrike, Chaberts vanga, Madagascar
bee-eater and the madagascar white-eye. Open habitat however had 6 unique species, the
African palm swift, Alpine swift, Cattle egret, Long-billed green sunbird, Madagascar
hoopoe and the madagascar kestrel. The species observed in open habitat represented a
mixture of both forest specialist and grassland generalist species. This is not congruent with
the literature which suggests that disturbance generally reduces species diversity and
typically replaces forest specialists with grassland generalists (Irwin *et al.*, 2010). A study
conducted by Peh *et al.*, (2006) suggested that nearby primary forest could act as a source
habitat, resulting in a steady influx of forest birds to degraded areas Thus, the bird
community we have observed may not be representative of the forest avifauna in degraded
lands that have no primary forests nearby. The results may also be biased due to the inherent
difficulties sampling in closed forest. Future studies should include a larger survey effort data
that will increase the chances of encountering those species with very low levels of
detectability.

5. **The butterfly diversity of the Ambalohonko Region**

**Introduction**

Butterflies are often used as indicator species as they are conspicuous and relatively
easy to identify and are far better documented than other invertebrate taxa. Madagascar has
been used previously in case studies for sampling tropical butterflies which provides a useful
means of assessing overall invertebrate diversity (Kremen, 1992; Kremen, 1994). There are
currently around 297 species of butterfly described from Madagascar, with 210 (70%) of
these being endemic (Sáfián *et al.*, 2009).

Previous studies into *Lepidopteran* diversity and abundance have shown that habitat
modification and loss have significant detrimental effects (Lawton *et al.*, 1998). Butterflies
are known to be good indicator species as they are extremely sensitive to minor changes
within their micro-habitats, especially with regards to altered light levels (Kremen, 1992).
They are therefore useful tools in assessing and monitoring the levels of habitat degradation
through forest clearance, reacting rapidly to any changes in light concentrations associated
with the loss of canopy cover (Steer *et al.*, 2009). On the nearby Comoros Islands, a study
conducted on the effects of habitat loss on the *Lepidopteran* communities showed that more
geographically widespread species replaced species which were more specialised and had
smaller distributions, in some cases restricting their distributions further and causing some
species to become endangered (Lewis et al., 1998). Monitoring over a long period of time can allow for us to understand better how the butterfly assemblage changes over the course of the seasons (Kunte, 1997).

**Aims and Objectives**

**Aim:** To examine the effect of habitat degradation on butterfly diversity and abundance.

**Objectives:**
1. To compare avian abundance between different habitat types from surveys.
2. To compare avian species richness between different habitat types from surveys.

**Methodology**

**Butterfly Community**

Surveys were carried out along three 400m transects (ZB, MA, TS; see figure 2 for location on map, and table 1 for transect descriptions) between the hours of 1000 and 1400 at the time of peak butterfly activity (Steer et al., 2009). Butterflies were surveyed using a modified version of the Pollard walk (Pollard, 1977), similar to that used by Sparrow et al. (1994), where researchers would walk slowly along a transect, counting the numbers of each species observed and identifying them visually. Many of the species found in the area are difficult to identify on the wing (notably the families Hesperidae and Lycaenidae), any individuals that could not be positively identified in this way were captured using a sweep net and identified using a set of photographic plates. Nomenclature primarily followed that of Safian et al. (2009).

**Statistical Analysis**

Butterfly species diversity among habitat types (open and closed forest) was assessed via the Shannon diversity index ($H'$).

$$H' = - \sum_{i=1}^{n} p_i \ln p_i$$

Where: $p_i$ is the proportion of the sample that belongs to the $i$th species (Stirling, 1999). It generally varies from 1 (low diversity) to 5 (high diversity) (Gering et al., 2002). The index takes into account both the number and evenness of species. The diversity value is increased through a greater number of species combined with a more even distribution (Stirling, 1999).

Analysis of bird species richness was carried out in EstimateS, using rarefaction curves to produce smoothed curves, and Chao I estimations of species richness were used to predict the likely number of species if curves reach asymptote.

The Shapiro-Wilk Test has been labelled as the preferred test for normal distribution due to its good power properties (Medes and Pala, 2003). It is recognised as the first statistical procedure to detect departures from normality due to skewness (Razali and Wah, 2011). The Shapiro-Wilk Test was selected to assess if variables had normal distributions as the procedure is regarded as optimal for sample sizes less than 2000 (Razali and Wah, 2011).
Kruskal-Wallis $H$ tests were used to determine whether a significant difference exists among the habitat types (primary, secondary and degraded). The Kruskal-Wallis $H$ test is a non-parametric test used to determine whether three or more independent groups are the same or different on some variable of interest. It is used in situations where an ordinal level of data or an interval or ratio level of data is available (Chan and Walmsley, 1997).

Both the Shapiro-Wilk Test Kruskal-Wallis conducted using IBM SPSS, version 17.0. All tests were two-tailed, and statistical significance was set at $p < 0.05$.

**Results**

The mean number of individuals found in two habitats is shown in figure 7.

![Figure 7. Mean number of individuals found in two differing habitat types (n=646) per transect.](image)

Kruskal-Wallis analysis between closed forest and open habitat shows a highly significant difference between habitat groups (Kruskall-Wallis $H= 67.491, \text{df}=1, p < 0.05$). Open habitat had a significantly higher mean abundance of $13.19 \pm 7.38$ (mean $\pm$ SE) with closed forest at $0.55 \pm 0.26$ (mean $\pm$ SE).
Figure 8. Rarefaction curves showing species richness for closed forest and open habitats. Dashed lines indicate species richness estimates, showing potential species richness and projected time until their asymptote is reached.

Analysis of species richness using rarefaction between habitat types (fig. 8) showed significant differences between the sites. At their respective equivalence points, closed forest (7) had a significantly lower species richness than open habitat (16). Projected species richness estimates place species richness at the same equivalence points at closed forest (8.35) and open habitat (16), significantly different again.

Table 4: Total number of bird species recorded in each habitat type. Shannon diversity index values are denoted by $H'$. List of species unique to either primary, secondary or degraded habitat.

<table>
<thead>
<tr>
<th></th>
<th>Closed Forest</th>
<th>Open Habitat</th>
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</thead>
<tbody>
<tr>
<td><strong>Total Species</strong></td>
<td>7</td>
<td>15</td>
</tr>
<tr>
<td><strong>$H'$</strong></td>
<td>1.70</td>
<td>1.82</td>
</tr>
<tr>
<td><strong>Unique Species</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Graphium evombar evombar</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Eurema brigitta pulchella</td>
<td></td>
<td></td>
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<tr>
<td>Colotis evanthe</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Byblia anvatara anvatara</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Acraea ranavalona</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Acraea masamba</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Neptis kikdeli</td>
<td></td>
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<tr>
<td>Danaus chrysippus orientis</td>
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</table>
Discussion

Butterfly abundance was significantly higher in open habitat (13.19 ± 7.38) as opposed to closed forest (0.55 ± 0.26). However, rarefaction estimates suggest that open habitat has reached asymptote and richness is unlikely to increase with additional sampling effort. Closed forest, however, has not reached predicted asymptote and would likely increase in richness with additional sampling effort. Closed forest richness estimates however, suggest that the plateau of new species discovery is likely to still be significantly lower than that of the currently observed open habitat richness. An increased abundance and richness in open habitat may be a function of many environmental variables such as increased early sun exposure facilitating increased activity or the presence of plantation and agriculture habitat which is likely to be favourable to lepidopteran species. Changes in species abundance and diversity are likely to be linked to both the environmental and physical effects that human disturbance has on the ecological and life history traits of species (Cleary et al., 2009). Habitat degradation increases the frequency and size of gaps in the canopy which are liable to favour some groups of species while hindering others. The loss of biodiversity in anthropogenically modified habitats reduces the number of species with a restricted area distribution. These specialised species are of conservation importance because they are vulnerable and prone to extinction (Spitzer et al., 1997). Eurema brigitta pulchella and Acraea masamba were observed solely in open habitat and are known to thrive in forest margins as well as transformed grasslands (Henning et al., 1997).

Although the Pollard method (Pollard, 1977) reduces the chance of pseudo replication by moving continuously along a transect, observer bias has to be taken into account. This is because many of the species that inhabit Nosy Be are small and difficult to identify on the wing, particularly Heteropsis spp., and the families Lycaenidae and Hesperiidae. Heteropsis need to be closely examined as many are dull coloured with differing numbers and positions of eye spots. Lycaenidae and Hesperiidae consist of mostly very small, fast moving species, and also need to be examined closely in the hand to be able to differentiate them. The method attempted to account for this is through capturing species not identifiable on the wing, however, to catch all individuals by sweep netting is difficult, even by a very experienced researcher. Lycaedinae and Hesperiidae tend to stay close to vegetation, so it is often difficult to capture them without damaging the nets. This clearly represents an under sampling bias for these groups. The more enclosed environment within forested transects also restricts the ability to sweep net effectively in dense vegetation, so for species which cannot be identified on the wing there is likely to be further disparity in sampling between forested habitats and the more open scrubland and plantations of the degraded habitats. As mentioned above, it is also probable that forest specialist species spend more time in the canopy, close to sunlight and available food, and are thus under sampled in closed forest habitat (Dumbrell and Hill, 2005).

Future study will look to focus to diversity of species in varying sub-habitats within each major habitat (Closed habitat and open habitat). As such a significant differences has been consequitively detected (Lorenti, 2015; Lorenti, 2016), characterising in greater detail the variability to species richness will be focused on outside of strict linear transects within the Ambalahonko area thereby detecting the effective of degradation on fine scale species richness.
References


