Birds in humanized landscapes:

São Tomé endemic birds' response to agricultural intensification

By

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Abstract

Main aim Assessing how the replacement of agroforestry systems, by more open agricultural practices affects São Tomé's birds abundance, diversity and distribution.

Location Agricultural matrix and montane rainforest in the northeast end of Obo Natural Park in the mountainous centre of the island of São Tomé.

Methods Within the study landscape four different land-use types were selected: primary forest, shade coffee, shade polyculture and annual agriculture representing a gradient of agricultural intensity and a total of 105 count stations was spread across the landscape. Data on bird species was collected from May-July 2008 using different day repeated point counts and vegetation structure around each point count was recorded. Species composition among different sites was explored using non-metric multidimensional scaling and linear models were used to assess the relationship between community composition, diversity, similarity to forest and abundance of different bird groups to landscape and local habitat variables.

Results Species abundance and diversity change varied according to land use, with shade polyculture being the most species rich land-use type whereas the rainforest had the lower number of species. Abundance of most guilds also varied according to land-use type and the same was true for endemic and recently arrived species. Bird community composition of annual agriculture was found to be more distinct from native forest than any of the shade plantations and edge effects, local variables and landscape variables were found to impact upon bird distribution and abundance across the landscape.

Main conclusions Agroforestry systems were found to support bird communities closer to ones in native forest than annual agriculture did. However, several species were simply absent from the agricultural matrix, highlighting that their conservation can only be achieved by the preservation of large tracks on native vegetation.

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Acronyms

AA - Annual agriculture

ECOFAC - European Programme for the conservation and sustainable use of forest ecosystems in Central Africa.

IUCN - International Union for the Conservation of Nature

MR - Montane Rainforest

SC - Shade coffee

SP - Shade polyculture

1. Introduction

1.1 – The issue

Agricultural induced habitat loss and the numerous undesirable environmental impacts associated with agricultural practices has led conservation scientists to identify agriculture as one of the major drivers of biodiversity loss (Sala *et al*., 2000; Norris, 2008).

Nowadays roughly one-third of the global land area is devoted to cultivated systems (Musters *et al*., 2000; Luck & Daily, 2003) and by 2050 the world's demand for food is expected to be the double that of present levels (Tilman *et al*., 2002) and as a consequence more area is expected to be converted to agricultural practices.

The future increase in agricultural area is predicted to be mostly concentrated in the tropics (MEA, 2005), where the majority of the planet's most important conservation areas are located (Myers, 1988; Myers *et al*., 2000; Brooks *et al*., 2006; Fa & Funk, 2007). One can therefore forecast that some of the biggest challenges with which conservation scientists will be faced with in the near future will be related with understanding the dynamics surrounding agroecosystems and how agricultural landscapes can be managed in order to accommodate both human needs and biodiversity.

The establishment of protected areas has traditionally been the main approach of conservation planners and has "successfully" devoted 11,5% of the planet's surface to conservation purposes (Rodrigues *et al*., 2006). Despite this being a significant achievement the existing network is still far from complete when it comes to both species and habitat representativeness (Soulé & Sanjayan, 1998; Rodrigues *et al*., 2006) and new areas for the expansion of the actual protected areas network are predicted to be almost inexistent in the near future (Musters *et al*., 2000). This emphasises the need to assess if human-managed systems are capable of sustaining biodiversity and if so, the need to infer how this potential can be maximised.

Despite the major changes in species distribution and abundance, as a consequence of human-induced changes associated with agricultural practices, having long been identified, substantial gaps still exist in our understanding of the relative impact of different land-uses on the native ecological communities (Lindenmayer *et al*., 2002). This is particularly true for tropical ecosystems since the bulk of ecological theory that tackles the synergies between agriculture expansion and biodiversity depletion have been developed in temperate areas (Waltert *et al*., 2004).

Recently, however, an increasing number of studies have pointed to multi-strata agroforestry systems as being able to accommodate high levels of species richness and abundance for several tropical groups, especially when compared with alternative land uses devoid of arboreal vegetation (Greenberg *et al*. 2000; Faria *et al*. 2006; Bos *et al* 2007). At the forefront of this debate has been the potential of coffee and cocoa shade plantations to retain original forest biodiversity but despite some of the most important cocoa and coffee producing areas being located in Africa, virtually no data on the topic exists for the continent (Rice & Greenberg, 2000; Komar, 2006).

The island of São Tomé in the Gulf of Guinea island system is known as being an exceptional centre of endemism and has a five century long history of agricultural induced habitat modification with much of it being steered by cocoa and coffee shade plantations (Fa & Just, 1994; Jones & Tye, 2006; Melo, 2006). Despite this, many of the endemics have been able to adapt to the agricultural landscapes with considerable tree cover associated with shade plantations (Peet & Atkinson, 1994) and no extinctions of endemic species have been documented on the island (Melo, 2006). In the last decades however, a shift from agroforestry systems to more open agricultural practices involving less tree cover has started to take place (Peet & Atkinson, 1994; Joiris, 1998; Vaz & Oliveira, 2007) and the impacts of this land-use transformation in the island native species is still largely unknown.

Assessing the relative impact of different agricultural practices is therefore essential to design conservation strategies which can best preserve the islands biodiversity while satisfying human needs.

1.2 – Thesis scope

This thesis will focus on São Tomé's bird species' response to different types of agricultural land-use. In order to do so, bird communities were sampled along a gradient of agricultural intensity, going from primary forest to shade coffee plantations to shade polyculture plantations and finally annual agriculture with the main aims of:

- Assessing how the replacement of agroforestry systems, with more open agricultural practices affects bird abundance, diversity and distribution across the studied landscape;
- Inferring if different feeding guilds respond differently to habitat modification;
- Assessing if land-use change will facilitate the spread of non-native bird species;
- Assessing which variables affect bird distribution within the studied landscape at both the local and landscape levels;
- Assessing for the influence of edge effects upon the local avifauna.

1.3 Thesis overview

In **Chapter 2 – Background** provides an introduction to the literature regarding biodiversity and agriculture putting emphasis on tropical regions and within those to the research surrounding shade plantations. Particular attention will be given to edge effects finishing with a description of São Tomé's biogeography, avifauna and agriculture.

Chapter 3 – Methodology starts with a brief description of the study area followed by a detailed presentation of the main methods used for both field data collection and statistical analysis. The main results are presented in **Chapter 4 – Results** and finally **Chapter 5 – Discussion** puts the results into the context of the broader literature emphasising the conservation considerations and policy implications of the presented work.

2. Background

2.1 – Biodiversity and agriculture

"After all of the considerable parks and reserves are established the majority of the world's biodiversity (including nematodes, arthropods, and the other small things than run the world) will exist in fragments of remaining habitats that exists within the agricultural matrix."

(Vandermeer (2007))

Since its development, agriculture has been one of the main drivers of habitat loss and habitat fragmentation (*Sisk, et al., 1994;* Ricketts & Imhoff, 2003*)* giving birth to new landscapes with different capacities for retaining the communities present in the original habitats.

Despite an intensive landscape transformation, some agricultural areas do however retain a remarkable amount of biodiversity. In Europe more than 50% of the continent's important conservation areas are associated with low-intensity farming (Bignal *et al*., 1996) and an increasing number of studies have identified some tropical agricultural landscapes as being able to accommodate as much as 50% of the original fauna (Balmford *et al*., 2005; Sekercioglu *et al.*, 2006). However, intensification in agricultural practices has been identified as reducing the ability of agricultural landscapes to accommodate wild species (Benton *et al.*, 2003; Matson & Vitousek, 2006). In the UK for example, ten million individuals belonging to ten farmland species are predicted to have disappeared from the countryside over the last two decades due to agricultural intensification (Krebs *et al*., 1999; Donald *et al.,* 2001).

Despite agricultural practices and biodiversity losses being linked in both temperate and tropical regions the way this link is made differs quite dramatically. While in temperate regions, and especially in Europe, landscapes are already dominated by

intensive agriculture, in most of the tropics landscape transformation is still underway and the intensity is much lower (Norris, 2008).

2.1.1 – Tropical biodiversity and agriculture

An increasing body of literature is highlighting the fact that tropical agricultural landscapes do not constitute featureless areas of unsuitable habitat for biodiversity and can indeed be remarkably rich in terms of species numbers (Greenberg *et al* 1997b; Matlock Jr. *et al*., 2002). Species composition in modified landscapes, however, has often been found to be highly dissimilar to that of the original habitat (Waltert *et al*, 2005; Norris, 2008, Harvey *et al.,* 2006) and the capacity of tropical agricultural landscapes to retain biodiversity is far from being uniform across different land-use types. Agroforestry for instance, is known as having a far greater capacity to accommodate biodiversity than palm oil or sugar cane plantations (Norris, 2008).

Studies seeking to understand the relative impact of different agricultural land-use types upon biodiversity have typically looked for intensity gradients (Hughes *et al*., 2002; Waltert *et al.*, 2005; Harvey *et al.,* 2006). From these studies a pattern of compositional change is beginning to be revealed in which much of the species compositional variation can be explained by differences in vegetation complexity (Heikkinen *et al*., 2004; Waltert *et al*., 2005). Tree cover (Hughes *et al*. 2002; Waltert *et al*., 2005; Harvey *et al*., 2006), overall landscape heterogeneity (Matlock Jr. *et al.*, 2002; Benton *et al*., 2003; Naidoo, 2004) and distance to natural habitat (Greenberg *et al*., 1997b) have been identified as playing a major roles in the retention of tropical biodiversity within agricultural landscapes.

Throughout the tropics birds have been a preferred taxa for studying the impacts of the conversion of natural areas into agricultural landscapes. Studies can be found for South America (Gascon *et al*., 1999; Hughes *et al*., 2002; Matlock Jr. *et al*., 2002; Sekercioglu *et al*. 2006), Africa (Naidoo, 2004; Waltert *et al*., 2005) and South-East Asia (Thiollay, 1995; Waltert *et al*., 2004; Marsden *et al*., 2006). Despite a low number of studies addressing this issue and a large geographic bias towards South America, common trends are emerging and generalisations can start to be made.

In relation to native forests, agricultural areas appear to experience a considerable decrease in the overall number of species (Thiollay, 1999; Naidoo, 2004; Waltert *et al*., 2005; Komar, 2006), a shift from more forest-interior species towards open or bush-land species (Hughes *et al*., 2002; Naidoo, 2004; Waltert *et al*., 2005) and a change from more habitat specific to more generalist species (Naidoo, 2004). Specific life history attributes seem also to be related with reduced resilience to habitat conversion including insectivory (Thiollay, 1995; Waltert *et al*., 2005), large body size (Thiollay, 1995) and restricted ranges (Waltert *et al*., 2004).

Myers (1991) describes small scale agriculture as the main agent of tropical deforestation. This land-use type has, however, been hugely neglected and very little is known about its impact upon tropical wildlife (Marsden *et al.*, 2006). When compared with alternative land-use types, bird community composition in small scale farming areas is highly dissimilar to those of other land use types (see Table 1). One study in Cameroon (Waltert *et al*., 2005) comparing near-primary forest, secondary forests, agroforestry and annual agriculture plots identified an overlap of only 27% between annual agriculture plots and near-primary forest (contrasting with the 62% similarity between agroforestry and near-primary forest) and a similar study in Uganda has identified an overlap of only 19% between intact forest and small-holder agricultural plots (Naidoo, 2004).

Table 1 – Bird community overlap between several land-use types and native forest within tropical landscapes.

Land-use type	Overlap with forest %	Region, country	Authors
Agroforestry system[a]	62[1]	African, Cameroon	Waltert *et al*., (2006)
Annual agriculture	27[1]	African, Cameroon	Waltert *et al*., (2006)
Annual agriculture	19[2]	Africa, Uganda	Naidoo *et al*., (2004)
Agricultural matrix[b]	54[2]	South America, Costa Rica	Hugles *et al*., (2002)
Banana plantations	50[2]	South America, Costa Rica	Matlok Jr. *et al*., (2002)
Agricultural matrix[c]	40[2]	South America, Nicaragua	Harvey *et al*., (2006)

Agroforestry system[d]	43-55[2]	South-East Asia, Indonesia	Thiollay, (1995)

[1] estimate based mean Sorensen index.

[2] estimate based on number of shared species.

[a] cocoa, coffee and plantain plantations

[b] landscape analysis – cattle pastures, coffee plots, mixed agricultural plots, gardens, thin riparian strips of native vegetation and small forest remnants.

[c] landscape analysis - riparian forest, secondary forest, forest fallows, live fences and pastures.

[d] agroforests dominated by rubber tree (*Hevea brasiliensis*), dammar (*Shoera javanica*), and durian (*Durio zibethinus*).

2.1.2 – Shade plantations

The potential role of agroforestry systems, in particular shade cocoa (*Theobroma cacao*) and coffee (*Coffea spp.*) for the conservation of tropical biodiversity has been the focus of considerable research (Greenberg *et al*., 1997a, 2000; Raboy *et al*, 2004; Tejeda-Cruz & Sutherland, 2004; Van *et al*., 2007; Hervé & Vital, 2007). Earlier work on this matter was probably stimulated by the Perfecto *et al*. (1996) hypotheses that migratory bird declines could be related with the decline in shade coffee plantation (Komar, 2006) and much of the subsequent work has had birds as a target group (Greenberg *et al*., 1997a, 1997b, 2000; Tejeda-Cruz & Sutherland, 2004; Bael *et al*. 2007).

The theory behind most of the studies undertaken is that systems which incorporate shade trees provide more structural complexity and resources than unshaded systems and are therefore capable of conserving forest organisms that would otherwise be displaced (Greenberg *et al*., 1997a; Rice & Greenberg *et al*, 2000). Shade management systems for both cocoa and coffee vary widely forming a gradient that goes from rustic management, where planting occurs under old secondary forest or thinned primary forest; planted shade; commercial shade where crops other than coffee and cocoa are scattered among shade trees and finally to specialised shade, where shade is created by a limited number of species (normally less than 3) (Rice & Greenberg, 2000). Despite large differences in floristic composition of the different shade systems the conservation value has been found to be comparable (Greenberg *et al*., 1997b; Tejeda-Cruz & Sutherland, 2004).

2.1.3 – Conservation value of shade plantations to birds

In a review on the conservation role of coffee plantations for birds, Komar (2006) analysed more than 45 studies and showed that most studies found a lower species richness and diversity in plantations than in nearby forest patches. However, some studies have found plantations to be as, or even more, species rich than natural forest and the majority reported them to be richer than agricultural systems which are associated with less tree cover. This inflation in the number of species is partly explained by the greater structural heterogeneity and floristic diversity of plantations (Greenberg *et al.* 1997b; Tejeda-Cruz & Sutherland, 2004) which represent an intermediate habitat in terms of disturbance between natural forest and agriculture habitats, thereby functioning as an ecotone (Komar, 1996). However, none of the studies reviewed by Komar (1996) took into account differences in species detectability between habitats which could have biased the results and despite some evidence that proximity to natural patches may have an effect on the number and abundance of species found within plantations (Tejeda-Cruz & Sutherland, 2004), a factor that most studies failed to take into account (Rice & Greenberg, 2000).

A general trend among the studies reviewed by Komar (2006) was the occurrence of species turnover where the loss of more specialised forest species in plantations was cancelled out by the addition of species characteristic of more open and disturbed habitats (Greenberg *et al*., 1997b; Tejeda-Cruz & Sutherland, 2004; Komar, 2006). Insectivores (Komar, 1996; Tejeda-Cruz & Sutherland, 2004) also seem to be depleted in plantations in relation to natural habitats.

As a consequence of most coffee-growing regions being located within biodiversity hotspots the potential conservation role of shade coffee has been highlighted (Tejeda-Cruz & Sutherland, 2004; Komar 2006) (Fig. 1). Most bird coffee research has, however, been conducted in the Neotropics (Greenberg *et al*., 1997a, 1997b, Tejeda-Cruz & Sutherland, 2004) with little or no information known for important coffee producing areas where a large number of Important Bird Areas can be found such as in Africa or South-East Asia (Komar, 2006).

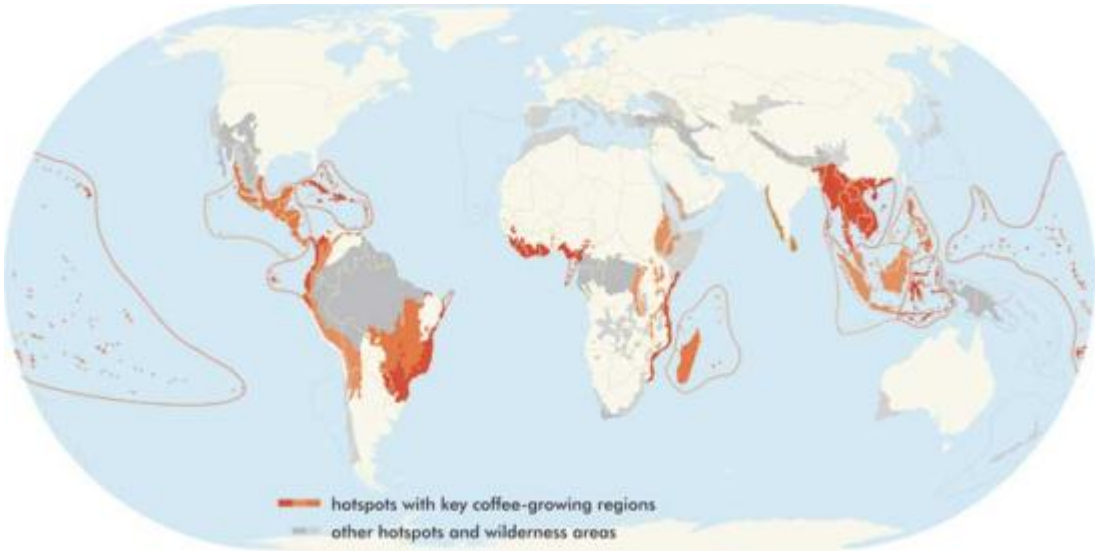

Figure 1 - Map showing the overlap between coffee producing regions and the biodiversity hotspots.

Source: www.conservation.org

The conservation value of coffee plantations is still arguable. Among 45 reviewed studies by Komar (1996) only eight Globally Threatened (6 IUCN Vulnerable and 2 Endangered) species were found within shaded coffee plantations and despite species numbers found within plantations being comparable to natural habitat, the number of more sensitive species to habitat modification was, without exception, lower (Komar, 1996; Greenberg *et al*., 2000; Tejeda-Cruz & Sutherland, 2004). The potential of shade plantations to act as population sinks to surrounding forest fragments is highlighted by Rice & Greenberg (2000) but the importance of shade plantations in acting as suboptimal habitats allowing periodic dispersal among nearby natural habitats was emphasised. This feature has lead several authors to promote shade plantations as corridors and buffer areas to optimal forest habitat (Komar, 1996; Rice & Greenberg, 2000; Tejeda-Cruz & Sutherland, 2004).

2.1.4 – Conservation value of shade plantations for other taxa

An increasing body of evidence seems to indicate that different groups respond differently to shade plantations. Pineda *et al*. (2005) compared species diversity of dung beetles, bats and frogs in natural forest and shade coffee plantation and found that beetle abundance and species richness was significantly greater in plantations

whereas frog species richness was one-fifth lower and no change was noticed for bats. Similar patterns for frog and bat species were found in cocoa plantations in the Brazilian Atlantic forest (Faria *et al*., 2007). The same plantations were found to constitute high-quality habitat for forest-associated species of frogs, lizards and bats but poor quality habitat for forest-dependent ferns (Faria *et al*., 2007). South-America cocoa plantations were also noted to constitute suboptimal habitat for both primates (Raboy *et al*., 2005) and sloth species (Vaughan *et al.*, 2007).

2.2 - Edge effects

One of the main consequences of agricultural expansion is a pronounced increase in habitat boundaries. These habitat boundaries have long been identified as having very large effects on species distributions and dynamics (Ries *et al*., 2004; Ewers & Didham, 2006a) which are commonly referred to as edge effects. Edge effects emerge as a consequence of a gradient of change in both biotic and abiotic factors (Murcia 1995; Sisk *et al.,* 1997) across a range of distances from the patch boundaries into habitat interiors which result from transitions between neighbouring habitats (Murcia, 1995).

Edge effects are described to be remarkably diverse, ranging from changes in species abundance (Sisk *et al., 1997;* Manu *et al*., 2007), alterations in trophic interactions and individual fitness (Paton, 1994) and hindrance of the movement of individuals among fragments (Pineda et al, 2005). From a conservation perspective edge effects are known to reduce the effective area of protected areas (Woodroffe & Ginsberg, 1998), to facilitate the invasion of exotic species (Didham *et al.,* 2007) and to impact on the meta-dynamics of fragmented populations (Lidicker Jr., 1999)

The significance of edge effects has led them to be one of the most researched areas in both general ecology and conservation and therefore a vast literature can be found on the matter (Lidicker Jr., 1999; Ries *et al*., 2004; Ewers & Didham, 2006a). A comprehensive picture of the mechanisms driving species response has, however, remained elusive (Murcia, 1995). Reasons for this range from poor study design and lack of consistent methodology (Murcia, 1995) to inappropriate statistically robust analysis (Ewers & Didham, 2006b) and the failure to take into

account the complexity of the interactions behind individual, and ultimately community, responses to edges (Murcia, 1995; Ries *et al*., 2004; Ewers & Didham, 2006b).

Multiple confounding factors have been identified as leading to different taxa exhibiting inconsistent responses to edges. Examples of those are fragment area (Laurance & Yensen, 1991; Ewers *et al.,* 2007, Manu *et al., 2007*), fragment shape (Ewers & Didham, 2006c), edge orientation (Murcia, 1995; Ries *et al*, 2004) and edge contrast (Ries & Debinski, 2001) all of which have been found to influence the strength of edge effects, but are rarely controlled for within edge effects studies. Of those, edge contrast is one of the main concerns in this study. Edge contrast is the difference in vegetation structure within the two habitat types that border to form an edge (Ries *et al*., 2004), and affects the movement of individuals across the border referred to as edge permeability (Ewers & Didham, 2006a).

2.2.1 – Edge contrast

General consensus exists on the premise that structurally similar edges (often referred to as "soft edges") are less of a barrier to the movement of species than "hard", high contrast ones (Ries *et al.,* 2004*;* Ewers & Didham, 2006a). This has held true for several taxa including, forest birds (Sisk *et al*., 1997; Castellón & Sieving*,* 2006), arthropods (Duelli *et al*., 1990), butterflies (Ries & Debinski, 2001) and was used by Pineda *et al.* (2005) to partly justify differences in species assemblages between montane rainforest and shade coffee plantations. Some studies have, however, failed to identify significant relationships between edge contrast and species response (Pimentel, 2006).

One major constraint of relating edge contrast and edge permeability to each other is the difficulty in collecting the required field data necessary to estimate migration movement parameters and rates (Ries & Debinski, 2001). Telemetry has been used to address this issue (Castellón & Sieving*,* 2006), however, a more common (and simpler) approach has been to compare variation in abundance of a species or group of species across a gradient of distance from habitat edges (Sisk *et al*., 1997; Pimentel, 2006).

2.3 – São Tomé

2.3.1 – Biogeography and climate

The 857 km² (47 km x 28 km) oceanic island of São Tomé (1 00 N, 7 00 E) lies 255 km West of Gabon and is the second largest island of the Gulf of Guinea island system (fig. 2). Its highest elevation is 2,024 m (Pico de São Tomé) and the climate is oceanic equatorial with mean annual temperatures ranging between 22-33°. Levels of precipitation are high, exceeding 7,000mm annually in the southwest and are mostly concentrated in the rainy season which is from September to May. The island has a dry season from July to August, known as the *gravana* (Jones & Tye, 2006).

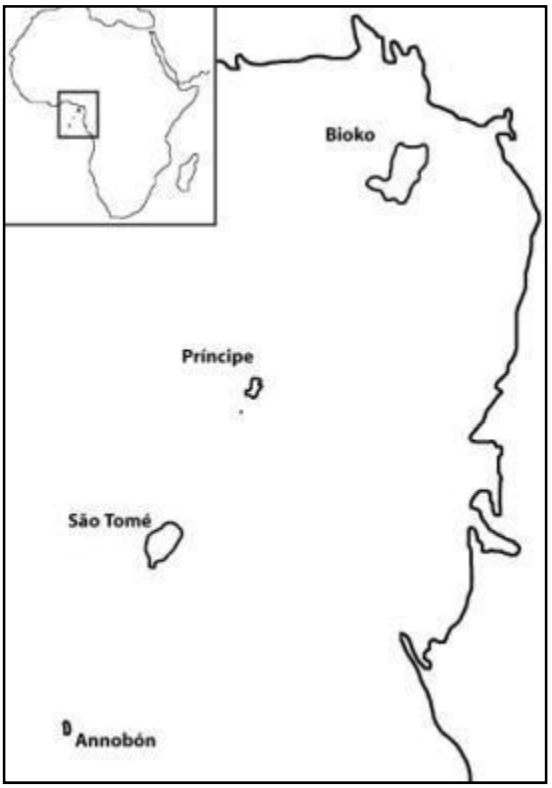

Figure 2 - Representation of São Tomé relative position to continental Africa and its adjacent islands. Adapted from Jones & Tye (2006).

2.3.2 – Island avifauna

The total number of breeding land bird species in São Tomé is still not precisely known, but is thought to be more than 50 (Jones & Tye, 2006). Out of this total, sixteen are endemic to the island and further 5 are shared only with the neighbouring island of Príncipe (Jones & Tye, 2006). Three endemic genera, *Amaurocicha* (fig. 3), *Dreptes* and *Neopiza* are also recognized to the island (Jones & Tye, 2006) and seven species that are shared with mainland Africa have evolved into endemic subspecies on São Tomé (Peet & Atkinson, 1994).

This level of endemism seems to have no parallel worldwide: the average number of endemic bird species in islands of less than 10,000 km² is just two with the mode being one (Melo, 2006), making São Tomé with its 16 endemics a large outlier. Due

to the high level of endemic species the island was classified by BirdLife International as one of the top 25% Endemic Bird Area (EBA) of Global Conservation Significance (Stattersfield *et al.,* 1998) and Collar & Stuart (1988) have classified the southwest forest of the island, where all the endemic species can be found, as the second most important forest in Africa for bird conservation, Madagascar included.

Figure 3 - Picture of a fledgling of São Tome short-tail (*Amaurocichla bocagei*), an endemic genus to the island.

Photo: Ricardo Rocha

2.3.3 – Agriculture in São Tomé

Rainforest is thought to have once fully covered the island (Melo, 2006) but soon after its discovery by the Portuguese in the 1470's land started to be cleared for agricultural purposes (Jones & Tye, 2006). Sugarcane was the first cash crop to be harvested on the island but by the 18th century coffee and cocoa were introduced and following the country's motto - "*Aumentemos a Produção*" (Lets increase the production) – more land was cleared, leading to most forest (below 1000-1200 m) being converted to shade plantations (Olmos & Turshak, 2007). Cocoa and coffee production declined during the 20th Century leading to the abandonment of many plantations, a process that became more pronounced following independence from Portugal in 1975 (Jones & Tye, 2006). As a consequence, large areas of the island reverted into secondary forest, resulting in a substantial increase in forest cover across the island (Jones & Tye, 2006).

In the last decade, however, the government of São Tomé and Príncipe undertook a land reform in which large state owned plantations were divided into smaller plots and distributed among ex-plantations workers. A substantial shift from agroforestry systems to small scale farming was experienced, resulting in the felling of a considerable number of shade trees and to the replacement of cocoa and coffee trees by banana and other food crops (Joiris, 1998).

Habitat choice among São Tomé's birds was studied by Peet & Atkinson (1994), who reported that plantations supported both endemic and non-endemic species, with several endemics having considerable population sizes in plantation areas. The number of endemics found in modified landscapes was reported to decrease in farmland habitats (Jones & Tye, 2006) where the more conspicuous species were considered to be non-natives to the island. This trend was also identified by Olmos & Turkshad (2007) who reported that a single, recently arrived species was more than twice as abundant in modified landscapes than the most abundant native species.

2.3.4 – Conservation in São Tomé

Practical conservation initiatives in São Tomé are rather recent to the island. Legislation was put forward in 1993, leading to the establishment of the São Tomé Obo Natural Park which protects an area of roughly 245 km^2 in the central and southwestern parts of the island (Jones & Tye, 2006). The initiatives for the establishment of the Park were supported by the European founded ECOFAC project which was established in 1992 with the aims of promoting the conservation and rational utilization of rainforests in seven central African countries and since then several conservation initiatives in both social and ecological areas have been supported.

3. Methodology

3.1 – Study area

The study was carried out in and around the northeast end of Obo Natural Park, in the mountainous centre of São Tomé. The region's landscape comprises an extensive area of relatively undisturbed montane forest which is part of the natural park, surrounded by old-growth secondary forest (*capoeira)* in areas that were formerly occupied by coffee shade plantations and by a highly heterogeneous agricultural matrix that constitutes the island's most important horticultural and coffee production region. This area differs from the rest of the island by its lower minimum temperatures, higher rainfall and humidity, and greater cloud and mist cover (Jones & Tye, 2006) which makes the area particularly favoured for agriculture practices (Oliveira, pers. comm.)

Within this landscape study sites were selected in four different land-use types: "intact" montane rainforest, annual agriculture, shade coffee and shade polyculture, representing the full range of land-use types in the area.

3.1.1 - Montane rainforest (MR)

The montane rainforest as described by Excel (1944) ranges from 800 m to 1400 m and is characterised by tall trees (30-40 m) with a dense canopy and high abundance of Pteriophytes and Bryophytes (fig. A1.1,appendix 1). The families Rubiaceae and Euphorbiaceae are the most common and *Trichilia grandifolia, Pauridiantha insularis, Pavetta montícola, Erytrococca molleri* and *Tabernaemontana stenosiphon* are the most characteristic species (Vaz & Oliveira, 2007). Although there is evidence of localised pitsawing (pers obs), most of this vegetation zone appears to be in its native state (Jones & Tye, 2006).

3.1.2 – Annual agriculture (AA)

This is a very dynamic habitat with intensive human influence. The annual agriculture plots are located in the buffer area of the natural park and constitute a recent habitat within the historical period of the island's occupancy. It is characterised by a substantial reduction in tree cover and its vegetation is predominantly composed of horticultural species with some introduced perennial fruit trees and remnant forest patches (fig. A1.2,appendix 1).

3.1.3 – Shade coffee (SC)

The 51 hectare coffee plantation surveyed in this study constituted the largest active plantation on the island. When compared with annual agriculture this habitat type has a substantial increase in vertical structure complexity due to the the 2-3 m coffee trees (*Coffea sp*) and the presence of shade trees (mostly *Erythrina sp.*) (fig. A1.3, appendix 1).

3.1.4 –Shade polycultures (SP)

This habitat is a consequence of the agricultural reforms that happened following the country's independence. This is a very heterogeneous habitat where coffee trees are grown alongside plantain (*Musa sp.)* and a diverse range of annual crops (fig. A1.4, appendix 1). A fair share of the shade trees have been felled either for wood or to give place to annual crops.

3.2 – Data collection

The selection of the four habitat types was made after advice from Monte Pico (local non government organisation), analysis of Google Earth ™ satellite images and extensive ground-truthing during the first week of field-work.

A total of 105 count stations were systematically placed through the target landscape in the following manner:

Twelve count stations (six in the case of shade coffee) were located 100m apart along transect lines in the centre of each studied land-use, located as far from land-use edges as possible (these will be referred to as core transects; fig. 5). Due to the

topology of the landscape it was not always possible to create straight-line transects. The different number of count stations in shade coffee was due to the relatively small area of the plantation, meaning it was not large enough able to accommodate more than six stations in a straight line with all stations a minimum distance of 150 m from the nearest habitat edge – a criterion that was imposed upon all core count stations. Time balancing (Nally & Horrocks, 2002) was used to compensate for the lower number of count stations in the coffee plantation land-use. Each of the coffee plantation stations was sampled ten times whereas each of the other land-use core stations was sampled just five times. Core count stations were located at least 100 m apart in order to avoid possible double counting of the same individual. Distance between count stations was measured using Global Position System (GPS) technology on a handheld (Garmin, Etrex Vista HCx).

The remaining count stations were located along seven transects crossing two land-use types and penetrating approximately 200 m into each neighbouring land-use (fig. 4). Each of these transects was comprised of nine count stations: one located at the land-use boundary and the remaining eight at four different distances from the boundary into neighbouring habitats. Stations were therefore located at: -200 m, -150 m, -100 m, -50 m, 0 m, 50 m, 100 m, 150 m and 200 m from the boundary. This range of distances is adapted from Pimentel (2006).

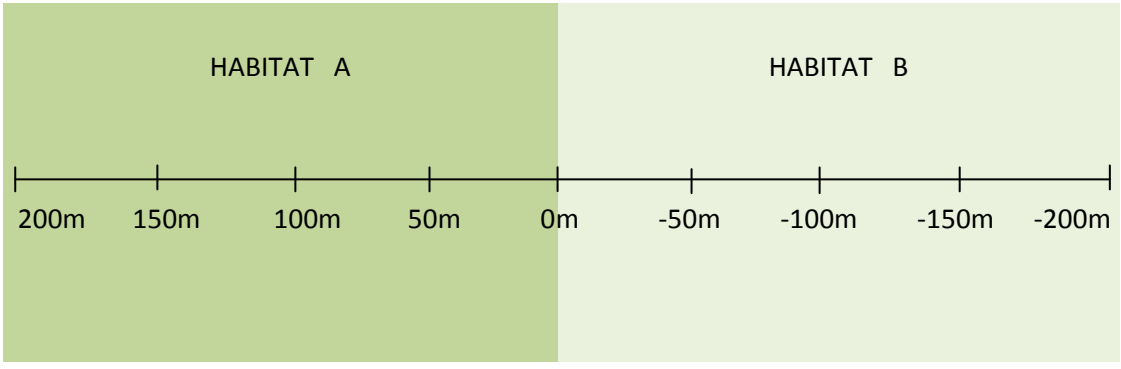

Figure 4 - Schematic representation of the transects across land-uses with distance of each count station from the boundary (0m).

The total number of transects was therefore the following:

Four core transects:

- Montane rainforest (12 stations, each sampled 5 times) ;

- Annual agriculture (12 stations, each sampled 5 times) ;
- Shade coffee (6 stations, each sampled 10 times);
- Shade polyculture (12 stations, each sampled 5 times).

Seven edge transects:

- 3 edge montane rainforest / annual agriculture (9 stations, each sampled 3 times);
- 2 edge montane rainforest / shade coffee (9 stations, each sampled 3 times);
- 2 edge shade coffee / shade polyculture (9 stations, each sampled 3 times).

The distribution of the transects across the landscape can be seen in figure 5.

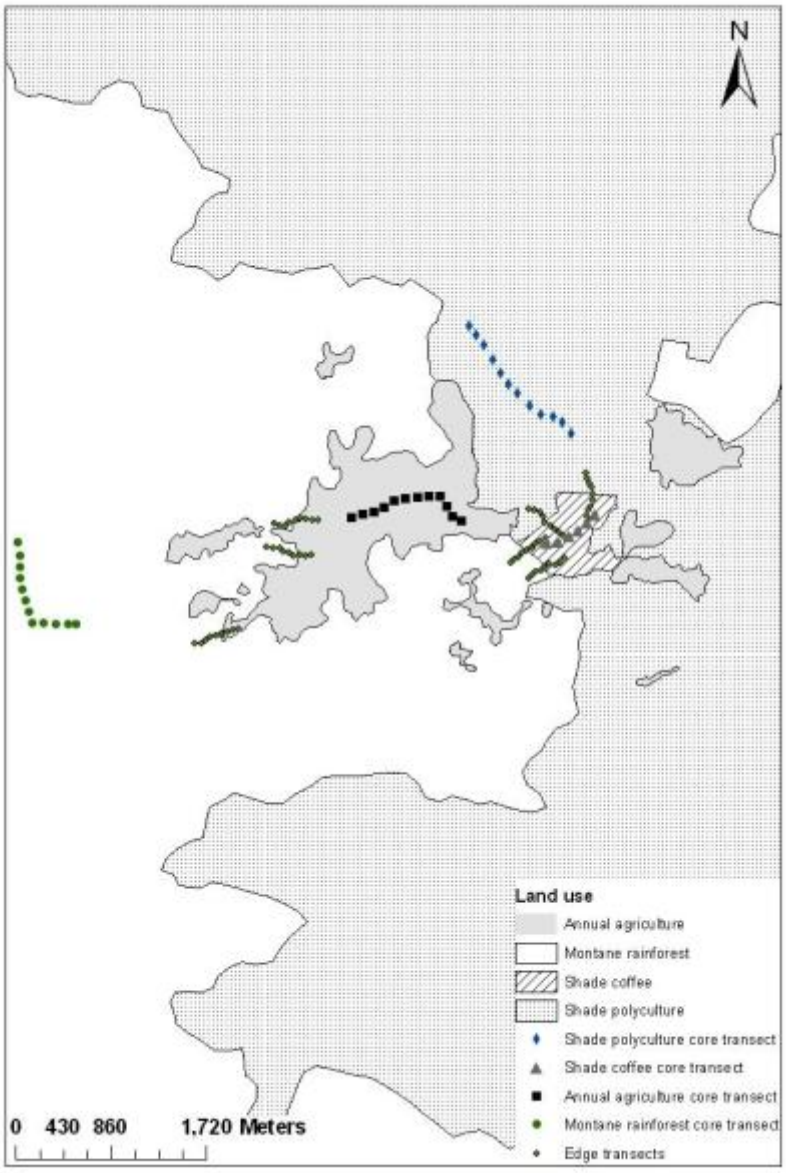

Figure 5 - Map of the analysed landscape where the distribution of the performed transects can be seen.

Transects followed abandoned paths whenever possible but several trails had to be opened specifically for this study. Bird surveys where always carried out in different days to trail opening.

Different day repeated sampling (Field *et al.,* 2002*)* was used to establish a more robust inventory of the bird assemblages at each sampled site. Core transects were sampled a total of 5 times (except for shade coffee ones which were sampled 10 times), four during the morning period and one during the afternoon (eight during morning period and two in the afternoon for shade coffee). Between-land-use

transects where sampled three times: two of those during the morning period and once during the afternoon. Same-day repeated surveys leads to underestimation of species richness (Field *et al*., 2002). The same transect was therefore only surveyed on different days with the order in which transects were sampled being randomized each time. With the aim of reducing time-of-day effects, the order in which the count stations within a transect where sampled was reversed at each visit.

All count stations where located between 800 m and 1400 m above sea level, the range span of the native montane rainforest as described by Excel (1944) allowing the direct comparison between the current state of the bird's assemblages among the different anthropogenic land-uses and the native habitat that would have previously occupied the entire area.

3.3 – Bird data

A one-week pilot study was carried out (from the 28th of April to the 5th May) in which the most appropriate survey length and data collection periods were selected based on area of habitat available, logistic and time constraints. Training was undertaken during this time (also continuously during field work) to estimate the distances at which birds were located with special attention given to train distance estimation in all land-use types covered by the study.

Diversity and abundance data was recorded by a pair of observers acting as one (Bibby *et al.* 2000). Field surveys were conducted between the 6th of May and 6th of July using fixed-radius point count method (Sutherland *et al.* 2004) with a survey period of 4 minutes during which all birds seen or heard within an approximate 25 m radius where recorded. A waiting period of 2 minutes prior to the actual survey time was used to locate the more cryptic individuals and to allow birds to recover from disturbance of the observers arriving at the site. The survey time of 4 minutes was selected with the intention of avoiding possible double counting of single individuals.

Due to the similar appearance and similar vocalizations of the African masked weaver (*Ploceus velatus peixotoi*) and the Village weaver (*Ploceus cucullatus nicriceps*), they were almost impossible to tell apart so were recorded as *Ploceus sp.*

and were considered as a single species for the analysis unless otherwise mentioned.

Surveys were confined to the periods 05.30-09.30 a.m. and 16-17.30 p.m. on days without strong rain or strong wind.

3.4 - Habitat data

In order to assess differences in vegetation structure and habitat complexity between the sampled land-uses, ten variables where visually estimated within a 10 m radius of each point station. All estimates were made by the same individual and the selected variables were the following: percentage understory canopy cover, mid-canopy cover and upper-canopy cover; vegetation density, recorded by counting the number of trees with a dbh (diameter at breast height) greater than 10 cm (in annual culture, shade coffee monoculture and shade polyculture count stations the number of coffee / plantain was counted but the data was not included in the analysis), number of tree species, maximum vegetation height, percentage of bare ground; leaf litter cover and finally the abundance of climbers and epiphytes estimated on a scale of 0 (none) to 3 (dense).

3.5 – Landscape data

A GPS handheld (Garmin, Etrex Vista HCx) was used to record the geographical coordinates of each count station and to produce a detailed map of the study landscape. The GIS data layers were used to calculate the distance of each sampling station to the control transect inside the forest, to the nearest human settlements and to determine the proportion of each land-use type within a 250m, 500m and 750m radius. These analyses were performed using ArcMap 9.0 (ESRI® ArcGIS™, 2004).

3.6 - Life history attributes

To assess the influence of life history-traits on the general distribution of bird species throughout the analysed landscape information on feeding guilds was collated from the literature where available (Christy & Clarke, 1998; Jones & Tye, 2006) and for those species where information could not be found surrogate species within the genera were used as a *proxy* in addition to personal observations. Additionally, species were classified according to level of endemism (endemic genus, endemic species, endemic subspecies or non endemic), origin (following the classification presented in Jones & Tye (2006) of native or recent colonisation) and IUCN threat status (appendix 1, tables A1.1 and A1.2).

3.7 - Statistical analysis

3.7.1 – Richness and species diversity

The number of observed species was counted for each sampling station and summed for each of the different land-uses. Shannon and Simpson diversity and Berger-Parker dominance indices were calculated for each sampling point. The first two indices were computed for each land-use type in EstimateS v.8.0 (Colwell, 2006) using 100 permutations and the Berger-Parker index, which is expressed as the proportional abundance of the most abundant species, was calculated in Excel using the formula:

$$d = \frac{Nmax}{N}$$

where N is the total abundance of all species and $Nmax$ is the number of the most abundant species (Magurran, 2004).

Sample-based randomized species accumulation curves were calculated using EstimatesS v.8.0 (Colwell, 2006) in order to assess sampling efficiency.

3.7.2 – Feeding guilds, endemics and recently arrived species

Per site abundance (measured as the mean number of recorded individuals per visit) was calculated for each feeding guild, for native and non native species and for

endemics and non endemics. Differences in abundance across land-use types were tested with an analysis of variance (ANOVA) for normally distributed data (followed by Tukey HSD) or Kruskal-Wallis for non-normally distributed data (followed by pairwise comparisons).

3.7.3 – α, β and γ diversity

Species diversity per count station (alpha diversity) was calculated as the mean number of species in every repeated visit to a specific count station. Gamma diversity is the overall species richness within a land-use type and beta diversity corresponds to the difference between alpha and gamma diversity, providing a measure of natural variation between count stations within a given land-use (Legendre *et al.,* 2005). Differences in avian biodiversity between land-uses over the three scales of diversity (α, β *and* γ) was assessed through a Chi-squared test.

Differences in beta diversity across count stations were inferred using Bray-Curtis similarity index. For that, the per visit mean number of recorded individuals for each species was calculated for all sampling stations with the following formula applied:

$$BC = 1 - \frac{\sum | \mathrm{Yij} - \mathrm{Yik} |}{\mathrm{Yij} - \mathrm{Yik}}$$

Where Y_{ij} refers to the abundance of species $_i$ in site $_j$ and Y_{ik} refers to the abundance of species $_i$ in site $_k$; the summation is over all species (Báldi & Kisbenedek, 1994). Calculations where made using *vegan* package in R v.2.7.1 software.

3.7.4 – Spatial autocorrelation

The similarity in species composition between stations is likely to be related to the distance they are apart. To explore this relation a Mantel test was carried out between the Bray-Curtis values and geographical distance.

3.7.5 Species assemblages

Species composition among different sites was explored using non-metric multidimensional scaling (NMDS). This technique was selected because it makes no assumption about the distribution of the data (Shaw, 2003) and therefore is widely used for analysis of community data (Tejeda-Cruz & Sutherland 2004; Naidoo, 2004;

Watson, 2004; Barlow *et al* 2004). The ordination was performed using a Bray-Curtis similarity matrix and was executed using the *vegan* packaged in R v.2.7.1 software.

The significance of count station groupings within the NMDS was assessed using a one-way multivariate analysis of variance (MANOVA) upon the first and second axis station scores and in order to assess differences in group dispersion, the Euclidean distance between each station and the centre of its group cluster was calculated and based on that a one-way ANOVA was carried out.

The community composition (as given by the NMDS first axis station scores) along the edge transects was modelled as function of edge type, distance to edge and their interaction using an analyses of covariance (ANCOVAs).

3.7.6 – Relationship between birds, landscape, edge and local habitat variables

The relationship between landscape, edge and local habitat variables as explanatory variables and the following response variables were investigated: community composition (as represented by the major NMDS axis), diversity (based upon Shannon index), similarity to forest controls (based on the Bray-Curtis similarity values) and abundance of endemics, recently arrived species and each different feeding guild were explored using linear models. The minimum adequate model was arrived at by fitting the maximal model and then using stepwise regression (specifying backward and forward selection) (Crawley, 2006).

Collinearity among variables is known to affect the efficiency of the models. Pairwise correlation was therefore used to reduce the number of variables. Following the approach adopted by Naidoo *et al*., (2004) it was chosen to eliminate one variable of each pair that had a correlation coefficient superior to 0.8 (appendix 1, table A1.3). Case-wise correlations were undertaken using STATISTICA version 8.0 (StatSoft, Inc., 2008).

4. Results

4.1 – Avifauna of the region

A total of 8764 individual recordings of 27 species from 17 families were made during the 429 samples from the different point count stations (table 1, appendix 1). This represents 56 % of the island's resident bird species. The number of native species (18) recorded within the survey periods was exactly double the number of recently arrived species (9). Noteworthy is the fact that only two of the island's endemic species - the Maroon pigeon *Columba thomensis* and the São Tomé Grosbeak *Neospiza concolor* failed to be registered in the study landscape**.**

Species accumulation curves reached a plateau for all four land-uses indicating that the 60 samples performed in each land-use where enough to provide a good picture of the communities (Fig 5).

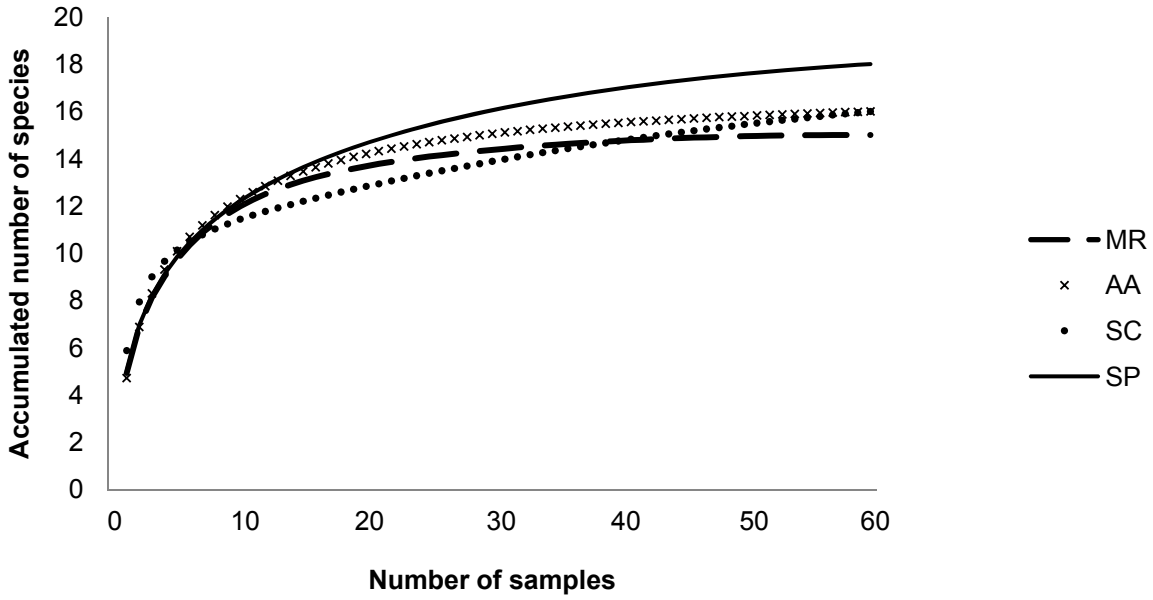

Figure 5 - Species accumulation curves based for bird species in the four studied land-use types. MR- montane rainforest, AA- annual agriculture, SC- shade coffee, SP- shade polyculture.

The overall mean number of recorded individuals per sampling station differed between habitats (one-way ANOVA $F_{3, 38} = 8.0$, $p < 0.01$) being significantly higher in shade coffee monoculture (33.3 ± 5.05; mean ± SE; Tukey HSD), followed by annual

agriculture (19.67 ± 3.2), shade polyculture (15.8 ± 1.74) and finally montane rainforest (14.5± 1.72).

In total the proportion of individuals recorded visually (65%) was almost double the proportion of individuals recorded by sound (35%) (fig. 6). A closer analysis shows that this tendency was not kept within each of the sampled land-uses. While most individuals where recorded visually within the anthropogenic habitats the pattern was reversed for the tropical rainforest where most records were made by sound.

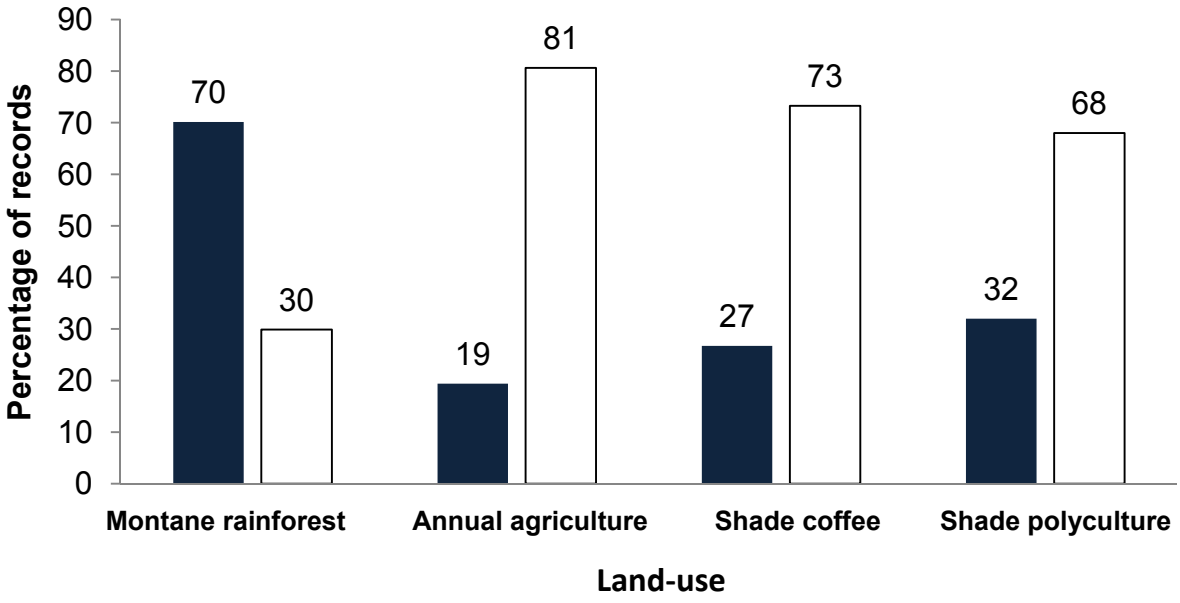

Figure 6 - Proportion of individuals recorded by vocalizations (■) and by visual sightings (□) within montane rainforest, annual agricultural plots, shade coffee monocultures and shade polyculture.

Out of the 9 IUCN threatened species 5 were recorded within survey period. Three of those, the IUCN Critically Endangered Dwarf Ibis (*Bostrychia bocagei*), the Giant sunbird (*Dreptes thomensis*) and the São Tomé Oriole (*Oriolus crassirostris*) where only recorded within montane forest whereas the other two, the Gulf of Guinea Trush (*Turdus olivaceofuscus olivaceofuscus)* and the Príncipe white-eye (*Zosterops ficedulinus feae)* where conspicuous thought the landscape (table 2).

Table 2 The five most endangered bird species and the land-use types they were recorded in.

Species	Status[1]	Land-use recorded in
Bostrychia bocagei	CR	Montane rainforest
Dreptes thomensis	VU	Montane rainforest
Oriolus crassirostris	VU	Montane rainforest
Turdus olivaceofuscus olivaceofuscus	NT	Montane rainforest, annual agriculture, ahade coffe and shade polyculture.
Zosterops ficedulinus feae	VU	Montane rainforest, annual agriculture, shade coffee, shade polyculture

[1]IUCN Threat categories: CR, Critically Endangered; VU, Vulnerable; NT, Near-Threatened.

Montane Rainforest (MR)

All but one of the 15 species recorded within this habitat were single island endemics at genus, species or subspecies level. The only exception was the Gulf of Guinea bronze-naped pigeon which additionally to São Tomé can be found in the nearby islands of Príncipe and Annobón. None of the recently arrived species was found within this habitat.

The forest assemblage was found to be dominated by insectivores species with the most abundant species being São Tomé speirops followed by the São Tomé prinia, Newton's yellow-brested sunbird and the Príncipe seedeater (fig. A2.1, appendix 2).

Annual agriculture (AA)

The farmland matrix is dominated by granivore species with just one species, the Common waxbill totalling almost half of the individuals recorded within this land-use.

Virtually no frugivores where found within this land-use and in relation to the montane rainforest there was a decrease in the abundance of São Tomé speirops and an increase in abundance of the IUCN vulnerable Principe white-eye (fig. A2.2, appendix 2).

Only one species, the Laughing dove was found exclusively within the annual agriculture plots and other two, the Common waxbill and the Palm swift had more than two thirds of the total recordings in this land-use (Fig. A2.2, appendix 2).

Shade coffee (SC)

This land-use sees a slight increase in the abundance of insectivores in relation to farmland but nevertheless the most common species are still part of the granivore guild with the genus *Ploceus* accounting for the bulk of the recorded individuals within this land-use (fig. A2.3, appendix 2).

Despite only being recorded twice the Bronze Mannikin was found only in this land-use and the São Tomé spinetail had more than two-thirds of the recordings in shade coffee (Fig. A2.3, appendix 2).

Shade polyculture(SP)

In relation to the other anthropogenic habitats the shade coffee plantation was characterised by a notoriously decrease in abundance of granivores and an increase in the abundance of insectivores. As for montane rainforest the São Tomé speirops is the most common species and when compared with the other shade plantation type the abundance of all shared species with the exception of the São Tomé speirops is lower (fig. A2.4, appendix 2).

No species was found exclusively within this land-use and only one, the Giant weaver, had more than two-thirds of the total recordings in shade polycultures (Fig. 7).

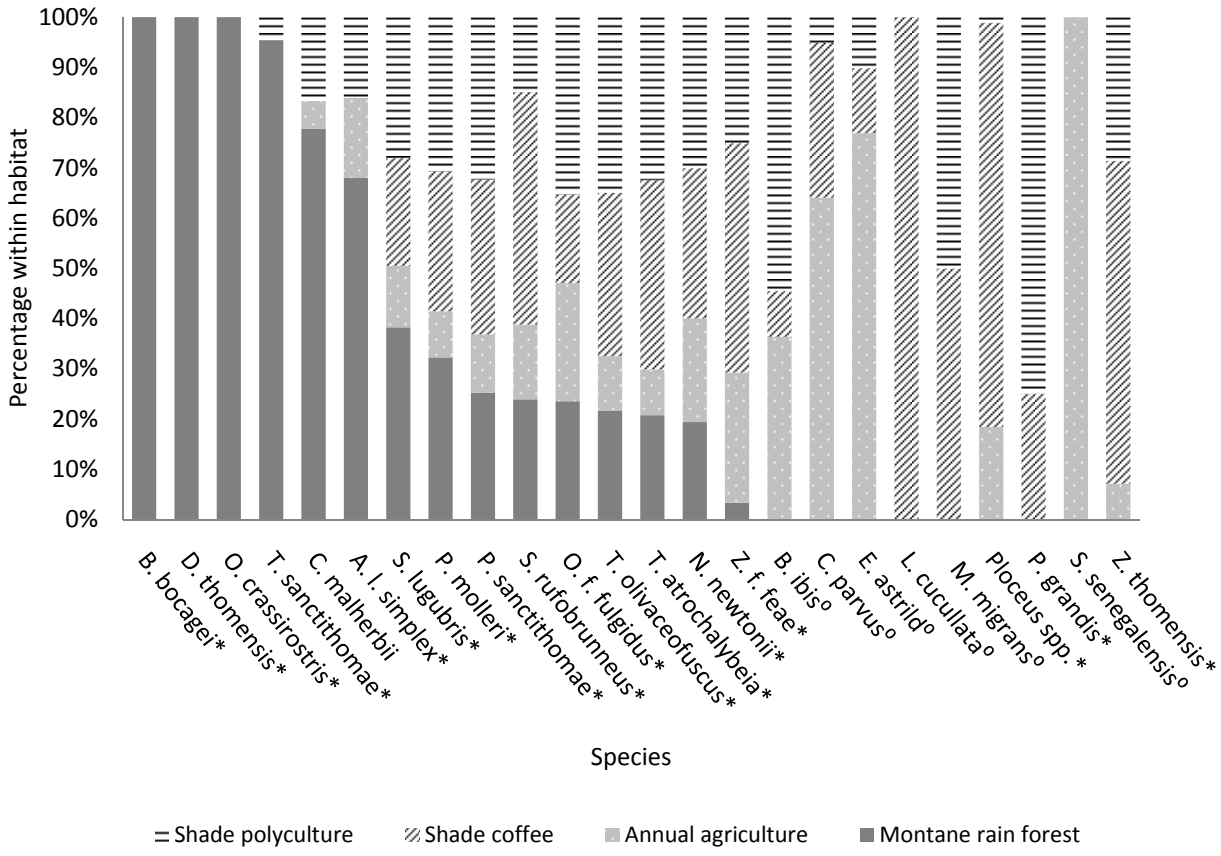

Figure 7 - Percentage of each species within the surveyed land-uses (only data of the core transects was considered for this analysis). * denotes and endemic genus, species or sub-species and º denotes a recent arrival.

4.2 – Species richness and abundance among the four land-uses

During survey time more species were recorded among the agricultural matrix point stations than within the forest. The overall number of recorded species was higher in shade polyculture (20 species), followed by shade coffee and annual agriculture (18) and lastly by montane rainforest (15) (Table 3).

Both Shannon and Simpson diversity indexes were, however, significantly higher for shade coffee plantation. On the other hand the Berger-Parker index was higher for montane rainforest and annual agriculture (0.42 and 0.43 respectively), reflecting the high contribution of a single species to the total number of recordings within those habitats.

Table 3 - Broad measures of species richness for the four different land-uses with standard error in brackets.

	Montane rainforest	Annual agriculture	Shade coffee	Shade polyculture	$F_{3, 38}$	*p*
Total number of observed species[1]	15	18	18	20		
% of endemics[1]	93	67	67	70		
% of recently arrived species[1]	0	27	33	27		
Shannon diversity index[2]	1.69^{c} (0.02)	1.69^{c} (0.03)	2.14^{a} (0.03)	1.91^{b} (0.03)	40.7	<0.001
Simpson diversity index[2]	3.91^{c} (0.07)	4.38^{c} (0.19)	8.50^{a} (0.14)	5.86^{b} (0.01)	196.6	<0.001
Berger-Parker dominance index	0.42^{a} (0.028)	0.43^{ac}(0.06)	0.32^{b} (0.03)	0.31^{ac} (0.01)	3.6	<0.005

[1] *Ploceus velatus peixotoi* and *Ploceus cucullatus nigriceps* are included; [2] Average of mean among runs values after 100 permutations; Standard errors are given in brackets; Different superscripts stand for significant differences.

Note: only data from the core transects is included in this analysis.

4.3 – Differences in *α, β and γ* diversity among the four land-use types

Alpha diversity was found to be low in all land-uses with between 4 to 6 species being found on average per station (table 4). Between-land-use differences in *α, ß* and γ were not significant (Pearson's Chi-squared test, X-squared = 6.8369, df = 6, $p < 0.3362$).

Table 4- Differences in *α, β and γ* diversity among the four land-use types. *α* diversity standard error is given in brackets.

	Montane rainforest	Annual agriculture	Shade coffee	Shade polyculture
α	4.95 (0.63)	4.06 (0.86)	5.9 (0.42)	5.1 (0.67)
ß	10.05	12.94	11.1	13.9
γ	15	17	17	19

Note: only data from the core transects is included in this analysis.

4.4 – Feeding guilds, endemics and recently arrived species

With exception to the omnivores, the different feeding guilds abundance (measured as the mean number of recorded individuals per sample site) has shown to vary among the different land-uses (fig. 8).

Insectivore abundance was significantly lower in annual agricultural with the number of recorded individuals decaying by roughly 50% in comparison to any of the other treatments. The mean number of recorded granivores was higher in shade coffee with almost 20 recordings per sample which contrasts to the 2.33 ± 0.20 (mean ± SE) individuals found per sample in shade polyculture (table 5). Nectarivores were found to have a small, but significant variation among land-uses. Frugivores on the other hand where significantly depleted outside the rainforest.

The number of recorded individuals of endemics species was significantly lower in annual agriculture whereas the number of individuals of recently arrived species was significantly higher.

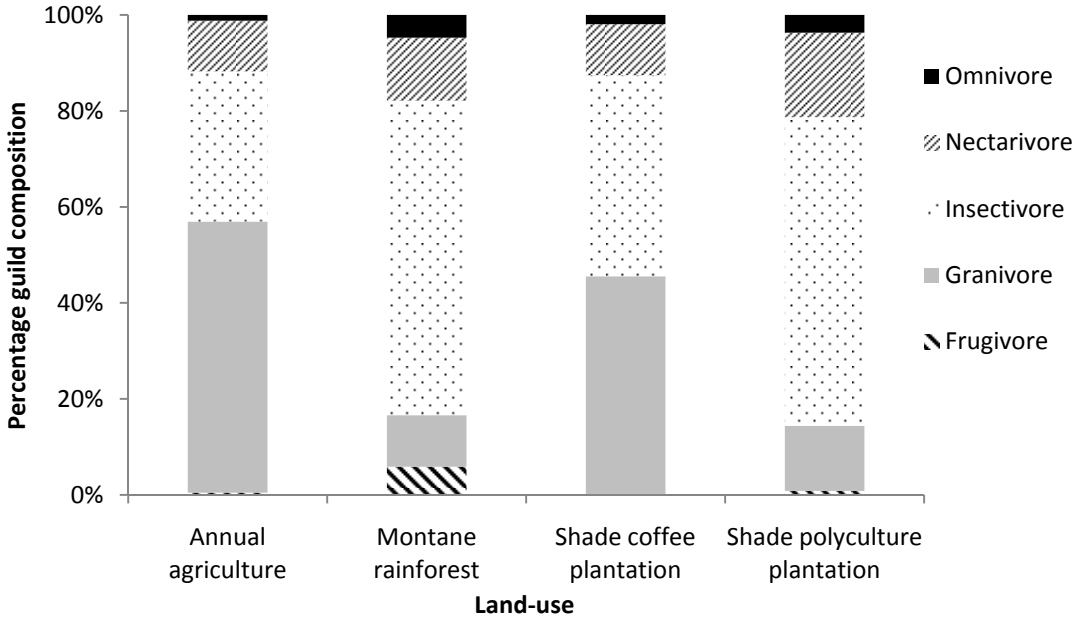

Figure 8 - Land-use composition by different feeding guilds.

Table 5 - Mean (±1SE) abundance of different feeding guilds and arrival times in each land-use type.

Bird guild or group	Montane rainforest	Annual agriculture	Shade coffee	Shade polyculture	Test statistic df= 3, 38	*P*
Insectivores	9.47 ±0.94^{a}	5.55±1.04^{b}	13±0.29^{a}	10.00±1.18^{a}	F = 7.1	<0.001
Granivores	1.55±0.17^{a}	11.85±2.24^{b}	19.22±6.28^{c}	2.23±0.5^{a}	F = 25.7	<0.001
Nectarivores	1.90±0.2	1.90±0.26	2.75±0.20	2.77±0.20	F = 7.1	<0.05
Omnivores	0.68±0.12	0.22±0.09	0.50±0.29	0.58±0.15	X^2= 7.6	n.s
Frugivores	0.85±0.22 ^{a}	0.08±0.04 ^{b}	0.13±0.02 ^{b}	0^{b}	X^2= 23.5	<0.001
Endemics	14.22±1.07^{a}	9.98±1.27^{b}	24.33±3.04^{a}	14.57±1.25^{a}	F = 7	<0.001
Non endemics	0.23±0.9^{a}	9±2.33^{b}	8.98±3.70^{b}	1.27±0.47^{a}	X^2= 30.2	<0.05
Natives	14.45±1.11^{a}	8.30±1.47^{b}	16.90±1.47^{a}	14.52±1.25^{a}	F = 7.2	<0.001
Recent colonisers	0^{a}	11.37±2.40^{b}	16.42±6.78^{b}	1.32±0.49^{a}	X^2= 33.7	<0.001

Note: only data from the core transects is included in this analysis; different superscripts stand for significant differences.

4.5 – Species assemblages

The NMDS based only on data from the core points resulted in a two dimensional final solution with a stress value of 17.5, which is within the range of 15-20 of most ecological studies (Naidoo, 2004). Grouping of same land-use count stations was significant ($F_{3, 38} = 9.6$, $p < 0.01$) showing that species composition is significantly different among the studied land-uses. Montane rainforest count stations formed a distinctive group, closer to the cluster of shade plantation count stations than with annual agriculture count stations (Fig.9)

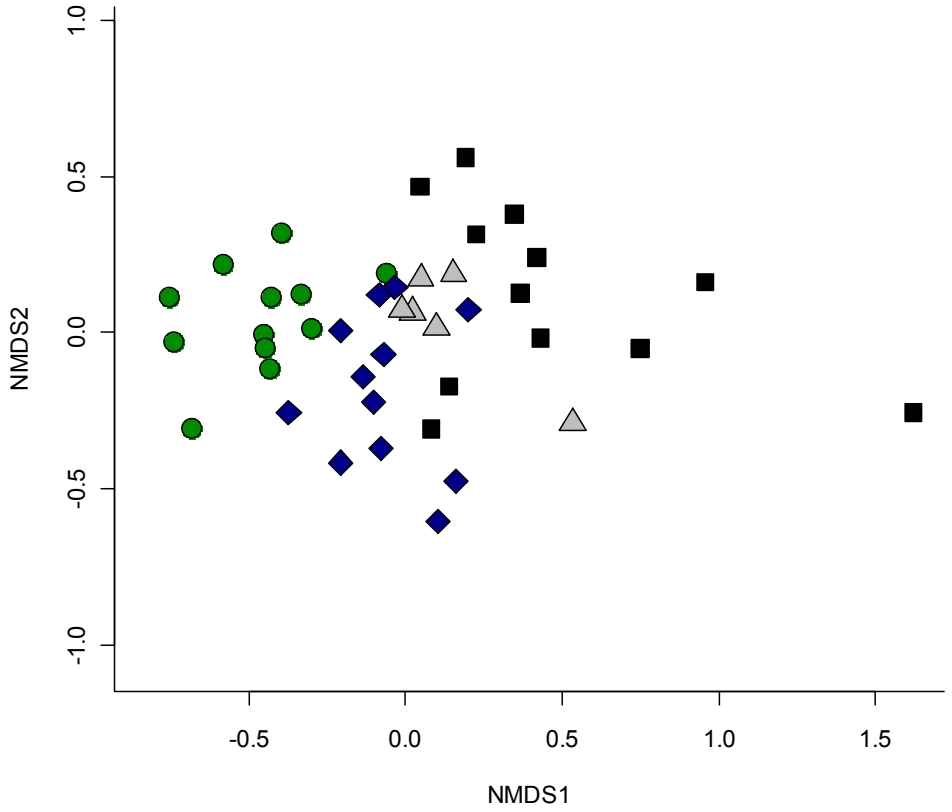

Figure 9 - Non-metric multi-dimentional scaling (NMDS) plot of the different stations among all analysed land-uses: montane rainforest (●), annual agriculture (■), shade coffee (▲) and shade polyculture (♦).

When edge stations were included in the NMDS (fig. 10), the goodness-of-fit decreased (stress value increased to 23.3). Nevertheless a one-way MANOVA using the stations scores extracted from the two-dimensional ordination revealed that grouping among some land-use stations was still significant ($F_{3, 110} = 16.8$, $p < 0.01$).

The computed Euclidean distances between each station and the centre of its group cluster has shown dispersion to be relatively similar in montane rainforest (0.34 ± 0.03; mean ± 1SE error), shade coffee (0.35 ± 0.18) and shade polyculture (0.33 ± 0.2) but much higher between the annual agriculture stations (0.48 ± 0.04); Differences were found to be significant ($F_{3, 110} = 2.82$, $p < 0.05$).

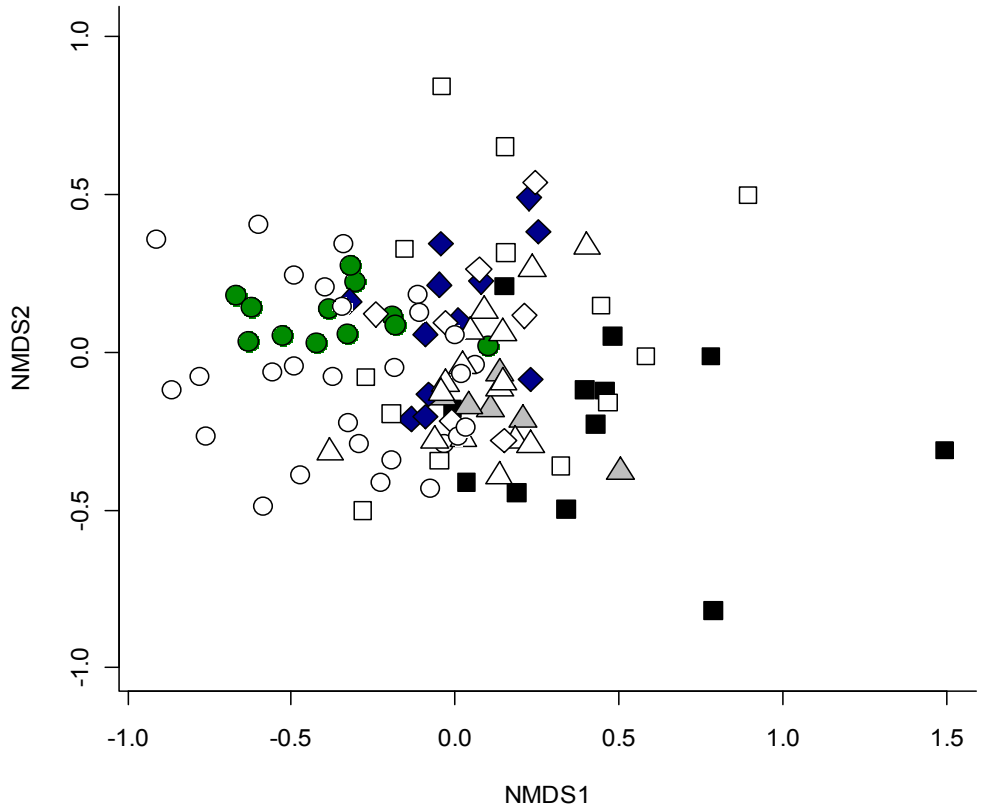

Figure 10 - Non-metric multi-dimensional scaling (NMDS) plot of the different stations among all analysed land-uses. Core transects are represented by: montane rainforest (●), annual agriculture (■), shade coffee (▲) and shade polyculture (♦). Edge transects are represented by: ○, □, Δ, ◊.

A Mantel test has shown the Bray-Curtis values to be correlated with geographical distance (Mantel test with 1000 permutations: r=0.05; P < 0.01).

4.6 – Vegetation variables

As expected, vertical structure complexity was higher for montane rainforest for which all the vegetations variables used as surrogates for structural complexity scored the highest (Table 6). Both shade plantation types presented similar values for most variables with the biggest difference being at the level of the mid-canopy

cover which was absent in shade coffee plantation. The percentage of understory canopy cover in shade coffee plantation (which was exclusively due to 2-3 m coffee trees) was comparable with the forest value but all the other variables reflect the considerably lower structural complexity of both shade plantations types when compared with the rainforest. Upper-canopy and mid-canopy values for montane forest were more than double in relation to both shade plantations and the contrast was even greater for the number of recorded tree species. The biggest difference was found at the tree density level which was nearly an order of magnitude higher for forest than for any of the shade plantations. Annual agriculture scored the lowest for most vegetation variables and presented almost negligible values of upper and understory canopy cover.

Table 6 - Descriptive statistics of vegetation variables based on estimates made on a 10 m radius around each point count station; mean is given with standard error in brackets. N is sample size.

Land-use	*N*	*Cover (%)*			*Number of species*	*Maximum vegetation height (m)*	Bare ground (%)
		Upper-canopy	*Mid-canopy*	*Understorey canopy*			
MR	12	68.3 (2.39)	45.0 (1.94)	20.0 (1.29)	6.58 (0.74)	29.2 (1.56)	0
AA	12	5.83 (0.70)	20.0 (1.29)	6.67 (0.74)	1.5 (0.35)	10.4 (0.93)	79.2 (2.56)
SC	6	30.0 (2.23)	0	20.0 (1.83)	1.83 (0.55)	30.8 (2.27)	0
SP	12	20.8 (1.32)	20 (1.29)	10.4 (0.93)	1.50 (0.35)	19.9 (1.28)	3.33 (0.527)

Note: only data from the core transects is included in this analysis.

The estimated values represent a gradient of vegetation complexity decline going from montane rainforest to shade plantations and finally to annual agriculture plots. The position in which one shade plantation type would be allocated in relation to the other is debatable because despite the higher values of upper and understory canopy cover, tree density, number of species and maximum vegetation height found in the coffee shade plantation, this land-use type lacks one vegetation strata – the mid-canopy which can be found in shade coffee plantation.

4.7 – Interaction between edge and distance to edge

The ANCOVAs carried out upon the count station scores along the NMDS first axis revealed no significant results for the interaction between edge distance and edge type (Table 7) (Fig. 11 and 12).

Table 7 - Tests statistics of the ANCOVAs assessing the effect of edge type, edge distance and their interaction on the community composition for montane rainforest and shade coffee edge transects.

Variables	Montane rainforest edge transects			Shade coffee edge transects		
	Coeff.	t	*p*	Coeff.	t	*p*
Edge type	-0.00005	**-0.143**	***n.s.***	**0.028**	**0.75**	***n.s.***
Edge distance	**-0.205**	**-2.728**	***< 0.01***	**0.00005**	**0.18**	***n.s.***
Interaction: edge type vs distance	**-0.0001**	**1.18**	***n.s.***	**-0.0004**	**-0.37**	***n.s.***
	$F_{3, 41} = 2.491$ $r^2 = 0.154$ $p < 0.01$			$F_{3, 32} = 0.6763$ $r^2 = 0.05$ $p < 0.01$		

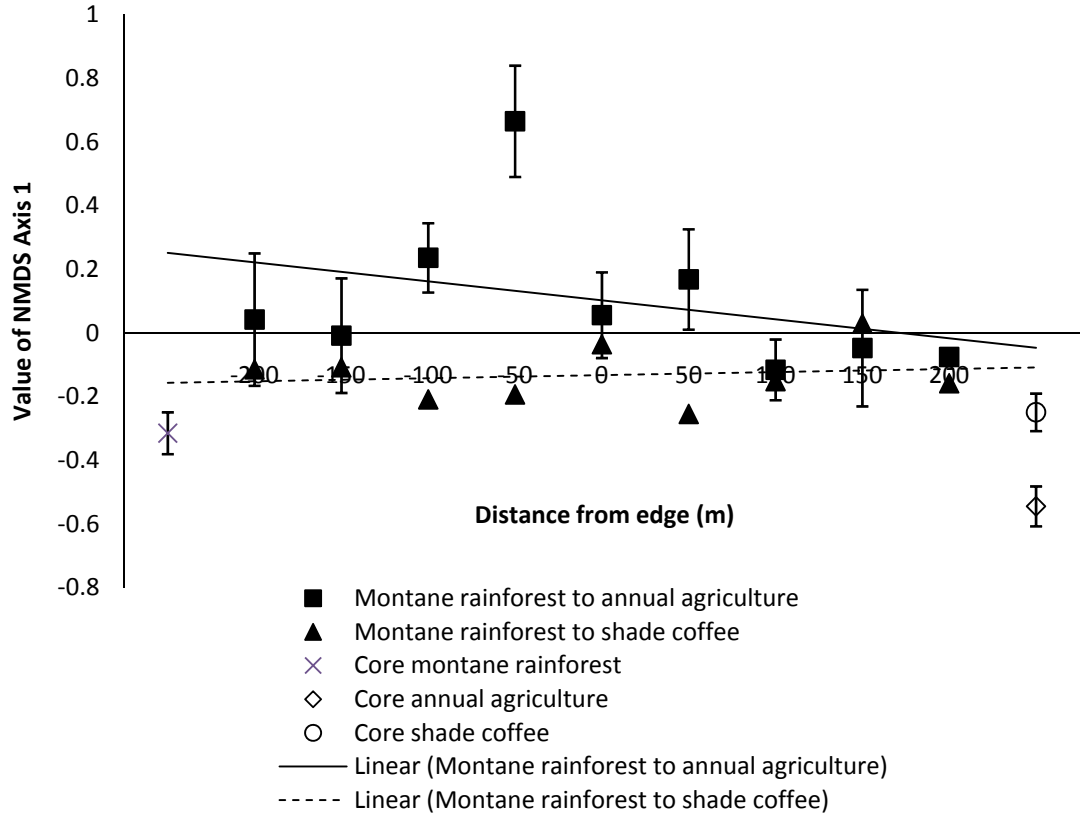

Figure 11 - Community composition[1] change along edge transects going from montane forest (200m) to shade coffee (-200) and from montane rainforest (200m) to annual agriculture (-200). Values represent the

mean among same distance stations of the same edge type transect and error bars represent standard errors. Standard errors were not calculated for the montane rainforest shade coffee transect due to insufficient number of samples (n=2).

[1]as given by the NMDS first axis station scores.

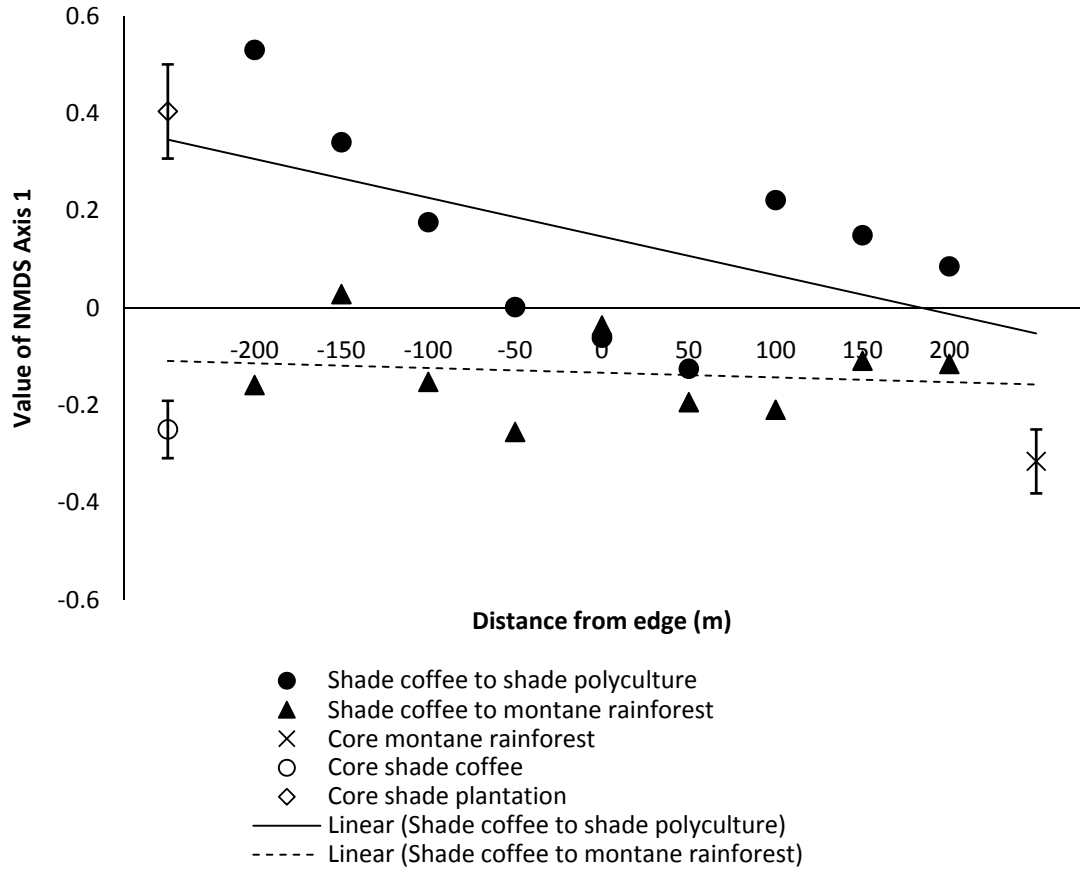

Figure 12 - Community composition[1] change along edge transects going from shade coffee (-200m) to shade polyculture (200) and from shade coffee (-200m) to montane rainforest (-200). Values represent the mean among same distance stations of the same edge type transect. Standard errors were not calculated due to insufficient number of samples (n=2).

[1]as given by the NMDS first axis station scores.

4.8 – Bird habitat relationships

4.8.1 – Similarity to forest, community composition and diversity

Edge type was found to have the strongest effect upon all three response variables (Bray-Curtis similarity to forest controls, community composition as given by the NMDS first axis and Shannon indices) (see table 7). No landscape variable was

retained in the minimum adequate linear model using similarity to forest as a response variable. Distance to forest controls was however significantly related with changes in community composition and the amount of agricultural land within a 250m buffer from the count station was found to negatively affect bird diversity.

Maximum vegetation height and percentage of bare ground where found to impact communities' similarity to forest in opposite directions with an increase in vegetation height being associated with a highest similarity to forest, whereas an increase in bare ground is associated with lesser overlap with forest controls (fig 13 a) and b)). Diversity however was found to be negatively influenced by increases in maximum vegetation height and bare ground (fig. 13 c) while being positively related to tree density.

Table 7- Effect of landscape, edge and local variables in bird communities' similarity to forest, composition and diversity (Shannon index). Regression coefficient (Coeff.), t-value (t) and p-value (*p*) given.

Variables	Similarity to forest			Community composition			Diversity		
	Coeff.	t	*p*	Coeff.	t	*p*	Coeff.	t	*p*
Landscape									
Distance to controls				-0.001	-3.292	< 0.01			
Distance to settlements							-9.98E-02	-1.787	n. s.
AA within 250m buffer							-2.93E-01	-1.997	< 0.01
Edge									
MR core	-1.04	-2.37	< 0.05	-3.07	-1.714	n. s.	-0.23	-1.566	n. s.
SC core	-0.03	-0.56	n. s.	0.49	2.478	< 0.05	-0.52	-2.927	< 0.01
MR-AA	-0.17	-3.84	< 0.01	0.44	2.491	< 0.05	-0.79	-10.09	< 0.01
MR-SC	-0.24	-5.06	< 0.01	0.62	3.92	< 0.01	-0.89	-7.518	< 0.01
SP-SC	-0.31	-6.56	< 0.01	0.64	3.992	< 0.01	-1.26	-8.362	< 0.01
Distance to edge	0.0008	1.12	n. s.						
Distance to edge:MR core	0.01	3.03	< 0.01	0.02	1.492	n. s.			
Distance to edge: AA core				-0.002	-0.813	n. s.			
Distance to edge: SC core	-0.00009	-0.09	n. s.	0.007	2.297	n. s.			
Distance to edge: MR-AA	0.0006	0.49	n. s.	-0.003	-0.737	n. s.			
Distance to edge: MR-SC	-0.001	-0.91	n. s.	-0.0008	-0.158	n. s.			
Distance to edge: SP-SC	-0.002	-1.44	n. s.	-0.003	-0.63	n. s.			
Local									
Max. vegetation height	0.02	2.885	< 0.01				-0.51	-3.407	< 0.01
Bare ground (%)	-0.006	-1.996	< 0.01				-0.01	-2.461	< 0.05
Tree density							0.04	2.223	< 0.01
Epiphyte low							-0.18	-2.617	< 0.01
Epiphyte medium							-0.08	-1.675	< 0.01

Epiphyte high			-0.11	-1.313	< 0.01
	R^2=0.926, *p*=<0.01 $F_{13,91}$=87.0	R^2=0.543, *p*=<0.01 $F_{12,92}$=9.12	R^2=0.872 *p*=<0.01 $F_{13,91}$=47.6		

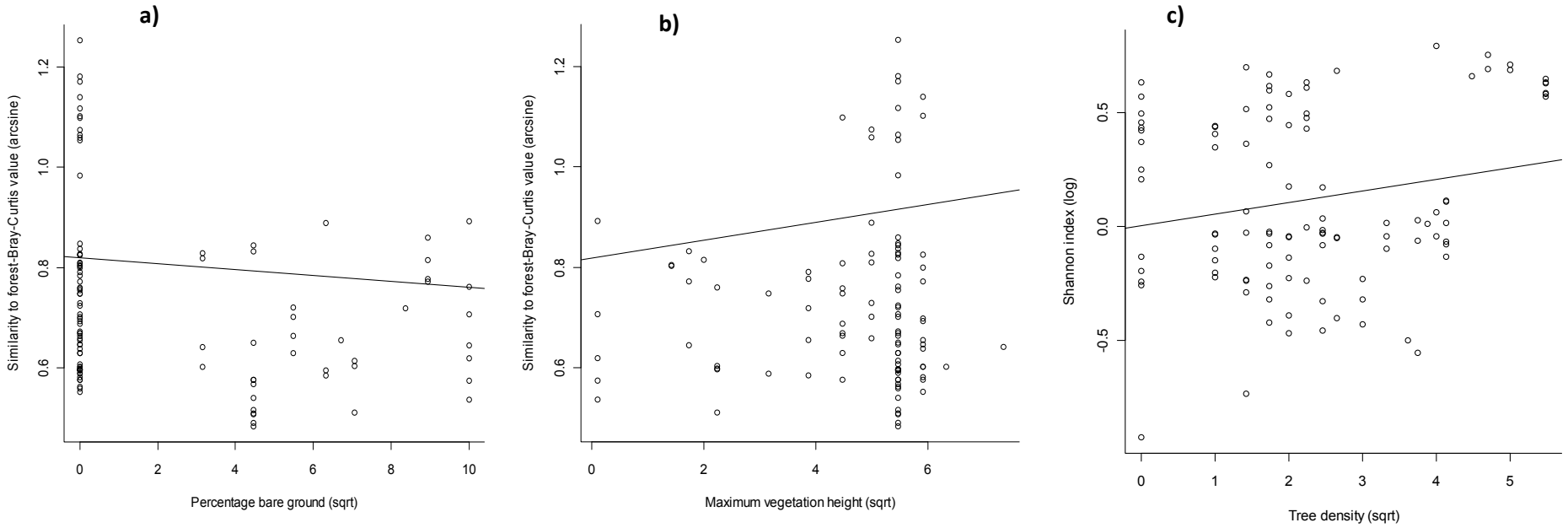

Figure 13 - Relationship between community similarity to forest based on a Bray-Curtis similarity matrix and a) percentage of bare ground, $p < 0.05$, $r^2 = -0.006$; b) maximum vegetation height, $p < 0.01$, $r^2 = 0.002$ and c) relationship between Shannon diversity index and maximum vegetation height, $p < 0.01$, $r^2 = 0.004$.

4.8.2 – Feeding guilds

Landscape, edge and local variables impact the different feeding guilds in different ways. No landscape variable was retained in the frugivore minimum adequate model, edge type however was showed to be significant and at a local scale maximum vegetation height was the variable that contributed the most (table 9).

Granivore abundance is significantly related with the area of annual agricultural and shade coffee plantation on a landscape scale and is negatively related with maximum vegetation height (fig. 14 c)) and density of epiphytes on a local scale. Also on a local scale insectivore abundance was found to be negatively associated with the percentage of bare ground and epiphytes and positively correlated with tree density (fig. 14 a) and b)).

Table 9 - Variables retained in the minimum adequate models of the different feedings guild abundance response to landscape, edge and local variables. Regression coefficient (Coeff.), t-value (t) and p-value (*p*) given.

Variable	Frugivore			Granivore			Insectivore			Nectarivore			Omnivore		
	Coeff.	**t**	***p***	**Coeff.**	**t**	***p***	**Coeff.**	**t**	***p***	**Coeff.**	**t**	***p***	**Coeff.**	**t**	***p***
Landscape															
Distance to controls				0.0002	1.9210	n. s.									
Distance to settlements				-0.459	-2.126	< 0.01	0.3032	1.9440	n. s.	-0.360	-2.021	< 0.05			
MR within 250m buffer				1.6460	2.8170	< 0.01	2.3335	2.6860	< 0.01				0.7504	2.7460	< 0.01
AA within 250m buffer				-3.405	-3.042	< 0.01									
AA within 500m buffer				6.6940	3.7160	< 0.01	3.7191	2.6610	< 0.01	-1.618	-2.731	< 0.01			
SC within 250m buffer										-2.219	-2.656	< 0.01			
SC within 500m buffer				4.5530	4.0970	< 0.01	5.2666	3.6230	< 0.01						
SP within 250m buffer							2.3148	3.3650	< 0.01	-1.701	-3.614	< 0.01			
Edge															
MR core	4.082	2.648	< 0.01	-3.068	-1.714	n. s.	-0.167	-0.553	n. s.	-4.031	-2.145	< 0.05	-0.327	-1.175	n. s.
SC core							0.0470	0.1150	n. s.	0.9542	1.9500	n. s.	0.3620	2.5810	< 0.05
MR-AA	0.3981	2.745	n. s.				-0.017	-0.101	n. s.	-0.422	-1.916	n. s.	0.1089	0.6160	n. s.
MR-SC	0.1242240	0.686	< 0.01				0.2094	0.7130	n. s.	0.2936	0.8850	n. s.	0.2550	1.5190	n. s.
SP-SC	-0.191925	-1.367	n. s.				-0.363	-0.928	n. s.	1.0072	2.1110	< 0.05			
Distance to edge										0.0002	0.6540	n. s.			
Distance to edge:MR core	-0.00328	-2.950	< 0.05							0.0023	1.5440	n. s.			
Distance to edge: AA core	0.0001278	0.0002451	< 0.01												

Variable	Frugivore			Granivore			Insectivore			Nectarivore			Omnivore		
	Coeff.	**t**	***p***	**Coeff.**	**t**	***p***	**Coeff.**	**t**	***p***	**Coeff.**	**t**	***p***	**Coeff.**	**t**	***p***
Edge															
Distance to edge: SC core	0.0011708	0.657	< 0.01	< 0.01						0.0005	0.6640	n. s.			
Distance to edge: MR-AA	0.0017152	3.644	< 0.01							-0.002	-3.744	< 0.01			
Distance to edge: MR-SC	0.001	2.997	< 0.01							-0.003	-3.947	< 0.01			
Distance to edge: SP-SC	0.0002731	0.699	n. 1.							0.0005	0.6110	n. s.	0.3212	2.2940	< 0.05
Local															
Max. vegetation height	0.066	3.07	< 0.01	-0.118	-2.342	< 0.05	0.0537	1.5820	n. s.						
Bare ground (%)							-0.053	-3.813	< 0.01						
Tree density							0.1570	3.334	< 0.01						
Mid canopy cover	0.003	2.00	< 0.05				-0.003	-1.807	n. s.						
Upper canopy cover		2.282								0.003	2.529	< 0.05			
Epiphyte low density	0.0686616	-1.482		-0.658	-2.938	< 0.01	-0.387	-2.856	< 0.01	-0.287	-1.735	n. s.			
Epiphyte medium density	0.1378988	2.197	< 0.05	-0.500	-2.627	< 0.01	-0.235	-2.270	< 0.05	-0.185	-1.810	n. s.			
Epiphyte high density	-0.143983		< 0.05	-0.521	-1.792	< 0.01	-0.455	-2.946	< 0.01	-0.130	-0.733	n. s.			
	R^2=0.719 $F_{17,87}$=13.42 *p*=<0.01			R^2=0.543 $F_{12,92}$=9.12 *p*=<0.01			R^2=0.609 $F_{10,94}$=14.6 *p*=<0.01			R^2=0.596 $F_{19.85}$=6.60 *p*=<0.01			R^2=0.210 $F_{6,38}$=4.34 *p*=<0.01		

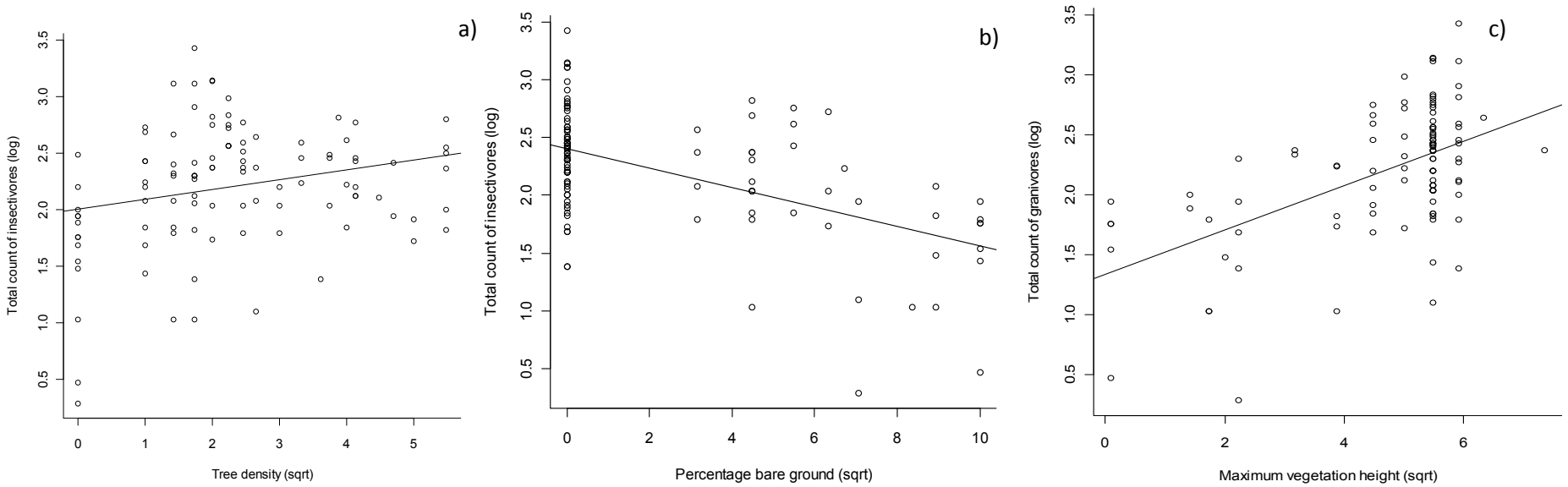

Figure 14 - Relationship between a) insectivores and tree density, $p< 0.01$, $r^2=0.1570$; b) insectivores and bare ground cover, $p< 0.01$, $r^2=-0.053$ and c) granivores and maximum vegetation height, $p< 0.01$, $r^2=-0.118$.

4.8.3 – Endemic and recently arrived species

Endemics abundance was shown to be significantly associated with landscape, edge and local variables (table 10). At the local scale tree density and maximum vegetation height had a positive relationship with abundance (fig. 15) whereas epiphyte density had a negative association. The edge montane rainforest and annual agriculture was also found to influence endemics' numbers and at the landscape level the area of shade coffee within a 500m buffer around each point was also significant.

Abundance of recently arrived species was negatively affected by both montane rainforest edges and at the landscape level was significantly affected by the area of both annual agriculture and shade coffee.

Table 10 - Effect of landscape, edge and local variables on endemic and recently arrived bird species.

Variable	Endemic			Recent		
	Coeff.	**t**	***p***	**Coeff.**	**t**	***p***
Landscape						
AA within 500m buffer				3.5960	2.7570	< 0.01
SC within 250m buffer				9.5220	5.3030	< 0.01
SC within 500m buffer	1.9153	3.5030	< 0.01			
Edge						
MR core	0.0773	0.3470	n. s.	-2.4050	-0.3530	n. s.
SC core	0.5917	3.7380	< 0.01	-1.1890	-1.1700	n. s.
MR-AA	-0.2923	-2.0470	< 0.01	-2.4520	-3.6960	< 0.01
MR-SC	0.0464	0.2310	n. s.	-2.1450	-2.5000	< 0.01
Distance to edge:MR core				0.0000	0.0000	n. s.
Distance to edge: AA core				-0.0018	-1.6650	n. s.
Distance to edge: SC core				0.0005	0.3280	n. s.
Distance to edge: MR-AA				-0.0065	-3.9150	< 0.01
Distance to edge: MR-SC				-0.0015	-0.7050	n. s.
Distance to edge: SP-SC				0.0027	1.1860	n. s.
Local						
Max. vegetation height	0.0891	2.2960	< 0.05			
Tree density	0.1153	2.2100	< 0.01			
Epiphyte low	-0.4243	-3.2110	< 0.01			
Epiphyte medium	-0.3834	-3.5090	< 0.01			
Epiphyte high	-0.5099	-3.5170	< 0.01			
		R^2=0.648			R^2=0.673	
		p=<0.01			*p*=<0.01	
		$F_{11,93}$=15.6			$F_{13,91}$=14.4	

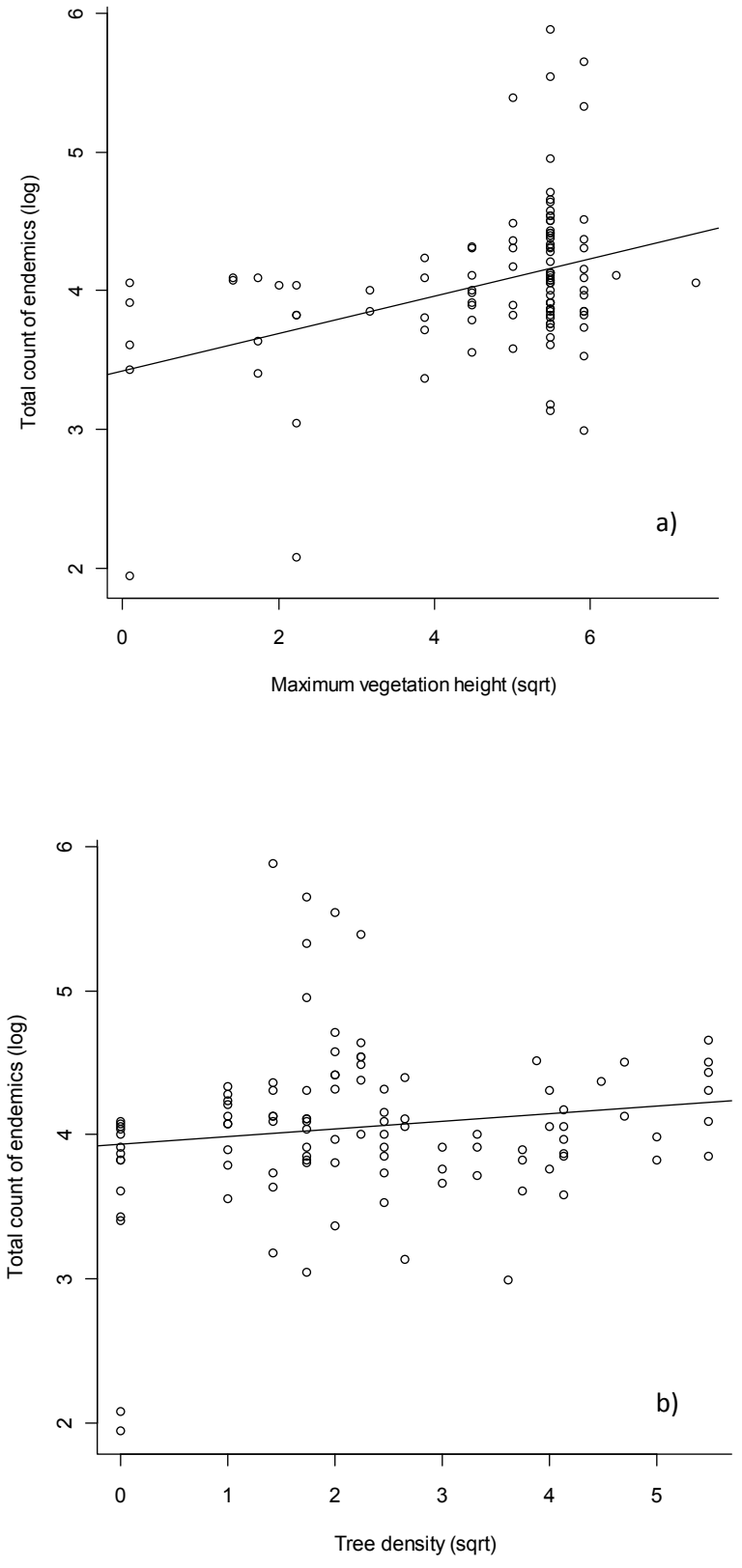

Figure 15 - Relationship between endemic bird species and a) maximum vegetation height, $p<0.05$, $r^2= 0.0891$ and b) tree density, $p<0.01$, $r^2= 0.1153$.

5. Discussion

The simple analysis presented here adds to the evidence that tropical native forest bird species are adversely affected by habitat modification. Consistent with previous studies (Thiollay, 1995; Tejeda-Cruz & Sutherland, 2004; Waltert *et al*, 2005; Harvey *et al.,* 2006) the analysed agricultural matrix showed evidence of supporting rich but modified bird communities. However, distribution in terms of species numbers across the analysed landscape differs from the ones found in past studies. As the extent of my knowledge goes, this is the first study specifically regarding the impact of agricultural practices in tropical ecosystems to report a higher number of species within the agricultural matrix than within its surrounding rainforest. This outcome is likely due to the relatively low native species pool and the proportionally high number of open habitat specialists introduced to islands. Despite being the habitat type with the lowest species richness recorded, montane rainforest was the one where more endemic species was found.

A large turn-over from forest-type species to more open-habitat has been found and this is consistent with the results of previous studies (Hughes *et al*, 2002; Naidoo, 2004; Waltert *et al*., 2005) and since most open habitat specialists present in the islands are considered to be introduced (Jones & Tye, 2006) this provides some evidence of the synergies between land-use change and the spread of non-native species (Didham *et al.,* 2007).

5.1 -Habitat impact in species assemblages

Birds within the studied landscape have been found to respond significantly to differences in land-use type. Count stations in both shade plantations have been shown to cluster closer to rainforest than rainforest to agricultural stations, suggesting thereby a more similar community structure. These results are consistent with other studies regarding the study of gradients of agricultural intensity within tropical landscapes (Naidoo, 2005; Waltert *et al*., 2005) showing that bird species respond to gradients of intensity and that communities are more dissimilar to forest in more intensive agricultural practices associated with lesser vegetational

complexity. This supports the hypothesis that São Tomé's native bird species will be negatively impacted by a replacement of shade plantations for more open agricultural areas.

5.1.1 Feeding guilds

Different groups of bird species seem to respond differently to the analysed land-uses. Insectivores are known to present marked responses to land-use change (Thiollay 1995; Waltert *et al*., 2005; Komar, 2006), which was confirmed for annual agricultural areas where insectivore's mean number of recordings per visit decayed by 50% in relation to forest controls, but was not confirmed for shade coffee and shade polyculture, where mean number of recorded individuals per visit differed little from forest. In fact it was even slightly superior for shade coffee plantations. Despite some evidence of frugivore depletion in agricultural landscapes (Tejeda-Cruz & Sutherland, 2004; Matlock Jr. *et al*., 2006), most literature referring to both this guild and omnivores presents them as being less impacted by land-use change and often even more common in agricultural areas such as shade plantations than in nearby forests (Naidoo, 2004; Waltert *et al*., 2005; Komar, 2006). Within the studied landscape this trend held true for omnivores, for which no significant differences in detection rate among the four land-uses was found, but false for the frugivores, for which a pronounced decrease in the number of detections was observed. This reduction in the detections was found despite the fact that a large number of fruit trees known to the used by the frugivores, such as *Ficus* spp., were scattered across the landscape in both annual agricultural areas and shade plantations. It is unlikely therefore that lack of food resources acts as the main driver of such a pronounced decrease in abundance and others reasons such as behavioural barriers towards entering the agricultural matrix (Harris & Reed, 2002) or behavioural responses to hunting pressure which is strong in the study area (pers. obs) may be playing a main role. As found in other studies granivores seem to be the winners when it comes to land-use changes leading to more open landscapes (Naidoo, 2004; Tejeda-Cruz & Sutherland, 2004; Waltert *et al*., 2005). For this guild a tremendous increase in the number of recordings was found for both shade coffee and annual agriculture.

5.1.2- Sensitive species

Previous studies regarding São Tomé avifauna's habitat use had already identified that endemic species do respond differentially to land-use change (Peet & Atkinson, 1994). The same can be seen by the results of this study. Out of the 27 species identified during field-work, 6 had more than 60% of their recordings within rainforest, and 3 of those, the IUCN Critically Endangered Dwarf Ibis and the IUCN Vulnerable Giant Sunbird and São Tomé Oriole were not once recorded in the agricultural matrix. These species can therefore be said to be extremely sensitive to habitat modification and their populations can be in serious danger if agricultural expansion continues to take place.

5.2 Landscape and local habitat variables

Both landscape and local scale variables were retained in the minimum adequate models highlighting the importance of considering these two scales in the description of bird diversity and abundance patterns across a given landscape.

At the landscape level two of the most interesting relationships are the ones between area of annual agriculture and recently arrived species and between area of shade coffee and recently arrived species. According to the model the abundance of this group is expected to increase with the increase of these two landscape features which is consistent with the hypothesis that land-use change would benefit non-native species spread.

At the local level, the positive regression coefficients of similarity to forest and tree density and of frugivore, insectivore and endemic abundance on tree density is consistent with previous findings regarding the influence of vegetation complexity in the variation of species across agricultural intensity gradients (Heikkinen *et al*., 2004; Waltert *et al*., 2005; Harvey *et al*., 2006). This has important policy implications since tree felling is common within the study landscape, including large tree poaching within the area of the Natural Park.

5.3 - Edge effect

One of the objectives of this study was to test the hypothesis that edge effect was one of the drivers of species distribution and abundance across the landscape. This has proven to be true with edge effect having a significant influence in similarity to forest, community composition, diversity and abundance of frugivores, insectivores, nectarivores, endemics and recently arrived species. Different edge types were also found to impact the referred parameters in different ways. Distance to edge failed to provide significant results, as found by Pimentel (2006), and this lack of significance in both studies may be due to the selected distance ranges; birds are known to be extremely mobile (Sutherland *et al*., 2004) and one of the main constraints in their sampling is related with the perverse consequences of double counting the same individual (Bibby *et al*., 1998). During field work this was taken into account and mental notes of birds' movements were kept in order to avoid possible double counting. This approach is, however, far from being reliable and consequently some birds must have been counted more than once probably at different distances. The small number of replicates for each land-use may also be presented as a justification for the lack of significant results.

5.4 Study limitations

One of the main constraints of the present work relates to the lack of replicates for the core transects and with the small number of replicates for the edge transects. More replicates would have allowed more rigorous statistical analysis and therefore more robust conclusions.

Another limitation relates to the dramatic decrease in the relative proportion of visual recordings in relation to sound recordings in montane rainforest in relation to other land-use types which seems to indicate that differences in detectability may have influenced the results. A more efficient and equitable way of recording bird data across the different habitats would have been to follow the approach used by Naidoo (2004) and record sounds at each sampling station and identify the vocalizations from the tapes. This approach however would only provide information about presence / absence of species within a point count and precious abundance information would be lost.

This study is also sensitive to the “shifting baseline syndrome” described by Dayton *et al*., (1998). If the selected core montane rainforest transect already had an altered community due to anthropogenic stresses the baseline for the comparison of the impact of the different land-use types upon original rainforest species would be incorrect and therefore an incomplete picture would be gathered. This is likely to be the case since one species, the Maroon Pigeon, is reported as having one of its strongholds within the core montane rainforest transect area (Christy & Clarke 1998) but despite all the repeated visits in both survey and non survey time the species failed to be recorded. This is worrying since the species is known to be heavily hunted (Carvalho, *pers comm*) and very little is known about its current distribution and population size.

5.5 Further work

Simple measurements of relative abundance and diversity such as the ones used in this study are not enough to capture the complex dynamics regarding bird use of humanized landscapes. There’s a need to move towards studies that take into account species interactions, differences in fitness and the long term viability of the populations and other than focussing on a single taxa, use a multi-taxa approach in such a way that comparable results can be provided. On São Tomé one interesting group to address would be the amphibians, which during the field work phase of this project have been identified to use the agricultural matrix for reproductive ends. One of those, the São Tomé endemic caecilian, has even been noticed to reach higher population densities in the agricultural areas associated with banana plantations than in forest (Uyeda, *pers comm.*).

Despite addressing the impacts of agriculture this thesis did not address the impacts of farming practices, such as fertilizer, pesticide and herbicide applications. Within the study landscape the backbone of the shade systems if formed by *Erythrina* spp. which is a nitrogen-fixing legume; as these are felled, there’s as need to apply chemical fertilizers (Oliveira, *pers. comm*.) and valuable ecosystem services, like pest control provided by insectivores birds are reduced, consequently resulting in a need for an increase in pesticide application. A more complete study on the impact of agricultural activities within the different land-uses would thereby provide a risk assessment that directly addresses the effects of agrochemicals.

5.6 Conservation considerations

Several findings of special conservation relevance have arisen from this project one of which regards the recording of the *Bostrychia bocagei*, the Dwarf olive ibis, within the study area at an altitude of roughly 1400 meters. This species is classified as "Critically Endangered, D1" (population <50 adults) by Birdlife International (2000) and has never been recorded at such a high altitude (Jones & Tye, 2006). Two other species of special conservation interest, the Critically Endangered *Lanius newtoni* (BirdLife International, 2000), and the Vulnerable *Amaurocichla bocagei* (BirdLife International, 2000) were also recorded within the proximity of the study area at an altitude of 1100 meters. For both species this constitutes the second highest altitude record (Dallimer *et al*., 2003; Olmos & Turshak, 2007) and for the *Amaurocichla bocagei* constitutes the first confirmed recording of reproduction activity in montane rainforest (fig. 3, p.14). These recordings considerably increase the conservation relevance of montane rainforest of the island and thereby of the scope of this study.

5.7 Policy implications

The present study highlights the enormous importance of tree density for the conservation of São Tomé bird species across the agricultural landscape and therefore suggests that land-use types that retain substantial amount of tree cover should be favoured for those that do not. Policies should be adopted in order to avoid tree felling and reforestation should be supported whenever possible.

Since bird's community similarity to forest has been seen to be higher in agroforestry systems than in annual agricultural plots, the adoption of this land-use type in the buffer area around the natural park may prove to be successful.

5.8 Conclusions

Both shade plantations have been found to support bird communities more similar to forest than annual agriculture. It can be said that the promotion of agroforestry systems would aid in the preservation of São Tomé's native bird species, however, several forest species simply do not occur within shade plantations and therefore the transformation of forest into shade plantations would be of limited value.

The discovery of the critically endangered species in the montane rainforest is a sign of hope. However, their conservation must be guaranteed and that will only be achieved by the conservation of extensive areas of native vegetation.

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Appendix 1

Figure A1.1 – Montane forest

Figure A1.2 – Annual agriculture

Figure A1.3 – Shade coffee plantation

Figure A1.4 – Shade coffee plantation

Table A1.1- Within survey time observed species

Species	English name	Endemism	Status[1]	Origin[2]	Feeding guild[3]	Habitat[4]
Aplopelia larvata simplex	Lemon dove	Endemic subspecies	LC	N	F	MR, AA, SP
Bostrychia bocagei	Dwarf ibis	Endemic species	CR	N	I	MR
Bubulcus ibis	Cattle egret		LC	RC	I	AA, SC, SP
Chrysococcyx cupreus insularum	Emerald Cuckoo	Endemic subspecies	LC	N	I	AA
Columba malherbii	Gulf of Guinea bronze-naped pigeon		LC	N	F	MR, AA, SP
Cypsiurus parvus	Palm swift		LC	RC	I	MR, AA, SC, SP
Dreptes thomensis	Sao Tome giant sunbird	Endemic genus	VU	N	N	MR
Estrilda astrild	Common waxbill		LC	RC	G	AA, SC, SP
Francolinus afer	Red-necked francolin		LC	RC	G	SP
Lonchura cucullata	Bronze Mannikin		LC	RC	G	SC
Milvus migrans	Black kite		LC	RC	C	AA, SC, SP
Nectarinia newtonii	Newton's yellow-brested sunbird	Endemic species	LC	N	N	MR, AA, SC, SP
Oriolus crassirostris	Sao Tome oriole	Endemic species	VU	N	O	MR
Onychognathus fulgidus fulgidus	Forest chestnut-wing starling	Endemic subspecies	LC	N	O	MR, AA, OF, SP
Ploceus cucullatus nicriceps	Village weaver		LC	RC	G	AA, SC, SP
Ploceus velatus peixotoi	African masked weaver	Endemic subspecies	LC	RC	G	AA, SC, SP
Ploceus grandis	Giant weaver	Endemic species	LC	N	G	AA, SC, SP

Prinia molleri	Sao Tome prinia	Endemic species	LC	N	I	MR, AA, SC, SP
Speirops lugubris	Sao Tome speirops	Endemic species	LC	N	I	MR, AA, SC, SP
Serinus rufobrunneus	Principe seedeater	Endemic species	LC	N	G	MR, AA, SC, SP
Streptopelia senegalensis	Laughing dove		LC	RC	G	AA
Terpsiphone atrochalybeia	Sao tome paradise flycatcher	Endemic species	LC	N	I	MR, AA, SC, SP
Turdus olivaceofuscus Olivaceofuscus	Gulf of Guinea thrush	Endemic subspecies	NT	N	O	MR, AA, SC, SP
Treron sanctithomae	Sao Tome Green Pigeon	Endemic species	LC	N	F	MR, SP
Ploceus sanctithomae	Sao Tome weaver	Endemic species	LC	N	I	MR, AA, SC, SP
Zosterops ficedulinus feae	Principe white-eye	Endemic subspecies	VU	N	I	MR, AA, SC, SP
Zoonavena thomensis	Sao Tome spinetail	Endemic species	LC	N	I	MR, AA, SC, SP

[1]IUCN Threat categories: CR, Critically Endangered; VU, Vulnerable; NT, Near-Threatened; LC, Least Concern.

[2] Origin following Jones & Tye (2006): N, Native; RC, Recent colonisation.

[3] Feeding guild: F, Frugivore; G, Gravinore; I, Insectivore; N, Nectarivore; O, Omnivore.

[4] Habitats where recorded: MR, Montane Rainforest; AA, Annual Agriculture; SC, Shade Coffee; Shade Policulture.

Table A1.2 - Outside survey time observed species

Species	English name	Endemism	Status[1]	Status[1]	Feeding guild[3]	Habitat[4]
Amaurocichla bocagei	São Tome short-tail	Endemic genus	VU	N	I	MR
Alcedo cristata thomensis	Malachite kingfisher	Endemic subspecies	LC	N	F	AA
Coturnix delegorguei	Harlequin quail	Endemic subspecies	LC	N	G	SP
Lanius newtoni	Sao Tome fiscal shrike	Endemic species	CR	N	I	MR
Otus hartlaubi	Sao Tome scops owl	Endemic species	VU	N	C	MR
Phaethon lepturus	White-tailed tropicbird		LC	N	F	AA
Serinus mozambicus	Yellow-fronted canary		LC	RC	G	AA, SC
Tyto alba thomensis	Barn owl	Endemic subspecies	LC	RC	C	AA
Vidua macroura	Pin-tailed widow		LC	RC	G	AA, SC

[1]IUCN Threat categories: CR, Critically Endangered; VU, Vulnerable; NT, Near-Threatened; LC, Least Concern.

[2] Origin following Jones & Tye (2006): N, Native; RC, Recent colonisation.

[3] Feeding guild: F, Frugivore; G, Gravinore; I, Insectivore; N, Nectarivore; O, Omnivore.

[4] Habitats where recorded: MR, Montane Rainforest; AA, Annual Agriculture; SC, Shade Coffee; Shade Policulture.

Table A1.3- Correlation values between habitat variables with correlation values above 0.8 in italic bold. Columns contain those variables that were not included in the models. Row names are the coded variable names, see appendix 1, table A1.4 for full variable name.

	AA within 500m	MR within 500m	SC within 500m	AA within 750m	MR within 750m	SC within 750m	SP within 750m	Abundance of climbers
AA250m	***1.00***	-0.16	-0.39	***0.98***	-0.20	-0.42	-0.41	0.19
MR250m	-0.16	***1.00***	-0.44	-0.16	***0.98***	-0.37	-0.60	0.09
SC250m	-0.39	-0.44	***1.00***	-0.37	-0.45	***0.94***	0.06	-0.13
SP250m	-0.53	-0.72	0.44	-0.52	-0.67	0.42	***0.90***	-0.20
AA500m	0.00	-0.16	-0.39	***0.98***	-0.20	-0.42	-0.41	0.19
MR500m	-0.16	0.00	-0.44	-0.16	***0.98***	-0.37	-0.60	0.09
SC500m	-0.39	-0.44	0.00	-0.37	-0.45	***0.94***	0.06	-0.13
SP500m	-0.53	-0.72	0.44	-0.52	-0.67	0.42	***0.90***	-0.20
AA750m	***0.98***	-0.16	-0.37	0.00	-0.21	-0.43	-0.41	0.07
MR750m	-0.20	***0.98***	-0.45	-0.21	0.00	-0.41	-0.54	0.18
SC750m	-0.42	-0.37	***0.94***	-0.43	-0.41	0.00	0.02	-0.18
SP750m	-0.41	-0.60	0.06	-0.41	-0.54	0.02	0.00	-0.12
dis.set	0.04	***0.85***	-0.52	0.09	***0.85***	-0.57	-0.53	0.06
uncc	-0.12	0.31	-0.04	-0.13	0.29	0.05	-0.23	0.25
mcc	0.07	0.48	-0.43	0.03	0.52	-0.43	-0.24	***0.89***
upcc	-0.45	0.43	0.06	-0.46	0.44	0.13	-0.14	0.03
mvh	-0.57	0.16	0.34	-0.61	0.20	0.41	0.04	0.11
aoc	0.19	0.09	-0.13	0.07	0.18	-0.18	-0.12	0.00
bg	0.72	-0.22	-0.20	0.75	-0.27	-0.23	-0.26	-0.16
edge.dist	-0.16	0.40	-0.34	-0.10	0.42	-0.41	-0.01	-0.10
distance.control	-0.09	-0.79	0.50	-0.12	-0.79	0.56	0.51	-0.15
aoe	-0.32	0.49	-0.28	-0.34	0.54	-0.31	0.02	***0.85***
td	-0.26	0.74	-0.26	-0.24	0.76	-0.27	-0.36	0.08

Table A1.4 – Full variable name for variable name codes.

Variable name code	Variable
AA250m	Proportion of AA within 250m
MR250m	Proportion of MR within 250m
SC250m	Proportion of SC within 250m
SP250m	Proportion of SP within 250m
AA500m	Proportion of AA within 500m
MR500m	Proportion of MR within 500m
SC500m	Proportion of SC within 500m
SP500m	Proportion of SP within 500m
AA750m	Proportion of AA within 750m
MR750m	Proportion of MR within 750m
SC750m	Proportion of SC within 750m
SP750m	Proportion of SP within 750m
MR core	Montane rainforest core area
AA core	Annual agriculture core area
SC core	Shade coffee core area
SP core	Shade polyculture core area
MR-AA	MR AA edge
MR-SC	MR-SC edge
SP-SC	SP-SC edge
dis.set	Distance to settlements
uncc	Understorey cover
mcc	Mid canopy cover
upcc	Upper canopy cover
mvh	Max. vegetation height
aoc	Abundance of climbers
bg	Bare ground (%)
edge.dist	Distance to edge
distance.control	Distance to edge
aoe	Abundance of epiphytes
td	Tree density

Appendix 2

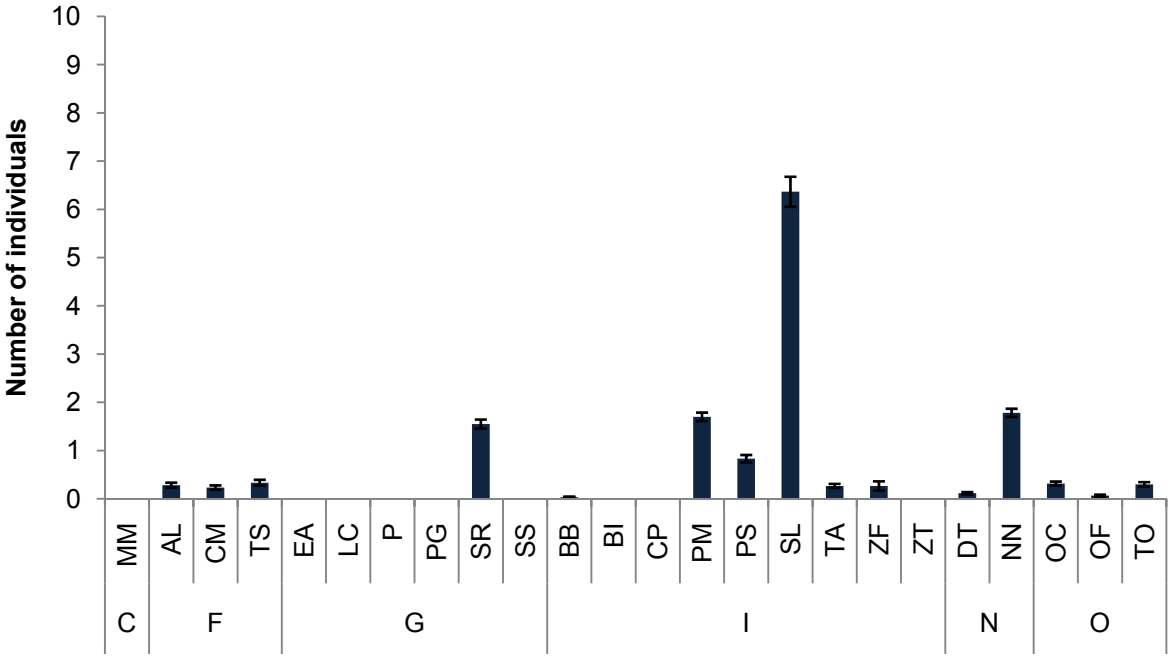

Figure A2.1 - Average number of recorded individuals within each montane rainforest point count grouped by feeding guild. Species names are coded by their initials, see table 1, p. for species names. Error bars indicate standard errors.

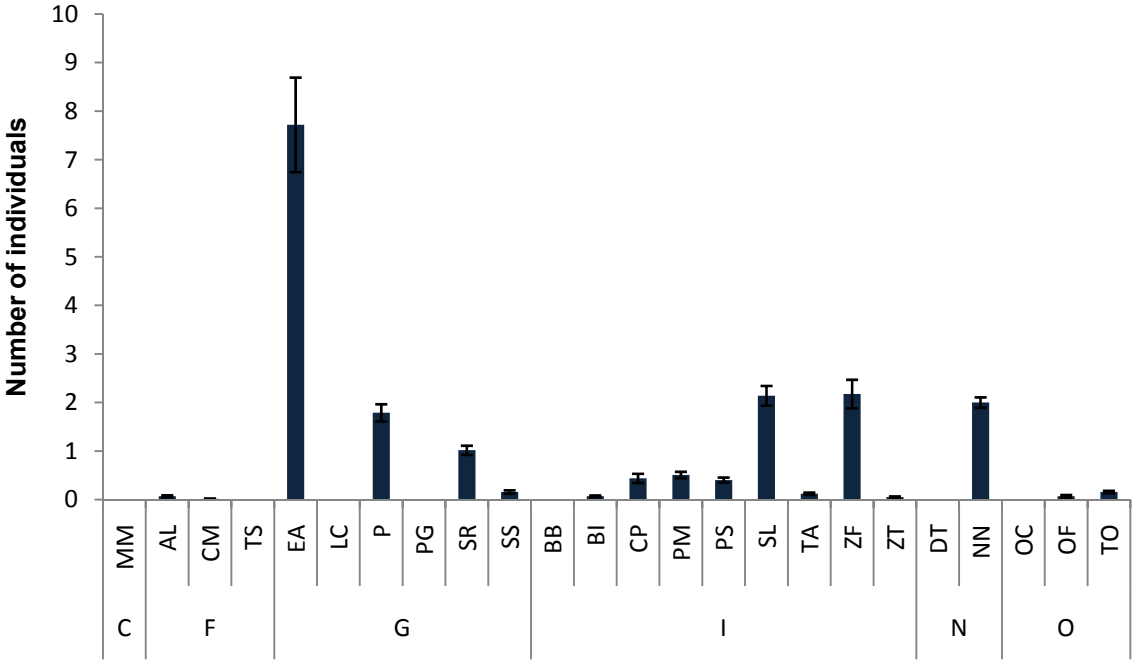

Figure A2.2 - Average number of recorded individuals within each annual agriculture point count grouped by feeding guild. Species names are coded by their initials, see table 1, p. for species names. Error bars indicate standard errors.

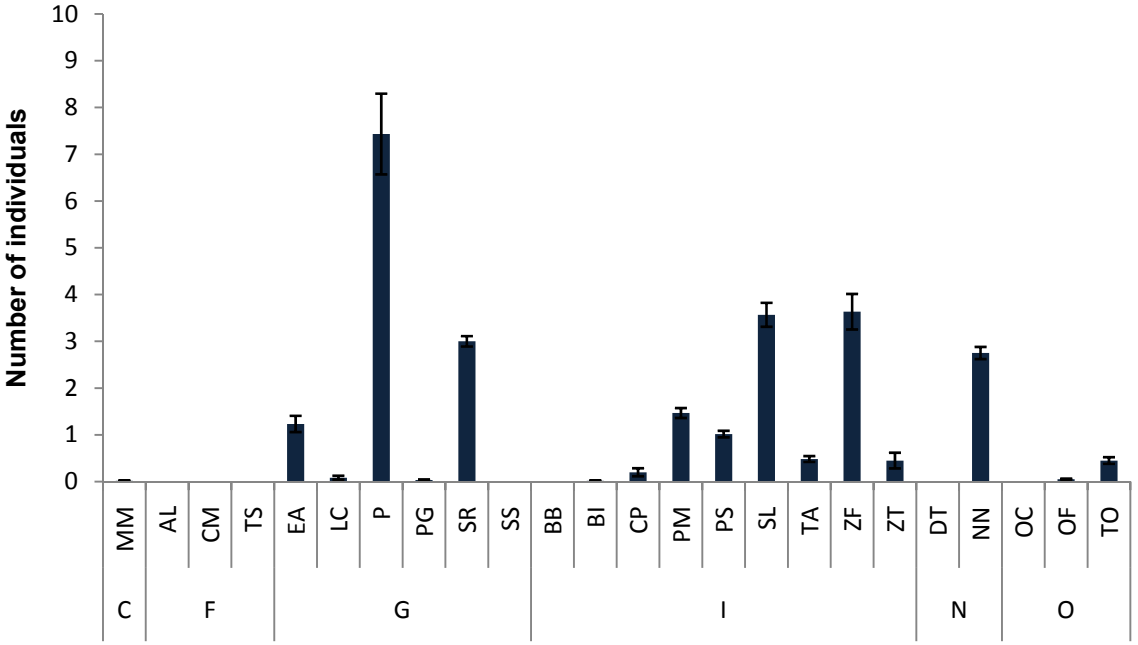

Figure A2.3 - Average number of recorded individuals within each shade coffee plantation point count grouped by feeding guild. Species names are coded by their initials, see table 1, p. for species names. Error bars indicate standard errors.

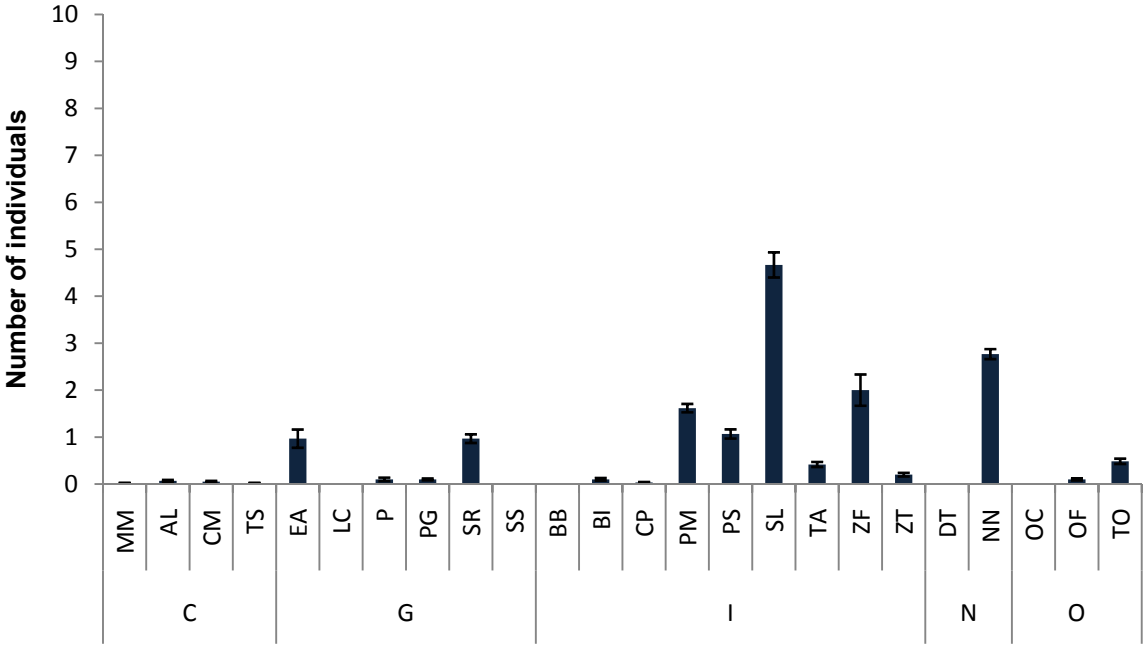

Figure A2.4 - Average number of recorded individuals within each shade policulture plantation point count grouped by feeding guild. Species names are coded by their initials, see table 1, p. for species names. Error bars indicate standard errors

Table A2.1 – Species code names for figs. 1-4, appendix 2, above.

Species	Species code
Aplopelia larvata simplex	AL
Bostrychia bocagei	BB
Bubulcus ibis	BI
Columba malherbii	CM
Cypsiurus parvus	CP
Dreptes thomensis	DT
Estrilda astrild	EA
Lonchura cucullata	LC
Milvus migrans	MM
Nectarinia newtonii	NN
Oriolus crassirostris	OC
Onychognathus fulgidus fulgidus	OF
Ploceus cucullatus nicriceps	P
Ploceus velatus peixotoi	P
Ploceus grandis	PG
Prinia molleri	PM
Speirops lugubris	SL
Serinus rufobrunneus	SR
Streptopelia senegalensis	SS
Terpsiphone atrochalybeia	TA
Turdus olivaceofuscus olivaceofuscus	TO
Treron sanctithomae	TS
Ploceus sanctithomae	PS
Zosterops ficedulinus feae	ZF
Zoonavena thomensis	ZT