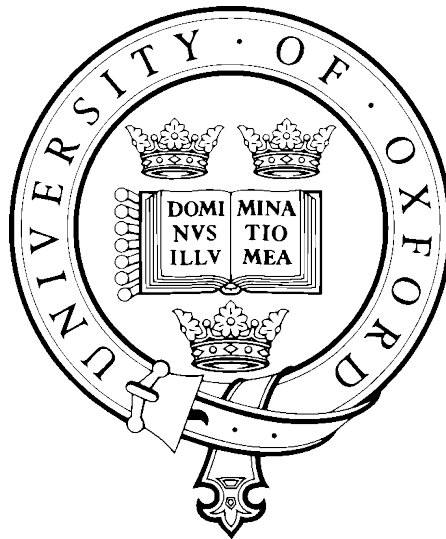


The impacts of human land-use change on avian diversity and associated ecosystem functions



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Abstract

Understanding the impacts of land-use change on biodiversity and the ecosystem services that it provides is of great importance given unprecedented growth of the human population. Past studies attempting to explore these impacts have described the overall structure of communities (i.e. species richness and trait diversity) across gradients of local scale degradation and fragmentation, and have sought to identify whether the loss of species following land-use change is non-random. Yet, despite a wealth of research we still lack a generalised understanding of how land-use change impacts on traits responsible for determining species sensitivity and their role within ecosystems, particularly for vertebrates. Moreover, despite the importance of niche-based processes in the assembly of communities, we have not yet elucidated whether these are important in mediating the collapse of communities in human-dominated landscapes. To fill these existing research gaps, I collated comprehensive avian species inventories from fragmented and degraded forests and compared their structure with communities existing in continuous forests.

In *Chapter 2*, I tested whether sensitivity of species to forest fragmentation varies between the temperate zone and the tropics and whether there are key differences in the size of fragments required to maintain ecosystem processes in these regions. I found that sensitivity to fragmentation varies according to functional group and body mass, with the prevalence of insectivores and large frugivores declining in relation to fragment size, particularly in tropical fragments smaller than 100 ha. In *Chapter 3*, I tested whether functional diversity and the mean position of trait diversity of insectivores and frugivores, changed across a gradient of intensifying land-use change. I found a decline in the functional diversity of forest species and a shift in the mean community traits for both forest and non-forest species. In *Chapter 4*, I tested whether the structure of tropical bird communities are influenced by species interactions in a fragmented landscape. I found increasing over-dispersion in functional and phylogenetic trait relatedness among species with decreasing fragment size, suggesting that competitive interactions are important in the disassembly of avian communities. In *Chapter 5*, I modelled the impact of forest cover change on ecosystem function across the Brazilian Amazon, focusing on seed dispersal by birds. Furthermore, I tested whether ecosystem function declined linearly with decreased forest cover after accounting for differences in the underlying pools of species. I found the lowest levels of functional diversity along the southern arc of deforestation and that the dispersal of large seeds showed some resilience to declining forest cover.

Taken together, my results suggest that the loss of species from communities in degraded and fragmented landscapes is strongly non-random. Insectivores and large frugivores are most sensitive to land-use change, with species located in the densest parts of trait space being most threatened by a decline in forest patch size, suggesting that species interactions regulate the collapse of avian diversity in human-modified forests. I conclude that land-use change has important implications for the provisioning of ecosystem services, including seed dispersal and the control of insect herbivores. The impact of future land-use change is likely to be mediated by the composition of the original pool of species and the amount of redundancy in the ecosystem services that they provide. I discuss the relevance of my findings to land-use management strategies and policy interventions, and in particular conclude that these should, where possible, maintain pristine forest patches above 1000 ha, improve connectivity among habitat patches, and ensure greater protection for logged and burnt forests. Future studies should focus on clarifying the link between shifts in vertebrate community structure and the functioning of forest ecosystems.

Thesis preface

Declaration and chapter authorship

The work presented in this thesis is my own with the following acknowledgements.

My supervisors, Dr Nathalie Seddon and Dr Joseph Tobias (University of Oxford) aided in the development of project ideas and also provided useful editorial comments on all chapters. Cagan Sekercioglu contributed habitat classifications for species present in the forest fragments in Chapter 2. Alexander Lees (ACL) provided species occurrence data for Chapter 4 and (together with Nargila Moura (NGM), Toby Gardner, Christian Andretti (CBA), Bradley Davis (BGWD), and Jos Barlow) provided species occurrence data used in Chapters 3 and 5. Hannah MacGregor and Bianca Darski aided with the collection of biometric data (used in Chapters 3–5) from bird specimens in the Natural History Museum, Tring and Museu Paraense Emílio Goeldi, Brazil, respectively.

Study locations

While Chapter 2 is global in focus, Chapters 3–5 utilise datasets collected across two localities in Brazil. Chapters 3 and 5 use bird community lists collected as part of the Rede Amazônia Sustentável (RAS) project. One area (1.9 million ha) was located in the municipality of Paragominas, another (~1 million ha) in the municipalities of Santarém and Belterra (hereafter, Santarém), all in the state of Pará, Brazil. Figure i shows the location of Santarém and Paragominas in relation to the rest of South America, and the location of the focal water basins within those regions.

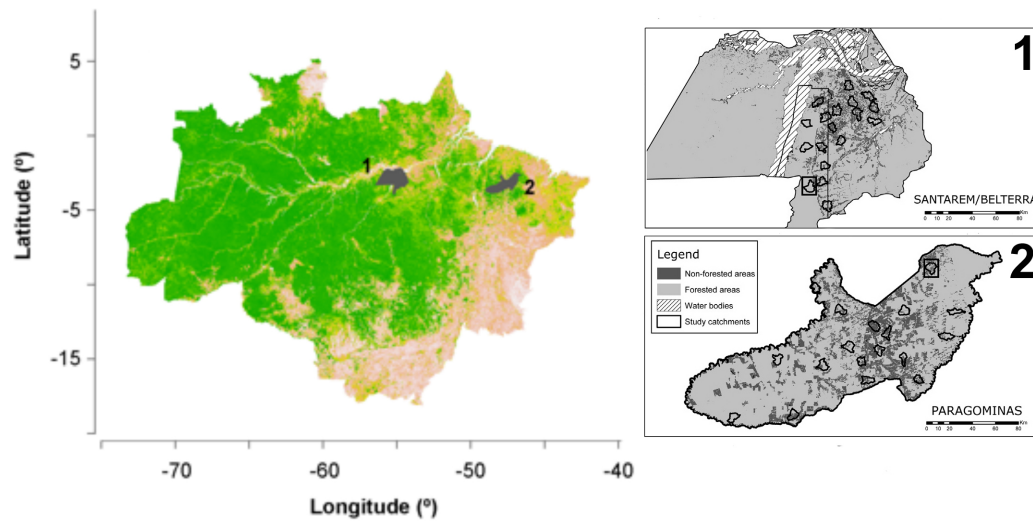


FIGURE i The location of (1) Santarém and (2) Paragominas in relation to the rest of the Brazilian Amazon, and the distribution of the water basins sampled as part of the Rede Amazônia Sustentável (RAS) project within these two study areas (adapted from Moura et al. 2014).

Meanwhile, Chapter 4 uses bird community lists from isolated forest patches located around Alta Floresta, Mato Grosso state, Brazil. Figure ii shows the location of Alta Floresta in relation to the rest of South America, and the spatial distribution of the focal forests.

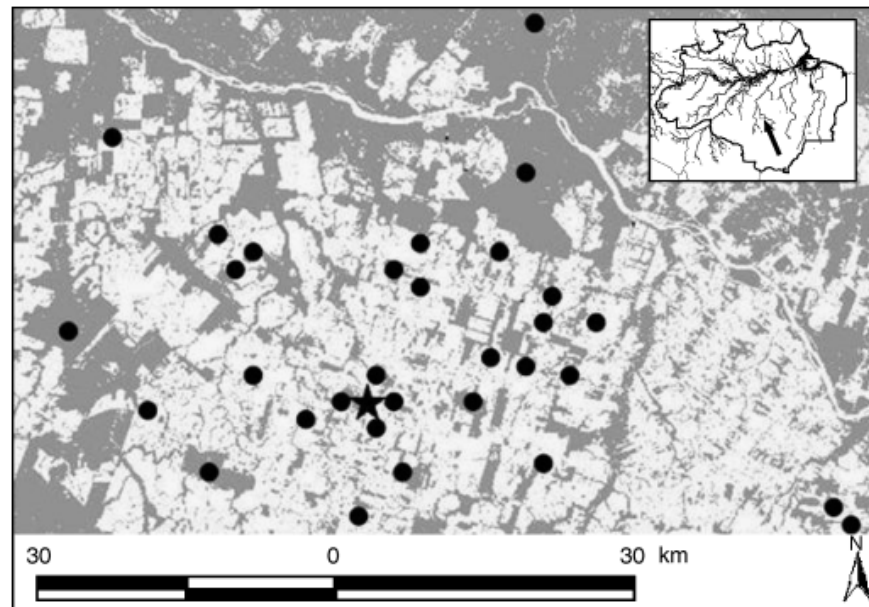


FIGURE ii Location of Alta Floresta in relation to the Brazilian Amazon, and location of the forest patches (circles) in relation to the town of Alta Floresta (star). Grey areas indicate forested areas (From Lees and Peres 2008a).

Supervisor declaration



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(Thesis submitted for the degree of Doctor of Philosophy by Tom P. Bregman)

I declare that the chapters presented in this thesis are substantially the work of Tom Bregman. Although I am a co-author on the publications arising from three chapters, my role in these chapters has been to aid in the development of project ideas, and to provide editorial comments. Data collection, management, analysis, figure preparation and the drafting of the chapters were entirely Tom's work. I therefore consent to the inclusion of this material in Tom's doctoral thesis.

Best wishes,

Dr Joseph Tobias

Departmental Lecturer

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1 General introduction

Understanding patterns of biodiversity remains a key challenge for ecologists and biogeographers (Gaston 2000), particularly in light of threats associated with the growing human population (McKee et al. 2003). Theory states that communities are constructed from pools of species assembled by evolutionary processes (Ricklefs 1987), mediated by local ecological factors (Lessard et al. 2012). The importance of these eco-evolutionary processes is thought to vary with scale (Angermeier and Winston 1998). At coarse spatial resolutions, evolutionary processes are thought to be dominant, while at fine spatial resolutions, species sensitivity to the local environment (Grinnell 1917), interspecific interactions (HilleRisLambers et al. 2012) and dispersal (Hanski 1999b) are thought to be of greater importance in determining community structure. We are beginning to link local ecological processes with the evolutionary influences of species pools to understand community structure (Lessard et al. 2012), but do not fully comprehend how these processes are impacted by human activities, despite these now dominating global land-cover (Vitousek et al. 1997).

It is known that land-use change has resulted in species extinction (Wilson et al. 1988), such that the global rate of species loss is now many times greater than the background level (Pimm et al. 2014). By altering the prevailing environmental conditions, land-use change, directly and indirectly, influences interspecific relationships (Tylianakis et al. 2008) and the propensity of species to cross landscapes (Desrochers and Hannon 1997, Lees and Peres 2009), changing the persistence and spatial distribution of individual species. Understanding how the impacts of land-use change on individual

species scale to determine the overall structure of communities will help us to predict future management trade-offs and develop pro-active conservation strategies.

Impacts of human land-use change: Species scale

If the conditions of altered habitats fall outside the range tolerable by the species, such as if key nutritional resources (e.g. fruiting trees) or nesting sites are no longer available, then species will be impacted directly. For example, some large understory insectivorous birds are obligate followers of ant-swarms (*Eciton* spp.), which only persist in large forest fragments (Willis 1979). The absence of the ant-swarms in small fragments means that understory insectivores often become extinct in fragmented landscapes (Lees and Peres 2008a). Furthermore, land-use change may also impact on other life-history characteristics important in maintaining viability of populations. For example, in the bog fritillary butterfly, *Proclossiana eunomia*, the tendency of individuals to disperse decreased in fragmented landscapes (Schtickzelle et al. 2006). The implications of decreased dispersal are clear: connectedness amongst populations will decline, reducing genetic flow and the strength of the ‘rescue-effect’ if populations do become locally extinct (Brown and Kodric-Brown 1977), meaning that metacommunity processes may be disrupted (Hanski 1999a).

Land-use change also impacts species indirectly through the alteration of biotic interactions (Fahrig 2003). For example, apex carnivores are particularly sensitive to land-use change (Woodroffe and Ginsberg 1998) due to their large size, low population densities and large home range requirements (Cardillo et al. 2005). The loss of these predators and their role as top-down ecosystem engineers following fragmentation causes an increase in the abundance of mesopredators (Prugh et al. 2009). For example, the removal of Coyotes from a native scrub landscape in southern California led to increased abundance of native and exotic mesopredators, in turn causing bird species

richness to decline (Crooks and Soulé 1999). Changes in the abundance of nest predators also have an impact on the breeding success of a variety of bird and mammal species (Vetter et al. 2013). In some cases, species responsible for nest predation in natural habitats decrease in abundance but are replaced by habitat generalists that increase predation pressures in edge habitats (Andren 1992). However, these biotic interactions may sometimes occur within trophic levels. For example, noisy miners (*Manorina melanocephala*) are aggressive native honeyeaters (Dow 1977) that specialise in occupying edge habitats (Clarke and Oldland 2007). By actively defending their territories they are known to exclude small passerines following land-use change, leading to reduced overall avian diversity across Australia's agricultural zones (Grey et al. 1998).

Impacts of human land-use change: Community scale

By impacting species on an individual-level, land-use change is known to alter the overall structure of communities. Specifically, the literature is dominated by studies describing the negative impacts of human-land use change on taxonomic and functional diversity (e.g. Lees and Peres 2006, Flynn et al. 2009, Moura et al. 2013). For example, across multiple taxa, Flynn et al. (2009) found that communities in agricultural habitats had lower species richness and functional diversity than communities in natural habitats. Furthermore, declines in species richness with decreasing habitat size are well documented, in line with species-area relationships (e.g. Lees and Peres 2006, Banks-Leite et al. 2012). However, habitat patches should not be treated as true islands (Janzen 1983); species are capable of dispersing between isolates, enabling them to maintain metapopulations at the landscape scale (Hanski 1998). These movements among patches mean that even small, degraded habitat isolates may contain species commonly associated with natural habitats, as long as larger, higher quality patches are located nearby. This “spillover effect” (Rand et al. 2006) has been found in numerous studies

(e.g. Lucey and Hill 2012) and highlights the importance of taking landscape structure into consideration when assessing the impact of human-land use change on the structure of communities (Moilanen and Hanski 1998).

Conversely, others have suggested that communities in landscapes dominated by human activities are similar to those in undisturbed habitats. For example, in their seminal paper, Daily et al. (2001) found that forest fragments remaining post-disturbance retained a substantial proportion of the avian community in a large forest fragment (measuring 227 ha). Furthermore, mammal communities in small forest fragments were similar to communities in the 227 ha Las Cruces forest as long as the intervening matrix was covered in coffee plantations (Daily et al. 2003). However, many of these studies are misleading due to their inappropriate choice of comparative baseline (Gillson et al. 2011). By comparing communities in particularly small habitat patches (0.3–25 ha) to those in slightly larger patches (in this case 227 ha), these studies are likely to be underestimating the loss of species following land-use change relative to a true, long-term natural baseline. Furthermore, by only assessing species richness, many approaches have overlooked possible interspecific impacts of land-use change, particularly on specialist species (Matthews et al. 2014).

Non-random extinction of species

There is now widespread evidence for the non-random extinction of species in communities from degraded and fragmented habitats, implying that species differ in their sensitivity to land-use change, with local ‘winners’ and ‘losers’ (Devictor et al. 2008). Trait-based methods (i.e. those that consider the phenotypic attributes of species) have been used extensively to provide insight into how species differ in their susceptibility to habitat fragmentation and degradation. For example, large body size (Peres 2001), limited dispersal ability (Lees and Peres 2009), and high trophic level (Didham et al.

1998b) are attributes associated with sensitivity to land-use change. This non-random loss of species means that communities existing in landscapes dominated by human activities are often found to have lower levels of trait diversity than we would expect if the loss of species were random (e.g. Flynn et al. 2009). Furthermore, the non-random response of communities is likely to lead to a disproportionate impact of land-use change on the functioning of ecosystems (Corbet 2000, Larsen et al. 2005, Karp et al. 2013b), particularly if the traits responsible for mediating species sensitivity are also important in determining the ecosystem service provided by the species.

Functional redundancy in disturbed ecosystems

However, even if the loss of species is non-random, highly redundant communities of ecosystem service providers (where multiple species provide the same ecosystem services) might maintain ecosystem functioning following disturbance (Memmott et al. 2004). This could occur if multiple species are involved in the top-down control of the same resources (e.g. the control of insects by avian insectivores) or if species adapt foraging routines (Kondoh 2003) to consume resources previously utilised by a species lost during disturbance. Furthermore, the invasion of exotic species into communities has largely been associated with negative impacts on native communities (e.g. Clavero et al. 2009), but they may also play an important role in the maintenance of some ecosystem processes or may promote the reestablishment of native species (Rodriguez 2006). Yet, in the majority of cases, communities show low levels of functional redundancy (e.g. Micheli and Halpern 2005, Petchey et al. 2007) and the impacts of land use change are likely to be exacerbated, rather than mitigated, by the invasion of non-native species (MacDougall and Turkington 2005). Therefore, we now understand that ecosystem resilience is highly dependent on the biotic context (i.e. the underlying species pool) (Mayfield et al. 2010), the extent to which similar species respond to biotic and abiotic

stressors, and the redundancy of traits among native species (Laliberte et al. 2010) and those invading post-disturbance.

Unexplored avenues: thesis themes

Despite numerous studies identifying non-random assembly of communities in landscapes dominated by human activities, there are at least two major avenues for further work to improve our understanding of the impacts of land-use change. First, we need to distil the nuanced impacts of land-use change (in particular, how trait diversity changes following land-use change); and second, we need to gain a better understanding of the underlying processes responsible for structuring communities in landscapes dominated by human activities.

Most studies have focused on explaining the impacts of land-use change on overall community metrics (i.e. community species richness and functional diversity), and have generally failed to explore the responses of individual traits to land-use change (but see Villeger et al. 2010). Grouping traits to generate overall measures of functional diversity is likely to provide misleading insights, given that processes structuring communities are thought to exert varying pressures on different trait axes (Trisos et al. 2014), and that each trait is likely to represent a different aspect of a species' life history. Therefore, to understand whether land-use change will have repercussions for ecosystem functioning, we must identify traits important in determining different species' responses to land-use change as well as the likely role of these traits in demarcating the services provided by species.

Furthermore, most studies have been focused at the local scale (e.g. Sisk et al. 1997, Norton et al. 2000, Lens et al. 2002, Lindenmayer et al. 2002) while studies at regional scales have either focused on temperate or tropical climatic zones (e.g. Gray et al. 2007, Newbold et al. 2013). We therefore do not fully understand whether patterns of

species responses described at the local scale can be generalised globally or whether they merely reflect local conditions and idiosyncrasies. Also, by focusing on the local scale, we have not yet explored how diversity differs across broad spatial extents. If we do not know how species vary in relation to current environmental conditions, then it is impossible to informatively predict the implications of future environmental change and put in place effective conservation strategies.

In addition, we also lack knowledge of the fundamental processes determining how communities disassemble in response to land use change. Competition has been identified as a key process in formation of communities; notably, it is thought to reduce the likelihood of species invasions when closely related ecological competitors are present (Fargione et al. 2003, HilleRisLambers et al. 2012). However, we are still unsure of whether this niche-based process plays an important role in the disassembly of communities following land-use change. Understanding the processes that structure communities is important as it allows for the development of conservation action plans that tackle the drivers of change rather than simply mitigating the impacts of land-use change after it has already occurred.

Model system

Forests

Forests are globally distributed and provide a number of essential services to humans (Edwards et al. 2014) yet continue to be exploited unsustainably (Meyfroidt and Lambin 2011). Aside from their important role in carbon storage (Pan et al. 2011), they stabilise soil structure (Oades 1984), generate microclimates (Vanwallegghem and Meentemeyer 2009) and drive atmospheric circulations (Zhao et al. 2001). Furthermore, the Earth's forests are paramount in maintaining human health (Chivian and Bernstein 2004) and considered important in mitigating the impacts of anthropogenic climate change (Bonan

2008). Historically, forests have been exploited for the extraction of timber but the land on which forests once stood has become important for the production of oil palm, soy and beef (Ghazoul and Sheil 2010). The financial incentives of these industries are driving extensive and unprecedented forest loss, fragmentation and degradation, particularly in the tropics. These processes not only directly reduce the ability of forests to provide these services but also impact upon species important in the maintenance of forest functioning (Sethi and Howe 2009). For example, vertebrate communities in forest fragments are generally less diverse (Estrada et al. 1993, Lees and Peres 2008a) with most species being less abundant (Pardini et al. 2005) leading to a decrease in the dispersal of seeds (Cordeiro and Howe 2003). This is likely to be exacerbated as trees reliant on vertebrates for seed dispersal are likely to face exaggerated density-dependent mortality as seeds may not be dispersed far from the parent tree (Cordeiro and Howe 2001). Overall, the extent of forests worldwide is diminishing at an alarming rate and the loss of key ecosystem service providers may prevent their regeneration even if destructive processes cease. Therefore, it is essential that we better understand how communities of forest-dependent ecosystem service providers are impacted by the disturbance and loss of forests.

Birds

Birds are a hyperdiverse group of species that perform a number of roles important in the maintenance of ecosystem functioning (Sekercioglu 2006b). They are especially important in forests as they regulate insect herbivory (Van Bael et al. 2003), disperse seeds (Sethi and Howe 2009, Anderson et al. 2011) and are actively involved in the pollination of a number of plant species (Bawa 1990). They are often described as mobile links as they connect various aspects of forests and consequently mediate the impacts of human land use change on the functioning of such systems (Lundberg and Moberg 2003,

Sekercioglu 2006b). Birds are the ideal model system for meeting the broad aims of this thesis as they are a particularly well-studied class of organisms and consequently detailed species lists across a range of human modified landscapes are available. Furthermore, we already have comprehensive distribution data for all species, a global phylogeny enabling relatively robust phylogenetic analysis, extensive datasets of life history and behavioural characteristics, and functional traits are easily measured and their role in mediating species biotic and abiotic interactions are well understood (Vandewalle et al. 2010).

Thesis overview

The main body of this thesis consists of four data chapters formatted for submission to peer-reviewed journals. The first has already been published in *Biological Conservation* and the third is currently in review. This layout means that each chapter is a stand-alone piece of work, improving accessibility for future readers, although this invariably adds a degree of repetition. Given that many references appear across all four chapters, I include a single reference list at the end of the thesis. The focus of each chapter is as follows:

Differences between tropical and temperate birds in terms of life histories are widely acknowledged (Skutch 1949, Stratford and Robinson 2005), yet we have so far assessed species responses to fragmentation at a local or regional scale. It is therefore difficult to determine how much information local results provide about general patterns. In *Chapter 2* I test whether temperate and tropical bird guilds differ in their sensitivity to a decrease in forest fragment size.

Despite knowledge that land-use change alters the richness of tropical bird communities, our understanding of the impacts of decreased diversity on functional diversity and ecosystem services remains poor. Furthermore, our knowledge of how land-use change impacts species with certain phenotypic traits remains limited, particularly in vertebrates. In *Chapter 3* I test whether land-use change alters functional

diversity and whether this habitat disturbance leads to a shift in the location of species persisting in trait-space.

Competition among species with similar ecological niches is known to regulate the assembly of biological communities. However, it is not clear whether similar processes operating in reverse can predict the collapse of communities and associated shifts in ecosystem function in the face of environmental change. In *Chapter 4* I test for the presence of a signal suggesting the importance of competition in the disassembly of communities in a fragmented tropical landscape.

Despite the clear need to understand the spatial extent and resilience of ecosystem services to land use change, we still do not have a framework that enables the forecasting of entire communities and the services that they provide at a regional scale. In *Chapter 5* I explore the usefulness of functional traits in predicting community structure across the Brazilian Amazon, testing whether trait diversity decreases linearly over a gradient of forest cover.

2 Global patterns and predictors of bird species responses to forest fragmentation: implications for ecosystem function and conservation

2.1. Abstract

The fragmentation of forests is a dominant human impact worldwide with major implications for the conservation and management of ecosystems. Although many studies have assessed the effects of fragmentation on biodiversity at local scales, our understanding of the ecological implications for different functional groups of organisms remains limited, particularly at global scales. Here, we use linear mixed models to explore patterns of occurrence and ecological function of 2844 bird species at 293 localities spanning five continents. We show that sensitivity to fragmentation varies according to functional group and body mass, with the prevalence of insectivores and large frugivores declining in relation to fragment size, particularly under 100 ha. However, the most severe effects were restricted to the tropics, whereas fragmentation had no significant impact on the basic ecological structure of temperate bird communities. We conclude that land-use change in tropical systems is likely to disrupt biotic processes, including seed dispersal and the control of insect herbivores. Our findings highlight the importance of latitude in mediating the impacts of habitat loss, and offer general guidelines for the minimum size of fragments required to prevent the collapse of key ecosystem processes in sensitive regions.

2.2. Introduction

Widespread declines in terrestrial biodiversity have largely been driven by the loss and fragmentation of natural habitats, particularly in forested regions (Turner 1996, Fahrig 2003). According to a range of forecasts, these processes are likely to accelerate as the growing demand for food and energy places increasing pressure on land over the next century (Sala et al. 2000, Godfray et al. 2010, McLaughlin 2011). Faced with the challenge of preventing biodiversity loss and maintaining ecosystem function in increasingly fragmented landscapes, it is therefore vital to understand how the effects of fragmentation vary across groups of organisms with different ecological roles (Lewis et al. 2013).

Most studies of fragmentation have focused on the extent to which the loss of species richness in forest isolates can be explained by factors such as species-area relationships (Lees and Peres 2006, Banks-Leite et al. 2012), edge effects (Murcia 1995), isolation (Beier et al. 2002), habitat complexity (Norton et al. 2000), and characteristics of the habitat matrix (Gascon et al. 1999, Martensen et al. 2012). However, extinction of species from forest fragments is potentially non-random in relation to functional traits, and thus patterns of species richness tell us little about the structure and function of assemblages surviving in human-modified landscapes (Robinson et al. 1992). In particular, several studies have highlighted the importance of dietary niche (Sekercioglu et al. 2002, Yong et al. 2011) and trophic level (van Nouhyus 2005, Murphy and Romanuk 2012) as factors influencing sensitivity to fragmentation, suggesting that this form of land-use change may have predictable effects on trophic processes integral to ecosystem function.

Although the importance of understanding anthropogenic effects on the function of forest ecosystems has been highlighted by numerous studies (van Nouhyus 2005, Sekercioglu 2012, Newbold et al. 2013), few generalities have emerged about the impacts

of fragmentation on different functional groups of organisms (Watson et al. 2005). For example, a decline in insectivores is thought to increase insect herbivory in degraded or fragmented forests (Skoczylas et al. 2007, Van Bael et al. 2008), whereas the opposite pattern of increased insectivory and reduced herbivory has been reported in others (Gonzalez-Gomez et al. 2006, De la Vega et al. 2012). Similarly, seed dispersal appears to decline in some fragments (Watson et al. 2003) yet increase in others (Donoso et al. 2004). One possible explanation for these inconsistent results is that spatial variation in life history traits such as dispersal (Salisbury et al. 2012) or the strength of biotic interactions (Schemske et al. 2009) drive latitudinal gradients in the impacts of forest fragmentation (Stratford and Robinson 2005). However, most studies have focused at the local level (e.g. Sisk et al. 1997, Norton et al. 2000, Lens et al. 2002, Lindenmayer et al. 2002), whereas analyses at regional scales have been restricted to the temperate zone or the tropics (e.g. Gray et al. 2007, Newbold et al. 2013). Thus, it is difficult to determine how much information local results provide about general patterns, and we still lack a global synthesis of functional responses to fragmentation, particularly in vertebrates (Vandewalle et al. 2010).

Here, we address this issue using a linear mixed modelling approach to test the impact of forest fragmentation on bird communities in 293 human-created forest fragments worldwide. For each of these fragments, we recorded its size and latitude. We then compiled functional trait data (see Appendix A: Supplementary methods and results) for all 2844 species reported as present in at least one study site. We focused on habitat preferences, feeding guild, dispersal and body mass as these traits are known to predict sensitivity to land-use change, and are also relevant to the functioning of ecosystems (Sekercioglu 2006b, Newbold et al. 2013).

Birds offer a useful system for understanding broad-scale fragmentation processes for three reasons. First, they are popular study species and readily surveyed,

and thus detailed species lists have been compiled for numerous forest fragments using standard techniques. Second, they are the best-known class of organisms and provide the most comprehensive dataset of interspecific variation in functional traits (Vandewalle et al. 2010). Third, they perform critical roles in forest systems, partly through top-down processes, such as predation on insect herbivores (Van Bael et al. 2003), and also through the ubiquitous plant–animal mutualisms underpinning pollination and seed dispersal (Sethi and Howe 2009, Anderson et al. 2011). Because of their key role in forest dynamics linked to plant dispersal and recruitment, birds are sometimes described as ‘mobile links’, connecting various parts of the landscape and mediating the resilience of forests to anthropogenic change (Lundberg and Moberg 2003, Sekercioglu 2006b).

Previous studies have shown that avian species richness declines with forest fragment size, approximately in accordance with classic species–area relationships (e.g. Lees and Peres 2006, Banks-Leite et al. 2012). Much less is known about how these relationships vary across latitude, and between functional groups. For example, dispersal limitation in tropical forest birds (Moore et al. 2008) is thought to be a major factor explaining their sensitivity to rainforest fragmentation (Lens et al. 2002, Barlow et al. 2006, Tobias et al. 2013a), whereas many temperate zone species with similar functional roles are migratory or highly dispersive (Salisbury et al. 2012) and thus potentially better able to withstand habitat fragmentation (Stratford and Robinson 2005).

Our goal in this study is to assess the effects of fragment size on different functional components of the bird community, particularly in relation to insect predation and seed dispersal. The methods are designed to shed light on broad-scale patterns relevant to the structure and functioning of human-modified ecosystems, rather than species conservation. Thus, we focus not only on species richness but the relative prevalence of different foraging guilds in the community. In addition, we use body mass to assess within-guild patterns of resilience as the impacts of anthropogenic change may

vary across different size classes of organisms, even within functional groups (Crooks 2002). Our approach allows us to test (1) whether sensitivity to fragmentation varies between the tropics versus the temperate zone, and (2) whether there are differences in the size of fragment required to maintain key ecosystem processes in both regions.

2.3. Materials and methods

Literature review

Using Web of Science and Google Scholar, we compiled all relevant research articles published between 1962 and June 2012 that contained the following key words: (bird* OR avian) AND (habitat fragmentation* OR forest fragmentation* OR habitat degradation* OR land-use change* OR habitat loss) AND (community structure* OR species list* OR community assembly* OR community list* OR community). We also conducted an extensive search of grey literature available in the Alexander Library, Department of Zoology, University of Oxford, one of the world's premier collections of ornithological literature. Citations in all articles and reports were examined for relevant information.

Articles were checked for comprehensive species inventories from fragmented or continuous forests worldwide. We restricted our search by applying the following criteria. (1) Broad-leaved or coniferous forest: We excluded very dry woodlands as these tended to involve scrub or stunted thorn forest, supporting communities with a different balance of ecological functions, and therefore likely to obscure general patterns relevant to forests. (2) Mature closed-canopy forest: We excluded fragments dominated by secondary forest, or forest that had recently been logged or burnt, so that the effects of fragmentation were not confounded with other processes. (3) Natural fragments: We excluded data from any natural forest fragments such as forest patches in natural grasslands or montane 'sky-islands'. (4) Consistent isolation: given the importance of

habitat isolation to metacommunity dynamics under fragmentation, we removed all non-isolated fragments from analyses, e.g. fragments connected to other blocks of forest by corridors or recent secondary regrowth (Ferraz et al. 2007). (5) Total community: We excluded all studies focusing on a subset of the avian community, e.g. only passerines or forest-dependent species. This is because we were interested in quantifying impacts of fragmentation on forest-dependent species relative to the entire community, and these impacts cannot be explored with incomplete datasets.

For each community, we compiled data on total species composition, habitat patch size and latitude, excluding sites for which any of these three details were unavailable. The final dataset (Appendix A) contained a total of 293 avian communities from 56 study regions (Fig. 1). Sites were classified as either ‘tropical’ ($<23.6^\circ$ N/S; $n = 113$) or ‘temperate’ ($>23.6^\circ$ N/S; $n = 180$).

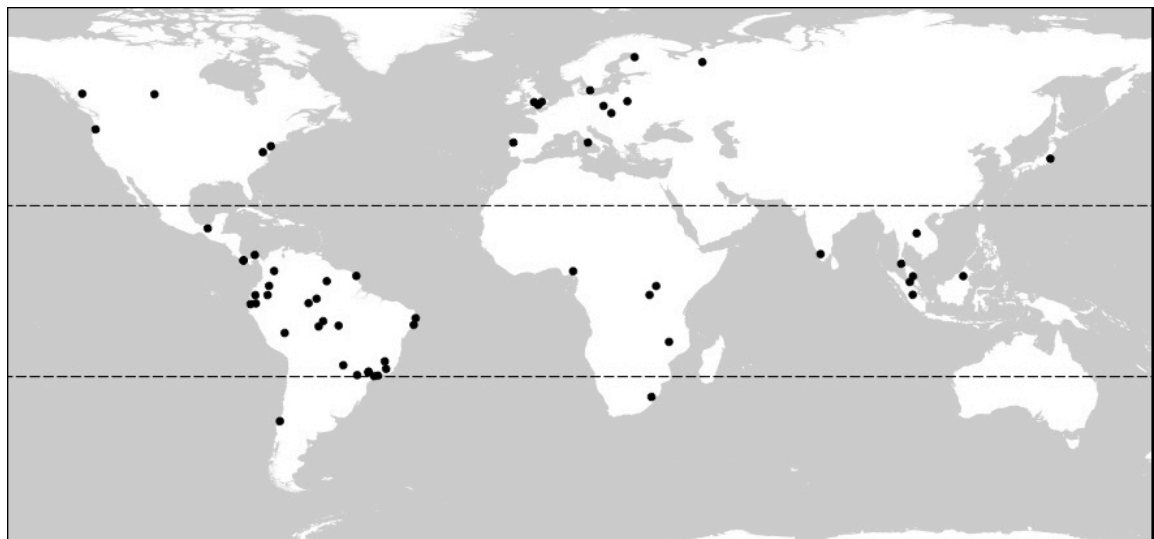


FIGURE 2.1. Distribution of 56 regions generating data for the 293 fragments included in this study. Dashed lines indicate the tropics of Capricorn and Cancer. Details of locations and data sources are provided in Appendix A.

Dealing with extinction filters and extinction debt

Examining responses to fragmentation at global scales is challenging because of the contrasting histories of land-use across latitudes, with the time elapsed since deforestation events being generally long in the temperate zone, and shorter in the tropics. These differences are potentially important as, in theory, fragmented bird communities only fully collapse after an extended time delay (Brooks et al. 1999, Ferraz et al. 2003), such that young fragments may still have to pay an ‘extinction debt’ (Tilman et al. 1994). Moreover, unlike recently cleared regions (e.g. much of the tropics), fragments in historically cleared regions (e.g. Europe, North America) may have already passed through an ‘extinction filter’ during which sensitive species were removed (Balmford 1996). To address these problems, we selected data from continuous forest habitat, defined as habitat patches >100 000 ha in extent, in both the tropical ($n = 20$) and temperate zones ($n = 9$). Although extensive primeval forest in the Nearctic region has lost at least two bird species through habitat loss and hunting (Fuller 2002), these extinctions affect different functional groups and are too rare to explain general patterns. Similarly, few recorded extinctions have occurred in extensive lowland tropical forests, where the component of extinct species is likely to be negligible in relation to overall species richness. Thus, we assumed that the avifauna of continuous forest in all regions was intact, providing a baseline against which fragment communities can be compared.

Functional traits

We classified species according to four key ecological or life-history traits: habitat preference, migratory behaviour, feeding guild and body mass (Appendix A: Supplementary methods and results). Species were assigned to one of two habitat categories according to whether they are primarily detected in (i) forest or (ii) non-forested habitats. We defined forest as any type of evergreen or deciduous woodland

lacking gaps between tree canopies. Bird species associated with these habitat types are henceforth termed ‘forest dependent’. To focus on functional groups typically associated with ecosystem dynamics in forest patches, rather than those restricted to the non-forest matrix, we used forest-dependent species as the focal group in analyses.

Species were assigned to one of two categories of migratory behaviour according to whether they were (i) long-distance migrants, or (ii) either sedentary or short-distance migrants. The latter category includes elevational migrants and species undertaking regional movements, seasonally or otherwise. Species were then allocated to one of eight categories of feeding guild—insectivores, frugivores, granivores, nectarivores, carnivores, herbivores, scavengers and omnivores—based on datasets of Sekercioglu et al. (2004a), with updates from Belmaker et al. (2012). Dietary categories represent the primary diet, with species categorized as omnivores if no primary diet was apparent. Finally, data on body mass were collated for each species, primarily from Dunning (2008). For a complete list of species with functional trait definitions, data and sources, see Appendix A.

We focused on breeding species as these form the predominant functional component of avian communities, either present year-round, or at least during the leaf-growing, flowering and fruiting season of plants. In contrast, non-breeding migrants often appear briefly or sporadically. In excluding non-breeding species from communities, long-distance migrants were thus retained in temperate communities (where they breed) but removed from tropical communities (where they occur as non-breeding visitors). To test whether our results held when focusing exclusively on resident species, we re-ran analyses with all migrants removed from both temperate and tropical zones (Appendix A: Sensitivity analyses).

Although the contribution of biodiversity to ecosystem functioning depends on a range of complex factors including network structure and trophic complementarity

among species (Poisot et al. 2013), the efficiency and stability of ecosystem function is expected to increase with species richness, partly as a natural consequence of limiting similarity among species (Naeem and Li 1997, Naeem et al. 2012). We thus used raw species richness as a basic measure of the contribution of each guild to ecosystem function. However, this measure may give a misleading impression of fragmentation effects across latitudes because overall species richness varies so dramatically with distance from the equator, in line with the latitudinal diversity gradient. Thus, we also calculated the proportional contribution of each guild to the total species richness within each community and used it as a response variable in my models.

Analytical approach

We used Generalized Linear Mixed Models (GLMM) to test for differences in sensitivity using the MCMCglmm package (Hadfield 2010) in R (R Development Core Team 2013). The advantage of a mixed modelling framework is that it allows us to consider the effects of multiple factors, including latitude, fragment size and dietary guild, as well as their interactions. In addition, we were able to explore non-linearity in effects by including fragment size as a quadratic term in models of both species richness and proportion; this tests for changes in the rates at which species were lost, or at which communities shifted in structure, across the fragmentation gradient. Finally, as some studies reported data for multiple communities clustered geographically, we accounted for potential spatial autocorrelation and pseudoreplication by including ‘study’ (i.e. publication) as a random effect in all models. Fragment size was $\log_{10}$ -transformed before analysis to ensure model residuals were normally distributed. We ran six sets of analyses.

Analysis 1. To test the effect of fragment size on raw species richness in key functional groups, we focused on three major guilds (frugivores, granivores and insectivores) of forest-dependent species. The response variable was the richness of

species in each of these guilds, with fragment size as the explanatory variable. We ran models for temperate and tropical communities separately as sample sizes for some guilds were too low in the temperate zone.

Analysis 2. To examine the effect of fragment size on the prevalence of forest-dependent species within communities, we ran a multi-predictor model with species grouped by habitat preference (forest and non-forest). The response variable was the proportion of forest-dependent species in the community, with fragment size and latitudinal zone included as covariates.

Analysis 3. To explore latitudinal variation in the functional composition of fragment communities, we subdivided each community into eight feeding guilds. We then calculated the proportion of each guild in each community, based on species richness, and then averaged across communities for four different datasets: (i) all species in all communities; (ii) forest-dependent species in all communities; (iii) all species in intact communities (iv) forest-dependent species in intact communities. To compare the proportions of key guilds between latitudinal zones in datasets (i–ii), we used arcsine transformation to normalize data then conducted t-tests. Sample sizes in datasets (iii–iv) were too small for this approach, so we used Mann-Whitney U-tests.

Analysis 4. To test the effect of fragment size on different functional groups, we subdivided forest-dependent species into categories of feeding guild and then calculated the proportion of each guild present in each community. We then re-ran the model described in analysis 2. Rather than including latitudinal zone as a covariate, models for temperate and tropical communities were run separately to improve model convergence and facilitate interpretation.

Analysis 5. To test for variation in the effects of fragmentation across scales, we classified sites into three broad size categories: 0–100 ha, 100–1000 ha and >1000 ha. We then compared the proportion of each feeding guild occurring in each category of

fragment size (see analysis 4). This approach is complementary to our analyses treating fragment size as a continuous variable, allowing us to compare fragments against extensive continuous forests where size has not been estimated.

Analysis 6. As proportional data calculated from patterns of species richness may conceal other more subtle impacts of fragmentation on functional traits, we modelled the effect of fragment size on body mass, to test the prediction that fragmentation has a disproportionate effect on large species. We used the same analytical framework as analysis 3 to test whether the average body mass of each guild of forest-dependent species differed among categories of fragment size.

Because of latitudinal gradients in the history of fragmentation, communities were not evenly sampled across spatial scales (Appendix A). Specifically, in the temperate zone all forests classified as >1000 ha were essentially continuous (>100 000 ha), whereas the same category in the tropical zone contained a mixture of very large fragments (1000–100 000 ha; $n = 10$) and continuous forests (>100 000 ha; $n = 10$). However, there was no significant differences in the proportion and average body mass of guilds in these two samples of tropical forests (Appendix A: Supplementary methods and results), suggesting that they are similar from a functional perspective. Thus, we pooled large fragments and continuous forests in analyses 4 and 5. Model results are presented using P and DIC (Deviance Information Criterion) statistics. Minimum adequate models can increase the rate at which Type I errors occur so we report the full model DIC and examine the relative importance of predictors by calculating the change in the DIC when each predictor is removed (for further details on these statistics see the extended methods section in the appendices) (Hadfield 2010). For all statistical tests a critical value of $P < 0.05$ was used.

2.4. Results

Broad-scale patterns

Species richness was far higher in tropical than temperate fragments (Fig. 2.2), in line with the latitudinal diversity gradient, and the effects of fragmentation also differed across latitudinal zones. In the tropics, species richness declined with decreasing fragment size in all three major guilds (Analysis 1; Fig. 2.2; Table A1). The decline with the decreasing fragment size in tropical fragments was steepest in insectivores, both in relation to frugivores and granivores (Fig. 2.2). Moreover, in both tropical and temperate fragments, we found that the rate of insectivore loss was non-linear across the gradient of fragment size, with rates increasing in small fragments (Fig. 2.2; Table A1). In contrast, the rate of decline in richness for frugivores and granivores was similar across the entire gradient of fragment size in the tropical zone. There were too few frugivores and granivores in the temperate zone to test whether this linearity holds outside the tropics.

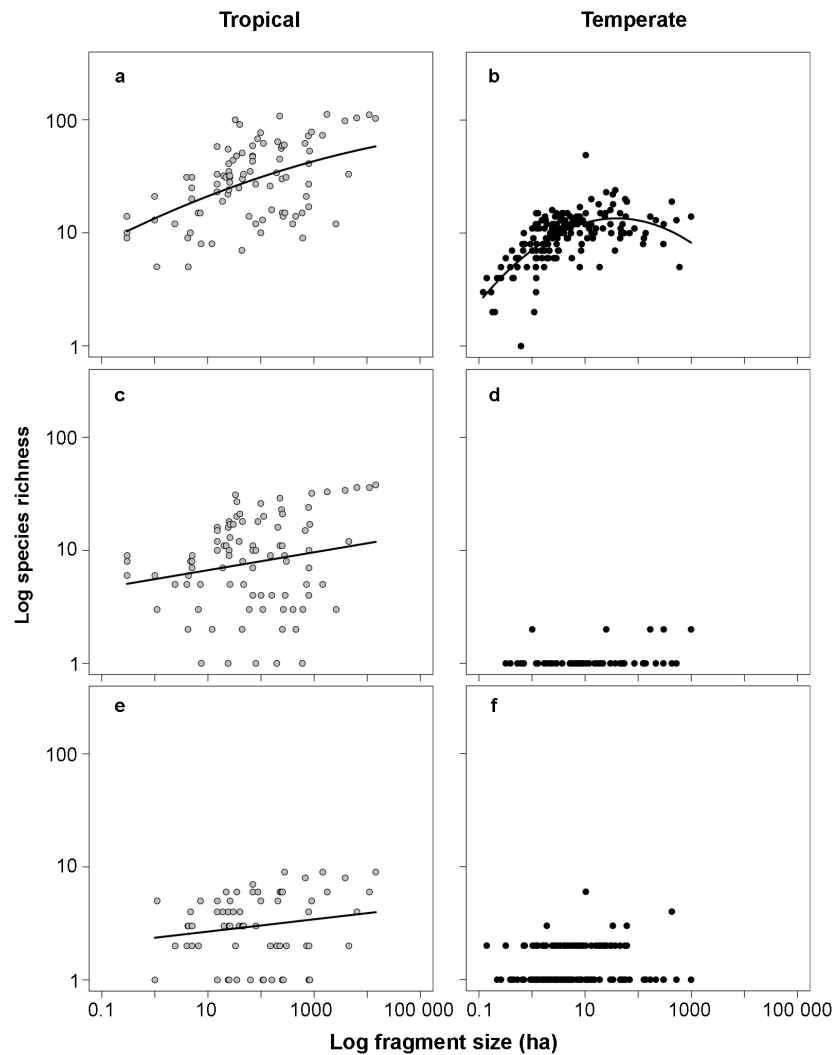


FIGURE 2.2. The effect of fragment size on species richness of forest-dependent insectivores (a,b), frugivores (c,d) and granivores (e,f) in avian communities occurring in tropical ($n = 93$) and temperate ($n = 171$) forest fragments. Species richness data are shown on a logarithmic scale; trend lines are plotted for significant relationships calculated using regression analysis in a mixed effects framework.

These three guilds — insectivores, frugivores and granivores — together make up approximately 90% of species richness within fragment communities in both tropical and temperate zones (Fig. 2.3b). Insectivores predominate throughout, but the relative contributions of the major guilds differ significantly across latitudes. Specifically, the frugivore component was significantly larger in the tropics whereas the granivore component was much larger in the temperate zone (Analysis 3). There were also dramatic latitudinal differences among minor guilds, with some being dominant in the

tropics (e.g. nectarivores) and others dominant in the temperate zone (e.g. herbivores). Although these general patterns take no account of fragment size, they were similar in intact forests (Fig. 2.3c). Thus, they are likely to reflect underlying gradients in guild structure caused by latitudinal differences in climate and resource availability (e.g. year-round availability of fruits and flowers in the tropics) (Belmaker et al. 2012, Kissling et al. 2012, Tobias et al. 2013a). Modelling habitat preferences revealed that the proportion of forest-dependent species occurring in fragment communities was strongly predicted by habitat patch size, with smaller fragments containing a relatively low proportion of forest-dependent species ($P = 0.002$; Fig. 2.3a; Table A2). However, we detected no effect of latitudinal zone or the interaction between latitudinal zone and fragment size (Table A2), suggesting that the effect of fragment size on the proportion of forest-dependent species is similar in the temperate zone and the tropics.

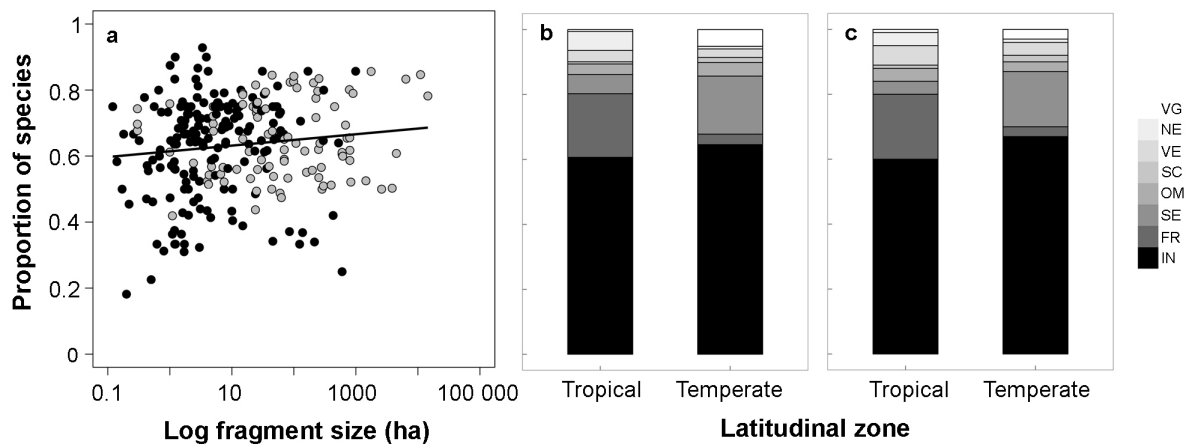


FIGURE 2.3. The effect of forest patch size and latitude on the structure and function of fragmented bird communities. (a) shows how the proportion of forest-dependent species occurring in each community varies with patch size in tropical ($n = 93$) and temperate ($n = 171$) fragments; the regression line is given for the pooled dataset ($P = 0.002$). Black circles represent temperate fragments; grey circles represent tropical fragments. (b) and (c) show the ecological structure of tropical and temperate communities; (b) shows the proportion of forest-dependent species belonging to each guild in forest fragments. (c) shows the proportion of all species (both forest dependent and independent) belonging to each of the guilds in continuous forests. Foraging guilds: insectivores (IN), frugivores (FR), granivores (GR), omnivores (OM), scavengers (SC), carnivores (CA), nectarivores (NE), herbivores (HE). We focused on insectivores, frugivores and granivores in our analyses because of their global dominance.

Structure and function of fragmented bird communities

Given that latitude drives variation in the guild structure of communities, particularly in relation to smaller guilds, it is difficult to compare the effects of fragmentation across both latitudinal zones simultaneously. To overcome this problem, we ran models separately for each latitudinal zone, and for each of the three major feeding guilds (Analysis 4). The results revealed distinct latitudinal patterns of sensitivity to fragmentation in forest-dependent birds mediated by feeding guild (Fig. 2.4; Table A3). In the tropics, decreasing fragment size was associated with a decrease in the insectivore component of avian communities, an increase in the granivore component, and no change in the frugivore component. As predicted by the increasing rates of insectivore loss in small fragments (Analysis 1), the decline in the proportion of insectivores was also non-linear, with an accelerating collapse of the insectivore component in small tropical fragments. In contrast, in the temperate zone, fragment size had no effect on the proportion of any of the three key functional groups, or changes in their proportions across the fragment size gradient.

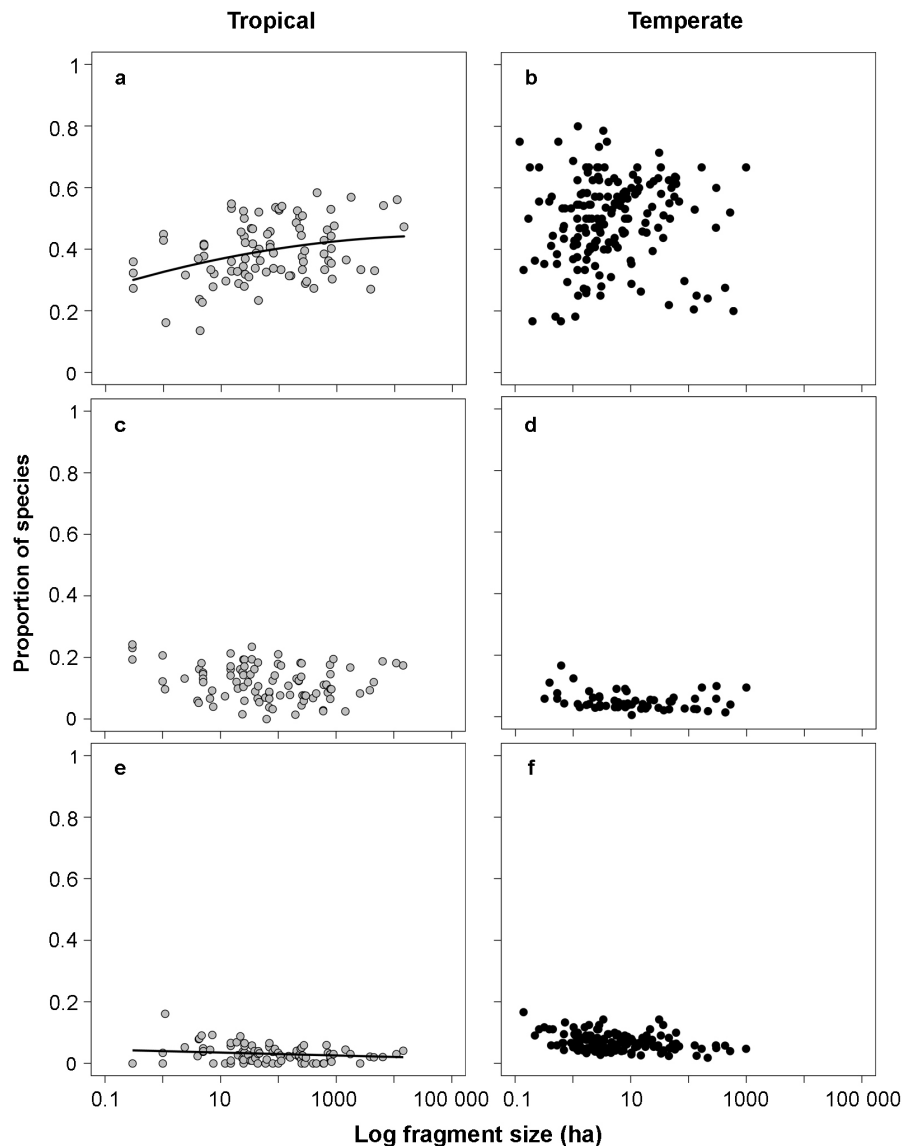


FIGURE 2.4. The effect of fragment size on the prevalence of forest-dependent insectivores (a,b), frugivores (c,d), and granivores (e,f) in avian communities occurring in tropical ($n = 93$) and temperate ($n = 171$) forest fragments. Data shown are the proportion of the total community of each fragment made up of each functional group. Trend lines are plotted for significant relationships (where $P < 0.05$) calculated using regression analysis in a mixed effects framework.

To provide further insight into the effect of climate zone, we compared guild-specific responses to fragmentation among small (<100 ha), mid-size (100–1000 ha) and large fragments (>1000 ha). We found no change in the proportion of any guild when comparing between small and mid-size fragments, or between mid-size and large

fragments, in either the tropics or the temperate zone. The only exception was that small fragments in the tropics contained a lower proportion of forest-dependent insectivores than did mid-size fragments (Table A4). Differences increased when comparing across widely contrasting fragment sizes in the tropics, with small fragments containing a lower proportion of insectivores and frugivores, and a higher proportion of granivores, than large fragments (Fig. 2.4; Table A5). In contrast, we detected no difference in the proportion of insectivores, frugivores and granivores in small and large fragments in the temperate zone.

When we ran similar tests of latitudinal differences in fragmentation effects mediated by body mass (Analysis 6), we found similar results. There was no change in the average mass of species when comparing between small and mid-size fragments, or between mid-size and large fragments, in either the tropics or the temperate zone (Table A6). Mass differences between the smallest (<100 ha) and largest habitat patch categories were more pronounced, and again highly variable among feeding guilds (Fig. 2.5; Table A7). The average mass of tropical frugivores and temperate insectivores was significantly lower in small forest patches (Fig. 2.6; Table A7). However, the average mass of granivore, temperate frugivore and tropical insectivore communities did not change significantly in the smallest forest fragments.

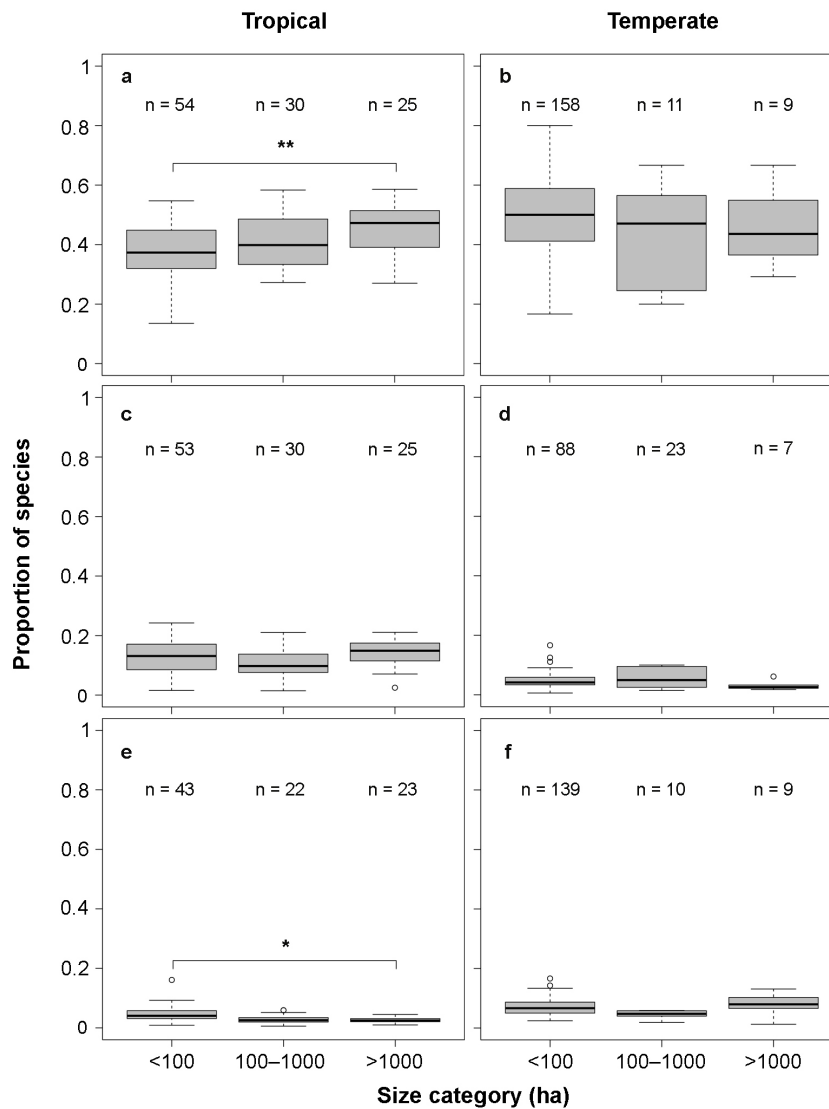


FIGURE 2.5. The effect of habitat patch size on the prevalence of different feeding guilds in avian communities inhabiting tropical ($n = 113$) and temperate ($n = 180$) sites, calculated for forest-dependent insectivores (a,b), frugivores (c,d) and granivores (e,f). Data are calculated as a proportion of total species richness in the community. Whiskers shown on Tukey boxplots include points within 1.5x interquartile range. Sample sizes vary as foraging guilds were absent from some communities. The largest category (>1000 ha) contained some ($n = 25$) large fragments in the tropical zone, but in the temperate zone it only contained continuous forest ($>100\,000$ ha; $n = 9$). Statistics are from ANOVA in a mixed effects framework (analysis 5); * $P < 0.05$, ** $P < 0.01$.

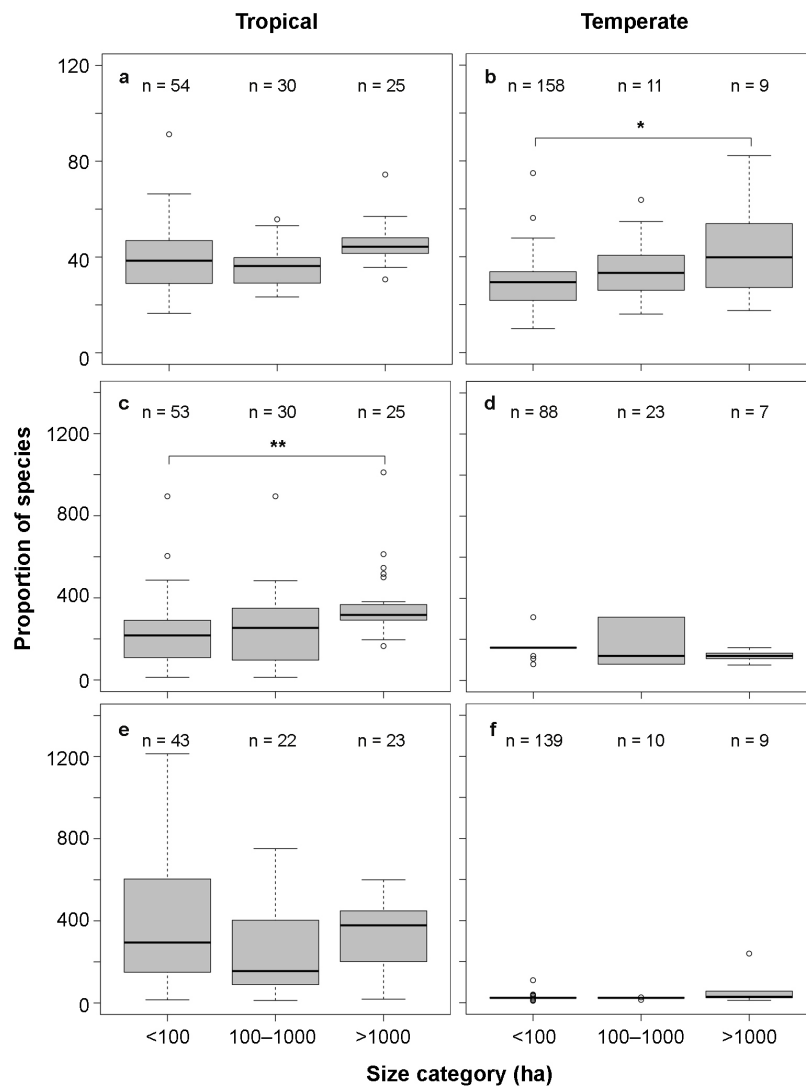


FIGURE 2.6. The effect of habitat patch size on the body mass of forest bird species in temperate ($n = 180$) and tropical ($n = 113$) communities, calculated for forest-dependent insectivores (a,b), frugivores (c,d) and granivores (e,f). Whiskers shown on Tukey boxplots include points within 1.5x interquartile range. Sample sizes vary as foraging guilds were absent from some communities. The largest category (>1000 ha) contained some ($n = 25$) large fragments in the tropical zone, but in the temperate zone it only contained continuous forest ($>100\,000$ ha; $n = 9$). Statistics are from ANOVA in a mixed effects framework (analysis 6); * $P < 0.05$, ** $P < 0.01$.

Sensitivity analysis

To test whether the difference in sensitivity between tropical and temperate guilds is driven by the removal of non-breeding migrants in tropical communities we re-ran analyses for temperate guilds after removing migratory species (Tables A10–A13).

Fragment size remained a significant predictor of the proportion of forest-dependent species and the trend still did not vary between communities in the tropical and temperate zones (Table A10). None of the guilds changed in their relative proportion as fragment size declined (Table A11 and A12). Furthermore, when we assessed how these effects are mediated by body size, we found that large insectivores were still sensitive to a decline in fragment size with the greatest change in small relative to large fragments (Table A13).

2.5. Discussion

We have shown that raw species richness and the proportion of forest species declines with fragment size, in accordance with numerous previous studies (e.g. Kattan et al. 1994, Beier et al. 2002, Fahrig 2003, Lees and Peres 2006). However, the effects of fragmentation varied both within and between feeding guilds, with implications for the functioning of fragmented ecosystems. Specifically, the proportion of insectivorous birds within tropical communities declined in relation to fragment size, whereas the proportion of frugivores and granivores did not. Similarly, the loss of insectivores (species richness and proportion) accelerated when fragments became very small, in line with classical theory on species-area relationships (Rosenzweig 1995), but rates of species loss in frugivores and granivores remained constant across the gradient of fragment size. In parallel, tropical fragmentation was associated with reduced body mass in frugivorous birds, but not in insectivores or granivores. The sensitivity of insectivores and large frugivores to forest fragmentation and disturbance has been reported from studies focused at the local scale (Sekercioglu et al. 2002, Barlow et al. 2006, Terborgh et al. 2008, Yong et al. 2011). Our results confirm that these impacts hold in forest fragments across the tropics, and yet are far less severe in the temperate zone. We conclude that the effect of fragmentation on the structure of forest bird communities is strongly mediated

by climate zone, and that general rules about ecosystem responses to land-use change can only be developed in the context of understanding this spatial variation.

Fragmentation has a clear impact on the species richness of temperate forest birds. This is in part because fragmentation reduces the breeding success of migrant species (Robinson et al. 1995), some of which appear unable to survive in smaller fragments (Wilcove et al. 1986). However, our results suggest that local extinctions from land-use change in temperate systems are a product of random species loss rather than guild-specific sensitivity to fragmentation. This makes sense for two reasons. First, the difference between forest and non-forest vegetation, and between the avian communities occupying these habitats, is often less dramatic in the temperate zone than the tropics (Salisbury et al. 2012). Thus, forest specialists may be replaced by non-forest species with similar ecological roles in smaller fragments. Second, the seasonally fluctuating climate of the temperate zone, as well as the climatic upheaval associated with recent ice ages, have selected for traits associated with broad climatic tolerance and dispersal (Dynesius and Jansson 2000). These traits increase the likelihood that temperate forest birds can accommodate higher levels of land-use change, and disperse across non-forest habitats. Our results support the view that temperate birds are therefore better equipped to survive in fragmented landscapes, and to recolonize fragments after local extinction (Stratford and Robinson 2005, Tobias et al. 2013a).

In contrast, tropical birds may be more sensitive than temperate birds to habitat fragmentation for several reasons. First, the patchy distribution of many tropical species (Meyer et al. 2008) means they are more likely to be absent from a particular fragment because they did not originally occur within its boundaries (Ferraz et al. 2007, Laurance et al. 2011). Second, factors such as low population density, large home ranges and slow reproductive output are also more prevalent in tropical species, increasing the likelihood of population extinction in small habitat patches, over a range of timeframes (Tobias et

al. 2013a). Third, adaptation to relatively stable climates and microhabitats may lead to narrower physiological tolerance in tropical birds (Janzen 1967). This is important because tropical forest fragments are often warmer, brighter and less humid than intact forests (Laurance et al. 2011), and thus potentially unsuitable for forest specialists (Stratford and Robinson 2005).

One of the most commonly cited predictors of sensitivity to fragmentation is dispersal limitation, as this reduces the potential for gap-crossing and re-colonisation after extinction events (Fahrig and Merriam 1994, Thomas 2000, Lees and Peres 2009), thus disrupting key metacommunity processes (Hanski 1999a). A growing number of empirical and comparative studies suggest that dispersal limitation is more extreme in tropical forest species (Moore et al. 2008, Salisbury et al. 2012). One obvious reason is that they tend to be highly sedentary due to climatic stability; another is that they are relatively specialised in terms of dietary (Belmaker et al. 2012) or microhabitat niches (Salisbury et al. 2012). Many tropical birds are therefore poorly adapted for sustained flight, and unlikely to disperse across a non-forest matrix where suitable resources may be absent. Thus, our finding that sensitivity to fragmentation peaks in the tropics may be explained by an array of mechanisms ranging from the purely spatial to intrinsic processes mediated by demography or ecological adaptation.

Similar mechanisms may also explain why sensitivity to fragmentation varies across functional groups. The least dispersive rainforest bird species are typically insectivores, many of which are year-round territorial (Tobias et al. 2011), and struggle to cross even narrow gaps such as roads (Laurance et al. 2004) or small stretches of water (Moore et al. 2008). This problem is compounded in some insectivores which also require large home ranges because of behavioural specialisation, e.g. army ant followers (Willis and Oniki 1978). In contrast, forest frugivores are generally non-territorial and highly dispersive because they are adapted to tracking temporally and spatially patchy

food resources across the landscape (Salisbury et al. 2012, Tobias et al. 2013a). Meanwhile, granivores do not tend to hold year-round territories, prefer more open habitats, and are relatively insensitive to fragmentation, often thriving in small habitat patches, agricultural areas, and the intervening landscape between forest fragments (Donoso et al. 2004, Lees and Peres 2008a, Sekercioglu 2012). Thus, it makes sense that the proportion of insectivores declines in smaller tropical fragments, while the proportion of frugivores remains constant and granivores increase. This pattern is also consistent with recent analyses identifying fragmentation as a key threat to insectivorous birds (Sekercioglu et al. 2002, Barlow et al. 2006, Yong et al. 2011), but a far less serious threat to frugivores than selective logging or hunting (Markl et al. 2012).

Although the proportion of frugivores in avian communities was not affected by fragmentation, we detected a strongly significant reduction in the size of frugivores in tropical fragments. This finding corroborates numerous studies suggesting that larger species are more extinction prone (e.g. Kattan et al. 1994, Bennett and Owens 1997). It may arise because habitat patches are too small to accommodate large home ranges, or because fragmentation is associated with higher human populations and improved access, leading to greater pressure from hunting, which disproportionately targets larger species. The synergistic effects of land-use change and hunting tends to exterminate populations of larger species from tropical fragments (Peres 2001), and may apply particularly to tropical frugivores simply because many of them are large and sought after as game species (e.g. guans, toucans, hornbills etc.). The disappearance of large-bodied frugivores highlights the importance of considering functional traits, including body size, as analyses focusing on raw or proportional species richness detected no impacts of fragmentation on tropical frugivores.

Implications for ecosystem processes

We have shown that the effects of fragmentation on functional groups of birds are most extreme in the tropics, largely coinciding with rainforest regions where deforestation and fragmentation reach their peak (Achard et al. 2002). The implication of these effects on ecosystems is unclear, particularly as rare species make up a greater proportion of tropical communities, suggesting that functional redundancy may be elevated in comparison with temperate communities (Petchey et al. 2007). While this implies that many species lost from tropical forest fragments may play a relatively minor functional role, an opposing viewpoint is that increased specialisation to microhabitats and trophic niches leads to greater complementarity in tropical systems, regardless of species rarity (Rosenberg 1997, Walker 2006, Tobias et al. 2013a). From this perspective, the complex architecture of tropical food-webs makes them vulnerable to disruption by fragmentation.

Very few studies have investigated the functional implications of fragmentation, and those that do have tended to focus on life-history traits with uncertain implications (e.g. Barbaro and van Halder 2009). In this study, we have focused on feeding guilds with direct links to the functioning of ecosystems. For example, insectivorous birds provide an important service through the regulation of phytophagous insects (Sekercioglu 2006a, Sekercioglu 2006b). Previous work confirms that avian insectivores reduce herbivory on tropical forest trees, and that increasing herbivory by experimentally excluding insectivores leads to reductions in plant growth and rates of photosynthesis (Van Bael et al. 2003, Mooney et al. 2010). Moreover, high species richness and functional diversity of insectivores is associated with reduced leaf damage and improved agricultural yield in agroforestry systems (Philpott et al. 2009). The pervasive decline of forest-dependent insectivores with reduced fragment size in the tropics fits a pattern reported previously (Sekercioglu et al. 2002, Barlow et al. 2006), and supports the view that the loss of

specialised insect predators may have major long-term impacts on plant diversity and biogeochemical cycling in human-modified tropical forests (Tobias et al. 2013a).

The loss of frugivores from tropical forests has been shown to reduce dispersal distances and recruitment of large-seeded tree species (Sekercioglu 2006a, Terborgh et al. 2008, Sethi and Howe 2009, Wotton and Kelly 2011), many of which are important for maintaining the complex vegetation structure and dark understorey associated with rainforest. Although our results confirm that avian frugivores as a guild are relatively insensitive to forest fragmentation, in line with previous research (Willis 1979, Markl et al. 2012), the pervasive decline we detect in large-bodied frugivores in tropical fragments has potentially critical implications for long-term ecosystem function. Notably, such frugivores are associated with the dispersal of larger seeds (Galetti et al. 2013), and thus the decline in frugivore size after fragmentation may lead to the decline or extinction of large-seeded trees, with implications for the composition and functioning of fragmented tree communities (Sekercioglu 2006a). Moreover, the loss of key agents of seed dispersal will theoretically reduce the potential for forest restoration when fragments are reduced in size.

Limitations of my approach

Although we deal with the possibility of ‘extinction filters’ in temperate birds by comparing fragments with continuous forests, my analyses ignore a range of factors influencing the outcome of land-use change. For example, we do not consider the timing and duration of habitat isolation, or the history of land-use cover underlying current habitat configurations, factors that can influence present-day patterns of biodiversity occurring in fragments (Brooks et al. 1999, Ferraz et al. 2003, Martensen et al. 2012, Ewers et al. 2013). It is therefore possible that we have underestimated extinction in younger or larger fragments, both of which are more frequent in the tropics, and perhaps

yet to repay their ‘extinction debt’ (Tilman et al. 1994). However, we have drawn our extensive dataset from both old and recent fragmentation events at all latitudes (Appendix A), suggesting that fragment history does not explain my results. Moreover, we do not focus on measures of species richness per se, but rather the extent to which fragmentation affects the proportion of different feeding guilds in a community, and we are not aware that any link has been identified as yet between fragment history and the ecological structure of communities (e.g. Krauss et al. 2010). Exploring this relationship may be a fruitful target for future studies.

A related issue is that, by focusing exclusively on species occurrence data, we may overlook changes in abundance, which rank among the most stochastic and unpredictable responses to fragmentation (Mac Nally 2007). Shifts in abundance can be influential given that species occurring in small fragments may survive in such low numbers as to be ‘functionally extinct’ (Galetti et al. 2013, Tobias et al. 2013a). The species richness of a guild may thus be artificially inflated by species that are rare or transient, contributing little to ecosystem function in small fragments. Clearly, it is important to move beyond the simplistic framework of species occurrence and patch size to explore the implications of an array of relevant factors, such as species abundance, fragment age and isolation (Laurance et al. 2011), and habitat heterogeneity within fragments (Andrén 1994). However, these questions are currently difficult to tackle in a global framework as the datasets required are not currently available. There is a critical need for long-term, standardized monitoring studies in forest fragments, particularly through large-scale experimental studies (e.g. Ewers et al. 2011, Laurance et al. 2011).

Implications for land-use management and conservation strategies

The loss of species from forest fragments is a longstanding issue of central importance for biodiversity conservation and management (Fahrig and Merriam 1994), including

reserve design (Burkey 1989). This is particularly true in tropical areas with rapidly growing economies such as Southeast Asia and the Brazilian Amazon, where land-use change is fragmenting natural habitat rapidly (Lepers et al. 2005). By pooling data from across a large number of fragments, we have shown that fragmentation has relatively little impact on the functional structure of bird communities in the temperate zone, where efforts to conserve ecosystem function can potentially succeed even in patchy habitats. In contrast, our results indicate that fragmentation changes the balance of functional traits, and removes key functional components, from tropical communities. These findings highlight the effect of land-use change on trophic processes (Valladares et al. 2006) and add weight to calls for maximising reserve size and connectivity in the tropics (Laurance 2005, Laurance et al. 2011). In particular, the decline of dispersal-limited species emphasises the importance of implementing strategies to promote dispersal and connectivity across the non-forest matrix. Management interventions enforcing the retention of vegetation corridors and encouraging mosaics of secondary regrowth in deforested landscapes would help to minimize the deleterious impacts of fragmentation, and allow species to recolonize fragments after local extinctions (Gascon et al. 1999, Gillies et al. 2011, Laurance et al. 2011).

We are cautious in making inferences about the size of habitat patches required to conserve biodiversity and ecosystem function because so much depends on local conditions, and in any case our analyses are based on fragment size categories that are too broad to be directly informative. However, our results suggest that tropical forest patches should be maintained above 1000 ha where possible, and that the functioning of smaller habitat patches in agroforestry systems will be improved if they can be maintained above 100 ha because they otherwise suffer a disproportionate loss of species controlling insect herbivory. We note that these estimates relate to the maintenance of basic ecosystem function; much larger patches of tropical forest may be required to

conserve populations of particular threatened or sensitive species (Laurance 2005). Clearly, conservation efforts in fragmented tropical environments should also aim to reduce hunting pressure and maintain sufficient patch size for large frugivores. Finally, our analyses suggest that policy interventions focused on land-use and biodiversity management are always worthwhile given that even tiny forest fragments retain a relatively full set of ecosystem functions, offering hope that fragmented tropical landscapes can be successfully managed and restored.

3 Land-use change and ecosystem function in tropical forests: impacts, thresholds, and implications for policy

3.1. Abstract

Species richness and functional diversity typically decline with increasing intensity of human land-use in tropical forests, yet the implications for key ecosystem functions are less clear. This particularly applies to vertebrates and their contributions to mutualistic networks and top-down processes, many of which are difficult to quantify. Here we use the functional trait structure of 330 bird communities, focusing on the distribution of specific foraging guilds across land-use gradients in Amazonia, to examine shifts in two ecosystem processes integral to the resilience and productivity of rainforests: seed dispersal and insect predation. We show that functional diversity of frugivores and forest-dependent insectivores declines with intensification of human land-use, particularly in secondary forest, agricultural areas, and pasture. In addition, shifts in mean community traits for both insectivores and frugivores reveal that the loss of forest species and recruitment of non-forest species in these habitats is non-random with regard to functional traits. Taken together, our results suggest that changes in bird community size and structure with rainforest degradation have implications for ecosystem function, with an apparent collapse in key services provided in intact forest. Furthermore, the maintenance of relatively constant functional diversity in logged and burnt primary forests highlights their importance to conservation planning.

3.2. Introduction

Land-use change in tropical forests is among the most severe threats to biodiversity worldwide (Green et al. 2005, Jetz et al. 2007), particularly as the effects will almost certainly reach even remote forested regions over the next century, driven by the growth of human populations and their increasing demand for timber, food and energy (Sala et al. 2000, Godfray et al. 2010, McLaughlin 2011). The familiar outcomes of this process are forest loss, fragmentation and degradation, all of which are known to cause steep declines in species richness across a wide range of taxonomic groups (Blair 1996, Blair and Launer 1997, Rybicki and Hanski 2013, Bregman et al. 2014, Pimm et al. 2014). However, the number of species surviving in a community is often a poor index of other functional aspects of biodiversity (Monnet et al. 2014), and thus the implications of tropical land-use change for ecosystem function are still largely unclear (Newbold et al. 2013, Barnes et al. 2014).

One method of overcoming this problem has been to focus on ecological traits as these may provide insight into the likely role of focal group diversity in ecosystem processes (de Bello et al. 2010). The typical approach uses dendrogram-based metrics, such as Petchey and Gaston's (2002) FD (functional diversity), to capture the total range of ecological traits present in a community. Previous studies applying this method in a range of systems have shown that increasing habitat degradation causes shifts in functional diversity (Villegger et al. 2010, Hidasi-Neto et al. 2012), which tends to decline in comparison with undisturbed habitat (Flynn et al. 2009, Ding et al. 2013, Boyer and Jetz 2014). In addition, the intensification of human land-use can reduce functional redundancy by lowering the number of species performing similar roles within the ecosystem (Lawton and Brown 1993, Laliberte et al. 2010).

While these findings highlight the dramatic effects of land-use change on the functional trait structure of ecological communities, there is currently a limit to what can

be inferred about ecosystem function. For instance, most previous studies pool all functional groups together within total communities, making it difficult to draw conclusions about any specific process (Villegger et al. 2010), particularly as the impacts of land-use change vary across functional groups (Vandewalle et al. 2010, Luck et al. 2013). Similarly, even when the focus is at the species-level, multiple functional traits are often pooled together randomly, regardless of whether they have similar or contrasting functions, making it difficult to tease apart the effects of land-use change on different processes (Trisos et al. 2014). In addition, two other methodological drawbacks hamper the interpretation of land-use effects. First, functional traits are often included as categorical variables, such as dietary type, potentially over-simplifying variation in function (Villegger et al. 2010). In reality, outwardly similar species tend to perform an array of slightly different roles within ecosystems (Hoehn et al. 2008), to the extent that losing just a few members of a particular functional group may have disproportionate consequences and cascading effects (Galetti et al. 2013). Second, variation in land-use is also generally reduced to categorical variables (e.g. Flynn et al. 2009, Edwards et al. 2013), with few studies sampling continuously across degradation gradients (Villegger et al. 2010).

Here, we investigate the effects of land-use change on the functional trait structure of 330 rainforest bird communities, sampled across two gradients of habitat disturbance in eastern Amazonia. Environmental data, including continuous measures of habitat degradation, were collected at all sampling points. We sampled a range of different land-uses, including primary forest, burnt forest, logged forest, secondary forest, and various types of agriculture and pasture. For each community, we compiled data on functional traits for two dietary guilds of birds—frugivores and insectivores—focusing on their habitat preferences and morphology. To capture specific aspects of the ecological niche, we measured biometric traits as continuous variables and

then used ordination techniques (Trisos et al. 2014) to partition into specific niche axes, including overall size, trophic traits (e.g. beak shape), locomotory traits (e.g. tarsus length) and dispersal traits (e.g. wing shape). The importance of beak shape and size as an index of dietary niche in birds has long been recognised (Hutchinson 1959, Schoener 1965, Lack and Gillmor 1971, Hsu et al. 2014, Tobias et al. 2014), but few studies have investigated how these key traits change across a gradient of disturbance. Once these separate niche axes were constructed, we applied standard functional diversity metrics to test the impact of land-use change on the niche structure of bird communities.

Most previous studies investigating the effects of land-use change on ecosystem function have focused on plants and microbes (Yao et al. 2000, Laliberte et al. 2010), whereas relatively little attention has been paid to vertebrates, largely owing to the difficulty of detecting highly mobile vertebrate species (Vamosi et al. 2009). Birds provide a useful system for understanding vertebrate responses to landscape degradation for three reasons. First, they are relatively conspicuous and readily surveyed (Gardner et al. 2008), such that species lists can be rapidly compiled for numerous transects using standard techniques. Second, they are the best-known class of vertebrate organisms and thus provide the most comprehensive dataset of interspecific variation in diet and life history (Vandewalle et al. 2010). Third, they perform essential functional roles in tropical forests (Sekercioglu 2006b), both through top-down processes, such as predation on insect herbivores (Van Bael et al. 2003, Mooney 2007), and also through the ubiquitous plant–animal mutualisms underpinning pollination and seed dispersal (Howe 1977, Terborgh et al. 2008, Sethi and Howe 2009, Anderson et al. 2011).

Our analyses focus specifically on the key ecosystem processes of insect predation and seed dispersal by assessing the effects of land-use change on insectivores and frugivores, respectively. Partitioning continuous biometric variables into trophic, locomotory and dispersal traits provides insight into specific responses, with trophic and

locomotory traits representing different aspects of the foraging niche (Trisos et al. 2014), and dispersal being especially relevant to seed dispersal dynamics (Terborgh et al. 2008) and the ability of species to persist in degraded habitats (Lees and Peres 2009). Although these trait axes are not direct measures of ecosystem processes, they nonetheless shed light on likely changes in ecological function (Petchey and Gaston 2006, Sekercioglu 2006b). Specifically, our approach allows us to test (1) the extent to which different types of land-use cause shifts or collapses in functional diversity relative to primary forests, and (2) whether habitat disturbance changes the position of functional diversity, based on traits relevant to ecosystem processes regulated by birds.

3.3. Materials and Methods

Study site and species

Data were collected during intensive field surveys in 2010–2011 across two study regions in eastern Amazonia as part of the Rede Amazônia Sustentável (RAS) project (Gardner et al. 2013). One area (1.9 million ha) was located in the municipality of Paragominas, another (~1 million ha) in the municipalities of Santarém and Belterra (hereafter, Santarém), all in the state of Pará, Brazil. The two regions differ in their history of human occupation, but encompass broadly similar land uses, with a mix of primary and secondary forest habitats interspersed with agricultural and silvicultural production areas. Both these agroforestry mosaics are typical of recently cleared and developed landscapes in Amazonia, and provide an ideal replicated framework for investigating the impacts of land-use change on biodiversity and ecosystem function.

Sampling within the two regions was based on a nested approach across multiple spatial scales (Gardner et al. 2013). 36 hydrological catchments were focused on, 18 per study region, each covering ~5000 ha in area. Within these catchments, detailed environmental information was collected at 377 transects, each 300 m in length with

three sampling points evenly spaced every 100 m. Two 15-minute bird surveys were undertaken at all points, identifying species seen and heard within a fixed 75 m radius. Paragominas was sampled between the 28th July and 20th November 2010, and 18th to 29th May 2011 (by ACL and NGM), while Santarém was sampled between 16th October 2010 and 8th February 2011 (by ACL, NGM, CBA, and BJWD). Transects were distributed using a stratified random sampling design within each catchment (with a sampling density of 1 transect per 400 ha) to ensure that effects of forest structure heterogeneity were captured. Before analyses, we removed 47 transects in rare habitat types (e.g. smallholder agriculture, plantations, fruticulture, reforestation) that were not easily classified within broader habitat categories, leaving a total sample of 330 transects. In total, 472 bird species were recorded across the two regions, with an average of 154 (max = 208; min = 97) species per catchment, and 39 (max = 79; min = 2) species per transect. For more details of survey methodology, full species lists and validation, including links to digital vouchers, see (Lees et al. 2012, 2013).

Habitat and trophic niche

Following (Bregman et al. 2014), we assigned all bird species identified during surveys into two categories of habitat choice depending on whether they were primarily detected in (i) forest or (ii) non-forested habitats (herein, forest and non-forest species respectively. See Appendix B for further details). Species were also classified into trophic niches based on (Sekercioglu et al. 2004a), with updates from Belmaker et al. (2012) and Salisbury et al. (2012). Eight major guilds were represented: insectivores, frugivores, granivores, nectarivores, carnivores, herbivores, scavengers and omnivores. Guilds were defined by their primary diet, which means for example that species were classified as insectivores if they were largely insectivorous but also consumed small amounts of nectar and fruit. Species were categorized as omnivores if no primary diet was apparent

(Appendix B). We focused on insectivores and frugivores because they were dominant in my study communities, providing good sample sizes. Specifically, insectivores were the most frequent guild with 206 species in Santarém and 188 species in Paragominas, whereas frugivores were the second most speciose guild with 67 species in Santarém and 72 species in Paragominas. In addition, both these guilds have clear links to important ecosystem processes, namely insect predation and seed dispersal, respectively (Tobias et al. 2013a). We conducted all analyses separately on insectivores and frugivores because they differ in their sensitivity to fragmentation (Lees and Peres 2008a, Bregman et al. 2014) and are likely to respond asymmetrically to land-use change.

Environmental conditions

Focusing on 330 transects used for bird surveys, standard environmental variables were collected, as described by Gardner et al. (2013). 20-year time-series of remote-sensing (Landsat, MODIS) images were visually inspected, and then verified data for each transect by interviewing local farmers. Key variables included age of deforestation; frequency and timing of forest degradation events; age and frequency of secondary forest regeneration; mapping of fire and logging scars; indices of deforestation and forest regeneration trajectories; cover of mechanized agriculture since 2000; and land-use intensity. Using these variables, transects were classified into one of five land-use types: undisturbed primary forest, disturbed primary forest, secondary forest, pasture and agriculture (for further classification details, see Appendix B). Agriculture is defined as ‘mechanised agriculture’ such as croplands, in contrast to non-mechanised agricultural land-uses (e.g. pasture). To provide an alternative assessment of land-use history, we also selected two continuous variables—mean canopy openness and basal area (area covered by trees and woody palms ≥ 10 cm) (Slik et al. 2010)—as proxies for habitat degradation.

I assumed that undisturbed forests typically have low mean canopy openness and basal area (Laurance et al. 2011).

Trait sampling

To generate a biometric dataset for all species recorded in study communities, skins stored at the Museu Paraense Emílio Goeldi, Belém, Brazil ($n = 1442$ skins), and the Natural History Museum, Tring, UK ($n = 470$ skins) were measured. To account for intra-specific variability, we attempted to measure at least two males and two females for each species. For 116 monomorphic species (31% of total) lacking four specimens of known sex, we used unsexed specimens. On average, we measured 6.9 ± 5.9 specimens per species (3.3 ± 2.6 males, 2.6 ± 1.3 females, 3.6 ± 6.3 unsexed). We generated a mean value for each morphological trait based on averaged data from the male and female (or unsexed) specimens for each species.

In all cases, seven morphometric measurements were taken to the nearest 0.01mm: beak length (distance from the anterior edge of the nostrils to the tip), beak width (at the anterior edge of the nostrils), beak depth (vertical height at the anterior edge of nostrils), wing length (carpal joint to wing tip, measured on a flattened wing), hand-wing index, tarsus length (distance from the middle of the rear ankle joint, i.e. the notch between the tibia and tarsus, to the end of the last scale of the acrotarsium), and tail length. The hand-wing index (Lockwood et al. 1998) is calculated as Kipp's distance (the distance between the tip of the longest primary/wing tip and the first secondary feather measured on the folded wing) divided by the wing length, multiplied by 100 (Claramunt et al. 2012b). See Appendix B for further details on measurement protocol.

We selected these seven biometric traits because they are functional, owing to their importance in mediating survival and fitness (Violle et al. 2007), and taking account of their link to ecosystem processes mediated by foraging behaviour and dispersal, e.g.

seed dispersal and the control of insect herbivores (Bregman et al. 2014, Edwards et al. 2014). Beak size and shape are key trophic traits related to the foraging niche in birds, predicting the size and type of prey or fruit items selected (Wheelwright 1985, Miles et al. 1987, Grant and Grant 2006). Similarly, tarsus length, tail length and wing length are locomotory traits strongly associated with substrate use and foraging manoeuvre (Miles and Ricklefs 1984, Miles et al. 1987, Claramunt et al. 2012a). Finally, the hand-wing index is a dispersal trait associated with dispersal limitation and gap-crossing ability (Dawideit et al. 2009, Claramunt et al. 2012b). These seven functional traits are often used, separately or in combination, to explore ecological and evolutionary patterns across assemblages of birds (Claramunt et al. 2012b, Tobias et al. 2014, Trisos et al. 2014).

Functional traits in birds are often strongly correlated, largely through their association with overall body size. To prevent these correlations biasing analyses towards detecting only processes associated with body size, we used ordination techniques to derive independent trait axes. Specifically, following Trisos et al. (2014), we generated three trait axes related to different ecological characteristics by applying a two-step principal components analysis (PCA) on morphological trait data. Initially, separate PCAs were undertaken on locomotory (tail, wing and tarsus length) and trophic (beak length, width and depth) traits. The second components of the locomotory PCA correlated with tarsus, wing and tail variation independent from body size, and the second PCA of the trophic PCA correlated with beak shape independent of body size. These PCAs were used as the locomotory and trophic trait axes, respectively. The first components of the trophic and locomotory PCAs correlated with body size, and thus we combined them in a second PCA to produce an axis of overall body size. A fourth axis (dispersal traits) consisted simply of the log-transformed hand-wing index (see Appendix B and Claramunt et al. 2012b).

Metrics of functional diversity

To assess changes in the structure and function of bird communities across the land-use gradient, we used Functional Diversity, FD (Petchey and Gaston 2002), and Functional Dispersion, F_{DIS} (Laliberte and Legendre 2010). These metrics were chosen because they offer complementary perspectives on variation in functional diversity, with FD performing relatively well in simulations (Kraft and Ackerly 2010), and F_{DIS} being sensitive to changes in the overall spread, or ‘dispersion’ of traits (Laliberte and Legendre 2010). The key difference is that FD is the sum of branch lengths of a given trait or set of traits, whereas F_{DIS} is the mean distance of all species to the community mean trait value. This means that, although FD is strongly correlated with species richness, F_{DIS} is not (Laliberte and Legendre 2010).

Analytical approach

Variation in functional diversity

Because insectivores and frugivores have been shown to respond differently to land-use change in tropical forests (Sekercioglu et al. 2002, Ferraz et al. 2007, Gray et al. 2007), we ran each of the following analyses separately for these guilds. We also divided all data sets into forest and non-forest species because (1) they are likely to show contrasting responses to forest extent and quality (Devictor et al. 2008) and (2) non-forest species play little role in the functioning of forest ecosystems (Sekercioglu et al. 2006).

We first (Analysis 1) assessed how raw FD of bird communities varied across different land-use categories. We then (Analysis 2) tested whether this variation deviated from random using an independent swap algorithm (Gotelli and Graves 1996). Specifically, the FD values derived for each community in the five land-use categories were compared to a null expectation drawn from 999 random communities with species richness equal to the observed community, and the probability of presence for each

particular species determined by its overall occurrence frequency across all communities (Gotelli and Graves 1996). We used the independent swap algorithm as it helps to control for underlying variation in species richness. Finally (Analysis 3), we assessed whether raw F_{DIS} varied across different land-use categories. Observed values of standardised FD and F_{DIS} were pooled for each of the land-use categories.

In analyses 1–3, we used two-tailed Wilcoxon signed-rank tests to compare among categories. We completed analyses 2–3 in two different ways regarding the treatment of trait data. First, we combined all measured traits into a single PCA, and, second, we conducted the two-step PCA described above to generate derived trait axes linked to specific ecological functions. We ran null model simulations (analysis 2) separately for Santarém and Paragominas as there were differences in species occurring in the two study regions. However, because the regional data were similar in their underlying sampling methods, degradation history, community structure (Moura et al. 2013), and habitat type (Gardner et al. 2013), we pooled the results for Santarém and Paragominas before running Wilcoxon analyses (that is, communities were partitioned by land-use category but pooled across study localities).

Variation in the trait structure of communities

Given that the five-way land-use categorisation used in analyses 1–3 is relatively crude, we also examined shifts in the functional trait structure of bird communities across a continuous land-use gradient (Analysis 4). To achieve this, we used a Generalised Linear Mixed Model (GLMM) technique to test for an association between the mean value for each trait axis and the quality of forest habitat. We used canopy openness and the basal area covered by trees as two simple indices of forest quality at each sampling location (Slik et al. 2010, Laurance et al. 2011). As canopy openness was correlated (inversely) with basal area, we combined these two variables with a PCA. The first PC for these

environmental variables explained 92% of variation (see Table B3 for eigenvalues), and was thus included in my GLMM as the explanatory variable. We allowed the relationship between the environmental PC score and functional traits to be both linear and non-linear, e.g. quadratic. We accounted for potential spatial autocorrelation by including catchment number (1–36) nested within region (Santarém or Paragominas) as random effects in the models. The best model was then selected using the Akaike information criterion, AIC (Burnham and Anderson 2002), with smaller values indicating a closer fit to the observed data. While combining species trait values into a mean value for each community may be relatively crude, its inclusion here allows informative conclusions to be drawn. Not only are we able to provide insight into how the size of occupied traitspace changes but also how this space shifts following land-use change.

All analyses were conducted in R (R Core Development Team 2014). FD was calculated using the ‘*Picante*’ package (Kembel et al. 2010), while F_{DIS} was calculated using the ‘*FD*’ package (Laliberte and Legendre 2010). Furthermore, the GLMMs were fitted using the ‘*glmmADMB*’ package (Skaug et al. 2011). For all statistical tests a critical value of $P < 0.05$ was used.

3.4. Results

Functional diversity

Across the Santarém landscape, trait diversity differed across the five habitat types in both insectivores and frugivores (Fig. 3.1). Specifically, there were large declines in diversity in pastures and agricultural habitats. For insectivores, the distribution of species in trait space was similar between primary and secondary forest, although there were fewer species in secondary forests. However, for frugivores, secondary forest habitats typically lost the larger species present in primary forests (Fig 3.1) and were also relatively

species poor. To formally test whether degraded habitats are capable of maintaining functional diversity, we compared metrics of functional diversity across five broad habitat types. Raw *FD* for insectivore and frugivore communities varied across land-use categories (Analysis 1, Fig. 3.2, Table B2), being lower in highly degraded habitats (secondary forest, pastures and agricultural habitats) than in primary forest. This effect was strong when forest and non-forest species were combined, and also when forest species were analysed in isolation. However, observed *FD* of non-forest insectivores remained similar across the degradation gradient, apart from in pastures where there was a significant increase in *FD* compared to primary forest communities (Fig. 3.2). Similarly, the *FD* of non-forest frugivores remained similar across the land-use gradient, apart from in mechanised agricultural habitats where there were very few occurrences of any frugivores. We note that patterns in raw *FD* are strongly correlated with species richness, and therefore may simply reflect variation in diversity within guilds and across land-use categories.

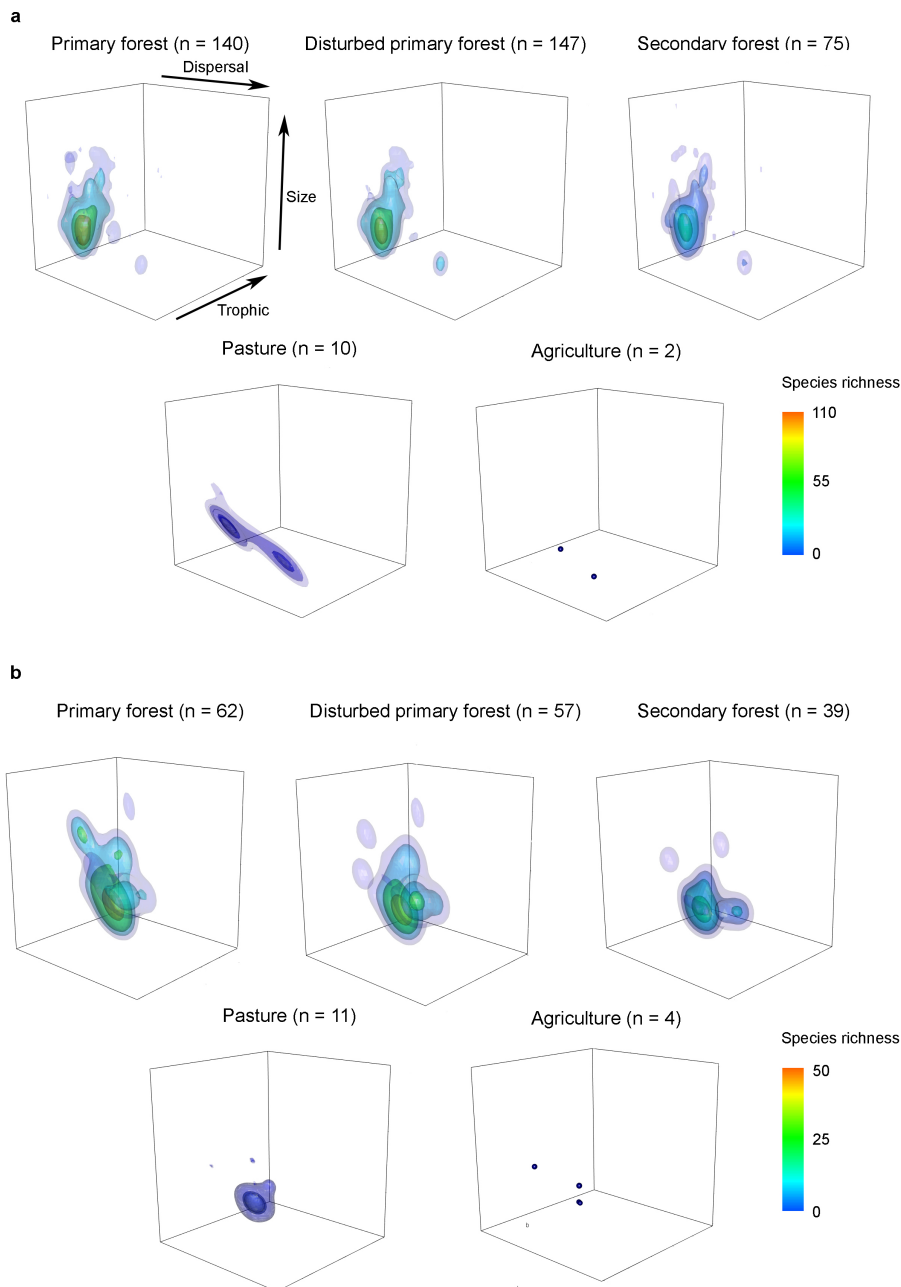


FIGURE 3.1. Functional trait distribution for forest-dependent (a) insectivorous and (b) frugivorous bird species across a land-use gradient in Amazonian forest. Data shown are exclusively from the Santarém landscape to avoid pooling data from different study sites, which overestimates variation in functional traits within land-use categories. Colours denote the density of species in a multivariate trait space (“morphospace”) described by three trait axes (dispersal traits, trophic traits, and overall size); isopleths denote the areas of morphospace containing 75%, 50% and 25% of species occurring in each land-use. Arrows in (a) indicate the direction of increasing values for each of the principal component axes (described in Table B5).

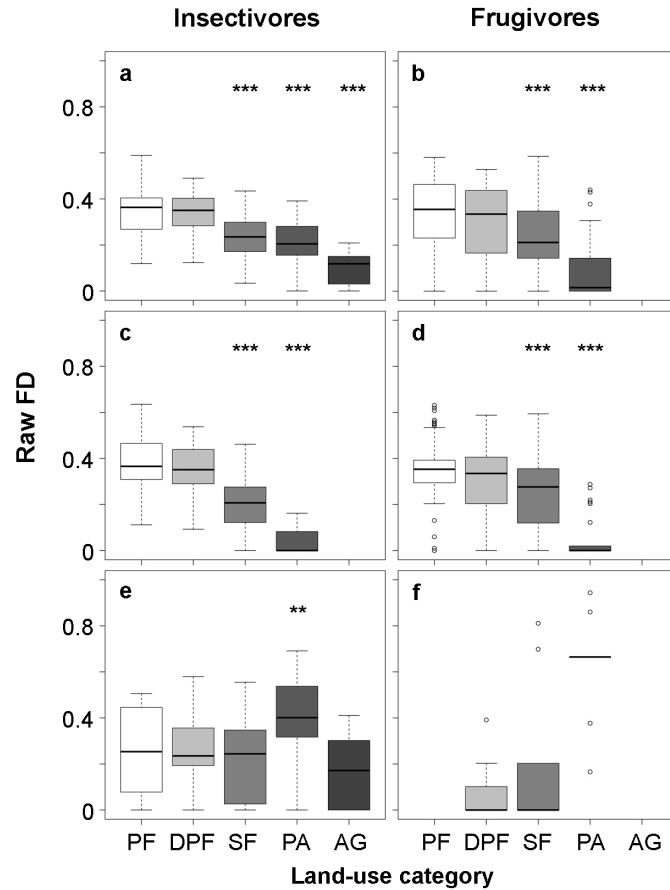


FIGURE 3.2. Observed Functional Diversity (*FD*) for insectivores and frugivores in 330 avian communities across five land uses: primary forest (PF), disturbed primary forest (DPF), secondary forest (SF), pasture (PA) and agriculture (AG). (a) and (b) show data for all species; (c) and (d) show data for forest species; (e) and (f) show data for non-forest species. Data from Santarém and Paragominas are pooled. Asterisks indicate that observed *FD* was significantly different from the null expectation (* < 0.05 , ** < 0.01 , *** < 0.001). All statistical results are from two-tailed Wilcoxon signed-ranks tests. Full results and sample sizes in each land use category are provided in Table B2.

Using a more nuanced null-model approach, and pooling forest and non-forest species to provide adequate sample sizes, we found that *FD* in insectivore communities was neither more nor less than expected by chance, regardless of position across the land-use gradient (Analysis 2, Fig. 3.3a). Although this approach revealed that observed F_{DIS} did change with land-use, it increased in pastures and decreased in secondary forest, in comparison with primary forest (Analysis 3, Fig. 3.3c, Table B4). These patterns were radically altered when we focused exclusively on forest insectivores, as *FD* in pastures

was no longer consistent with my null model, and instead significantly lower than anticipated (Fig. 3.3e), while F_{DIS} was significantly reduced in pastures than in primary forest (Fig. 3.3g).

Regardless of our analytical approach—that is, whether forest and non-forest species were pooled or not—habitat degradation was associated with declining FD and F_{DIS} in frugivores. In degraded habitats (secondary forests and pastures), FD was significantly lower than null expectations (Analysis 2, Fig. 3.3, Table B3) and F_{DIS} was lower than in undisturbed habitats (Analysis 3, Fig. 3.3, Table B4). There were too few species of forest insectivore and frugivore to calculate FD and F_{DIS} for mechanised agriculture (Fig. 3.3).

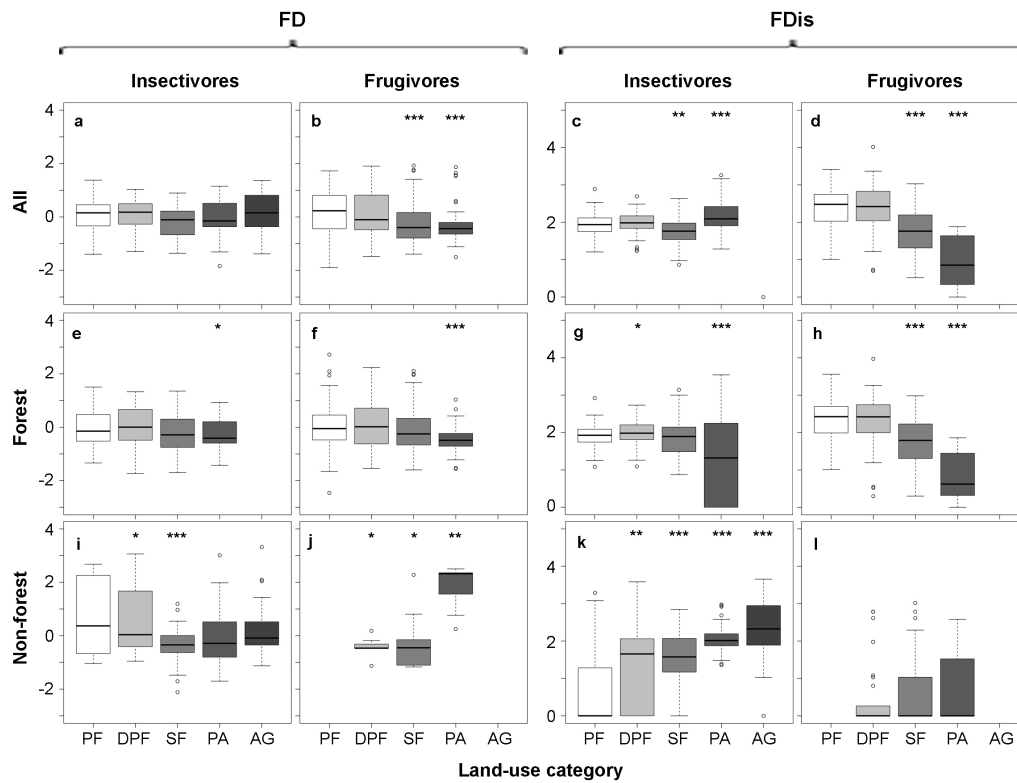


FIGURE 3.3. Standardized effect size of Functional Diversity (FD) and Functional Dispersion (F_{DIS}) for insectivores and frugivores in 330 avian communities across five land uses: primary forest (PF), disturbed primary forest (DPF), secondary forest (SF), pasture (PA) and agriculture (AG). Data from Santarém and Paragominas are pooled. Asterisks indicate that observed FD was significantly different from null expectations, or that observed F_{DIS} was significantly different from F_{DIS} in primary forest (* < 0.05, ** < 0.01, *** < 0.001). All statistical results are from two-tailed Wilcoxon signed-ranks tests. Full results and sample sizes in each land use category are provided in Table B3 and B4.

When we focused on individual traits, we found that the overall trends described above concealed idiosyncratic patterns (Fig. 3.4, Tables B7–B9). For example, body size and locomotory traits contributed disproportionately to the low FD of forest insectivore and frugivore communities in pasture habitats, while trophic traits were neither over- nor under-dispersed. In contrast, dispersal traits had higher FD than expected under my null expectation (Fig. 3.4). Changes in the F_{DIS} of individual traits mirrored the collapse of FD . Specifically, the F_{DIS} of all traits, apart from dispersal traits in insectivores, declined in secondary forests and/or pastures compared with primary forests (Fig. 3.4). In contrast, the F_{DIS} of dispersal traits was similar in secondary forest and primary forest communities, and increased in pastures (Fig. 3.4j, Table B11).

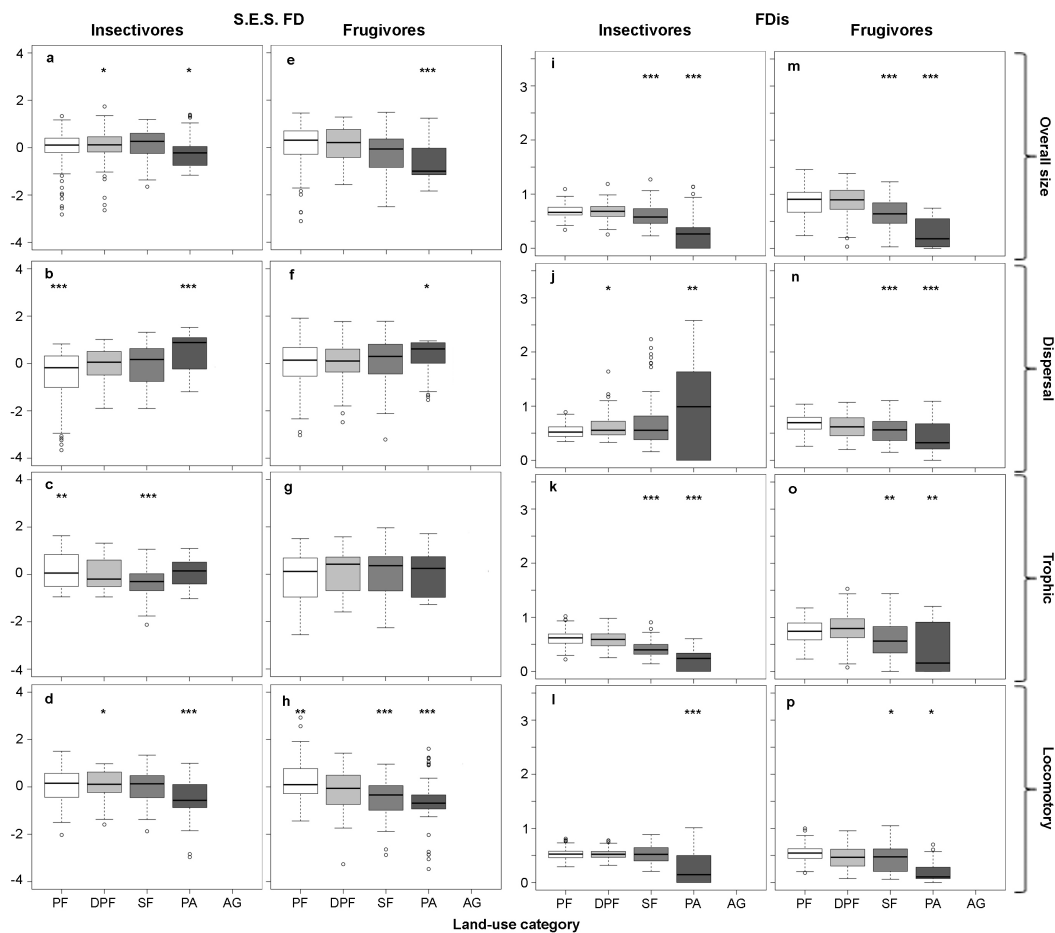


FIGURE 3.4. Standardized effect size (SES) of Functional Diversity (FD) and Functional Dispersion (F_{DIS}) for four functional traits of forest insectivores (a–d, i–l) and frugivores (e–h, m–p) in 330 avian communities across five land uses: primary forest (PF), disturbed primary forest (DPF), secondary forest (SF), pasture (PA) and agriculture (AG). Data from Santarém and

Paragominas are pooled. Asterisks indicate that observed FD was significantly different from null expectations, or that observed F_{DIS} was significantly different from the primary forest F_{DIS} (* <0.05 , ** <0.01 , *** <0.001). All statistical results are from two-tailed Wilcoxon signed-ranks tests. Full results and sample sizes in each land use category are provided in Table B8 and B11.

Community structure in relation to functional traits

To assess how disturbance changes the average position of trait diversity, we used a GLMM approach, removing the quadratic form of my disturbance proxy where it did not improve model fit (i.e. where model AIC was not smallest when the quadratic variable was included). These analyses revealed systematic shifts in mean trait values for both forest (8 models) and non-forest species (8 models) across the land-use gradient (Fig. 3.5, Table B14). In some (5 of 16) cases, four of which (80%) were restricted to insectivore communities, the best fitting models included disturbance as a quadratic term (Fig. 3.5, Table B14) suggesting that the rate of change in mean community traits varied across the gradient. The only model for frugivores retaining the quadratic explanatory variable was focused on overall size in non-forest species, but in this case the linear explanatory variable was not significant suggesting a v-shaped relationship reaching a minimum at intermediate levels of disturbance (Fig. 3.5e, Table B14). We attribute this to the presence of large toucans and ground-dwelling species in forested areas, and the ability of large pigeons and parrots to disperse across disturbed landscapes. However, in most (11 of 16) cases, ten of which (91%) occurred in frugivores and non-forest insectivores, the best fitting models included only the linear term, suggesting that the rate of change in mean community traits remained similar across the disturbance gradient (Fig. 3.5).

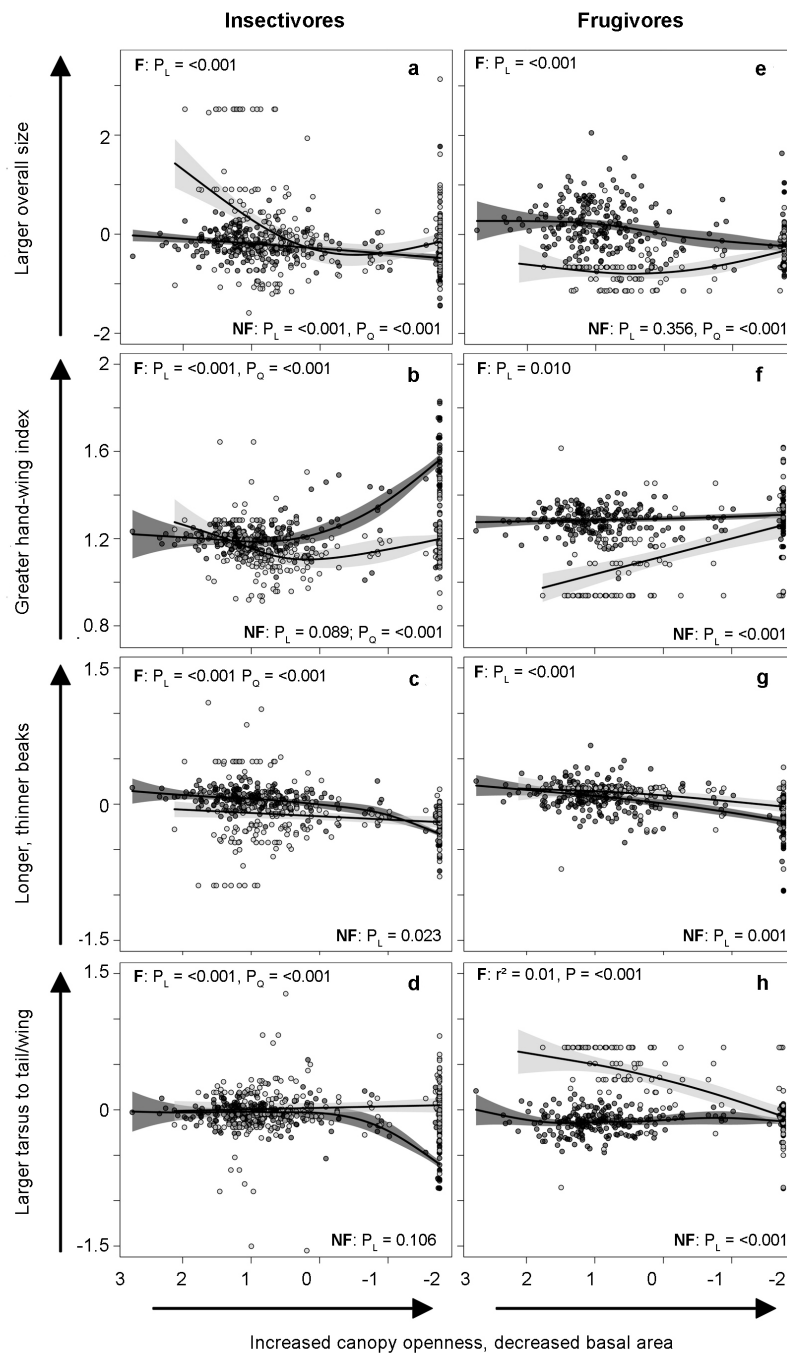


FIGURE 3.5. Change in functional traits in 330 avian communities distributed across a land-use gradient for insectivores (a–d) and frugivores (e–h). Forest species (8 models) are shown in dark grey; non-forest species (8 models) in light grey (lines show model fit from Generalised Additive Modelling; shaded areas show standard error ± 1.96 S.E). All data (traits and environmental variables) were derived from principal components analyses; see Tables B5, B6 and B13. Significance (P) values for linear (P_L) and quadratic (P_Q) forms of the environmental PC axis are from Generalised Linear Modelling (see Table B14 for more details).

3.5. Discussion

By combining detailed datasets of species occurrence, morphological traits and land-uses, we have assessed the impacts of anthropogenic change on two functionally important avian guilds in Amazonian rainforest. In line with previous studies (Flynn et al. 2009, Luck et al. 2013), we find that intensification of habitat disturbance and human land-use reduces both functional diversity and the dispersion of functional traits around the mean community trait. However, our analyses reveal that different processes play out in insectivore and frugivore communities, as well as in different aspects of the ecological niche, with contrasting implications for ecosystem services. By tracing these changes across land-uses using both categorical and continuous variables, we find that ecosystem function is more effectively retained in disturbed primary (i.e. logged or burnt) rainforests than in secondary forest, and that—unsurprisingly—all measures of functional diversity collapse when land-use intensifies to agriculture and pasture.

Impacts of land-use change on functional diversity

Previous studies generally agree that secondary forests and agricultural areas lose a large proportion of tropical forest biodiversity (Kleijn et al. 2001, Peh et al. 2006, Barlow et al. 2007, Koh and Wilcove 2008, Chazdon et al. 2009b, Gibson et al. 2011). For example, working at the same study regions in eastern Amazonia, Moura et al. (2013) found that secondary forests, pastures and areas of mechanised agriculture supported less than half the forest bird species present in undisturbed primary forest. In contrast, primary forests disturbed by human processes (i.e. where forest still stands after burning or logging) are capable of supporting similar levels of biodiversity to that found in undisturbed primary forest (Edwards et al. 2013, Moura et al. 2013). These findings have led to suggestions that logged and burnt rainforests are currently undervalued in biodiversity conservation

strategies because they are likely to maintain high levels of ecosystem function in relation to other human land-uses (Edwards et al. 2014).

Our analyses support this view by revealing that, when all forest and non-forest species were considered together, disturbed primary forests supported similar levels of FD , relative to undisturbed primary forests. In addition, the trait structure of communities in disturbed primary forests was also similarly dispersed around mean trait values (F_{DIS}). These patterns held for frugivores and insectivores, although the impact of increasing land-use intensity was more severe in frugivorous communities, with significant declines in FD in secondary forest and pastures, and diversity dropping to almost zero in agricultural areas (Fig. 3.1). These findings highlight the collapse in FD and F_{DIS} of frugivores in disturbed habitats, presumably reflecting the relative rarity of fruiting trees (Luck and Daily 2003). This lack of resources explains why the colonisation of disturbed habitats by non-forest frugivores does not offset the loss of forest species and thus fails to maintain functional diversity.

The impacts of land-use change on forest-dependent insectivores were essentially similar, with the total volume of trait space declining significantly in secondary forests, and collapsing in pastures and agricultural habitats (Fig. 3.1). This outcome is consistent with the findings of Moura et al. (2013), who showed that the species richness of forest insectivores declined with increased human disturbance across the Santarém and Paragominas landscapes. Indeed, several previous studies have emphasised that rainforest insectivores are highly sensitive to land-use change, often dropping out of fragmented habitat because of their poor dispersal abilities and adaptation to the forest interior (Canaday 1996, Sekercioglu et al. 2002, Tobias et al. 2013a).

Nonetheless, the functional effects of land-use intensification were less pronounced in insectivore communities compared with frugivore communities. In particular, both FD and F_{DIS} were more resilient to land-use change (Fig. 3.3), largely

because non-forest insectivores occurred widely in all habitats, with their FD remaining consistent (Fig. 3.2e) and their F_{DIS} increasing in degraded habitats (Fig. 3.3k). Breaking this down into individual traits, it is notable that both the standardised FD (Fig. 3.4b) and F_{DIS} (Fig. 3.4j) of dispersal traits in forest insectivores increase with land-use intensification, presumably because communities in disturbed habitats retain some low-dispersal species yet contain high-dispersal forest generalists. Yet, in all other traits for forest insectivores, F_{DIS} shows a strong decline in degraded landscapes (Fig. 3.4). Similarly, standardised FD values for overall size and locomotory traits were less than expected in pastures. Taken together these results suggest that although the diversity of dispersal traits in forest insectivores appears resilient to degradation, the collapse of the diversity of other traits is likely to have important implications for the provisioning of ecosystem services. Furthermore, we also emphasise that the overall raw FD for insectivores (forest and non-forest species combined) declined dramatically with increasing degradation (Fig. 3.2a), and the degree to which non-forest species replace forest species by effectively controlling populations of phytophagous insects in forest remnants and young regrowth remains largely unknown.

Variation in trait distribution

We detected shifts in the mean traits of both insectivore and frugivore communities with increasing intensity of human land-uses, corroborating previous studies showing that bird extinction in degraded and fragmented habitats is non-random with respect to functional traits (Turner 1996, Henle et al. 2004, Senior et al. 2013). The change in position of some traits in morphological space, particularly in forest insectivores, was best described by a quadratic fit with the linear term also significant. In these cases, the rate of change in the position of traits occurred most rapidly at high levels of disturbance. This finding suggests that degraded landscapes are sensitive to further small

increases in disturbance, as these have the potential to cause major changes in the position of trait diversity.

Focusing on individual traits, we found that large species were particularly sensitive to habitat degradation, in accordance with previous studies in tropical systems (Cleary et al. 2007, Newbold et al. 2013). Specifically, for insectivores and forest frugivores there was a decline in the overall mean size of species persisting in degraded habitats (Fig. 3.5, Table B14). We attribute these shifts to a suite of factors well known to increase the susceptibility of large species to local extinction in degraded or fragmented environments, including their large area requirements (Kattan et al. 1994), lower reproductive rates (Diamond et al. 1987) and their increased susceptibility to hunting (Peres and Palacios 2007, Markl et al. 2012).

Our models also reveal that the mean hand-wing index increases across the degradation gradient, particularly for forest insectivores (Fig. 3.5b) and non-forest frugivores (Fig. 3.5f). These results indicate that better dispersers typically dominate communities in degraded habitats. Forest insectivores, in particular, are often highly dispersal-limited, with many species unable or unwilling to make prolonged flights across open areas (Moore et al. 2008, Lees and Peres 2009). These species generally have a smaller hand-wing index, and tend to suffer both increased extinction and reduced likelihood of subsequent recolonisation in fragmented or degraded forests (Canaday 1996, Stratford and Stouffer 1999, Sekercioglu et al. 2002, Lees and Peres 2008a, Tobias et al. 2013a), presumably contributing to the patterns we detect.

We found that the beaks of forest and non-forest species were on average shorter, wider and deeper in degraded habitats than in undisturbed habitats (Fig. 3.5c). Among insectivores, we attribute this shift to the loss of species with very long and slender beaks (e.g. *Campylorhamphus*, *Galbula*), and the prevalence of species with shorter,

wider beaks (e.g. *Stelgidopteryx*), whereas for frugivores it can perhaps be explained by a shift to open-country oscine passerines (e.g. *Thraupis*, *Euphonia*).

Finally, the average length of tarsus to tail/wing decreased for forest insectivores in degraded habitats (Fig. 3.5d), again suggesting a shift in the dominant foraging behaviour within communities. Community-wide changes in locomotory traits are presumably driven by the loss of terrestrial and semi-terrestrial insectivores (e.g. *Sclerurus*, *Formicarius*) with long legs and short wings, and colonisation by dispersive arboreal or open country insectivores (e.g. *Tyrannus*) with short legs and long wings (Barlow et al. 2006). In frugivores, a similar decrease in tarsus to tail/wing ratio of non-forest species with disturbance (Fig. 3.5h) is largely driven by both of ground-dwelling frugivores (e.g. *Psophia*), and the continued presence of canopy frugivores.

Taken together, the functional trait structure of insectivore communities in the most degraded habitats (i.e. small tarsus to tail/wing ratio, increased dispersal, and short, wide beaks) suggests that those species present typically feed on the wing, or by gleaning canopy foliage. This perhaps reflects the addition of many mobile, opportunistic, non-forest species, in conjunction with the removal of insect predation services provided by dispersal-limited, gap-shy, forest-dependent species (Lees and Peres 2009). Similarly, the forest frugivores persisting in degraded habitats appear to be those better suited to increasingly patchy resources (increased dispersal, smaller-bodied, lower nutritional requirements). Our findings are therefore consistent with previous studies suggesting that understory insectivores and large canopy frugivores are particularly prone to extinction in degraded rainforests (Kattan et al. 1994, Barlow et al. 2006, Ferraz et al. 2007, Gray et al. 2007, Hamer et al. in press).

Limitations and considerations

One issue often overlooked in studies of functional diversity is abundance, and in particular the changes in abundance following disturbance that often vary across species in unpredictable ways (Mac Nally 2007). This variation can potentially lead to some species becoming ‘functionally extinct’ wherever their abundance drops so low that they contribute little to ecosystem processes (Galetti et al. 2013, Tobias et al. 2013a). We partly address this issue by excluding species that are either transient or rare from the dataset as a whole, but—in common with almost all previous studies focused on avian functional traits—we do not explicitly account for changes in species abundance. Further work should incorporate more informative abundance measures within and between avian communities.

Another important factor to consider is whether changes in functional traits in disturbed environments actually matter. It could be argued, for example, that the decline in frugivore *FD* with habitat degradation is simply a response to the lower range of fruits available, in which case frugivores may continue to provide the dispersal function in full. While this is undoubtedly true to an extent, it remains highly likely that some frugivore species disappear before their food plants, meaning that demand for seed dispersal outstrips supply. Moreover, as many rainforest mammals, including primates, are unlikely to disperse across unsuitable habitat, birds are often the main vector of dispersal for animal-dispersed trees in degraded or secondary forests (Sekercioglu 2006b). From this perspective, the ability of forests to regenerate fully relies on the presence or proximity of avian seed dispersal agents, and is therefore reduced by shifts in the functional trait structure of avian communities away from the state encountered in intact forest. Once components of functional diversity in frugivores are lost, the chances of recolonisation by associated food plants are much reduced.

A related limitation of this study is the lack of temporal replication. Fluctuations in species richness and abundance over time are widespread in nature (Boulinier et al. 1998) but adequate time series for communities remain rare (Debinski and Holt 2000). We note that further changes in bird community structure and composition are perhaps likely in this system because land-use change has been a relatively recent phenomenon (Gardner et al. 2013), suggesting that many communities still owe an extinction debt (Tilman et al. 1994, Wearn et al. 2012). Thus, even if bird communities are currently capable of maintaining ecosystem function in degraded habitats, local extinctions may further reduce functionality in future.

Implications for ecosystem function and land-use management

Because of their key role in forest dynamics such as plant dispersal and recruitment, frugivorous birds are critical ‘mobile-links’ connecting various parts of the landscape and mediating the resilience of forests to anthropogenic change (Wunderle 1997, Lundberg and Moberg 2003, Sekercioglu 2006b). Consequently, the loss of frugivores from forest systems can alter the structure of tree communities and impede regeneration in rainforests (Terborgh et al. 2008, Sethi and Howe 2009). In addition, the important role played by insectivorous forest birds in the top-down regulation of herbivores (Marquis and Whelan 1994) may be affected by species loss, leading to increased leaf damage, and hence reduced photosynthesis and plant growth in degraded habitats (Dunham 2008, Van Bael et al. 2008).

Our results show that functional diversity of frugivores and forest insectivores declines notably in secondary forests, and more significantly in habitats used intensely by humans (e.g. pastures, mechanised agriculture). Nonetheless, functional diversity in these avian guilds was similar in both disturbed and undisturbed primary forest, and shifts in average values for an array of functional traits only occurred at high levels of disturbance.

The frugivore component of avian communities was removed almost entirely in highly degraded habitats. However, it is possible that a similar loss of forest insectivores in these habitats is offset by an influx of non-forest species, which help to maintain the top-down control of herbivores. This remains to be demonstrated (De la Vega et al. 2012), and is perhaps unlikely because non-forest insectivores tend to be generalists (Clavel et al. 2011), and may be unable to capture many types of cryptic herbivorous insects in patches of forested habitat.

Rainforest degradation through the intensification of human land-use is a common global phenomenon, and thus our results may have important implications for the sustainable management of tropical ecosystems. In particular, they suggest that disturbed (logged and burnt) primary forest, and to a lesser extent secondary forest, are important in both maximising species richness and maintaining ecosystem function in landscape mosaics (Edwards et al. 2013, Moura et al. 2013, Edwards et al. 2014). Given that communities of avian insectivores and frugivores in secondary forest showed intermediate levels of functional diversity, we conclude that secondary forest can act as an important buffer against more severe forms of land-use change, increasing the potential for abandoned agricultural landscapes to regenerate rainforest (Chazdon et al. 2009b).

It is increasingly noted that standard approaches to understanding environmental change based solely on species richness are likely to conceal important implications for ecosystem function, leading to calls for functional diversity to be more explicitly considered in biodiversity conservation strategies (Devictor et al. 2010, Naeem et al. 2012). Our results add weight to these calls, and also highlight the importance of going beyond the usual approach of calculating functional diversity by pooling all traits for entire communities. We have shown that contrasting processes play out in different dietary guilds of birds, and in different functional traits, and that these underlying

patterns can be obscured when data sets are treated as homogeneous units. Future analyses need to consider these factors before inferring the likely implications of land-use change for ecosystem function.

4 Species interactions mediate the collapse of biodiversity and ecosystem function in fragmented rainforest

4.1. Abstract

Competitive interactions among species with similar ecological niches are known to regulate the assembly of biological communities. However, it is not clear whether such forms of competition can predict the collapse of communities and associated shifts in ecosystem function in the face of environmental change. Here we use phylogenetic and functional trait data to test whether communities of two ecologically important guilds of tropical birds—frugivores and insectivores—are structured by species interactions in a fragmented Amazonian forest landscape. We found that forest patch size, quality and degree of isolation influence the phylogenetic and functional trait structure of communities, with small, degraded or isolated forest patches having an increased signature of competition (i.e. phylogenetic and functional trait over-dispersion in relation to null models). These results suggest that local extinctions in the context of fragmentation are non-random with a consistent bias towards more densely occupied regions of niche space. We conclude that the loss of biodiversity in fragmented landscapes is mediated by niche-based competitive interactions among species, with potentially far-reaching implications for key ecosystem processes, including seed dispersal and plant-damage by phytophagous insects.

4.2. Introduction

Niche-based processes are often proposed to be dominant factors in explaining the assembly of local communities (Diamond 1975, Tilman 1982, Segre et al. 2014). For example, constraints on coexistence among competing species may regulate the sequence of invasion into a community, as well as its eventual diversity (Fargione et al. 2003, HilleRisLambers et al. 2012). However, it is less clear whether similar processes influence the opposite process of community disassembly (Rader et al. 2014), that is, the sequence and extent of local extinction resulting from habitat degradation or fragmentation. Thus, we still lack a basic understanding of whether declines in biodiversity driven by land-use change are predictable on the basis of species interactions, particularly interspecific resource competition (Tilman 1982, Debinski and Holt 2000, Mayfield et al. 2010).

One of the most prominent impacts of ongoing land-use change is the fragmentation of natural habitats (Fahrig 2003, Brooks et al. 2006, Mitchell et al. 2014). However, despite decades of research on the impacts of this process, the role of species interactions remains contentious. Some studies argue that fragmentation may reduce interspecific competition by generating new niche axes (Buchmann et al. 2013) or limiting the scope for monopolisation, thus allowing more species to co-occur (e.g. Levin 1974, Atkinson and Shorrocks 1981). Other studies provide evidence that competition is elevated in smaller fragments (Feeley 2003, Bennett et al. 2014), which seems plausible if the overall pool of resources declines in parallel with fragment size, increasing the likelihood of resource limitation and competitive exclusion. However, these alternative hypotheses have rarely been tested directly, with most studies focusing on the extent to which local extinction is mediated by species traits rather than species interactions (Robinson et al. 1992, McKinney and Lockwood 1999, Smith and Knapp 2003, Banks-Leite et al. 2012, Senior et al. 2013).

The focus on traits has been instrumental in identifying species attributes associated with sensitivity to habitat fragmentation, including large body mass and limited dispersal ability (Lees and Peres 2008a, 2009). A trait-based approach can provide further valuable insight because functional traits (e.g. beak shape in birds, seed size in plants) have clear implications for fitness through their impact on growth, reproduction and survival, and also provide an objective measure of an organism's role in ecosystem function (Diaz and Cabido 2001, Violle et al. 2007). However, while general patterns of susceptibility to extinction are of fundamental importance (Ewers and Didham 2006, Senior et al. 2013), they tell us little about the underlying processes structuring the collapse of ecological communities in degraded or fragmented environments, or the extent to which such processes are mediated by competitive interactions among species.

Previous studies have attempted to address this question by estimating how community structure varies across environmental or disturbance gradients (e.g. Cornwell and Ackerly 2009, Katabuchi et al. 2012). However, very few empirical tests have focused on communities in habitat fragments, with previous studies apparently limited to taxonomic groups where direct competitive or territorial interactions are diffuse, such as butterflies (Pavoine et al. in press) or other insects (Rader et al. 2014). Moreover, those studies of community assembly focusing on disturbance gradients have either ignored functional traits (Brunbjerg et al. 2012), or combined such traits into multivariate metrics (Mayfield et al. 2005, Dehling et al. in press). The results have been mixed, perhaps because both methods essentially combine different functional groups of species, and pool the effects of different assembly processes with contrasting implications for community structure. In particular, some functional groups and functional traits may be structured by habitat filtering (i.e. species occurrence is driven by their tolerance of environmental factors, predicting that communities are underdispersed with respect to

phylogenetic or trait structure) whereas others may be structured by limiting similarity (i.e. species which share similar resources cannot co-occur, predicting that communities are overdispersed with respect to phylogenetic or trait structure) (Webb et al. 2002, Cavender-Bares et al. 2009). It is now known that focusing on an assortment of guilds and traits can cause these opposing assembly patterns to be merged, reducing the signature of species interactions and providing the inaccurate impression of neutrality (Aiba et al. 2013, Trisos et al. 2014).

Here, we address these issues by testing for the role of interspecific competition in structuring communities of avian dietary guilds across a fragmentation gradient in a humid tropical forest landscape. We restricted our analyses to two key functional groups—frugivores and insectivores. These groups provide an essential service in maintaining ecosystem functioning, as they are critical for the top-down control of phytophagous insects (Van Bael et al. 2008) and seed dispersal (Howe 1977, Da Silva et al. 1996), respectively. We first assessed the phylogenetic structure of insectivore and frugivore communities, assuming that evolutionary history provides a broad proxy for ecological similarity across multiple axes (Cavender-Bares et al. 2009). We then used functional traits to test for the signature of competition in specific niche axes.

Direct competition is most frequent among bird species of similar size, diet and foraging behaviour, leading to the classic view that such competition will limit co-occurrence on these axes, resulting in communities with greater dispersion in body size, beak traits and foraging behaviour than expected by chance (MacArthur 1958, Hutchinson 1959, Schoener 1965, Lack and Gillmor 1971). We therefore considered such axes separately, allowing us (1) to focus on traits most likely to be structured by competition, and (2) to disentangle processes applying independently to different aspects of the ecological niche. The hypothesis that competition increases in fragmented landscapes predicts that as habitat patch size decreases, dispersion in the phylogenetic

and trait structure of communities will increase. Given that communities may be structured at the local scale by both limiting similarity and habitat filtering (Trisos et al. 2014), we tested for over- and under-dispersion of all phylogenetic and trait-based analyses.

Our aims were to understand: (1) the importance of competition in structuring communities across a fragmented landscape; (2) the effects of key properties of habitat patches (i.e. patch size, isolation and habitat quality) on the relative importance of competition; and (3) whether these associations are influenced by foraging guild and morphological traits. These goals are particularly urgent in tropical forests, where habitat fragmentation is currently proceeding most rapidly (Hansen et al. 2013), and with greatest potential impact on biodiversity and ecosystem function (Tobias et al. 2013b, Edwards et al. 2014).

4.3. Methods

Study site and community sampling

Our study focused on forest patches around the Amazonian town of Alta Floresta, Mato Grosso state, Brazil (c. 09°53' S; 56°28' W). Although rates of deforestation have recently slowed, the region suffered a rapid reduction in forest cover from 91.1% to 41.7% between 1984 and 2004; a loss of 3,600 km² of primary rainforest cleared primarily for cattle pasture (Michalski et al. 2008). The Alta Floresta study landscape is characterized by variable sized forest patches embedded in an agricultural matrix (Lees and Peres 2006, 2008a). We used data from standardised surveys carried out between June 2004 and June 2006 (Lees and Peres 2006, 2008a) to compile bird community data from 30 forest patches ranging in size from 1 to 14,476 ha. We then compared these against a single 'continuous' forest site—Rio Cristalino State Park—part of an extremely extensive area

of forest reaching the south bank of the Amazon far to the north. Sampling intensity was standardised across sites, with the same number of point counts (30) undertaken by the same observer, in the same season, for each forest patch regardless of size. More time was therefore spent per area of forest in larger fragments, and thus fewer rare species are likely to be overlooked in small fragments (see rarefaction curves in Lees and Peres 2006). Thus, our analyses are conservative with respect to declines in species richness or abundance with forest patch size. For further details of sampling, selection of fragments and a map of the study area, see thesis preface and Lees and Peres (2006,2008).

The total list of birds recorded in all study sites comprised 336 species, (excluding aerial insectivores, nocturnal species, and waterbirds. We note that this total differs from the totals reported by Lees and Peres (2006,2008), but is correct) (Appendix C, Figs. 4.1 & 4.2), representing seven major guilds: insectivores, frugivores, granivores, nectarivores, carnivores, scavengers and omnivores (Appendix A). Guild membership was based on (Sekercioglu et al. 2004a), with updates from Belmaker et al. (2012) and Salisbury et al. (2012). Guilds indicate primary diet (i.e. an insectivore can be a largely insectivorous species that also consumes small amounts of nectar and berries). Species were categorized as omnivores if no primary diet was apparent (Appendix C). We simplified trophic levels to one guild each because they dominated our study communities (Fig. 2), whereas other guilds did not provide adequate sample sizes for guild-specific analyses of community assembly (Appendix A). Restricting analyses within guilds avoids the problem that strong interactions are unlikely between trophic levels or guilds because of divergent foraging niches (Trisos et al. 2014). Moreover, separate analyses are useful because tropical insectivores and frugivores differ in the level of competitive interactions within guilds (Jankowski et al. 2012), as well as their responses to variation in fragment size, isolation and quality (Sekercioglu et al. 2004b, Sekercioglu 2007, Lees and Peres 2008b, Bregman et al. 2014).

For each forest fragment, landscape characteristics were extracted from satellite images by Lees and Peres (2006, 2008) using Fragstats v. 3.3 (McGarigal et al. 2002) and ArcView 3.2. The variables selected were patch size (ha), distance to the nearest source forest patch (>1000 ha) and the proportion of closed canopy forest within the fragment. We selected 1000 ha as the threshold for source communities because tropical forest patches >1000 ha tend to support communities with similar functional trait structure to continuous forests and may provide temporary refugia for most fragmentation-sensitive species (Bregman et al. 2014).

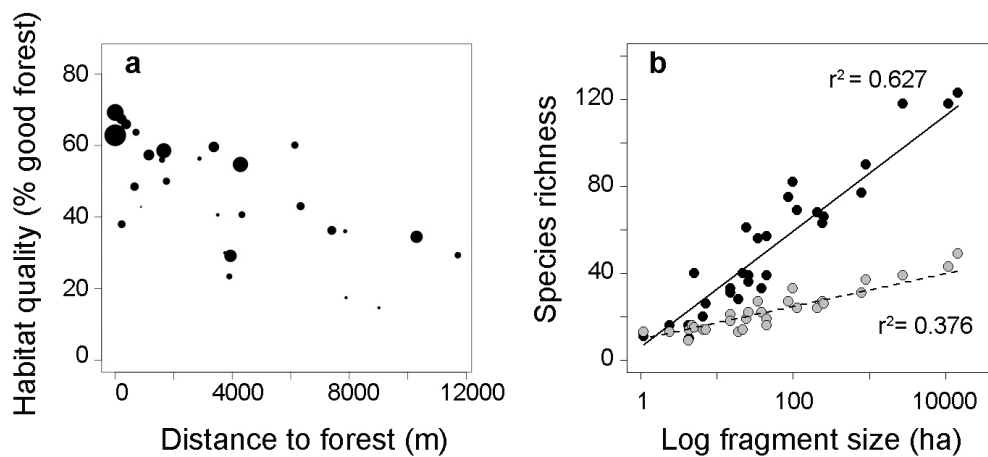


FIGURE 4.1. Basic characteristics of study forest fragments and their bird communities. (a) The relationship between habitat quality and fragment isolation. Circles represent fragments ($n = 30$) scaled by relative size. Habitat quality is plotted as the total percentage of the fragment composed of good quality forest (i.e. tall stature, closed canopy); distance to forest is the straight-line distance from each fragment to the nearest forest >1000 ha (distance is given as zero if fragment >1000 ha). (b) The association between species richness and habitat patch size. Black circles represent the total richness of insectivores in each forest fragment; grey circles represent the total number of frugivores. Lines show model fit from a linear regression: solid line = insectivores; dashed line = frugivores. r^2 values are pseudo- r^2 , calculated following McFadden (1974).

Trait sampling

To provide insight into the ecological function of species, we collected morphological data for all 336 species in the species pools, from specimens held in the Museu Paraense Emílio Goeldi, Belém, Brazil ($n = 1180$ skins), the Natural History Museum, Tring, UK ($n = 493$ skins), and the Museum of Natural Science, Louisiana State University, USA (n

= 185 skins). We took seven measurements: bill length, width and depth, wing length, length of primary projection, tarsus length and tail length (see Appendix C for details on measurement technique). Using a combination of wing length and primary projection we calculated the hand-wing index (Claramunt et al. 2012b). To account for intra-specific variability, we attempted to measure at least two males and two females for each species, using additional unsexed specimens for 130 monomorphic species (39% of total) lacking 4 specimens of known sex. On average, we measured 7.9 ± 8.9 specimens per species (3.7 ± 4.0 males, 2.7 ± 1.6 females and 1.7 ± 5.2 unsexed). Before analysis we averaged the data from unsexed, male and female specimens to create a species average for each morphological trait.

The relationship between avian morphology and ecology is well established: beak shape is related to the dietary niche (Miles et al. 1987, Grant and Grant 2006) while wing shape, tarsus and tail length are proxies for locomotion, i.e. dispersal limitation, substrate use and foraging technique (Miles et al. 1987, Dawideit et al. 2009, Claramunt et al. 2012b). Given that contrasting assembly processes can act on different ecological niche axes, causing problems for multivariate tests of community assembly (Trisos et al. 2014) we dealt separately with each category of functional trait in my analyses. Thus, we generated trait axes and then used the univariate metrics detailed in Kraft and Ackerly (2010) to test hypotheses related to assembly processes. Specifically, following Trisos et al. (2014), we used a two-step principal components analysis (PCA) on averaged morphological traits to generate four trait axes related to different ecological characteristics. Initially, PCAs were separately undertaken on locomotory (tail, wing and tarsus length) and trophic (beak length, width and depth) traits. The second components of the locomotory (short tarsus:long tail/wing) and trophic (short beak:wide/deep beak) PCAs were used as the respective trait axes. The first components of each of these PCAs were then combined in a second PCA to form an axis of overall body size. The dispersal

trait was the final axis and consisted of the log-transformed hand-wing index (see Appendix C and Claramunt et al. 2012b). We adopted this approach as it accounts for correlation between traits and outperformed standard ordination techniques (e.g. Ingram and Shurin 2009) by explaining a greater amount of variation (Trisos et al. 2014). The trait axes generated are described as ‘functional’ due to their importance in governing species interactions and fitness within ecosystems (Violle et al. 2007).

Phylogeny construction

To generate a phylogenetic tree we pruned a recently published global phylogeny of ~10,000 bird species (Jetz et al. 2012) to the regional species pool (336 species). We used the *TreeAnnotator* program in the package *BEAST* (Drummond et al. 2012) to derive the maximum clade credibility (MCC) tree from 1000 randomly selected trees provided by Jetz et al. (2012) (available at <http://www.birdtree.org>). The MCC tree was estimated following a 10% burn in (i.e. the first 100 sampled trees were removed), with a posterior probability threshold of 0.5 and median node heights. The MCC tree is the best-supported tree from my sample of 1000 alternative trees. The Jetz et al. (2012) tree is based largely on molecular data, but also includes >3000 species placed on the basis of taxonomic information. This includes 58 species (17%) of the regional species pool. Uncertainty in the phylogenetic relationships inferred from this tree affect both molecular and taxonomic placements. However, this uncertainty is far greater within clades and much reduced in community phylogenies such as ours, as these typically contain single members of particular genera and thus mainly estimate relationships between clades.

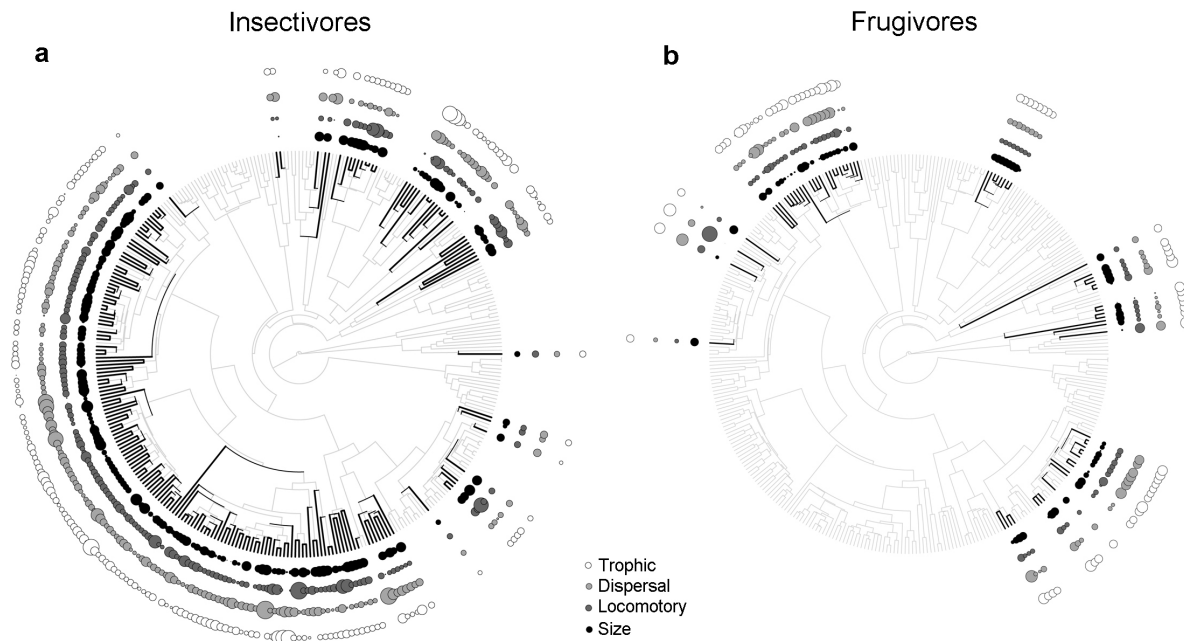


FIGURE 4.2. Phylogenetic history and functional traits of 336 bird species surveyed in rainforest at Alta Floresta, Brazil. Each branch of the phylogram represents one species present in at least one of 31 avian communities (30 habitat patches; 1 intact forest). Black branches represent the insectivore (a; $n = 196$) and frugivore (b; $n = 68$) feeding guilds. Circles at branch tips represent functional traits, sized in relation to trait axes derived through principal component analyses (PCA). Correlates of trait axes are summarized in tables C1 and C2.

Statistical tests

Using phylogenetic and trait data sets, we generated three complementary metrics capable of accurately detecting the presence of non-random assembly processes when combined with Wilcoxon signed ranks tests at the meta-community level (Kraft and Ackerly 2010, Aiba et al. 2013). Specifically, we extracted mean nearest taxon distance (MNTD) from a global phylogeny of birds (Jetz et al. 2012), as this phylogenetic metric is sensitive to both habitat filtering and competition (Webb et al. 2002); we used the variance (VAR) of biometric traits as a measure sensitive to both habitat filtering and competition (Kraft and Ackerly 2010, Aiba et al. 2013); and the standard deviation of successive neighbour distances divided by the trait range (SDNDR) as a measure sensitive

to even spacing in functional traits and thus the signature of competitive interactions (Kraft and Ackerly 2010).

Observed metrics for each community were compared to a null expectation generated by drawing 999 random communities from the species pool using a standard independent swap algorithm (Gotelli and Graves 1996). Specifically, species richness was constrained for each patch, with species weighted by their overall occurrence frequency (the proportion of patches in which they occurred across the study landscape). To construct the species pool, we included all species identified in the study patches and the adjacent tract of continuous forest (Lees and Peres 2006). Given that community-wide patterns in different functional traits can be structured by contrasting assembly processes (Trisos et al. 2014), we ran all analyses separately for each of the trait axes derived by PCA.

If niche traits are conserved on a phylogeny and competitive interactions are dominant in shaping community structure, then co-occurring species should be less related than by chance. To test for phylogenetic signal of my trait axes, we calculated Blomberg's K (Blomberg et al. 2003) and compared observed values to a null model predicting a random distribution of traits across tips of my phylogeny. In essence, it tests whether species' traits are more similar to one another than would be expected under a random draw from across the phylogeny. We found a significant phylogenetic signal in all of the trait axes in both insectivores and frugivores (Table C3). Values of K ranged from 0.47 to 1.38, suggesting intermediate to high phylogenetic conservatism. The majority of traits exhibited intermediate conservatism (K values <1), indicating that they are more conserved than a random association of traits and phylogeny but less conserved than a model of trait association under a model of Brownian motion. However, overall size for insectivores and the locomotory trait in frugivores exhibited K values >1 indicating that those traits are more conserved than expected under a model of Brownian

motion. Traits therefore appear to be non-randomly distributed across the tips of the phylogeny, and we test for competition in both functional trait and phylogenetic structure separately.

To summarise, we addressed the two main aims of my study using a three-stage approach (Aim 2 was explored in Analyses 2a and 2b):

Analysis 1.—To test for the dominant community assembly process operating at landscape levels, we compared the observed values for MNTD, VAR and SDNDR against my null expectation. Specifically, we used two-tailed Wilcoxon analyses to test whether the observed values were above or below the null expectation. Two-tailed tests are necessary for MNTD and VAR, as these metrics are sensitive to both competition and habitat filtering (opposite effects on structure). Previous studies (e.g. Kraft and Ackerly 2010) have used one-tailed Wilcoxon analyses for SDNDR, as this metric is designed specifically to detect competition, but we elected to use more conservative two-tailed tests to minimize type I error. In this analysis, all patches were grouped together irrespective of patch and landscape attributes.

Analysis 2a. To test whether communities are assembled differently in forest patches of varying size, quality and isolation we performed regression analyses with standardised estimates of the phylogenetic and trait metrics as the response variables. We selected the best models to generate average estimates for the most important explanatory variables. Specifically, we first fitted an initial GLMM, where we included fragment size, proportion of fragment covered in good forest and distance to forest larger than 1000 ha, using the *lme4* package (Bates and Maechler 2009). Input variables were standardized using Gelman's (2008) approach implemented in the *arm* package (Gelman et al. 2009). We then use the dredge function in the *MuMIn* package (Barton 2009) to derive a complete set of models before selecting those which are within 2 AIC_c (AIC corrected for the number of model parameters) (Burnham and Anderson 2002)

values of the top model to derive average parameter estimates. Using these model averaging techniques decreases the likelihood that important explanatory variables are left out of my final model, but may lead to overparametrization if too many variables are included and sample sizes are small (Grueber et al. 2011). Therefore, to limit the possibility of including estimates for factors in biologically meaningless models we only generated estimates from those models that are <2 AICc (AIC adjusted by the number of variables included) away from the best model (Burnham and Anderson 2002).

Analysis 2b. While the GLM technique used in Analysis 2a allows us to test whether standardised metrics changed following habitat fragmentation, it gives no evidence of whether the metrics are significantly structured. For example, a standardised metric may exhibit a significant decline as fragment size increases, but if the values are not significantly different from my null expectation then there is no evidence that the process being examined is important in driving community assembly. We therefore split the communities into the top 50% and bottom 50% for each of the habitat properties (i.e. size, quality and isolation) and ran two-tailed Wilcoxon analyses for SDNDR, MNTD and variance. For all statistical tests a critical value of $P < 0.05$ was used.

Validating assumptions

Apparent correlations among predictors entered in multivariate models (e.g. patch size and quality; Fig. 1) may violate the assumption of independence in GLMs. Thus, we tested for multicollinearity by estimating the variance inflation factor (VIF) for each predictor using the ‘usdm’ package. In addition, as our community data may be spatially autocorrelated, we tested for independence of all trait data by implementing a Mantel test in the ‘ade4’ package, to compare observed values to null values based on 999 replicate Monte-Carlo draws. All statistical tests were carried out in R (R Core Team 2014).

4.4. Results

Landscape-wide assembly processes

When we ran analyses across the study landscape as a whole, observed values for the phylogenetic metric, MNTD, were not significantly different from my null expectation for either frugivores or insectivores (Analysis 1, Table 4.1). This suggests that, when all communities are pooled and forest patch characteristics ignored, there is neither phylogenetic over- nor under-dispersion. Species are therefore neither more nor less phylogenetically related than expected by chance.

In contrast, when we focused not on phylogeny but on functional traits, we found evidence for competition in frugivores and both habitat filtering and competition in insectivores (Analysis 1). Specifically, there was a signature of competition in all traits in frugivores and all but dispersal traits in insectivores, with observed SDNDR values being significantly lower than expected under our null model (Table 4.1). Evidence for habitat filtering in insectivores consisted of lower than expected variance in beak shape, locomotory and dispersal traits.

TABLE 4.1. Wilcoxon analyses (two-tailed) of insectivore and frugivore communities ($n = 30$) testing whether observed MNTD, SDNDR and variance are significantly different from the null expectation. Asterisk (*) indicates that the observed values were significantly different from the null model and provide evidence for competitive interactions. The standardized effect size (SES) is provided along with the standard error (SE).

Metric	Trait	Insectivores				Frugivores			
		Mean SES (SE)	No. plots < expectation	V	Wilcoxon P	SES (mean + SE)	No. plots < expectation	V	Wilcoxon P
MNTD	Phylogeny	0.100 (0.982)	14	176	0.253	-0.050 (0.982)	13	270	0.452
SDNDR	Locomotory	-0.220 (0.946)	24	426	<0.001*	-0.248 (0.646)	28	457	<0.001*
	Trophic	-0.206 (0.821)	22	409	<0.001*	-0.127 (1.296)	25	404	<0.001*
	Dispersal	0.252 (0.916)	11	174	0.886	-0.153 (1.024)	19	329	0.024*
	Overall	-0.513 (0.942)	22	419	<0.001*	-0.336 (0.841)	28	451	<0.001*
	Locomotory	-0.025 (0.853)	21	349	0.015	-0.044 (0.663)	15	263	0.543
Variance	Trophic	-0.034 (0.892)	19	335	0.035	0.757 (0.934)	0	0	<0.001*
	Dispersal	-0.051 (0.833)	26	393	<0.001	-0.109 (1.007)	16	222	0.839
	Overall	0.272 (0.765)	5	28	<0.001*	-0.029 (0.769)	6	71	<0.001*

Influence of habitat properties on community assembly processes

Overall effect of patch size.—Patch size was an important predictor of community structure in both phylogenetic and functional metrics, for both insectivores and frugivores (Analysis 2a, Table C4). Patch size was retained in all models within 2 AICc values of the best model (see Table 4.2, Table C4), suggesting that it has a strong influence on community assembly processes in focal guilds. Although this evidence implies that patch size alters the relative dominance of competition and habitat filtering, it is not specific regarding the relationship between patch size and these community assembly processes.

TABLE 4.2. Schematic summary of results from model averaging. Circles show the explanatory variables retained in the final model output within 2 AICc of the best model, divided into significant predictors (black circles) and non-significant predictors (white circles). Estimates are provided for significant relationships (see Table C4 for full model output). *FS*, fragment size. *D*, distance to forest >1000 ha. *GF*, proportion of the fragment classified as good forest.

Feeding guild	Trait	Environmental variables				Estimates and interactions
		<i>FS</i>	<i>D</i>	<i>GF</i>	<i>INT</i>	
Insectivores MNTD	Phylogeny	●				<i>FS</i> , -0.580
	Locomotory	○	○	○		
Insectivores SDNDR	Trophic	○	○		○	
	Dispersal	○		○		
	Overall	●		○		<i>FS</i> , 0.884
	Locomotory	○	○	○		
Insectivores Variance	Trophic	●	○			<i>FS</i> , -0.852
	Dispersal	○	○		○	
	Overall	●				<i>FS</i> , -0.297
Frugivores MNTD	Phylogeny	●	○	○	●	<i>FS</i> , 1.111; <i>FS</i> * <i>GF</i> , -2.013
	Locomotory	○	○			
Frugivores SDNDR	Trophic	●	○	○	●	<i>FS</i> , 1.212; <i>D</i> * <i>FS</i> , -2.455, <i>FS</i> * <i>GF</i> , -2.729
	Dispersal	○	●	○	●	<i>D</i> , -1.752; <i>D</i> * <i>FS</i> , 2.364
	Overall		○	○		
	Locomotory			○		
Frugivores Variance	Trophic	●		○		<i>FS</i> , -1.032
	Dispersal	○		○		
	Overall	○	○	○		

Effect of patch size on competition.—In insectivore communities, both phylogenetic and functional trait data provided evidence that the signature of competition was stronger in small forest patches (Analysis 2a). Specifically, MNTD was positive in small patches (average SES MNTD = 0.662, $P = 0.022$), suggesting that species were less evolutionarily related than expected by chance, whereas MNTD was negative in larger patches (average SES MNTD = -0.461 , $P = 0.018$). These results are consistent with a phylogenetic signature of competition in smaller patches and of habitat filtering in larger patches. The evidence from functional traits differed slightly in that we detected competition across all forest patch sizes: standardized SDNDR values for overall body size, beak shape and locomotory traits were generally more negative than expected (Table C5). However, linear regression (Analysis 2a) revealed that standardized SDNDR of overall body size increased with patch size (Fig. 4.3, Table 4.2, Table C4), again suggesting that competition increased as patch size decreased.

In frugivore communities, we found slightly different patterns of community assembly in relation to habitat patches. Focusing on phylogenetic structure, MNTD was positively correlated with habitat patch size, and significantly underdispersed in small patches (Fig. 4.3, Table C5), suggesting that competition was absent from all patches—large and small—whereas habitat filtering was most prevalent in small patches. However, functional traits (Analysis 2a) suggested that competition played a role in structuring frugivore communities in all patch sizes. Specifically, in all habitat patches, SDNDR was lower than expected on all trait axes, while the variance in overall size and beak shape was greater than expected in smaller habitat patches (Table C6). The linear regression (Analysis 2a) revealed that SDNDR of one functional trait—beak shape—decreased with patch size (Table C4), while variance of beak shape increased

with patch size, suggesting that the relative role of competition increased as patch size decreased.

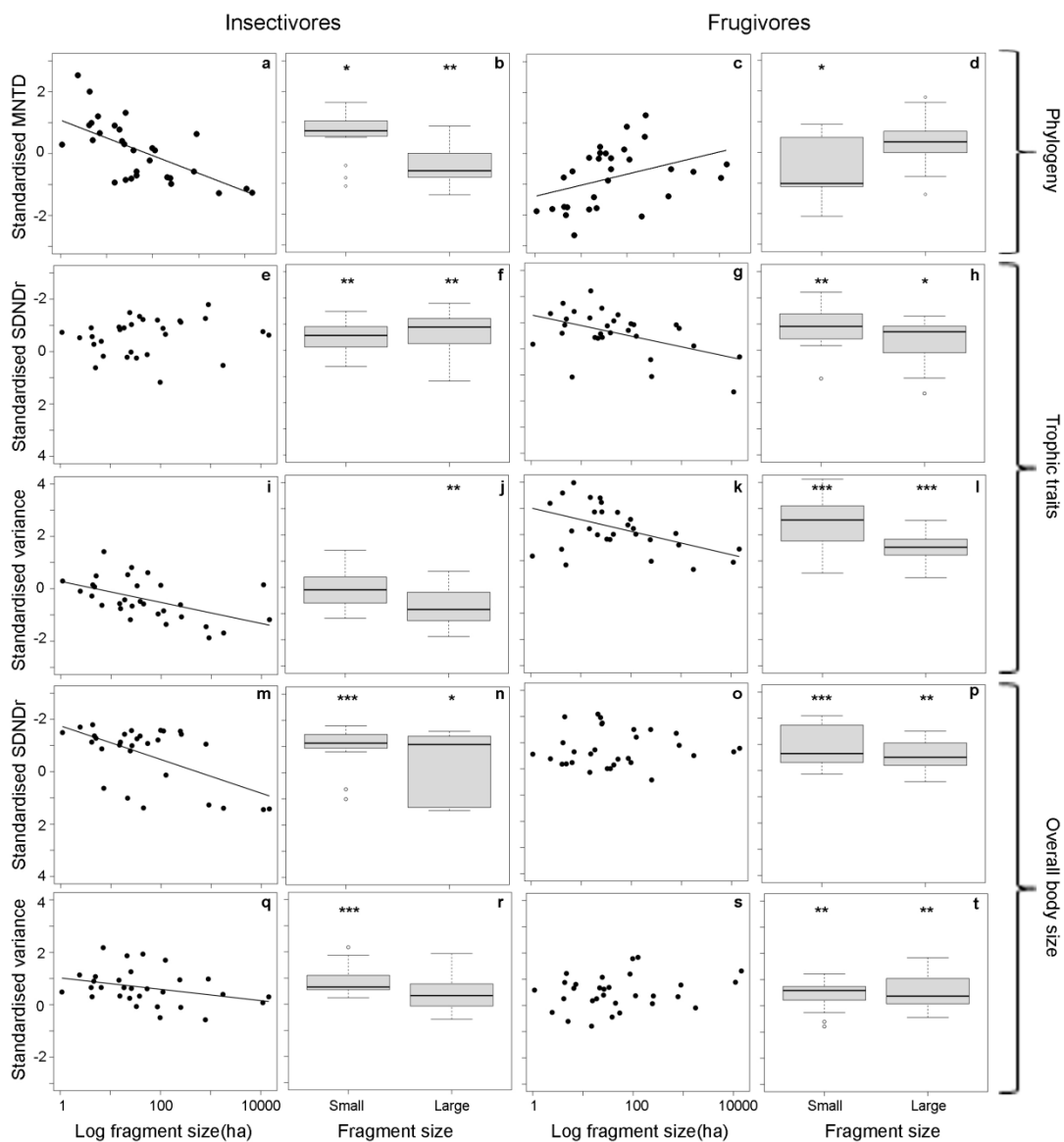


FIGURE 4.3. The structure of avian communities ($n = 30$) in relation to the size of forest fragments (Small fragments ($n = 15$), 1–26 ha; large fragments, 33–14476 ha ($n = 15$)). Structure was calculated from phylogenies (a–d) and from functional traits (e–t), including trophic traits (beak shape; e–l) and overall body size (m–t). Results from dispersal traits and locomotory traits are given in Fig. C2 as neither provided evidence for the changing importance of competition in structuring insectivore or frugivore communities. Regression lines for significant relationships (where $P < 0.05$) show the fit of a model with fragment size as the sole predictor of structure. Final model outputs are included in Appendix C. Note that plots for standardized SDNDR are reversed to facilitate interpretation (increased values indicate increased support for competition).

We note that the linear relationship between the foraging trait and patch size was mediated by the interaction between patch size, isolation and the proportion of the patch covered in closed-canopy forest (Table 4.2, Table C4). Specifically, evidence of competition increased as patch size declined unless forest quality was high or isolation low. Furthermore, for the dispersal trait (hand-wing index), we also found a significant decline in standardized SDNDR with increasing isolation, except in larger patches (Table 4.2, Table C4). In other words, the signal of competition was strongest in isolated habitat patches, unless the size of the patch was large.

Evidence for habitat filtering.—For insectivores, the evidence for habitat filtering was strongest in larger habitat patches, where MNTD and standardized variance for beak shape were lower than expected (Fig. 4.3, Table C5). In smaller fragments, we found no evidence for habitat filtering, with the only significant result being that standardized variance for overall body size was greater than expected, opposite to the prediction of filtering (Fig. 4.3, Table C5). For frugivores this general pattern was reversed, with MNTD being significantly lower than expected in small, but not large, fragments (Fig. 4.3, Table C6). Moreover, we found a significant decline in the standardized variance in beak shape with patch size (Fig. 4.3, Table 4.2), again providing evidence of stronger habitat filtering in small fragments. However, there was no evidence for habitat filtering in any functional trait as the standardized variance of communities was not significantly lower than that expected by chance in either small or large patches (Fig. 4.3, Table 4.2, Table C6).

Validating assumptions.—Maximum VIF across continuous predictor variables entered into our GLMs was 2.3 (Appendix C, Table C11). All variables were therefore <4 VIF, consistent with statistical independence and confirming that our model assumptions were unlikely to be violated by multi-collinearity. In addition, we found no

evidence of spatial autocorrelation across the landscape for any response variable (Appendix C, Table C12).

4.5. Discussion

Our results indicate that the structure of bird communities in tropical rainforest is strongly influenced by habitat fragmentation, in broad agreement with previous studies concluding that disturbance alters assembly processes (Grime 1979, Chesson and Huntly 1997, Violle et al. 2010). The most consistent pattern detected was that of significant over-dispersion of functional trait structure in communities surviving in smaller habitat patches, with this signature reduced or absent in larger habitat patches. This pattern of over-dispersion held true in both insectivorous and frugivorous birds, but varied in relation to patch characteristics, particularly habitat quality and degree of isolation. In addition, we found that the phylogenetic structure of insectivore communities was over-dispersed in smaller habitat fragments. These findings are consistent with the hypothesis that extinctions associated with habitat fragmentation are biased towards species co-occurring with other species that are closely related, or located in denser regions of trait space.

Species interactions are theoretically strongest among lineages with shared recent ancestry and similar ecological niches (Darwin 1859, Diamond 1975), and thus a pattern of over-dispersion in trait or phylogenetic data is generally interpreted as evidence of limiting similarity mediated by interspecific competition (Cavender-Bares et al. 2009). There may be problems with this interpretation when current-day communities are compared against null communities randomly generated from regional species pools, because in this case a pattern of over-dispersion may simply reflect the dominant geographical mode of speciation (allopatric speciation) rather than species interactions *per se* (Warren et al. in press). Our analyses circumvent this problem because they are

essentially comparing the structure of fragmented communities against the species pool occurring at the same location pre-fragmentation. Thus, shifts in structure are likely to reflect local processes playing out over the decades since fragmentation, rather than regional processes operating at biogeographic scales.

Another challenge is posed by spatial scale. Numerous studies testing for the pattern of over-dispersion in animal or plant communities have concluded that the importance of competition varies with spatial resolution, typically increasing at smaller spatial scales when habitat is not fragmented (Weiher and Keddy 1995, Wiens 2011). This scale-dependency of competition is expected as related species coexisting at small scales (e.g. 1 ha) are more likely to interact directly than those coexisting within larger grain sizes (e.g. 100 km²) (Cavender-Bares et al. 2006, Swenson et al. 2007). Our study corroborates this finding by revealing increased evidence for competition in communities with decreasing patch size across a smaller span of scales (90% of forest patches <1000 ha). Given that bird species are relatively mobile, they are more likely to interact directly within habitat patches at these smaller scales, and thus my finding of decreasing trait dispersion with increasing patch size is not likely to be explained simply because species richness builds up through the accumulation of non-interacting competitors.

We found that patch size was the most important driver of community disassembly processes in this hyper-fragmented landscape, being retained in all models and a significant predictor in all but one. Declines in species richness with forest fragment size are ubiquitous (Galli et al. 1976, Blake and Karr 1987, Dos Anjos 2004) in accordance with classical species-area relationships (Arrhenius 1921, MacArthur and Wilson 1967, Lees and Peres 2006, Banks-Leite et al. 2012). In theory, this process of defaunation can play out without any regulation by competitive interactions among related species. Species may simply be lost from smaller fragments because of area constraints, most commonly when the minimum area requirement for viable populations

are not met (Lees and Peres 2008a). In addition, it is often noted that a species present at the regional scale may be absent from a habitat patch simply because it never occurred within that particular local area of habitat, perhaps due to a lack of specialised microhabitat (Fahrig 2003). Extinction in smaller patches can also be driven by environmental changes. For example, smaller habitat patches typically have a reduced ‘core’ area, with edge effects increasing the detrimental and synergistic impacts of increased sunlight, wind, desiccation, fire and selective logging (Kattan et al. 1994, Laurance et al. 1998). These factors contribute to shifts in abiotic conditions as habitat patch size decreases, eventually leading to local extinctions when the environment becomes unsuitable for any particular species (Laurance et al. 2011). Finally, changes in patch and landscape structure may also drive extinction through interactions across trophic levels, for example through mesopredator release (Crooks and Soulé 1999).

Under a simple model of species-area relationships, community disassembly is assumed to be neutral, with species dropping out randomly through extinction as patch size declines (MacArthur and Wilson 1967). Alternatively, habitat changes associated with fragmentation seem likely to drive community disassembly through habitat filtering in smaller patches, leading to the loss of species with particular traits, and thus producing a signature of clustering in phylogenetic and functional traits (Ockinger et al. 2010). Our results suggest that neither of these processes predominates because fragmented communities are structured by the opposite pattern of increasing dispersion in phylogenetic and functional traits with decreasing habitat patch size. Importantly, we also detect increasing dispersion of functional traits with lower habitat quality and greater isolation of fragments from other major forest patches. These patterns are not expected under a purely neutral model of community disassembly through area constraints or habitat filtering, and instead are more consistent with a process driven largely by species interactions within trophic levels. If, after fragmentation, communities become over-

dispersed with respect to traits, this must suggest that the original community had a less dispersed structure (i.e. from more clustered positions near to other species). Here, this effect operates over an ecological, rather than an evolutionary time frame, and thus implies that species remaining outcompeted those lost during disturbance.

It is worth emphasising that the direction of relationships detected in this study are contrary to the prediction that the signature of interspecific competition will decrease in disturbed habitats, largely because lower species richness will relax competition for resources (Grime 1979, Huston 1979, Buchmann et al. 2013). Likewise, they are not consistent with the idea that habitat disturbance destabilizes community structure through environmental stochasticity and penetration of matrix conditions into habitat patches (Ewers and Didham 2006), leading to community structure in disturbed forest patches that is more variable or random when compared with intact tracts of forest (Didham et al. 1998a). Instead, our results suggest that habitat disturbance and fragmentation is associated with significant over-dispersion in community structure as a result of species interactions.

There are two major routes by which species interactions may lead to increased over-dispersion in community structure with decreasing patch size and habitat quality. First, reduced availability of resources could increase competition between pairs of species with similar ecological niches, causing one of them to drop out from the community. Second, changes in environmental conditions may increase mortality rates and reduce population size in some species, increasing their susceptibility to the negative impacts of competitive interactions (Turner 1996, Chesson and Huntly 1997). The intensity of such interactions may be reduced in less disturbed habitat fragments—i.e. those with a larger area of intact forest—if these offer an abundance of resources and a broader range of microhabitats. In all cases, non-random patterns in local extinctions driven by species interactions are likely to be masked when species recolonize from

adjacent source populations, potentially explaining why over-dispersion of communities is accentuated with increasing isolation of habitat patches, as in these cases immigration from source populations is theoretically reduced.

An alternative possibility is that over-dispersion arises because of a shift from specialist to generalist bird species as habitat patch size declines (Bregman et al. 2014). Communities in intact forests are dominated by forest specialists, which are thought to occupy narrower niches, potentially making communities more tightly packed. In contrast, bird communities in small or highly disturbed forest patches tend to contain a larger proportion of generalists, with wider niches, potentially leading to greater spacing between species. While it is difficult to rule out this possibility, there is no a priori reason why broader niches should have centroids that are more highly dispersed as they could simply be more overlapping with similar centroids, leading to greater clustering. Our results are therefore consistent with the view that extinctions driven by the reduction in size, habitat quality and connectedness of rainforest fragments are non-random, and mediated by niche-based interactions among related species.

The signature of over-dispersion in communities varied across guilds and traits, perhaps because of differences in ecological characteristics between insectivores and frugivores. In the aftermath of forest fragmentation, the abundance and availability of invertebrates may decline, leading to food shortages (Burke and Nol 1998, Zarette et al. 2000, Sekercioglu et al. 2002). The consequent decrease in the foraging success of insectivorous birds may intensify competition among species, thus potentially explaining my finding of over-dispersion in body size, an important proxy for prey-choice (Hespenheide 1971). Insectivore communities change dramatically with habitat fragmentation as many forest specialists disappear, whereas the same process has limited impact on tropical frugivores, many of which are tolerant of land-use change (Bregman et al. 2014). Nonetheless, rapid shifts in tree species composition in fragmented

landscapes (Michalski et al. 2007) may lead to local extinctions of key food resources. This could result in increased competition among frugivores, particularly in smaller habitat patches where resources are most limited, thus potentially explaining our finding of over-dispersion in beak shape with declining patch size (Carnicer et al. 2009). We also found over-dispersion in frugivore body size, which correlates with gape size and thus the maximum fruit size in the diet (Wheelwright 1985). Finally, evidence of competition among frugivores inhabiting small, isolated habitat patches was also detected in the dispersal trait axis. It is not apparent why competition should be mediated by this axis, but it may capture the signal of competition from other traits (i.e. species with similar dispersal traits may share other life history characteristics related to resource acquisition).

Given that many of the traits assessed here are representative of a species' role or 'function' in the ecosystem (Petchey and Gaston 2006) our results provide an insight into how key ecosystem processes are affected by fragmentation. Given that the loss of biodiversity causes shifts and declines in the functioning of ecosystems (Loreau et al. 2001), it has long been proposed that declines in species richness associated with habitat fragmentation will affect ecosystem function (e.g. Klein 1989, Laurance et al. 2011, Bregman et al. 2014). Our findings add an extra dimension to this idea by revealing that local extinction caused by habitat fragmentation leads to communities in functionally important groups becoming progressively over-dispersed in smaller habitat patches. Such over-dispersion may be interpreted as maximising the resilience and diversity of functionality because it suggests a relatively broad spread, or complementarity, of functional traits. However, ecosystem processes could be disrupted by the 'thinning out' of species with niches clustered around important optima. In the case of insectivorous birds this may result in reduced predation of certain classes of herbivorous insects, leading to increased leaf damage, and thus reduced photosynthesis and plant growth in smaller fragments (Van Bael et al. 2003, Mooney et al. 2010, Tobias et al. 2013a). In

frugivorous birds, it could lead to reduced dispersal of seeds in important size classes, with important implications for the long-term demographics and recruitment of trees and shrubs in fragmented landscapes (Terborgh et al. 2008, Sethi and Howe 2009, Galetti et al. 2013). Similar effects could occur broadly across numerous guilds of vertebrates and insects, although further studies are required to determine whether the deterministic disassembly processes identified in this study are detrimental to ecosystem function.

The influence of interspecific competition on the trait structure of communities may be most pronounced in tropical birds because of a number of factors, including high species richness, niche saturation, and intense year-round territoriality (Jankowski et al. 2012). For this reason, our results are perhaps less relevant to temperate zone habitats, where competition is theoretically reduced. However, even if the relevance of our findings is restricted in this way, they nonetheless highlight the role of species interactions in mediating the impact of tropical forest fragmentation on biodiversity, with likely implications for ecosystem function. This point is timely and potentially critical because the fragmentation of tropical forests is currently rampant, with poorly understood implications for more than half of the world's terrestrial biodiversity (Pimm and Brooks 2013).

The effects of forest fragmentation have been studied intensively over recent decades, with most previous analyses focusing on species richness or species traits, and ignoring interactions among species. The main exceptions are studies dealing with food webs and other interactions between different trophic levels (Galetti et al. 2013, Martinson and Fagan 2014). Our results suggest that interactions among species within the same trophic level can play an important yet overlooked role in regulating the effects of fragmentation on biodiversity and ecosystem function. This finding provides empirical support for theoretical models suggesting that species interactions mediate community disassembly in disturbed or fragmented environments (Chesson and Huntly 1997,

Buchmann et al. 2013, Lasky and Keitt 2013). The fact that responses to fragmentation appear to depend on interactions among species suggest that the likelihood of local extinction is not only governed by intrinsic traits, environmental variables or matrix conditions, but also on the identity of species co-occurring in the same habitat patch. This adds a new perspective to recent debates about whether the responses of multiple species to fragmentation are interdependent (Didham et al. 2012). Further studies should investigate the functional implications of community structure in fragments, in particular the extent to which over-dispersion of functional traits influences ecosystem function.

5 The future of ecosystem processes: predicting the long-term impact of land-use change on seed dispersal in the Brazilian Amazon

5.1. Abstract

Deforestation is known to negatively impact biodiversity, with major implications for ecosystem functioning. However, due to a focus on community structure at a local scale, we do not have a clear understanding of how the non-random loss of species, following land-use change, alters the provisioning of key ecosystem services across regions. Here, we use extinction data from well-surveyed sites in eastern Brazil to model the impacts of forest cover change on seed dispersal services provided by frugivorous birds across the Amazon basin. Our model predicted that, in general, areas with low forest cover, under current and future forest estimates, support fewer frugivorous species, have lower functional diversity and lack species capable of dispersing large seeds relative to intact forests. However, the relationship between seed dispersal and forest cover was non-linear, with areas containing >50% forest cover retaining large frugivores despite extensive deforestation. Yet, in areas where forest cover was below 50%, the rate of large frugivore loss rapidly increased. We conclude that while a decline in forest cover is likely to have dramatic effects on the diversity of frugivores in many regions of the Brazilian Amazon, the resilience of some frugivores capable of dispersing large seeds may mitigate some of the most damaging impacts of land-use change as long as overall forest cover remains above 50%. Our findings highlight the importance of understanding spatial context (i.e. probability of forest cover change) and community composition (i.e. pool of species providing ecosystem services) when delimiting areas for conservation prioritisation.

5.2. Introduction

Biodiversity is thought to play a critical role in providing and regulating a range of ecosystem processes and services (Balvanera et al. 2006, Naeem et al. 2012). For example, plants are important in the storage of carbon (McPherson and Sundquist 2009), frugivores in the dispersal of seeds and the regeneration of forests (Wunderle 1997), animal pollinators in the maintenance of crop yields (Groeneveld et al. 2010), and insectivorous birds in the control of agricultural pests (Karp et al. 2013a). Land-use change is known to impact species richness (Moura et al. 2013), yet the extent to which ecosystem processes and services are altered in degraded landscapes remains poorly understood (Aizen et al. 2012). Therefore, we still do not fully comprehend how land-use change will impact the resilience and functioning of ecosystems (Cariveau et al. 2013).

Ecological trait-based approaches have largely replaced the use of simple species richness curves for the assessment of land-use change impacts (Mason and de Bello 2013). Species richness curves intrinsically assume that all species in a community are impacted randomly and contribute equally to ecosystem functioning. In contrast, trait-based approaches provide insight into the extinction-proneness of species (Newbold et al. 2013, Senior et al. 2013), and the range and redundancy of traits existing in post-disturbance communities (Pillar et al. 2013). Furthermore, given that many traits are linked to specific ecosystem services (Violle et al. 2007), this approach can provide a useful insight into the resilience and functioning of ecosystems after land-use change. One way of using species traits is to combine them and calculate functional diversity metrics (Mason and de Bello 2013), which generally represent the spread or variety of traits existing within a community (Mason et al. 2005). However, if there is a direct link between a trait and a specific ecosystem service, then we can use that to estimate the impact of land-use change.

Most studies examining changes in biodiversity following land-use change have focused on community structure at a local scale (i.e. across individual land-use gradients) (Flynn et al. 2009, Laliberte et al. 2010, Barragan et al. 2011, Cariveau et al. 2013, Edwards et al. 2013). Typically, communities of species providing ecosystem services decrease in their trait diversity and redundancy across gradients of human land-use change (Laliberte et al. 2010). For example, Edwards et al. (2013) showed a decline in functional richness across a land-use gradient, from undisturbed forest, through logged forest to oil palm plantations. The problem with these studies is that they only test the impact of land-use change on a single pool of species and therefore do not account for variation in overall trait diversity and redundancy between communities in different localities. Another limitation of previous work is that it has focused on predicting species occurrence based on information gained elsewhere in their range (Berglund-Sonnhammer et al. 2006). As a result, it has not been possible to predict the spatial extent of species for which we have no detailed prior knowledge (Keil et al. 2013 but see, Newbold et al. 2014). A third problem is that even where prior information is available for the entire list of hypothetical species, we have yet to consider the impacts of land-use change on both continuous ‘response’ (a trait that determines a species response to a pressure) and ‘effect’ (a trait that describes a species role within an ecosystem) traits (Hooper et al. 2005). Therefore, we still do not have a framework that enables the forecasting of entire communities and the services that they provide at the regional scale.

Here, we address these issues by modelling the integrity of communities of seed dispersing birds, i.e. frugivores, in relation to forest cover across the Brazilian Amazon. Avian seed dispersers (frugivores) offer a useful system for examining the impacts of land use change on the provisioning of ecosystem services for a number of reasons. First, they are well-known study group and can be surveyed with relative ease. Second, birds are the best-known class of organisms and provide the most comprehensive dataset

of interspecific variation in functional traits (Vandewalle et al. 2010). Furthermore, previous studies have highlighted the importance of these traits in determining species persistence in human-dominated landscapes (Lees and Peres 2008a, 2009) and the role of another trait, gape width, in controlling the maximum seed size that may be dispersed by avian frugivores (Wheelwright 1985). Third, they play a key role in maintaining the stability of forest environments through their role as ‘mobile-links’ in plant seed dispersal, thus connecting various parts of the landscape and mitigating the impacts of anthropogenic change (Lundberg and Moberg 2003, Sekercioglu 2006b).

We explore the effects of forest cover on bird species ranges to identify the implications of land-use change for the conservation of individual species, and then estimate ecosystem function as the integrity of frugivore communities, quantifying integrity along two key axes: (1) functional diversity, which represents the spread and variety of traits; and (2) maximum gape width, which indicates the maximum seed size that is capable of being dispersed (Wheelwright 1985). We model extinction probability as a function of forest cover using information on patterns of avian extinction at a well-studied site in eastern Brazil (Santarém), where detailed species lists have been compiled using standard techniques along a forest cover gradient (Moura et al. 2013). Using a mixed modelling approach we derive species probability of occurrence based on body size and dispersal ability (using the hand-wing index, Lockwood et al. 1998, Claramunt et al. 2012b) across forest cover gradients. We focused on these traits as they are known to be useful in predicting species sensitivity to land-use change (Sekercioglu et al. 2006, Newbold et al. 2013). They can therefore be used to predict the effects of changes in forest cover for species occurring in Amazonia but that are not present at Santarém. We then (1) explore the effects of forest cover on bird species ranges and hence community composition, (2) determine how our metrics of ecosystem integrity change along land-

use gradients, and (3) map how the provisioning of seed dispersal varies spatially across the Brazilian Amazon under current and future predicted forest cover.

5.3. Methods

Study sites and frugivore community sampling at the local scale

Intensive surveys of avian frugivores were undertaken in 2010–2011 at two study sites in eastern Amazonia (Fig. 5.1a) as part of the Rede Amazônia Sustentável (RAS) project (Gardner et al. 2013). One area (1.9 million ha) was located in the municipality of Paragominas, another (~1 million ha) in the municipalities of Santarém and Belterra (hereafter, Santarém), all in the state of Pará, Brazil. Both regions contain mosaics of agroforestry and mature rainforest, providing an ideal replicated baseline from which to extrapolate broader-scale impacts of land-use change on biodiversity and ecosystem function across Amazonia as a whole.

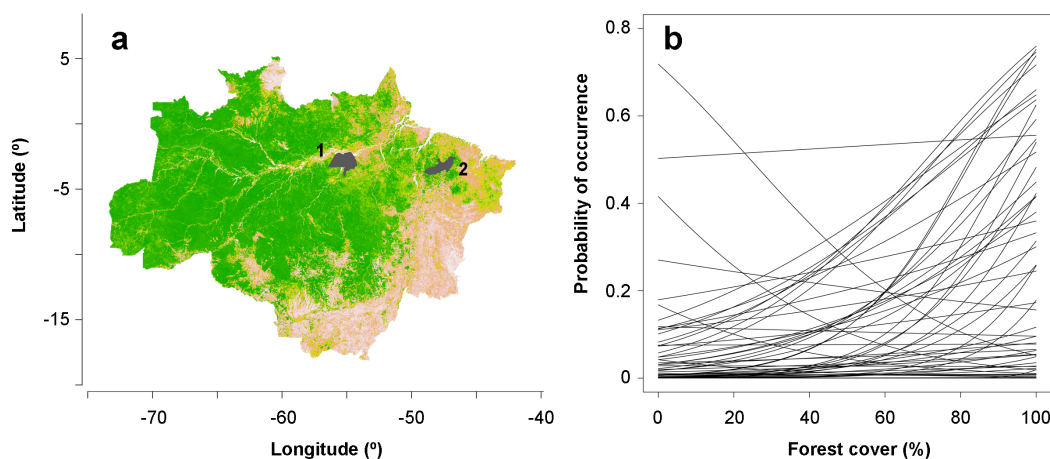


FIGURE 5.1. (a) Map of current (2010) forest cover captured using the MODIS multiband sensor (DiMiceli 2010) across the Brazilian Amazon showing the spatial location of (1) Santarém and (2) Paragominas, and (b) incidence functions for frugivores ($n = 77$) across the forest cover gradient in Paragominas. The curves represent the probability of occurrence for each species found in Paragominas, generated using Generalised Additive Mixed Modelling with forest cover as the explanatory variable and species presence/absence in Santarém as the response variable. Transect and Water Basin ID were included as random effects to account for spatial autocorrelation.

Sampling within the two regions was based on a nested approach across multiple spatial scales (Gardner et al. 2013). We focused on 36 hydrological catchments, 18 per study region, each covering ~5000 ha in area. Within these catchments, detailed environmental information was collected at 377 transects, each 300 m in length with three sampling points evenly spaced every 100 m. Two 15-minute bird surveys were undertaken at all points, identifying species seen and heard within a fixed 75 m radius. Paragominas was sampled between the 28th July and 20th November 2010, and 18th to 29th May 2011 (by ACL and NGM), while Santarém was sampled between 16th October 2010 and 8th February 2011 (by ACL, NGM, CBA, and BJWD). Transects were distributed using a stratified random sampling design within each catchment (with a sampling density of 1 transect per 400 ha) to ensure that effects of forest structure heterogeneity were captured. Before analysis, we removed 47 transects in rare habitat types (e.g. smallholder agriculture, plantations, fruticulture, reforestation) that were not easily classified within broader habitat categories, leaving a total sample of 330 transects. In total, 96 frugivore species were recorded across the two regions (67 at Santarém, 72 at Paragominas), with an average of 32 (max = 48; min = 18) species per catchment, and 8 (max = 21; min = 0 species per transect). For more details of survey methodology, full species lists and validation, including links to digital vouchers, see (Lees et al. 2012, Lees et al. 2013, Moura et al. 2013).

Hypothetical frugivore communities at the regional scale

At the regional scale, we generated hypothetical frugivore communities by overlaying geographical range polygons derived from published digital GIS shape-files (Ridgely et al. 2005) for all seed-dispersing frugivorous bird species recorded in the Brazilian Amazon. We excluded major seed predators (e.g. parrots), which destroy the reproductive parts of seeds and are thus unlikely to play a major role in seed dispersal (Appendix D). This left

a total of 193 seed-dispersing species occurring in the Brazilian Amazon. We limited the spatial extent of my focal area to the Brazilian Amazon for three reasons. Firstly, it has already faced a rapid rate of deforestation that is predicted to continue over the course of this century. Secondly, the Amazon supports a large proportion of the world's biodiversity (Da Silva et al. 2005), and is essential for the storage of human-produced carbon (Pan et al. 2011). Finally, it is one of the only regions in the world for which we have high quality data on current and future forest loss (e.g. Rosa et al. 2013). We follow Rosa et al. (2013) in using the extent of the Amazon described by Red Amazónica de Información Socioambiental Georreferenciada (available at <http://raisg.socioambiental.org>). Hypothetical bird species lists were generated by subsampling species' ranges at a 1km² resolution. Species whose ranges overlapped the centre of 1km² grid cells across the Brazilian Amazon were included in the hypothetical list for that locality.

Measurement of functional traits

For each Amazonian frugivore, morphological data were collected from specimens in the Museu Paraense Emílio Goeldi, Belém, Brazil (254 skins), the Natural History Museum, Tring, UK (467 skins) or from species caught in field studies (See Appendix D for details). Seven measurements were taken: unflattened wing chord, hand-wing index (Claramunt et al. 2012b), tarsus and tail length plus bill length, width and depth (see Appendix D for details on measurement technique). Where possible, we measured at least two males and two females for each species in order to account for intra-specific variability. For 55 species (28% of total) lacking 4 specimens of known sex, we used unsexed specimens. We measured a mean $\pm$ SD of 5.7 ± 5.6 specimens per species (2.3 ± 1.6 males; 1.8 ± 1.2 females; 1.6 ± 4.8 unsexed). Before analysis, we averaged data from unsexed, male and female specimens to create a species average for each morphological

trait. We used gape width as a proxy for the maximum seed size that can be dispersed by frugivores (Wheelwright 1985), the log-transformed hand-wing index as a measure of dispersal (Claramunt et al. 2012b), and overall body size as a measure of species metabolic requirements and extinction-proneness in human-dominated landscapes (Bregman et al. 2014). To generate the trait axis of overall size we used a two-step principal components analysis (PCA) on averaged morphological traits (Trisos et al. 2014). Specifically, PCAs were first separately performed on locomotory (tail, wing and tarsus length) and trophic (beak length, width and depth) traits. The first components of the trophic and locomotory PCAs are correlated with body size, and thus we combined them in a second PCA to produce an axis of overall body size. We adopted this approach as it accounts for correlation between traits and outperformed standard ordination techniques (e.g. Ingram and Shurin 2009) by explaining a greater amount of variation (Trisos et al. 2014). These traits are described as ‘functional’ due to their importance in governing species interactions and fitness within ecosystems (Violle et al. 2007).

Forest cover

We used forest cover data to ‘scale-up’ from Santarém and Paragominas as they were available at a fine enough resolution to link with individual transects, were obtainable for the entirety of the Brazilian Amazon, and are known to be important in determining the composition of avian communities following land use change (Lees and Peres 2008a, Bregman et al. 2014). To provide an estimate of forest cover at the transect scale, we used the proportion of forest cover at a 250 m² resolution available from an image captured with the MODIS multiband sensor in 2010 (DiMiceli et al. 2011), the year in which the original bird surveys were undertaken. The forest cover at the three sampling sites along each transect was averaged to give an overall forest cover value for each transect.

To provide an estimate of forest-cover at a regional scale, we averaged forest cover available at a 250 m² resolution to find the mean forest cover for each 1km² grid cell. Then, using the pre-PPCDAM (business as usual land-use scenario) forest cover projection of Rosa et al. (2013), we completely removed forest cover from cells which had a 60% chance of being deforested by 2050 (other deforestation projections are provided in Appendix D). The model introduced by Rosa et al. (2013) is a dynamic, spatially explicit model that considers uncertainty and the contagious nature of deforestation, and offers an appropriate snapshot into future Amazonian forest cover. While there may have been a recent slowdown in the rate of Amazonian deforestation (INPE 2013), the long-term rate depends on international demand for cattle and soya products (Nepstad et al. 2006). This demand is not slowing and unless beef yields can be increased significantly to open up pastures previously used for cattle ranching, for the production of soy, rates of deforestation are likely to increase once more (Macedo et al. 2012). We chose the 60% forest loss scenario as it captures many of the most pertinent predicted impacts of future land-use change and is unlikely to over-exaggerate broad implications for the provisioning of ecosystem services (Fig. D1).

Species' probability of occurrence at local and regional scales

Using these functional traits and estimates of forest cover, we quantified the probability of each species being present at the transect scale to objectively determine the likelihood of a species occurring at any locality, given the prevailing forest cover. Specifically, using the transects sampled at Santarém and Paragominas, occurrence weightings were generated based on overall body size and hand-wing index, using a generalised additive mixed modelling (GAMM) approach implemented in the package “*gamm4*” (details on model selection can be found in *Model selection* section in Appendix D), using R (R Core Development Team 2014). Specifically, this approach allowed us to estimate the

probability of a species being present at a given forest cover depending on the species traits. The advantage of using a mixed modelling framework is that it allows us to combine the effects of functional traits in determining species occurrence. Furthermore, by making the assumption that species with similar traits respond in a similar way to changing forest cover, this approach allowed us to predict the occurrence of species for which we have trait data but not direct information of their relationship with forest cover. Finally, the hierarchical structure of the dataset meant that multiple observations were made within the same water basins. Therefore, to account for any spatial autocorrelation or pseudoreplication we included ‘transect’ and ‘water basin’ as nested random effects when training the predictive model. In the mixed model, occurrence (species presence/absence along a transect) was the response variable with standardised forest cover, overall body size and the log-transformed hand-wing index included as explanatory variables. In addition to ‘transect’ and ‘water basin’, we included ‘species ID’ as a random effect. This allowed us to create an overall weighting for body size and dispersal across the gradient of forest cover. To test the usefulness of my model in predicting species weightings and to determine the most appropriate training dataset we used model cross-validation. We evaluated our predictions using model AUC (area under receiver operating characteristic curve) (Liu et al. 2011), using the package ‘AUC’, for each species using Santarém occurrences to predict Paragominas occurrences and vice versa. By comparing these evaluative metrics we were able to objectively decide which model was more appropriate for predicting communities across the Brazilian Amazon. The average model AUC value when using Santarém as the training dataset was 0.74 compared to 0.64 when the Paragominas dataset was used. I therefore used the model trained using the Santarém dataset to make predictions across the Brazilian Amazon. These evaluative values are comparable with other studies that have attempted to predict species occurrence at fine resolutions (e.g. Keil et al. 2013). However, the model did not

perform equally well for all species. AUC values varied between 0.53 and 0.98, presumably reflecting neutral processes (i.e. random dispersal and extinction) (Hubbell 2001) and the effect of other species-specific traits that may be important in mediating sensitivity to forest cover change.

Finally, to estimate the probability of a species occurring in a 1 km² cell, we used the model output from my transect scale GAMM to modify species likelihood of occurrence using the prevailing forest cover. Generally, species had a greater probability of occurrence with increased forest cover (Fig. 5.1b). This relationship was steeper in large, dispersal-limited frugivores, with a stronger decline in probability of occurrence as forest cover declined.

Species range size: current and future

To determine the area of species' hypothetical ranges, we counted the number of 1 km² cells that the predicted range overlapped. Species were only considered to be present where their range overlapped with the centre of the cell. Then, to find the size of species ranges after accounting for the influence of forest cover, we summed the probability of occurrence for every species in each of the 1 km² cells for the Brazilian Amazon under current and predicted future forest cover. To calculate the proportion of each species' range predicted to be occupied we divided the re-estimated range sizes by the total area of each species' hypothetical range.

Functional diversity across Amazonia

The size of the Amazon and complexity of my analyses meant that it was too computationally intensive to calculate gape size and functional diversity at a fine resolution. Instead, we calculated the mean of the 1 km² probabilities to generate a single value at a 2500 km² resolution for each species. Based on these averaged probability of

occurrences we drew 100 communities (to minimise noise in my predictions associated with the random draw of species) for each 2500 km² cell and calculated functional diversity (F_{RIC} ; i.e. the amount of functional space occupied by the minimum convex hull connecting traits in each community) (Cornwell et al. 2006, Vileger et al. 2008) for each predicted community and mean gape size for the 5 largest species in each community. We used F_{RIC} as my measure of functional diversity as it is more sensitive to the spread of species morphology than other functional diversity metrics (i.e. the tree-based FD metric of Petchey and Gaston 2006). This is important if we want to emphasise species with particularly divergent morphologies, as they may provide unique services within an ecosystem. For example, highly mobile frugivores are especially important in dispersing seeds long distances, reducing density-dependent impacts on the survival of seedlings. We averaged the values of F_{RIC} of all species and mean gape size for the five largest species from the 100 predicted communities to create a single value for each of the two metrics in every 2500 km² cell.

Testing linearity in functional collapse

To test whether our proxies for ecosystem service provision changed along a gradient of forest cover, we performed regression analyses with functional diversity and the mean gape size of the five largest species as the response variables at a 2500 km² resolution. This resolution was chosen as it is fine enough to provide an insight into appropriate areas for conservation prioritisation and allowed the efficient computation of functional diversity and mean gape width measures across a broad extent. Forest cover was included as the explanatory variable in both its linear and quadratic forms to test whether any change in F_{RIC} and gape width were non-linear. To account for spatial autocorrelation between cells, we used spatial correlation structures in the GLM framework (see Zuur et al. 2009 for further details). We tested the usefulness of different

correlation structures ('corSpher', 'corLin', 'corRatio', 'corGaus' and 'corExp') using overall model AIC (Burnham and Anderson 2002) in the package 'nlme' (Pinheiro et al. 2014). 'corRatio' and 'corLin' were the most appropriate correlation structures for gape width and F_{RIC} respectively (Tables D1 and D2). For the final GLMs a critical value of $P < 0.05$ was used.

5.4. Results

Species range size: current and future

As expected, after considering forest cover, species were predicted to occupy ranges far smaller than those hypothesised by their coarse distribution maps (Fig. 5.2). Under current forest cover, our model predicted that species, on average, occupied 77.5 % of their hypothetical ranges (Fig. 5.2). Furthermore, the proportion of hypothetical species ranges occupied under current forest conditions varied highly between species with proportional range occupancy varying between 45.2% and 98.5% (Fig. 5.2). Under the business as usual scenario of Rosa et al. (2013), average range occupancy decreased to 75.3% by the year 2050 relative to the hypothetical range in 2010. Overall, 131 (68%) species exhibited a decrease in their predicted range occupancy after accounting for future forest cover change. Of the species that showed a decline in range occupancy under forest cover change, the average range decline across the Brazilian Amazon was 4.5%, although, 53 species of the 131 declining species (40%) exhibited range declines greater than 5% by 2050. Evidently, the size of species' ranges matters when we predict the impact of future land use change. Specifically, species with larger ranges tended to show the greatest range contraction between 2010 and 2050 (Fig. 5.2b). This is likely to be because their ranges have a greater chance of overlapping regions most likely to be deforested by 2050.

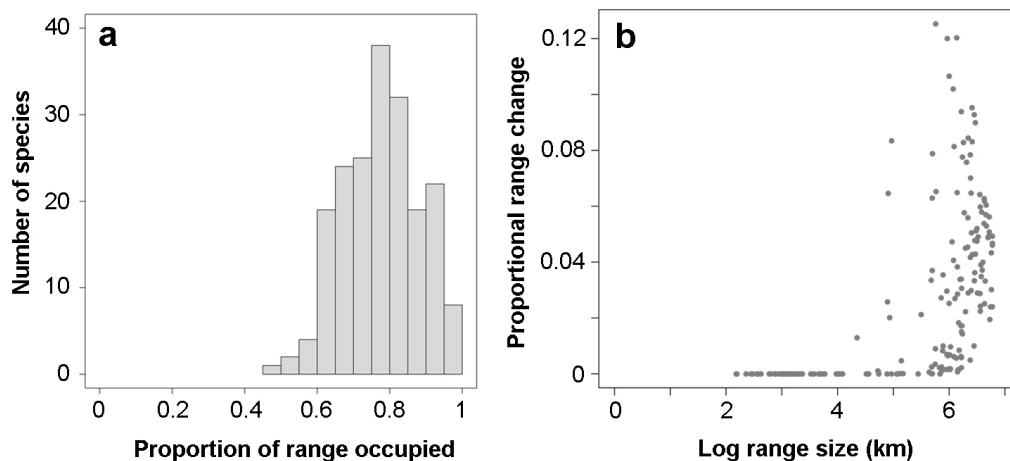


FIGURE 5.2. (a) Current (2010) range occupancy for frugivorous bird species ($n = 193$) in the Brazilian Amazon. The bars show the proportion of species hypothetical ranges (i.e. geographical range polygons derived from published digital GIS shape-files from Ridgely et al. 2005) predicted to be occupied after taking into consideration current (2010) forest cover. This was calculated by summing the individual probabilities of occurrence for each species at a 1 km² resolution across their range before dividing by the area of the total geographical range polygon. (b) The change in predicted proportion of species ranges occupied between 2010 and 2050 against the size of species ranges.

Collapse in functional richness and ecosystem services

After accounting for spatial autocorrelation among cells, we found that both functional diversity and mean gape size of the five largest species declined non-linearly with a decrease in forest cover (Fig. 5.3, Table 5.1). The rate of decline in functional richness was slightly faster for a given amount of forest loss where the proportion of forest cover in each 2500 km² grid cell was high (Table 5.1). However, the mean gape size for the five largest species appeared resilient to a decline in forest cover as long as the total proportion of forest in each cell was high. In other words, the rate of decline of gape size was greater at low forest covers, particularly below 50% (Fig. 5.3, Table 5.1).

TABLE 5.1 The effect of forest cover on overall community functional richness (F_{RIC}) and the mean gape width of the five largest species. The estimates are from generalised linear mixed models with correlation structures as per the framework of Zuur et al. (2009). Forest cover was included in both its linear and quadratic forms.

Response variable	Environmental variable	Estimate	Standard Error	t value	P value
F_{RIC}	Forest cover	0.578	0.045	12.736	<0.001
	Forest cover ²	0.002	<0.001	3.337	0.001
Gape width	Forest cover	0.163	0.010	15.690	<0.001
	Forest cover ²	-0.001	<0.001	-10.22	<0.001

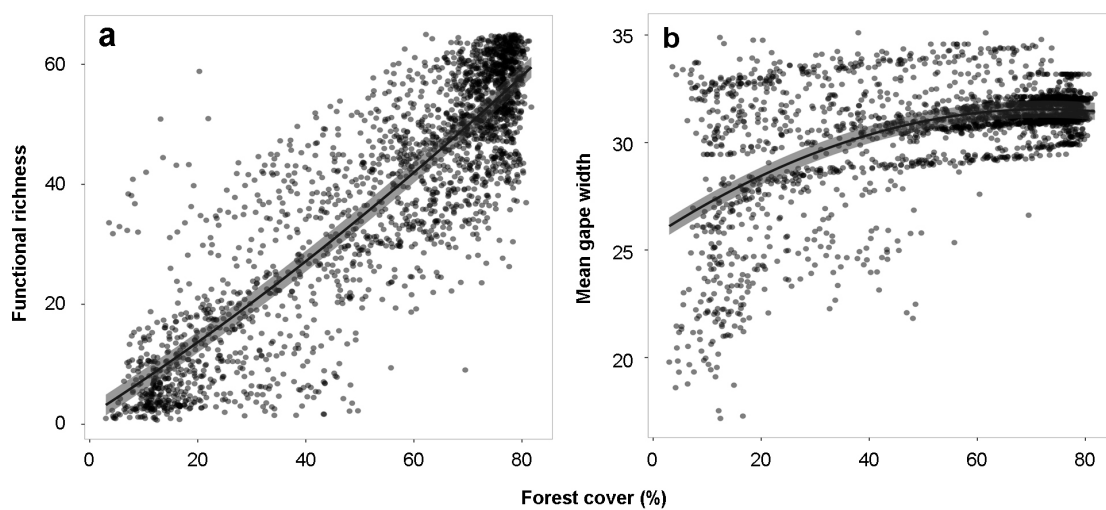


FIGURE 5.3. Scatterplots showing the predicted relationship between functional traits in frugivorous birds and forest cover across the Brazilian Amazon: (a) functional diversity, given as functional richness (F_{RIC}), and (b) mean gape size (in mm) of the largest 5 species. Data points were generated by averaging the functional richness and mean gape size of 100 random community draws at a 2500 km² resolution. The probability of any species being present in the communities was driven by the prevailing forest cover and species traits. Therefore, each data point represents faunal and environmental data from a grain size of 2500km². Lines show the fit of spatially corrected GLMs with standard error indicated by grey shading ± 1.96 S.E.

The spatial distribution of functional diversity around the Brazilian Amazon was strongly correlated with forest cover, with diversity greatest in central and western regions of the Amazon under 2010 forest cover (Fig. 5.4a). The mean gape width of the five largest species was more homogenous across the study region, perhaps because the hypothetical ranges of some particularly large species extend beyond the extent of forest cover. In

particular, Mato Grosso state and the lower reach of the Amazon River are predicted to support communities that have particularly large gape widths. This is largely driven by the presence of the world's largest Toucan, *Ramphastos toco*, a species commonly found in open-forests and savannahs (Ragusa-Netto 2008).

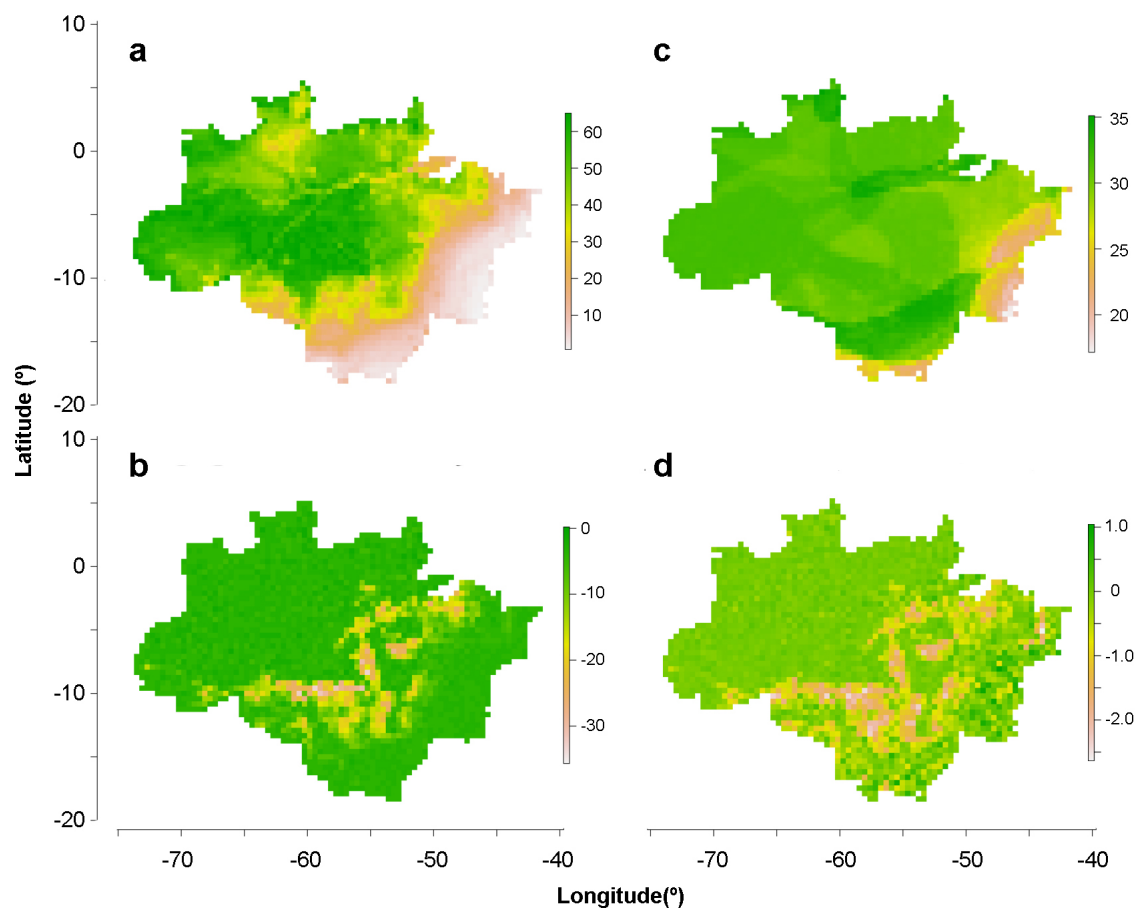


FIGURE 5.4. Mapping changes in an ecosystem process over time across the Brazilian Amazon in terms of predicted functional richness (F_{RIC}) of seed dispersers in (a) 2010 and (b) 2050, and mean gape size (in mm) of the largest 5 species (c) in 2010 and (d) 2050. Maps compare the current situation (a, c) with the predicted future scenario in 2050 (b, d) at a grain size of 2500km². (b) and (d) therefore represent the change in functional richness and mean gape size respectively.

Under future forest cover change, the overall pattern of declining functional diversity is accentuated. Predicted deforestation of eastern and southern forest areas means that, by 2050, functionally diverse regions shrink further towards the west of the Brazilian Amazon. The impact of forest cover change on the mean gape size of the five largest

species was more nuanced. In line with declines in functional diversity, the mean gape size of the five largest frugivores shows a decline in central southern and eastern areas by 2050. However, changes in this metric over larger scales were mediated by the underlying species pool and apparent redundancy in traits. Specifically, the mean gape width of the five largest species showed a notable decrease in 2050 only where forest loss was high and the underlying community pool did not contain large-gaped species capable of occupying open habitats.

5.5. Discussion

In this study we assessed the effects of forest cover loss on the likely future provisioning of seed dispersal services across the Brazilian Amazon. Specifically, we used species local-scale occurrence to estimate the probability that a species will occur at any locality across its range. Using this predictive model, we show that a decline in forest cover will impact the provisioning of seed dispersal services, by reducing the proportion of species ranges that are likely to be occupied. In areas heavily impacted by deforestation, trait diversity is reduced and many large frugivores are absent. However, we also find that the relationship between average gape width and forest cover was non-linear, with large frugivores retained in areas where forest cover was greater than 50%.

Land-use intensification reduces species richness and functional diversity (Flynn et al. 2009) by altering environmental conditions (Foley et al. 2005) and biotic interactions (*Chapter 4*). This loss of species diversity has important implications for ecosystem functioning (Spehn et al. 2005, McIntyre et al. 2007, Garcia and Martinez 2012), particularly where redundancy in service provision among species is low (Naeem 1998) or where extinctions are biased towards a particular group of service providers (Kleijn and Raemakers 2008). In line with studies at a local scale (e.g. Luck and Daily 2003, Lees and Peres 2008a), our model predicts the decline of seed dispersers as forest

cover decreases across the Brazilian Amazon. The diversity of this important bird guild has previously been shown to mediate the number and richness of seeds being dispersed in tropical landscapes (Garcia and Martinez 2012) and so the decline of this group of species is likely to have broader consequences for ecosystem functioning (Cordeiro and Howe 2001). Responses to land-use change are now known to vary within avian feeding guilds, with large frugivores often found to be particularly sensitive to forest fragmentation and degradation (Kattan et al. 1994, Bregman et al. 2014). Given that species within guilds vary in the services they provide, the non-random loss of species may exacerbate the impact of land-use intensification on certain ecosystem processes. Here we found that the mean gape size of the five largest species in the predicted communities declined as forest cover decreased, with the rate of decline greatest at low forest cover. In this case, the biased loss of species important in the dispersal of large seeds (Galetti et al. 2013) is likely to impact large trees by increasing the influence of density-dependent processes (i.e. Janzen-Connell effect) (Janzen 1970, Connell 1971). Specifically, this process decreases the potential for dispersal away from the parent plant, resulting in higher seedling mortality.

Furthermore, the non-linear response of gape width to the decline in forest cover has important implications for conservation prioritisation. If our intention is to maintain ecosystem functioning then these results suggest that there is value in protecting landscapes already impacted by deforestation. Our model predicted that there is some redundancy in the gape width of the largest species, particularly where forest cover is above 50% (Fig. 5.3). In terms of maintaining seed dispersal services, land-sharing strategies (Phalan et al. 2011) may therefore be a useful strategy. However, if conservation practitioners do not act to prevent the loss of forest cover in already degraded landscapes across broad scales then the resilience of large-seeded trees to climate change impacts may be compromised. This is because many of the species with

large gape widths are also essential in long-distance seed dispersal (Jordano et al. 2007), which is important for the adjustment of tree species ranges (Trakhtenbrot et al. 2005) and the recolonisation of pastures following land-use change (Zimmerman et al. 2000). Through this loss of important seed dispersers, human-dominated landscapes could therefore act as an insurmountable barrier for the tree species undergoing range adjustment under climate change. In order to prevent such a scenario, conservationists and policy-makers must interact to ensure that pockets of intact habitat are maintained across the deforestation frontier. These pockets of forest will contain important seed dispersers and act as important regeneration nuclei (Chazdon et al. 2009a) in the future.

Aside from the impacts on the structure of tree communities and the functioning of ecosystems, the results from this study suggest that land-use intensification has important implications for how we estimate patterns of diversity and threats to the persistence of individual bird species. There is an urgent need to describe species realised ranges at a scale that will aid decision-making related to the placement of protected areas. However, we are often reliant on polygons of species distributions that are too coarse to be used alone when we predict occurrence at a fine resolution (Lawes and Piper 1998, Rahbek 2005, Hurlbert and Jetz 2007). For example, studies continue to predict occurrence of species at 0.5 and 1 degree resolutions based solely on coarse range polygons (Davies et al. 2007, Rahbek et al. 2007), yet we know that species do not occur throughout their hypothetical ranges, which are often generated using a few point observations. Instead, a number of processes act on the regional pool of species to determine the species found at a given locality, including; competition, habitat filtering and neutral processes such as random dispersal and extinction (Hubbell 2001, Pigot and Tobias 2013, Segre et al. 2014). Despite not explicitly accounting for neutral processes, my model predicts that realised species ranges are much smaller than their hypothetical range distributions. Furthermore, discrepancies between hypothetical and predicted

ranges varied interspecifically. This has significant implications for the use of species range sizes when classifying species based on extinction-risk (i.e. the IUCN red list). The importance of appropriate scaling is acknowledged in the IUCN red list guidelines and the use of area curves is suggested (Kunin 1998). As species vary in their ecological characteristics, they are likely to occupy different proportions of their hypothetical ranges and so these area curves need to be adjusted for each species using a correction factor (IUCN Standards and Petitions Subcommittee 2014). However, detailed information is rarely available for all species in a community and so we should continue to develop techniques that utilise species traits and knowledge gained at a local scale to trim coarse range polygons (Newbold et al. 2014). When exploring biodiversity-carbon tradeoffs (Venter et al. 2009), it is essential, particularly at finer spatial resolutions, that we use species ranges that have been trimmed according to the prevailing environmental conditions and ecological characteristics. The framework described here not only allows refinement of species ranges but also can be used to weight the importance of locations for each species based on their probability of occurrence. Incorporating species' nuanced responses to forest cover change will minimise the possibility of prioritising areas that are not functionally important. Interestingly, we found that 60% of frugivores in the Amazon basin showed range contractions of less than 5% between 2010 and 2050. This pattern is likely a result of uneven predicted deforestation across the Amazon. Instead, the conversion of forests to agricultural land-uses is predicted to be most severe in southern and eastern regions of the Amazon. Therefore, much of species ranges will occur outside areas at risk from land-use change and are therefore not predicted to suffer from large decreases occupancy of their ranges. This spatial insurance is likely to mean that even though deforestation is a prominent process shaping the prevailing land-cover across the Amazon, it is unlikely to directly lead to the extinction of frugivores in the region. However, the synergistic impact of climate and land-use change is yet to be fully

elucidated and so we cannot rule out much larger decreases in species ranges by 2050 (Brando et al. 2014, Laurance et al. in press.).

Opportunities and limitations

The framework presented here has provided an insight into how key functional traits are currently distributed and the hypothetical impacts of land-cover change on these distributions. However, there are a number of improvements that may be made to improve the accuracy of these predictions and the usefulness of this framework for conservation practitioners. Firstly, our model predicts the probability of species occurrence rather than abundance, yet some of the most pervasive and unpredictable impacts of land-use change are known to involve changes in species abundance (Mac Nally 2007). Given that abundance is a key factor influencing species responses to land-use change (Davies et al. 2000) and also the actual services that a species provides (Kremen 2005), it is essential that future studies include abundance as both a response and effect trait (Violle et al. 2007). In addition to acquiring a better understanding of how service provider communities change under different environmental conditions, we must begin to empirically test how these changes affect the actual functioning of ecosystems. For example, we must develop models that predict change in sapling recruitment given knowledge of seed disperser community structure. These models should not only include avian species but also other taxa important in the dispersal of seeds (Herrera 2002). Due to the importance of fruiting trees for frugivores, we have used forest cover as my single predictor of seed disperser occurrence. However, for different functional groups, other factors are likely to play a more important role in delimiting species realised ranges. For example, dispersal-limited understory insectivores, important in the top-down control of phytophagous insects (Sekercioglu 2006b) are typically ‘gap-averse’ (Lees and Peres 2009)

and so the degree of habitat fragmentation along with overall forest cover should be included. If these advances can be incorporated into the framework described here then we will be in a stronger position to predict the synergistic impacts of climate and land-use change on the resilience of forest ecosystems globally.

6 General discussion

Each of the data chapters presented here includes a section detailing the implications and limitations of that specific study and I do not simply repeat these findings here. Instead, I use this chapter to draw together this body of work, present the key take-home messages and discuss the broader implications of my findings. I also take time to present a few research avenues that should be explored before applying the lessons learnt here for the conservation of biodiversity. I began this thesis seeking to address whether landscapes dominated by human processes are useful in the protection of biodiversity and the ecosystem services it provides. By using birds, a well-studied model organism, and a mixture of degraded and fragmented study systems, in a trait-based framework, I have provided insights into the possible implications of current and future land-use change. Taken as a whole, this body of work confirms that the loss of avian diversity from landscapes dominated by human activities is non-random in regards to species ecological traits, corroborating an array of recent studies (Didham et al. 1998b, Devictor et al. 2008). I have also demonstrated that the persistence of important ecosystem service providers in degraded and fragmented landscapes depends not only on species morphology, but also on interspecific competition, the spatial configuration of suitable habitat, and climatic zone.

The non-random loss of species

The consistent loss of large frugivores from human-dominated landscapes throughout this thesis corroborates a wealth of studies highlighting this group's extinction-proneness (e.g. Kattan et al. 1994, Bennett and Owens 1997). In *Chapter 2*, frugivore communities in

smaller patches tended to have a lower body mass; in *Chapter 3*, communities of forest frugivores and insectivores tended to have smaller bodies relative to species in continuous forests. I therefore reaffirm the importance of the synergistic relationship between resource limitation in small patches and the pressure of hunting on large frugivores (Peres 2001). It is not surprising that degraded and fragmented landscapes, with varying distributions of fruiting trees (Donati et al. 2011), cannot support larger species as they tend to have higher metabolic rates and so a larger food base is required to support each individual in a population. Furthermore, large frugivores (e.g. guans, toucans, hornbills, etc.) are most at risk from hunting, as they are sought as game species, and access for hunters is typically improved in degraded and fragmented landscapes.

The results of this thesis suggest that an ability to disperse across unsuitable habitat is important in determining whether a species is capable of persisting in landscapes dominated by human activities. For example, in *Chapter 2*, tropical forest-dependent insectivores, known to be dispersal-limited (Moore et al. 2008, Lees and Peres 2009), were lost disproportionately as fragment size declined. Furthermore, in *Chapter 3*, I showed that insectivores and frugivores persisting in degraded habitats were more mobile than those that were sensitive to land-use change. Being able to move across disturbed landscapes has two benefits: firstly, dispersive species are able to forage and trap-line resources across large areas to meet their nutritional needs (Watson 2013). Secondly, mobile species are more likely to cross gaps in suitable habitat and recolonise areas after extinction events (Fahrig and Merriam 1994, Thomas 2000, Lees and Peres 2009), therefore preventing the disruption of important metapopulation processes in disturbed landscapes.

The biased loss of large species and dominance of dispersive species in post-disturbance habitats suggests that limitation of resources is important in structuring communities in degraded and fragmented landscapes. The finding of increased over-

dispersion of traits in *Chapter 4* is consistent with the view that competitive interactions play a stronger role in defining insectivore and frugivore communities in isolated forest patches than in continuous forests. This is also supportive of the view that resource limitation is important in fragmented landscapes, as species that are morphologically similar are likely to compete most strongly for resources where they are limited. However, this is contrary to the suggestion that lower species richness will relax competition for resources (Huston 1979, Grime 2001), or the idea that habitat disturbance destabilises the formation of communities such that structure is more random in disturbed relative to intact forests (Didham et al. 1998a, Ewers and Didham 2006). Yet, the results from this thesis strongly imply that disturbance leads to limitation of resources and consequent strengthening of competitive interactions between species in feeding guilds. These interactions are likely to be mediated by species' underlying resource requirements and sensitivity to other human pressures (i.e. hunting).

The most pervasive impacts of land-use change on forest dependent birds are likely to be constrained to the tropics. In *Chapter 2*, I found that a decrease in patch size led to important changes in the structure of tropical communities but that changes to the structure of temperate bird communities were minimal. Tropical species are likely to be more sensitive to land-use change as they are typically patchily distributed (Meyer et al. 2008), occurring at low population densities, with large home ranges, and slow reproduction rates (Tobias et al. 2013a). Furthermore, their adaptation to relatively stable climates and microhabitats may lead to a narrower physiological tolerance of disturbed habitats (Janzen 1967), that are typically warmer, brighter and less humid than intact forests (Laurance et al. 2011). Similarly, I expect that competitive interactions will be less important in structuring temperate relative to tropical communities, as these experience less niche saturation and fewer cases of year-round territoriality (Jankowski et al. 2012).

Implications for ecosystem functioning and resilience

At face value, the results from this thesis are opposing in their prediction of ecosystem resilience in the face of land-use change. On one hand, the collapse of frugivore communities in pastures and mechanised agricultural habitats suggests that the provisioning of avian seed dispersal is greatly impacted by changing land cover. Moreover, the non-random loss of large frugivores is likely to have particularly negative implications for large-seeded trees (Galetti et al. 2013), affecting the composition and functioning of post-disturbance tree communities. Similarly, even though forest insectivore species were replaced by open habitat insectivores following conversion to pastures and mechanised agricultural habitats (*Chapter 3*), it is not certain whether invading species are capable of providing the same control of insect herbivores as those species that become locally extinct following land-use change.

On the other hand, in a modelling exercise the maximum gap width of the largest species remained high when forest cover was above 50% at a 2500 km² spatial resolution, suggesting some resilience in the dispersal of large seeds at this relatively coarse scale. Furthermore, the biased loss of species from the densest part of trait space could suggest that the impact of fragmentation of fragmentation on the provisioning of ecosystem services is minimal as it implies a degree of complementarity between functional traits. However, ecosystem services could still be disrupted by the ‘thinning out’ of species with niches clustered around important optima. For frugivorous birds, this could lead to the reduced dispersal of seeds in certain size classes while in insectivores, certain groups of insects may be partially or completely released from top-down control. However, even if the loss of complementary species does not impact the provisioning of ecosystem services by birds, by reducing the redundancy of the

remaining communities, ecosystem function could be at greater risk from other synergistic processes, such as climate change (Mokany et al. 2014).

Therefore, the results from this thesis overwhelmingly support the view that land-use change is associated with the disassembly of communities and the loss of ecosystem services from degraded and fragmented landscapes. It could also be argued that the loss of bird functional diversity following land-use change is not important, as there is no demand for certain services in those landscapes. For example, the loss of frugivores from agricultural areas may not be detrimental to the functioning of the ecosystem, as the range of fruit sizes that need to be dispersed is likely to be reduced. Yet, despite this, it is clear that maintaining the diversity of ecosystem service providers is of great economic importance, and of significance to the structural resilience of forests and reforestation efforts. For example, the loss of top-down control of insect herbivores, if it were to occur, is likely to have financial implications for plantation holders (e.g. Takekawa and Garton 1984, Greenberg et al. 2000) while avian seed dispersers are of great importance to the regeneration of native trees on land previously exploited by humans (Robinson and Handel 1993).

Management implications

Conservation in the Tropics

With many global forest commodities (such as palm oil, soya and beef) originating from tropical regions, it is clear that this is where many conservation efforts need to be focused. This is particularly pertinent given that studies, such as this, have identified species in the tropics to be more sensitive to land-use change. However, conservation efforts are only likely to be effective if they are developed and implemented by a wide-range of stakeholders (Senior et al. in press). Species richness and endemism are

commonly factors considered important in prioritising forests, yet care must be taken to ensure that we do not overlook the maintenance of ecosystem functioning.

Forests don't have to be pristine to maintain biodiversity

The similarity of avian community structure in disturbed relative to undisturbed primary forests in *Chapter 3* suggests that ecosystem functioning is maintained in these habitats. This corroborates recent studies that highlight the usefulness of selectively logged and burnt forests in maintaining biodiversity (Putz et al. 2012, Moura et al. 2013, Edwards et al. 2014). Despite similar levels of functional diversity in disturbed and undisturbed primary forests, it is essential that we fully understand the impacts of logging and burning on individual species, particularly those that are rare and sensitive to disturbance, to ensure that the entire pool of species is represented in the focal landscape or within a wider management unit. Such species are often highly dependent on old-growth forest, so the only way to maintain viable populations involves the protection of pristine forests. Yet, the protection of such forests involves the forgoing of economic profitability associated with timber extraction or other human activities. One proposed strategy that maintains suitable areas of old growth forest is known as the 'retention approach', where keystone trees or blocks of forest remain protected within the matrix of disturbed forest in logging concessions. This means that logging can still provide economic gain for landowners, while patches of old growth forest within the landscape provide suitable habitat for species sensitive to land use change.

Patches are islands but they still offer hope in a sea of agriculture

While forest patches, isolated in agricultural matrices offer fewer opportunities for forest species, particularly those that are dispersal-limited (Lees and Peres 2008a), they appear to be useful in maintaining biodiversity for some functional guilds (*Chapter 2*) relative to

pastures and areas of mechanised agriculture. Furthermore, the biased loss of closely related individuals with a decline in patch size, suggests that the loss of ecosystem functioning may not be as bad as potentially feared. I therefore suggest that forest patches in agricultural landscapes be protected, not only for their intrinsic biodiversity value but also because of their economic value in supporting key ecosystem service providers.

General caveats

Aside from a poor understanding of whether species associated with disturbed or open habitats will replace forest species in terms of their ecological roles, there are several other caveats associated with the approaches used in this thesis. Many of these are not specific to this thesis but appear in many other studies that seek to identify the impacts of land-use change on species diversity and ecosystem functioning. Firstly, the choice of traits in studies of functional diversity is important (Petchey and Gaston 2006) as they may respond differently to changes in land use change. For example, there could be a decrease in diversity of one trait axis but an increase in diversity in another, which could change how we view the overall impact of land use change. In this thesis, I have largely overcome this issue by examining changes in functional diversity across four different trait axes. Despite the importance of these traits in determining species occurrence and role within ecosystems being fairly well resolved, it is still unclear the exact role that each trait plays in defining specific ecosystem services. For example, gape size is important in determining the maximum fruit size that can be consumed by avian frugivores, but we are unsure how a change in this trait affects the distribution of fruit sizes that are consumed by species. Specifically, a species may have a large gape width but may only consume fruits with diameters many times smaller. Similarly, another caveat associated with this thesis is my focus on species occurrence despite the fact that some of the most

pervasive and unpredictable impacts of land use change are manifested in terms of changes in species abundance (Mac Nally 2007). This adds extra complexity to our interpretation of the impacts of land use change on ecosystem functioning because it is not only the overall composition of ecosystem service providers that matters to overall functioning, but also how many individuals there are of each species to perform their role within the ecosystem. For example, there may be a dramatic decline in the functional diversity of frugivores in terms of the community composition in degraded landscapes, but the species that remain may be released from competitive interactions and could occur at much higher abundances. Therefore, if these remaining species, occurring at higher abundances, are relatively generalist in terms of their consumption of fruit, then there may be little or no impact on the dispersal of seeds.

Avenues for future research

To overcome the caveats of this thesis and further our knowledge of how we can best preserve ecosystem services provided by birds in landscapes dominated by human activities, there are two main research avenues that should be explored. First, we should explore how the loss of avian functional diversity translates directly to a loss of ecosystem functioning. For example, using existing data, we should test whether the distribution of traits in frugivore communities affects the richness and number of saplings growing in degraded landscapes in the Santarém and Paragominas landscapes. Based on the findings from this analysis, future fieldwork should explore the relationship further by (1) estimating the abundance of seed dispersers across the landscape, and (2) identifying the specific seeds that individual species of birds are responsible for dispersing. Thus, this approach will allow us to gain a better understanding of the direct impacts of land-use change on the functioning of forest ecosystems. Specifically, it will provide an insight into the resilience of mutualistic and top-down relationships. For

example, by constructing detailed networks, we will be able to state with reasonable certainty the redundancy among frugivores in providing seed dispersal services. Similarly, this approach will enable us to determine whether the protection of the most functionally and phylogenetically unique species, two criteria often proposed for use in conservation prioritisation, will be efficient in the protection of ecosystem functioning.

Following on from these studies we should assess the extent to which different management strategies maintain species identified as important in the provisioning of ecosystem services. Specifically, more work needs to be undertaken to identify keystone structures in degraded landscapes. While the importance of isolated fruiting trees in agricultural landscapes for frugivores is fairly well resolved (Slocum and Horvitz 2000, Carriere et al. 2002, Guevara et al. 2004), their importance for other feeding guilds is still little known (but see Fischer and Lindenmayer 2002, Manning et al. 2006). Specifically, such trees may improve connectivity (Harvey et al. 2006) between larger tracts of forest or could supply an abundance of prey for insectivores. Furthermore, given that I have found that competition plays an important role in defining community structure in human-altered landscapes, future studies should identify the behavioural mechanisms by which species could exclude one another across gradients of land-use change.

7 General conclusion

In this thesis I set out to explore how land-use change acts to structure avian communities. By showing that the loss of species in degraded and fragmented landscapes is strongly non-random I have highlighted the importance of moving beyond the use of species richness when prioritising areas for conservation. Insectivores and large frugivores appear most sensitive to land-use change while species located in the densest parts of trait space are also threatened by a decline in forest patch size. These biased responses are likely to have major implications for the functioning of ecosystems and it is of utmost importance that we determine the structure of landscapes that promotes ecosystem functioning while meeting the demands of a growing global economy.

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Appendix A: Chapter 2 Supplementary Methods and Results

Contents

Expanded materials and methods: Fig. A1.

Expanded results: Tables A1–A7.

Sensitivity analyses: Fig. A2, Tables A8–A13.

Expanded materials and methods

Dataset structure

Location, size and species richness data for all forest fragments included in this study are provided in Supplementary data 1, along with information on data sources. The structure of our dataset raises issues of non-independence as different studies contain different numbers of fragments (1–56). To explore the underlying patterns, we modelled changes in species richness within studies (Fig. A1). This analysis revealed some consistent differences across studies, presumably relating to biogeography or habitat type. However, the basic positive relationship between fragment size and species richness was retained within studies.

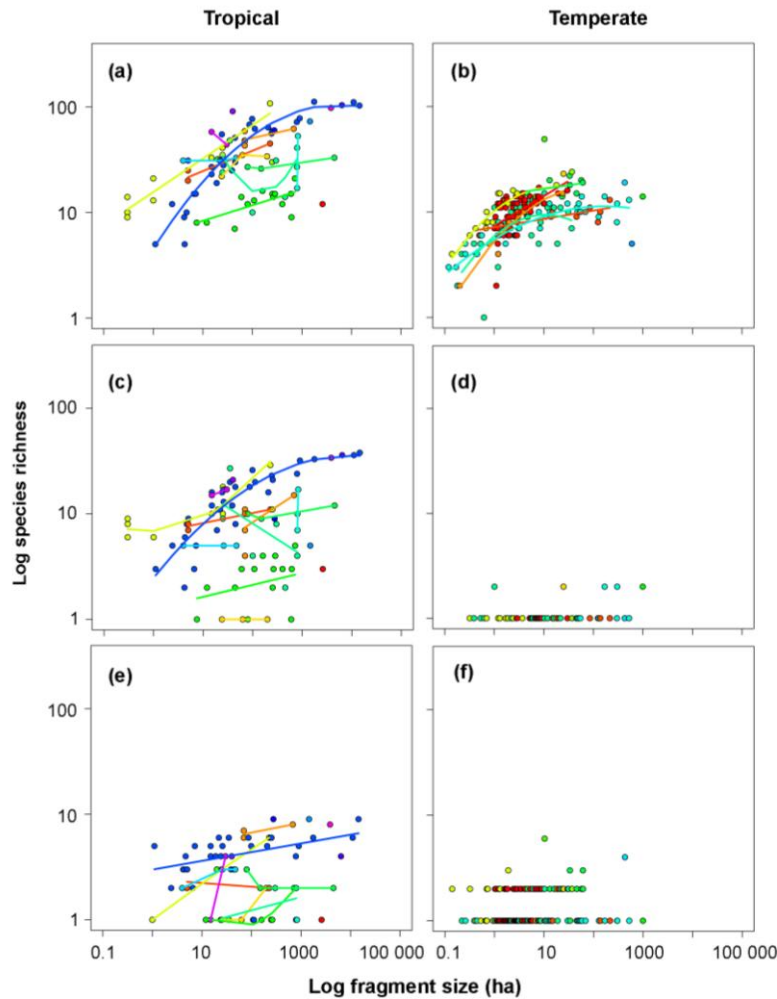


FIGURE A1. The effect of fragment size on species richness of forest-dependent insectivores (a,b), frugivores (c,d) and granivores (e,f). Each study is colour coded; lines represent linear model fit for each region.

Justification of avian traits

The most widely acknowledged threats to species in fragmented habitats are (1) breakdown of metacommunity processes, (2) reduction of area for viable populations, and (3) hunting. The first point is typically related to reduced connectivity and gene flow among populations, and is thus associated with dispersal limitation. The second and third points are exacerbated by large body size, as this is associated with increased home range size, decreased population density, slower reproductive output and higher hunting pressure. To explore the role of intrinsic differences in dispersal ability and size, we

quantified a number of life history and ecological traits in birds: feeding guild, the openness of preferred habitat, migration and body mass (see Supplementary data 2). All of these traits are associated with dispersal ability, while the last is also a direct measure of size.

Feeding guild: Among tropical birds, insectivores are generally weaker dispersers than non-insectivores (Levey and Stiles 1992; Moore et al. 2008) and their ranges may be constrained by narrow geographic barriers (Salisbury et al. 2012). This pattern is thought to be explained by the relatively stable availability of arthropods over space and time, which favors a more sedentary life-style (Levey and Stiles 1992). In contrast, the greater dispersal of frugivores and nectarivores, is thought to be due to the more patchy and seasonal distribution of plant resources (e.g. Holbrook et al. 2002).

Habitat openness: Within the tropics, species occurring in more structurally dense habitats (e.g. forest) are expected to be less dispersive than more open-habitat specialists (e.g. grassland) (Weir et al. 2009). Empirical studies show that forest species may fail to cross even narrow breaks in habitat easily crossed by open country species (Moore et al. 2008). This may arise both because of the behavioral inhibition of forest species to cross gaps (Harris and Reed 2002) and because the adaptations required for movement amongst dense vegetation (e.g. short and rounded wings) are not favourable for long distance flight (Stratford and Robinson 2005). In addition, dense habitats tend to be relatively dark and climatically stable, suggesting that species adapted to such habitats may avoid the strong light and fluctuating environmental conditions associated with the disturbed matrix between forest fragments (Stratford and Robinson 2005).

Migration: Species undertaking long-distance migrations are expected to have greater dispersal capacities than those moving only short distances, or that are year-round residents (Levey and Stiles 1992). While migratory species are adapted for sustained flight, sedentary species often lack the behavioral and morphological characteristics required for even relatively short distance movements (Harris and Reed 2002).

Body mass: Dispersal ability is generally positively correlated with body mass (Paradis et al. 1998), suggesting that large-bodied organisms may be better equipped to cope with habitat fragmentation. However, large species typically require larger home ranges to meet their nutritional demands, and thus have lower population density. In addition, they are more often the target of hunters. Thus, body size may be related in contrasting ways to fragmentation, depending on whether the positive effects of dispersal are outweighed by the negative effects of reduced area or increased hunting pressure.

Analytical approach

To control for nonindependence of fragments within studies, we included study region as an effect in our models. We used MCMCglmm (Hadfield 2010) to assess the impact of fragment size at three scales: (i) with all guilds grouped, (ii) across feeding guilds, and (iii) within feeding guilds. Model convergence was determined by visually inspecting the trend output and assessing whether there was any autocorrelation between consecutive chain values. The number of iterations for all analyses was 1 300 000, with a burn in of 100 000 and samples taken from the posterior distribution every 12 iterations. Using this approach, there were no identifiable trends in the chain output and autocorrelation between consecutive chain values was small. We set uninformative priors, allowing model fit to be strongly influenced by data.

Results are presented using model deviance information criterion (DIC) and P values. The change in DIC was used as a metric of model support, balancing the number of effective parameters with the overall fit of the model. An overall increase of 2 on this scale when a parameter is removed is an indication that the term makes a significant contribution to the model fit. P values for between-category analyses represent the standardised (scaled to the total number of iterations) number of times one category is higher or lower than the other.

Expanded results

TABLE A1. The effect of fragment size on species richness of tropical and temperate bird communities. The main effect in all cases was fragment size. E.S.S. = effective sample size; Δ_{DIC} = change in model DIC value when the main effect is removed from the model; the quadratic term (Fragment size²) tests for non-linearity in relationships.

Focal guild	Explanatory variable	Estimate (Conf. interval)	E.S.S	<i>P</i>	Δ_{DIC}	Full model random effects		
						Variance estimate		
						Full model DIC	Study (Conf. interval)	Residual (Conf. interval)
Tropical insectivores	Fragment size	0.430 (0.317 to 0.551)	10000	<0.001	91.48	-53.41	0.090 (0.032 to 0.168)	0.027 (0.018 to 0.036)
	Fragment size ²	-0.046 (-0.075 to -0.017)	10000	0.001	9.49			
Tropical frugivores	Fragment size	0.198 (0.326 to 0.766)	10000	<0.001	48.94	-8.708	0.168 (0.058 to 0.303)	0.043 (0.029 to 0.059)
Tropical granivores	Fragment size	0.097 (0.049 to 0.149)	9809	<0.001	15.11	-31.77	0.083 (0.026 to 0.154)	0.030 (0.019 to 0.042)
Temperate insectivores	Fragment size	0.348 (0.290 to 0.405)	10000	<0.001	131.47	-188.69	0.041 (0.006 to 0.090)	0.018 (0.014 to 0.022)
	Fragment size ²	-0.102 (-0.133 to -0.070)	10000	<0.001	2.31			

TABLE A2. The effect of habitat patch size and latitudinal zone on the proportion of forest-dependent species persisting in forest fragments. E.S.S. = effective sample size; Δ_{DIC} = change in model DIC value when the main effect is removed from the model.

Focal guild	Main effects	Estimate (Confidence interval)	E.S.S	<i>P</i>	Δ_{DIC}	Full model random effects variance estimate		
						Full model DIC	Study (Confidence interval)	Residual (Confidence interval)
Forest species	Fragment size	0.196 (0.066 to 0.321)	2266	0.002	31.21			
	Latitudinal zone (Tropics)	−0.150 (−0.678 to 0.376)	6377	0.573	−0.56	15 512.41	0.404 (0.191 to 0.652)	0.035 (0.01 to 0.064)
	Interaction	0.085 (−0.065 to 0.240)	2700	0.273	−0.19			

TABLE A3. The effect of fragment size on the component of tropical and temperate bird communities made up of key guilds, with species richness of guilds calculated as a proportion of overall species richness in the community. The main effect in all cases was fragment size. All fragments <100 000 ha were included in the analysis. Conf. interval = confidence interval; E.S.S = effective sample size; Δ_{DIC} = change in model DIC value when main effect is removed; the quadratic term (Fragment size²) tests for non-linearity in relationships.

Focal guild	Explanatory variable	Full model random effects						
		Estimate (Conf. interval)	E.S.S	<i>P</i>	Δ_{DIC}	Full model DIC	Variance estimate	
							Study (Conf. interval)	Residual (Conf. interval)
Tropical insectivores	Fragment size	0.457 (0.228 to 0.683)	6649	<0.001	19.80	10757.40	0.102 (0.023 to 0.206)	0.031 (0.002 to 0.061)
	Fragment size ²	−0.059 (−0.113 to −0.011)	5243	0.024	2.10			
Tropical frugivores	Fragment size	0.033 (−0.043 to 0.105)	5885	0.391	−1.30	6453.52	0.290 (0.096 to 0.546)	0.005 (0.0001 to 0.015)
Tropical granivores	Fragment size	−0.166 (−0.296 to −0.039)	3236	0.011	5.04	2381.88	0.178 (0.014 to 0.407)	0.009 (0.0001 to 0.031)
Temperate insectivores	Fragment size	0.080 (−0.038 to 0.197)	2268	0.179	0.08	5065.14	0.462 (0.109 to 0.969)	0.004 (0.0001 to 0.014)
Temperate frugivores	Fragment size	−0.184 (−0.611 to 0.228)	2095	0.373	−0.69	557.23	0.290 (0.0001 to 1.011)	0.0179 (0.0001 to 0.071)
Temperate granivores	Fragment size	−0.196 (−0.416 to 0.022)	1212	0.082	1.30	1549.62	0.039 (0.0001 to 0.023)	0.007 (0.0001 to 0.022)

TABLE A4. Differences between communities in the proportion of forest species belonging to focal foraging guilds, where communities in small fragments (<100 ha) are compared with communities in midsize fragments (100–1000 ha) (reference group). E.S.S = effective sample size; Δ_{DIC} = change in model DIC value when the main effect is removed from the model.

Focal guild	Main effects	Estimate	E.S.S	<i>P</i>	Δ_{DIC}	Full model DIC	Full model random effects	
							Variance estimate	
							Study	Residual
Tropical insectivores	Fragment size (<100 ha)	−0.234 (−0.409 to −0.06)	73 173	0.011	5.82	17478.56	0.042 (0.001 to 0.090)	0.052 (0.017 to 0.091)
Tropical frugivores		−0.014 (−0.190 to 0.164)	5603	0.880	−2.20	9796.96	0.208 (0.084 to 0.358)	0.004 (0.0001 to 0.013)
Tropical granivores		0.286 (−0.017 to 0.599)	2958	0.065	1.26	3327.02	0.097 (0.011 to 0.207)	0.007 (0.0001 to 0.026)
Temperate insectivores		0.162 (−0.160 to 0.478)	3168	0.317	−1.81	5845.85	0.315 (0.107 to 0.589)	0.004 (0.0002 to 0.013)
Temperate frugivores		−0.047 (−0.778 to 0.687)	1994	0.887	−2.72	649.28	0.182 (0.0002 to 0.607)	0.018 (0.0001 to 0.072)
Temperate granivores		0.447 (−0.180 to 1.083)	1057	0.162	−0.47	1864.46	0.029 (0.0001 to 0.115)	0.001 (0.0001 to 0.022)

TABLE A5. Differences between patches in the proportion of forest species belonging to feeding guilds, where patches are binned into three size categories: <100 ha, 100–1000 ha and >1000 ha. Main effect = the size category compared against fragments >1000 ha; Conf. interval = confidence interval; E.S.S = effective sample size; Δ_{DIC} = change in model DIC value when main effect is removed.

Focal guild	Main effects (fragment size)	Full model random effects						
		Estimate (Conf. interval)	E.S.S	P	Δ_{DIC}	Variance estimate		
						Full model DIC	Study (Conf. interval)	Residual (Conf. interval)
Tropical insectivores	<100 ha	–0.392 (–0.624 to –0.163)	7463	0.002	5	11 490.17	0.073	0.046
	100–1000 ha	–0.116 (0.365 to 0.122)	8598	0.341			(0.001 to 0.145)	(0.009 to 0.087)
Tropical frugivores	<100 ha	–0.102 (–0.336 to 0.137)	9173	0.386	–2.75	6278.86	0.437	0.005
	100–1000 ha	0.067 (–0.326 to 0.183)	9448	0.605			(0.131 to 0.823)	(0.0001 to 0.018)
Tropical granivores	<100 ha	0.380 (0.009 to 0.747)	5562	0.040	2.19	2366.59	0.14	0.009
	100–1000 ha	0.053 (–0.353 to 0.477)	5022	0.816			(0.001 to 0.325)	(0.0001 to 0.034)
Temperate insectivores	<100 ha	0.066 (–0.432 to 0.568)	5072	0.784	–1.69	5942.73	0.290	0.004
	100–1000 ha	–0.1 (–0.654 to 0.449)	3776	0.710			(0.102 to 0.526)	(0.0001 to 0.01)
Temperate frugivores	<100 ha	–0.058 (–0.799 to 0.692)	2083	0.878	–2.69	649.39	0.182	0.019
	100–1000 ha	–0.285 (–1.356 to 0.745)	2595	0.579			(0.0001 to 0.605)	(0.0001 to 0.075)
Temperate granivores	<100 ha	0.464 (–0.159 to 1.089)	1046	0.139	0.281	1877.93	0.031	0.007
	100–1000 ha	0.645 (–0.06 to 1.325)	1178	0.064			(0.0001 to 0.122)	(0.0001 to 0.023)

TABLE A6. Differences between communities in the average mass of closed forest species belonging to each of three focal guilds, where communities in small fragments (<100 ha) are compared to communities in midsize fragments (100–1000 ha) (reference group). E.S.S is the effective sample size; Δ_{DIC} is the change in model DIC value when the main effect is removed from the model.

Focal guild	Main effects	Estimate	E.S.S	<i>P</i>	Δ_{DIC}	Full model DIC	Full model random effects – variance estimate	
							Study	Residual
Tropical insectivores	Fragment size (<100 ha)	–0.002 (–0.048 to 0.044)	100 000	0.932	–5.02	–198.82	0.006 (0.002 to 0.011)	0.008 (0.006 to 0.010)
Tropical frugivores		–0.125 (–0.285 to 0.026)	100 000	0.116	1.83	–4.85	0.015 (0.003 to 0.035)	0.054 (0.042 to 0.066)
Tropical granivores		–0.066 (–0.225 to 0.097)	100 000	0.420	–0.94	73.66	0.054 (0.012 to 0.107)	0.097 (0.068 to 0.129)
Temperate insectivores		–0.0001 (–0.008 to 0.008)	100 000	0.986	0.65	–454.81	0.017 (0.005 to 0.034)	0.0001 (0.003 to 0.007)
Temperate frugivores		–0.052 (–0.181 to 0.083)	100 000	0.432	–5.32	0.34	0.183 (0.086 to 0.295)	0.042 (0.027 to 0.059)
Temperate granivores		0.002 (–0.054 to 0.056)	100 000	0.946	–2.21	–379.30	0.114 (0.045 to 0.204)	0.004 (0.003 to 0.005)

TABLE A7. Differences in the average mass of forest-dependent species, comparing among three size categories of habitat patch: <100, 100–1000, and >1000 ha. Main effect = the size category compared against fragments >1000 ha; Conf. interval = confidence interval; E.S.S = effective sample size; Δ_{DIC} = change in model DIC value when main effect is removed.

Focal guild	Main effects (fragment size)	Estimate (Conf. interval)	E.S.S	<i>P</i>	Δ_{DIC}	Full model DIC	Full model random effects variance estimate	
							Study (Conf. interval)	Residual (Conf. interval)
Tropical insectivores	<100 ha	0.050	10 000	0.101	-4.4	-201.54	0.007 (0.002 to 0.012)	0.008 (0.005 to 0.001)
		(-0.117 to 0.009)						
	100–1000 ha	-0.052 (-0.117 to -0.110)	10 000	0.123				
Tropical frugivores	<100 ha	-0.264	10 000	0.009	0.72	70.83	0.051 (0.011 to 0.1)	0.096 (0.067 to 0.126)
		(-0.452 to -0.07)						
	100–1000 ha	-0.199 (-0.415 to 0.011)	10 000	0.070				
Tropical granivores	<100 ha	-0.069	10 000	0.488	-3.19	-1.34	0.182 (0.087 to 0.294)	0.042 (0.027 to 0.058)
		(-0.242 to 0.108)						
	100–1000 ha	-0.008 (-0.208 to 0.183)	10 000	0.943				
Temperate insectivores	<100 ha	-0.162	9673	0.0374	1.94	-170.97	0.020 (0.003 to 0.036)	0.021 (0.016 to 0.026)
		(-0.309 to -0.02)						
	100–1000 ha	-0.038 (-0.208 to 0.124)	10 000	0.6494				
Temperate frugivores	<100 ha	0.130	10 000	0.110	4.22	-169.08	0.020 (0.006 to 0.041)	0.004 (0.002 to 0.005)
		(-0.037 to 0.293)						
	100–1000 ha	0.067 (-0.11 to 0.23)	10 000	0.415				
Granivores– temperate	<100 ha	-0.090	9933	0.523	-2.05	-390.34	0.107 (0.044 to 0.189)	0.004 (0.003 to 0.005)
		(-0.385 to 0.210)						
	100–1000 ha	-0.092 (-0.397 to -0.202)	9969	0.527				

Sensitivity analyses

For the categorical analysis, we grouped tropical communities in fragments greater than 1000 ha with those in continuous forests (there were no temperate fragments between 1000 and 100 000 ha). To test whether community structure differed between these large fragments and continuous forests we re-ran Analysis 5 (for tropical communities) but compared the proportion of each guild of forest-dependent species in large fragments to those in continuous forests. This revealed no significant differences in the proportion of major guilds (Fig. A2; Table A8) or their average mass (Fig. A2; Table A9).

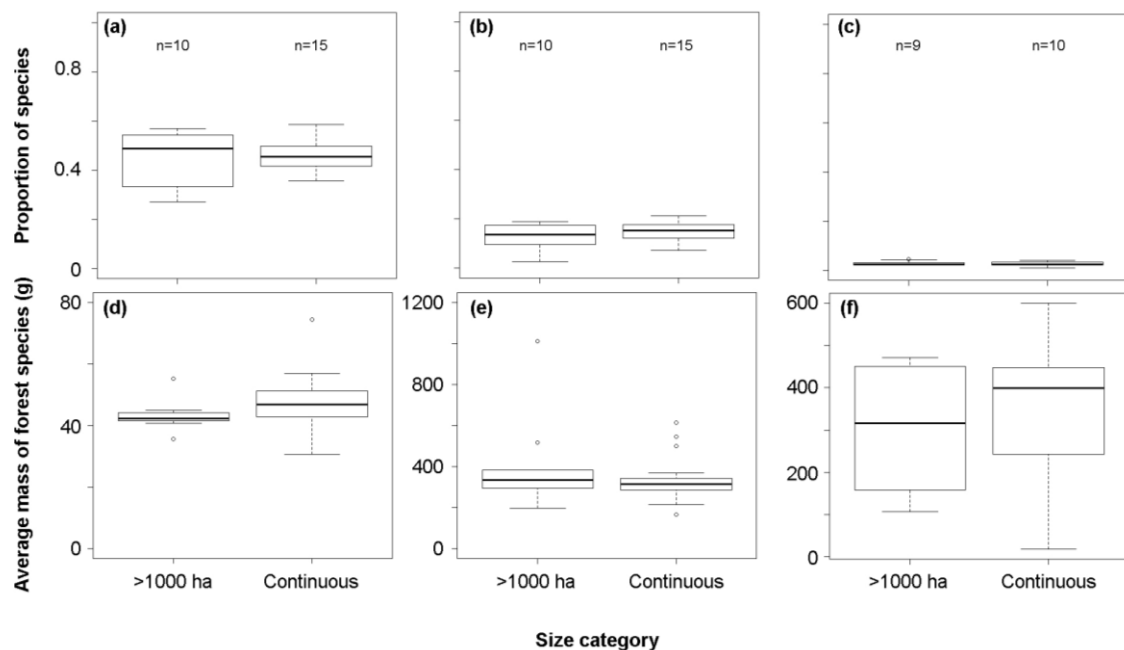


FIGURE A2. The effect of fragmentation on the proportion of species (a–c) and average body mass (d–f) grouped by feeding guild in large tropical forest fragments ($n = 10$) and continuous forest ($n = 15$). Whiskers shown on the Tukey boxplots include points within the 1.5x interquartile range. Data are shown for insectivore (a,d), frugivores, (b,e) and granivores (c,f). The sample sizes represent the number of communities where any of the focal guild were present.

TABLE A8. Differences between communities in the proportion of forest species belonging to focal foraging guilds, where fragments (>1000 ha) are compared with communities in continuous forest (reference group). E.S.S = effective sample size; Δ_{DIC} = change in model DIC value when the main effect is removed from the model.

Focal guild	Main effects	Estimate	E.S.S	<i>P</i>	Δ_{DIC}	Full model DIC	Full model random effects	
							Variance estimate	
							Study	Residual
Insectivores		−0.140 (−0.399 to 0.125)	53 583	0.282	−0.50	8391.57	0.096 (0.0003 to 0.190)	0.018 (0.0001 to 0.073)
Frugivores	Fragment size (>1000 ha)	−0.148 (−0.469 to 0.170)	37 615	0.350	−0.60	4873.08	0.159 (0.0002 to 0.333)	0.018 (0.0001 to 0.081)
Granivores		0.126 (−0.245 to 0.501)	10 253	0.496	−1.50	1454.05	0.018 (0.0001 to 0.069)	0.013 (−0.0001 to 0.049)

TABLE A9. Differences between communities in the average mass of closed forest species belonging to each of three focal guilds, where fragments (>1000 ha) are compared to communities in continuous forest (reference group). E.S.S = effective sample size; Δ_{DIC} = change in model DIC value when the main effect is removed from the model.

Focal guild	Main effects	Estimate	E.S.S	<i>P</i>	Δ_{DIC}	Full model DIC	Full model random effects	
							Variance estimate	
							Study	Residual
Insectivores	Fragment size (>1000 ha)	−0.011 (−0.065 to 0.043)	10 426	0.698	−5.90	−97.99	0.007 (0.001 to 0.013)	0.001 (0.0001 to 0.004)
Frugivores		0.001 (−0.117 to 0.133)	8023	0.956	−58.42	−62.82	0.045 (0.0004 to 0.025)	0.006 (0.0004 to 0.025)
Granivores		−0.010 (−0.136 to 0.115)	10 000	0.868	−4.61	−63.61	0.140 (0.055 to 0.253)	0.004 (0.0003 to 0.011)

TABLE A10. The effect of a decline in fragment size and latitudinal zone upon the proportion of forest-dependent species after migrants have been removed. E.S.S = effective sample size; Δ_{DIC} = change in model DIC value when the main effect is removed from the model.

Focal guild	Main effects	Estimate (Confidence interval)	E.S.S	<i>P</i>	Δ_{DIC}	Full model DIC	Full model random effects – variance estimate	
							Study (Confidence interval)	Residual (Confidence interval)
Forest species	Fragment size	0.256 (0.112 to 0.396)	10 056	<0.001	39.97			
	Latitudinal zone (Tropics)	0.344 (−0.159 to 0.858)	38 898	0.184	−2.74	13767.95	0.351 (0.168 to 0.575)	0.024 (0.0004 to 0.047)
	Interaction	0.013 (−0.143 to 0.174)	11 977	0.870	−1.66			

TABLE A11. The effect of a decline in fragment size on the proportion of three focal guilds in temperate communities after migrants have been removed. E.S.S = effective sample size; Δ_{DIC} = change in model DIC value when the main effect is removed from the model.

Focal guild	Main effects	Estimate (Confidence interval)	E.S.S	<i>P</i>	Δ_{DIC}	Full model DIC	Full model random effects Variance estimate	
							Study (Confidence interval)	Residual (Confidence interval)
Temperate insectivores		0.124 (−0.012 to 0.267)	1882	0.086	1.94	3105.79	0.261 (0.040 to 0.593)	0.005 (0.0001 to 0.017)
Temperate frugivores	Fragment size	−0.347 (−0.823 to 0.129)	1306	0.150	−0.32	440.13	0.341 (0.0001 to 1.200)	0.020 (0.0001 to 0.081)
Temperate granivores		−0.196 (−0.449 to 0.063)	1018	0.133	−1.07	1326.79	0.243 (0.0002 to 0.693)	0.008 (0.0002 to 0.026)

TABLE A12. Differences between communities in the proportion of forest species belonging to three focal foraging guilds, where fragments are binned into three size categories and migrants removed; <100 ha, 100–1000 ha and >1000 ha (This is used as the reference group and is a pool of large fragments >1000 ha and continuous forest). E.S.S = effective sample size; Δ_{DIC} = change in model DIC value when the main effect is removed from the model.

Focal guild	Main effects	Estimate	E.S.S	P	Δ_{DIC}	Full model DIC	Full model random effects	
							Variance estimate	
Temperate insectivores	Fragment size (<100 ha)	0.197 (−0.275 to 0.681)	5994	0.404	−2.51	3461.94	0.180 (0.035 to 0.374)	0.005 (0.0002 to 0.016)
	Fragment size (100–1000 ha)	0.128 (−0.434 to 0.680)	4230	0.647				
Temperate frugivores	Fragment size (<100 ha)	0.337 (−0.815 to 1.550)	2740	0.560	−2.72	494.58	0.309 (0.0002 to 0.977)	0.022 (0.0001 to 0.087)
	Fragment size (100–1000 ha)	0.003 (−1.381 to 1.428)	2104	0.985				
Temperate granivores	Fragment size (<100 ha)	−0.387 (−0.909 to 0.150)	1996	0.140	−1.1	1552.05	0.118 (0.0001 to 0.343)	0.008 (0.0002 to 0.026)
	Fragment size (100–1000 ha)	−0.891 (−1.676 to 0.071)	1202	0.060				

TABLE A13. Differences between communities in the average mass of forest species belonging to three focal foraging guilds, where fragments are binned into three size categories and migrants removed; <100 ha, 100–1000 ha and >1000 ha (This is used as the reference group and is a pool of large fragments >1000 ha and continuous forest). E.S.S = effective sample size; Δ_{DIC} = change in model DIC value when the main effect is removed from the model.

Focal guild	Main effects	Estimate	E.S.S	<i>P</i>	Δ_{DIC}	Full model random effects		
						Variance estimate		
						Full model DIC	Study	Residual
Temperate insectivores	Fragment size (<100 ha)	−0.267 (−0.457 to −0.082)	913	0.016	3.30	−4.45	0.016 (0.003 to 0.033)	0.054 (0.043 to 0.064)
	Fragment size (100–1000 ha)	−0.141 (−0.361 to 0.104)	1000	0.226				
Temperate frugivores	Fragment size (<100 ha)	0.115 (−0.027 to 0.275)	1000	0.126	0.75	−454.88	0.017 (0.006 to 0.034)	0.0001 (0.0001 to 0.0001)
	Fragment size (100–1000 ha)	0.115 (−0.023 to 0.276)	1000	0.126				
Temperate granivores	Fragment size (<100 ha)	−0.134 (−0.423 to 0.171)	1000	0.400	−2.80	−380.98	0.124 (0.050 to 0.221)	0.004 (0.003 to 0.005)
	Fragment size (100–1000 ha)	−0.133 (−0.415 to 0.166)	1000	0.410				

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**Community information: source, location, size and species richness
for all forest fragments**

Study ¹	Community	Latitude ²	Longitude ²	Size (ha) ³	Species richness
1	1A	64.57	25.48	Continuous	61
2	2A	−22.2	−48	40	235
3	3A	−22.45	−48.09	1451	200
4	4A	−8.29	−60.57	Continuous	435
5	5A	17.325	−92.445	4500	100
5	5B	17.325	−92.445	150	83
5	5C	17.325	−92.445	80	80
6	6A	41	12.6	302.48	20
6	6B	41	12.6	168.91	21
6	6C	41	12.6	68.84	18
6	6D	41	12.6	58.36	22
6	6E	41	12.6	54.69	16
6	6F	41	12.6	53.59	16
6	6G	41	12.6	50.7	20
6	6H	41	12.6	47.83	20
6	6I	41	12.6	46.71	20
6	6J	41	12.6	29.99	19
6	6K	41	12.6	13.99	15
6	6L	41	12.6	13.27	16
6	6M	41	12.6	13.02	18
6	6N	41	12.6	7.96	17
6	6O	41	12.6	3.18	14
6	6P	41	12.6	2.65	12
6	6Q	41	12.6	2.58	14
6	6R	41	12.6	1.69	11
6	6S	41	12.6	1.5	12
6	6T	41	12.6	0.26	9
6	6U	41	12.6	525.64	25
6	6V	41	12.6	298.72	17

6	6W	41	12.6	128.2	17
6	6X	41	12.6	10.97	14
6	6Y	41	12.6	8.4	12
6	6Z	41	12.6	1.7	12
6	6AA	41	12.6	0.56	8
6	6AB	41	12.6	0.45	9
6	6AC	41	12.6	0.43	7
6	6AD	41	12.6	0.26	6
6	6AE	41	12.6	0.18	3
6	6AF	41	12.6	0.17	6
6	6AG	41	12.6	0.12	4
7	7A	−1.08	−75.9	Continuous	292
7	7B	−1.08	−75.9	Continuous	292
8	8A	−2.1	−62.37	Continuous	454
9	9A	−3.4	−79.15	250	80
9	9B	−1.1	−79.3	138	94
9	9C	−3.59	−80.57	20	111
10	10A	−19.38	−43.53	24	64
10	10B	−19.38	−43.53	63	78
10	10C	−19.38	−43.53	198	70
11	11A	2.755	−59.55	6400	192
12	12A	8.47	−82.57	227	231
12	12B	8.47	−82.57	25	93
12	12C	8.47	−82.57	25	70
12	12D	8.47	−82.57	15	62
12	12E	8.47	−82.57	1	29
12	12F	8.47	−82.57	1	49
12	12G	8.47	−82.57	0.3	39
12	12H	8.47	−82.57	0.3	31
12	12I	8.47	−82.57	0.3	33
13	13A	−1	102.32	Continuous	109
14	14A	−23.32	−45.405	47	91

14	14B	−23.32	−45.405	26	84
14	14C	−23.32	−45.405	4	84
15	15A	−23.44	−46.51	Continuous	82
16	16A	−23.17	−51.15	681	134
16	16B	−23.17	−51.15	70	105
16	16C	−23.17	−51.15	70	129
16	16D	−23.17	−51.15	70	113
16	16E	−23.17	−51.15	70	106
17	17A	−14	35	720	58
17	17B	−14	35	7.5	25
17	17C	−14	35	108	39
17	17D	−14	35	263	39
17	17E	−14	35	80	31
17	17F	−14	35	600	35
17	17G	−14	35	600	39
17	17H	−14	35	44	30
17	17I	−14	35	253	45
17	17J	−14	35	614	27
17	17K	−14	35	12	27
17	17L	−14	35	400	44
17	17M	−14	35	285	52
17	17N	−14	35	160	51
17	17O	−14	35	60	43
18	18A	44.65	−123.41	Continuous	74
19	19A	−7.45	−35	356.93	77
19	19B	−7.45	−35	322	57
19	19C	−7.45	−35	500	21
19	19D	−7.45	−35	387.85	34
20	20A	51.46	−1.18	0.14	12
20	20B	51.46	−1.18	0.32	17
20	20C	51.46	−1.18	0.42	17
20	20D	51.46	−1.18	0.53	17

20	20E	51.46	−1.18	0.66	15
20	20F	51.46	−1.18	0.68	15
20	20G	51.46	−1.18	0.7	21
20	20H	51.46	−1.18	0.7	23
20	20I	51.46	−1.18	0.73	15
20	20J	51.46	−1.18	0.93	15
20	20K	51.46	−1.18	1.2	24
20	20L	51.46	−1.18	1.3	32
20	20M	51.46	−1.18	1.7	26
20	20N	51.46	−1.18	1.9	24
20	20O	51.46	−1.18	2.4	34
20	20P	51.46	−1.18	2.4	27
20	20Q	51.46	−1.18	2.9	31
20	20R	51.46	−1.18	3.7	28
20	20S	51.46	−1.18	6.9	25
20	20T	51.46	−1.18	18	37
21	21A	2.53	101.52	830	175
21	21B	2.53	101.52	800	48
22	22A	40	−75	0.2	12
22	22B	40	−75	0.8	17
22	22C	40	−75	1.2	12
22	22D	40	−75	2	20
22	22E	40	−75	3	22
22	22F	40	−75	4	25
22	22G	40	−75	7.5	31
22	22H	40	−75	10	33
22	22I	40	−75	24	38
23	23A	52.17	−2.3	10.3	170
24	24A	1.45	−75.5	30	142
24	24B	1.45	−75.5	15	106
25	25A	51.46	−1.18	430	69
26	26A	−20.45	−55	Continuous	329

27	27A	52.24	−0.14	1.18	22
27	27B	52.24	−0.14	2.88	19
27	27C	52.24	−0.14	7.98	32
27	27D	52.24	−0.14	4.15	21
27	27E	52.24	−0.14	3.44	25
27	27F	52.24	−0.14	1.01	19
27	27G	52.24	−0.14	1.81	20
27	27H	52.24	−0.14	2.9	14
27	27I	52.24	−0.14	9.05	26
27	27J	52.24	−0.14	1.03	17
27	27K	52.24	−0.14	1.53	24
27	27L	52.24	−0.14	1.6	17
27	27M	52.24	−0.14	1.22	25
27	27N	52.24	−0.14	1.72	30
27	27O	52.24	−0.14	1.61	21
27	27P	52.24	−0.14	2.42	26
27	27Q	52.24	−0.14	1.92	11
27	27R	52.24	−0.14	7.51	24
27	27S	52.24	−0.14	5.96	32
27	27T	52.24	−0.14	7.19	26
27	27U	52.24	−0.14	8.79	23
27	27V	52.24	−0.14	3.1	25
27	27W	52.24	−0.14	1.99	22
27	27X	52.24	−0.14	1.1	11
27	27Y	52.24	−0.14	10.3	34
27	27Z	52.24	−0.14	1.23	24
27	27AA	52.24	−0.14	5.4	24
27	27AB	52.24	−0.14	1.54	22
27	27AC	52.24	−0.14	3.46	27
27	27AD	52.24	−0.14	4.04	21
27	27AE	52.24	−0.14	1.83	23
27	27AF	52.24	−0.14	2.9	26

27	27AG	52.24	−0.14	3.88	20
27	27AH	52.24	−0.14	1.28	22
27	27AI	52.24	−0.14	19.42	29
27	27AJ	52.24	−0.14	4.18	26
27	27AK	52.24	−0.14	1.18	16
27	27AL	52.24	−0.14	4.56	29
27	27AM	52.24	−0.14	2.12	14
27	27AN	52.24	−0.14	7.95	28
27	27AO	52.24	−0.14	1.19	14
27	27AP	52.24	−0.14	3.53	21
27	27AQ	52.24	−0.14	2.63	18
27	27AR	52.24	−0.14	2.41	21
27	27AS	52.24	−0.14	5.29	20
27	27AT	52.24	−0.14	1.06	21
27	27AU	52.24	−0.14	1.76	24
27	27AV	52.24	−0.14	2.42	26
27	27AW	52.24	−0.14	6.64	24
27	27AX	52.24	−0.14	5.27	32
27	27AY	52.24	−0.14	5.85	21
27	27AZ	52.24	−0.14	5.85	21
27	27BA	52.24	−0.14	1.18	12
27	27BB	52.24	−0.14	1.37	19
27	27BC	52.24	−0.14	2.03	11
27	27BD	52.24	−0.14	2.73	22
27	27BE	52.24	−0.14	29.92	34
28	28A	−3.32	−64.58	35	115
28	28B	4.1	102.4	Continuous	204
29	29A	−1.05	29.62	Continuous	278
30	30A	49.11	19.06	Continuous	57
31	31A	54.32	−107.15	Continuous	51
31	31B	54.32	−107.15	Continuous	62
32	32A	4.14	116.28	Continuous	214

33	33A	−9.53	−56.28	1.1	31
33	33B	−9.53	−56.28	2.4	38
33	33C	−9.53	−56.28	4.2	38
33	33D	−9.53	−56.28	4.3	37
33	33E	−9.53	−56.28	4.7	44
33	33F	−9.53	−56.28	5.04	75
33	33G	−9.53	−56.28	6.6	45
33	33H	−9.53	−56.28	7.2	54
33	33I	−9.53	−56.28	15	70
33	33J	−9.53	−56.28	15	75
33	33K	−9.53	−56.28	19	58
33	33L	−9.53	−56.28	22	68
33	33M	−9.53	−56.28	24.2	105
33	33N	−9.53	−56.28	26	88
33	33O	−9.53	−56.28	26	76
33	33P	−9.53	−56.28	34.9	103
33	33Q	−9.53	−56.28	38.7	74
33	33R	−9.53	−56.28	44.6	98
33	33S	−9.53	−56.28	45	75
33	33T	−9.53	−56.28	87	127
33	33U	−9.53	−56.28	99	145
33	33V	−9.53	−56.28	112	115
33	33W	−9.53	−56.28	207.5	122
33	33X	−9.53	−56.28	243	126
33	33Y	−9.53	−56.28	254	116
33	33Z	−9.53	−56.28	787	136
33	33AA	−9.53	−56.28	900	164
33	33AB	−9.53	−56.28	1763	197
33	33AC	−9.53	−56.28	11034	198
33	33AD	−9.53	−56.28	14476	218
33	33AE	−9.53	−56.28	Continuous	227
34	34A	−21.45	−43.175	277	152

35	35A	38.35	-77.22	Continuous	79
36	36A	1.41	31.44	Continuous	108
37	37A	54.47	-127.1	33.28	33
37	37B	54.47	-127.1	56.96	35
37	37C	54.47	-127.1	4.64	29
37	37D	54.47	-127.1	45.76	24
37	37E	54.47	-127.1	60.95	30
37	37F	54.47	-127.1	23.52	26
37	37G	54.47	-127.1	61.28	31
37	37H	54.47	-127.1	33.6	31
38	38A	41	-8	0.39	9
38	38B	41	-8	1.22	10
38	38C	41	-8	2.2	16
38	38D	41	-8	2.85	16
38	38E	41	-8	3.36	14
38	38F	41	-8	5.19	19
38	38G	41	-8	6	21
38	38H	41	-8	10.3	25
38	38I	41	-8	21.66	18
38	38J	41	-8	0.22	11
38	38K	41	-8	0.53	13
38	38L	41	-8	1.01	16
38	38M	41	-8	2.83	15
38	38N	41	-8	4.76	17
38	38O	41	-8	8.62	16
38	38P	41	-8	11.91	19
38	38Q	41	-8	15.98	24
38	38R	41	-8	31.09	14
38	38S	41	-8	0.62	6
38	38T	41	-8	1.19	8
38	38U	41	-8	1.89	12
38	38V	41	-8	2.76	9

38	38W	41	−8	5.62	11
38	38X	41	−8	7.92	11
38	38Y	41	−8	18.81	11
38	38Z	41	−8	36.52	16
39	39A	5.56	−74.11	3848	349
40	40A	10.235	76.865	2600	36
41	41A	9.9935	−79.44	Continuous	252
42	42A	5.5	8.5	Continuous	177
43	43A	15.95	103.41	Continuous	184
44	44A	7.56	99.16	Continuous	109
45	45A	8.47	−82.57	227	89
45	45B	8.47	−82.57	5	53
45	45C	8.47	−82.57	5	48
45	45D	8.47	−82.57	5	61
46	46A	−9.31	−35.54	25	86
46	46B	−9.31	−35.54	795	61
46	46C	−9.31	−35.54	454	24
46	46D	−9.31	−35.54	300	105
46	46E	−9.31	−35.54	800	102
46	46F	−9.31	−35.54	100	19
47	47A	−9.73	−61.795	Continuous	470
47	47B	−9.73	−61.795	Continuous	312
48	48A	55.4	13.19	37	47
49	49A	−11.54	−71.18	Continuous	325
50	50A	63.2	44.2	Continuous	39
51	51A	4.7	−52.4	Continuous	257
52	52A	36.56	140.35	Continuous	65
53	53A	51.12	16.9	25	37
54	54A	52.4	23.5	Continuous	91
55	55A	−35.9	−72.5	600	25
56	56A	−29.2	30.15	215	54
56	56B	−29.2	30.15	138.2	40

56	56C	-29.2	30.15	123.8	39
56	56D	-29.2	30.15	85	37
56	56E	-29.2	30.15	45.8	41
56	56F	-29.2	30.15	15	38
56	56G	-29.2	30.15	3	36
56	56H	-29.2	30.15	1.72	26
56	56I	-29.2	30.15	0.5	33
56	56J	-29.2	30.15	1.7	31

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List of study species with associated trait data

Species name	Habitat ¹	Migration ²	Body mass (g) ³
<i>Abroscopus superciliaris</i>	1	1	6.5 ^a
<i>Aburria aburri</i>	2	1	21.4 ^a
<i>Acanthidops bairdii</i>	1	2	16 ^a
<i>Accipiter badius</i>	2	3	131.2 ^a
<i>Accipiter bicolor</i>	2	2	287.5 ^a
<i>Accipiter cooperii</i>	2	3	429.7 ^a
<i>Accipiter gentilis</i>	1	2	866 ^a
<i>Accipiter gularis</i>	1	3	120.7 ^a
<i>Accipiter melanoleucus</i>	1	1	699 ^a
<i>Accipiter minullus</i>	1	1	87.4 ^a
<i>Accipiter nisus</i>	1	2	220.8 ^a
<i>Accipiter poliogaster</i>	1	2	1369.4 ^b
<i>Accipiter rufiventris</i>	1	1	198 ^a
<i>Accipiter striatus</i>	1	3	130.6 ^a
<i>Accipiter supervilosus</i>	1	1	99.5 ^a

<i>Accipiter tachiro</i>	1	1	381 ^c
<i>Accipiter trivirgatus</i>	1	1	265 ^a
<i>Accipiter ventralis</i>	1	2	233 ^a
<i>Accipiter virgatus</i>	1	2	117 ^a
<i>Aceros comatus</i>	1	1	1470 ^a
<i>Aceros corrugatus</i>	1	1	1590 ^a
<i>Aceros undulatus</i>	1	1	2214.6 ^a
<i>Acestrura beliodor</i>	2	1	3 ^b
<i>Acridotheres fuscus</i>	2	2	82.8 ^a
<i>Acridotheres javanicus</i>	3	1	100 ^a
<i>Acridotheres tristis</i>	3	2	116.4 ^a
<i>Acrocephalus arundinaceus</i>	3	3	30 ^a
<i>Acrocephalus dumetorum</i>	2	3	11.2 ^a
<i>Acrocephalus palustris</i>	3	3	11.5 ^a
<i>Acrocephalus schoenobaenus</i>	3	3	11.8 ^a
<i>Acrocephalus scirpaceus</i>	3	3	12.3 ^a
<i>Actenoides concretus</i>	1	1	73.5 ^a

<i>Adelomyia melanogenys</i>	2	2	48.4 ^a
<i>Aegithalos caudatus</i>	2	1	8.6 ^a
<i>Aegithina lafresnayei</i>	1	1	12 ^f
<i>Aegithina tibia</i>	1	1	12 ^a
<i>Aegithina viridissima</i>	1	1	13.4 ^a
<i>Aegolius funereus</i>	1	2	137.2 ^a
<i>Aerodramus fuciphagus</i>	1	1	10.7 ^a
<i>Aerodramus maximus</i>	1	1	17 ^a
<i>Aethopyga mystacalis</i>	1	1	6 ^a
<i>Aethopyga saturata</i>	1	1	5.3 ^a
<i>Aethopyga siparaja</i>	2	1	6.7 ^a
<i>Agamia agami</i>	2	1	567 ^a
<i>Agapornis pullarius</i>	3	1	39.3 ^a
<i>Agelaius phoeniceus</i>	3	3	50.8 ^a
<i>Agelaius ruficapillus</i>	2	3	32 ^a
<i>Agelastes niger</i>	1	1	700 ^a
<i>Aglaeactis cupripennis</i>	2	1	7.6 ^a

<i>Agelaiocercus coelestis</i>	2	1	5.6 ^a
<i>Aix galericulata</i>	2	1	10.7 ^a
<i>Ajaia ajaja</i>	3	2	1490 ^a
<i>Alauda arvensis</i>	3	2	37.3 ^a
<i>Alcedo atthis</i>	3	1	168.9 ^a
<i>Alcedo euryzona</i>	1	1	43.1 ^a
<i>Alcedo leucogaster</i>	1	1	14.5 ^a
<i>Alcedo meninting</i>	2	1	20.4 ^a
<i>Alcedo quadribrachys</i>	2	1	34.9 ^a
<i>Alcippe brunneicauda</i>	1	1	14.3 ^a
<i>Alcippe poiocephala</i>	2	1	20.7 ^a
<i>Alectoris rufa</i>	3	1	527.9 ^a
<i>Alethe choloensis</i>	1	1	41.3 ^a
<i>Alethe diademata</i>	1	1	31.5 ^a
<i>Alethe fuelleborni</i>	1	1	52 ^a
<i>Alethe poliocephala</i>	1	1	32.4 ^a
<i>Alethe poliopteryx</i>	1	1	35.2 ^a

<i>Alophoixus bres</i>	1	1	35.3 ^a
<i>Alophoixus finschii</i>	1	1	24.2 ^a
<i>Alophoixus ochraceus</i>	1	1	41.5 ^a
<i>Alophoixus pallidus</i>	2	1	46 ^a
<i>Alophoixus phaeocephalus</i>	1	1	32 ^a
<i>Amandava amandava</i>	2	1	9 ^a
<i>Amaurornis flavirostra</i>	2	1	86.2 ^a
<i>Amaurornis phoenicurus</i>	2	1	180 ^a
<i>Amaurospiza moesta</i>	1	1	13.6 ^a
<i>Amazilia amabilis</i>	2	1	4.3 ^a
<i>Amazilia amazilia</i>	2	1	4.3 ^a
<i>Amazilia beryllina</i>	2	1	4.5 ^a
<i>Amazilia candida</i>	1	1	3.8 ^a
<i>Amazilia cyanifrons</i>	2	1	5 ^a
<i>Amazilia edward</i>	2	1	4.7 ^a
<i>Amazilia fimbriata</i>	2	1	4.9 ^a
<i>Amazilia franciae</i>	1	1	5 ^a

<i>Amazilia lactea</i>	2	1	4.6 ^a
<i>Amazilia tzacatl</i>	1	1	4.8 ^a
<i>Amazilia versicolor</i>	1	1	4.1 ^a
<i>Amazilia yucatanensis</i>	2	1	3.9 ^a
<i>Amazona aestiva</i>	2	1	451 ^a
<i>Amazona albifrons</i>	2	1	206 ^a
<i>Amazona amazonica</i>	2	1	370 ^a
<i>Amazona autumnalis</i>	2	1	416 ^a
<i>Amazona farinosa</i>	1	1	626 ^a
<i>Amazona festiva</i>	1	1	430 ^a
<i>Amazona kawalli</i>	1	1	327.4 ^f
<i>Amazona mercenaria</i>	1	1	340 ^a
<i>Amazona ochrocephala</i>	2	1	476.9 ^a
<i>Amazonetta brasiliensis</i>	3	1	500 ^a
<i>Amblycercus holosericeus</i>	1	1	70.4 ^a
<i>Amblyospiza albifrons</i>	2	1	41.6 ^a
<i>Ammodramus aurifrons</i>	3	1	17.8 ^a

<i>Ampeliceps coronatus</i>	1	1	87.9 ^a
<i>Ampelioides tschudii</i>	1	1	78.4 ^a
<i>Ampelion rubrocristatus</i>	2	1	66.3 ^a
<i>Anabacerthia amaurotis</i>	1	1	19.2 ^a
<i>Anabacerthia variegaticeps</i>	1	1	27.8 ^a
<i>Anabazenops fuscus</i>	1	1	39 ^b
<i>Anairetes parulus</i>	2	1	6.2 ^a
<i>Anas crecca</i>	3	2	341.9 ^a
<i>Anas platyrhynchos</i>	3	2	843.4 ^a
<i>Anas sparsa</i>	3	1	981.5 ^a
<i>Ancistrops strigilatus</i>	1	1	35.9 ^a
<i>Andigena hypoglaucha</i>	1	1	312.8 ^a
<i>Andropadus ansorgei</i>	1	1	18.7 ^a
<i>Andropadus curvirostris</i>	1	1	24.9 ^a
<i>Andropadus gracilirostris</i>	1	1	31.2 ^a
<i>Andropadus gracilis</i>	1	1	21.6 ^a
<i>Andropadus importunus</i>	2	1	27.4 ^a

<i>Andropadus latirostris</i>	1	1	26.2 ^a
<i>Andropadus masukuensis</i>	1	1	26.8 ^a
<i>Andropadus milanjensis</i>	2	1	37.4 ^a
<i>Andropadus montanus</i>	1	1	32.6 ^a
<i>Andropadus nigriceps</i>	1	1	35 ^d
<i>Andropadus tephrolaemus</i>	1	1	38.3 ^a
<i>Andropadus virens</i>	1	1	26.6 ^a
<i>Anbima cornuta</i>	3	1	3150 ^a
<i>Anbinga anbinga</i>	3	1	1235 ^a
<i>Anbinga melanogaster</i>	3	2	1480.1 ^a
<i>Anisognathus igniventris</i>	2	1	34 ^a
<i>Anisognathus lacrymosus</i>	1	1	31 ^a
<i>Anisognathus somptuosus</i>	1	1	44.5 ^c
<i>Anorrhinus austeni</i>	1	1	1079.5 ^f
<i>Anorrhinus galeritus</i>	1	1	1172 ^a
<i>Anthracoceros albirostris</i>	2	1	892.9 ^a
<i>Anthracoceros malayanus</i>	1	1	1050 ^a

<i>Anthracothorax nigricollis</i>	2	1	7 ^a
<i>Anthracothorax prevostii</i>	2	1	6.4 ^a
<i>Anthreptes collaris</i>	2	1	7 ^a
<i>Anthreptes fraseri</i>	1	1	11.3 ^a
<i>Anthreptes gabonicus</i>	2	1	9.6 ^f
<i>Anthreptes malacensis</i>	2	1	11.3 ^a
<i>Anthreptes rectirostris</i>	1	1	10.6 ^a
<i>Anthreptes rhodolaema</i>	2	1	13.1 ^a
<i>Anthreptes simplex</i>	1	1	8.9 ^a
<i>Anthreptes singalensis</i>	1	1	8.2 ^a
<i>Anthus bogotensis</i>	3	1	25.1 ^a
<i>Anthus hodgsoni</i>	1	1	116.8 ^a
<i>Anthus novaeseelandiae</i>	3	1	24.1 ^a
<i>Anthus pratensis</i>	1	3	5.5 ^a
<i>Anthus trivialis</i>	2	3	23.3 ^a
<i>Antilopbia galeata</i>	2	1	21.5 ^a
<i>Anurolimnas castaneiceps</i>	1	1	126 ^a

<i>Anurolimnas viridis</i>	2	1	64.3 ^a
<i>Apalis binotata</i>	1	1	8.2 ^a
<i>Apalis chapini</i>	1	1	8.8 ^a
<i>Apalis chariessa</i>	1	1	9.2 ^f
<i>Apalis cinerea</i>	1	1	9.9 ^a
<i>Apalis flavida</i>	2	1	8.2 ^a
<i>Apalis jacksoni</i>	1	1	8.7 ^a
<i>Apalis melanocephala</i>	1	1	8.8 ^a
<i>Apalis nigriceps</i>	1	1	15.6 ^a
<i>Apalis porphyrolaema</i>	1	1	8.4 ^a
<i>Apalis rufogularis</i>	1	1	8.3 ^a
<i>Apalis ruwenzorii</i>	1	1	9.9 ^a
<i>Apalis thoracica</i>	2	1	11.1 ^a
<i>Apaloderma aequatoriale</i>	1	1	58 ^f
<i>Apaloderma narina</i>	1	1	67.7 ^a
<i>Apaloderma vittatum</i>	1	1	55 ^a
<i>Aphantochroa cirrochloris</i>	1	1	7.7 ^a

<i>Apbelocoma californica</i>	2	1	85.7 ^a
<i>Aplonis panayensis</i>	1	1	56.2 ^a
<i>Aplopelia larvata</i>	2	1	155 ^a
<i>Apus affinis</i>	1	3	22.6 ^a
<i>Apus apus</i>	1	3	37.6 ^a
<i>Apus caffer</i>	1	3	22.1 ^a
<i>Apus horus</i>	1	2	27.5 ^a
<i>Aquila chrysaetos</i>	3	2	4248 ^a
<i>Aquila pomarina</i>	1	3	1359.4 ^a
<i>Ara ararauna</i>	1	1	1125 ^a
<i>Ara chloropterus</i>	1	1	1214 ^a
<i>Ara couloni</i>	1	1	250 ^a
<i>Ara macao</i>	2	1	1015 ^a
<i>Ara manilata</i>	2	2	372 ^a
<i>Ara maracana</i>	1	1	256 ^a
<i>Ara nobilis</i>	2	1	150.9 ^a
<i>Ara severa</i>	1	1	343 ^a

<i>Arachnothera affinis</i>	1	1	26.4 ^a
<i>Arachnothera chrysogenys</i>	1	1	23.2 ^a
<i>Arachnothera crassirostris</i>	1	1	16 ^a
<i>Arachnothera flavigaster</i>	1	1	38.4 ^a
<i>Arachnothera longirostra</i>	1	1	12.5 ^a
<i>Arachnothera robusta</i>	1	1	25.4 ^f
<i>Aramides cajanea</i>	1	1	397 ^a
<i>Aramides saracura</i>	1	1	540 ^a
<i>Aramus guarana</i>	3	1	1080 ^a
<i>Aratinga aurea</i>	2	1	84.6 ^a
<i>Aratinga chloroptera</i>	2	1	146 ^a
<i>Aratinga erythrogenys</i>	2	1	151 ^a
<i>Aratinga jandaya</i>	2	1	89.4 ^a
<i>Aratinga leucophthalmus</i>	2	1	158 ^a
<i>Aratinga nana</i>	1	1	76.9 ^a
<i>Aratinga pertinax</i>	2	1	84 ^a
<i>Aratinga solstitialis</i>	2	1	119.6 ^a

<i>Aratinga weddellii</i>	1	1	108 ^a
<i>Arborophila charltonii</i>	1	1	269.3 ^a
<i>Arborophila chloropus</i>	1	1	250 ^a
<i>Archilochus colubris</i>	2	3	3.1 ^a
<i>Ardea alba</i>	3	2	871.3 ^a
<i>Ardea cocoi</i>	3	1	1752.4 ^a
<i>Ardea herodias</i>	3	2	2523.4 ^a
<i>Ardea sumatrana</i>	3	1	2024 ^a
<i>Ardeola bacchus</i>	3	2	304.9 ^a
<i>Argusianus argus</i>	1	1	1959.9 ^a
<i>Arremon abeillei</i>	2	1	6.8 ^a
<i>Arremon aurantirostris</i>	1	1	34.5 ^a
<i>Arremon flavirostris</i>	1	1	26.1 ^a
<i>Arremon semitorquatus</i>	1	1	28.1 ^f
<i>Arremon taciturnus</i>	1	1	24.8 ^a
<i>Arremonops chloronotus</i>	1	1	27.4 ^a
<i>Arremonops conirostris</i>	2	1	35.9 ^a

<i>Arremonops rufivirgatus</i>	2	1	22.4 ^a
<i>Arundinicola leucocephala</i>	3	1	13.8 ^a
<i>Asio clamator</i>	2	1	443.3 ^a
<i>Asio otus</i>	1	2	296.6 ^a
<i>Asio stygius</i>	1	1	565 ^a
<i>Athene noctua</i>	3	1	168.9 ^a
<i>Atlapetes albinucha</i>	1	1	33.8 ^a
<i>Atlapetes atricapillus</i>	1	1	44.6 ^a
<i>Atlapetes brunneinucha</i>	1	1	44 ^a
<i>Atlapetes leucopterus</i>	2	2	20.8 ^a
<i>Atlapetes rufinucha</i>	1	1	21.5 ^a
<i>Atlapetes torquatus</i>	1	1	42 ^a
<i>Atticora fasciata</i>	2	1	13 ^a
<i>Atticora melanoleuca</i>	3	1	10.8 ^a
<i>Attila bolivianus</i>	2	1	39.5 ^a
<i>Attila cinnamomeus</i>	3	1	38.8 ^a
<i>Attila citriniventris</i>	1	1	33.7 ^a

<i>Attila phoenicurus</i>	1	3	32.3 ^a
<i>Attila rufus</i>	1	1	42.6 ^a
<i>Attila spadiceus</i>	1	1	39.1 ^a
<i>Attila torridus</i>	1	1	46 ^a
<i>Aulacorhynchus haematopygus</i>	1	1	194.4 ^a
<i>Aulacorhynchus prasinus</i>	1	1	170.6 ^a
<i>Automolus dorsalis</i>	1	1	35.4 ^a
<i>Automolus infuscatus</i>	1	1	32.9 ^a
<i>Automolus leucophthalmus</i>	1	1	34.5 ^a
<i>Automolus melanopezus</i>	1	1	31.4 ^a
<i>Automolus ochrolaemus</i>	1	1	40.2 ^a
<i>Automolus rubiginosus</i>	1	1	39.8 ^a
<i>Automolus rufipileatus</i>	1	1	34.8 ^a
<i>Ariceda cuculoides</i>	1	1	277 ^a
<i>Ariceda jerdoni</i>	2	1	363 ^a
<i>Ariceda leuphotes</i>	2	3	194 ^a

<i>Avocettula recurvirostris</i>	2	1	4.2 ^a
<i>Baeopogon clamans</i>	1	1	46 ^f
<i>Baeopogon indicator</i>	1	1	45.8 ^a
<i>Bailloni bailloni</i>	1	1	146 ^a
<i>Bambusicola thoracica</i>	1	2	866 ^a
<i>Bartramia longicauda</i>	3	3	158.9 ^a
<i>Baryphthengus martii</i>	1	1	165 ^a
<i>Baryphthengus ruficapillus</i>	1	1	141.7 ^a
<i>Basilenterus cinereicollis</i>	1	1	13 ^f
<i>Basilenterus coronatus</i>	1	1	16.3 ^a
<i>Basilenterus culicivorus</i>	1	1	10.5 ^a
<i>Basilenterus flaveolus</i>	1	1	13.2 ^a
<i>Basilenterus fraseri</i>	2	1	11.6 ^a
<i>Basilenterus fulvicanda</i>	1	1	14.9 ^a
<i>Basilenterus hypoleucus</i>	2	1	11.4 ^a
<i>Basilenterus leucoblepharus</i>	1	1	15.1 ^a
<i>Basilenterus luteoviridis</i>	1	1	12.9 ^a

<i>Basileuterus rivularis</i>	1	1	13.5 ^a
<i>Basileuterus rufifrons</i>	2	1	10.4 ^a
<i>Basileuterus trifasciatus</i>	1	1	9.7 ^a
<i>Basileuterus tristriatus</i>	1	1	12 ^a
<i>Bathmocercus cerviniventris</i>	1	1	15.8 ^a
<i>Batis capensis</i>	2	1	11.7 ^a
<i>Batis diops</i>	1	1	12.7 ^a
<i>Batis fratrum</i>	1	1	11.9 ^a
<i>Batis mixta</i>	2	1	12.2 ^a
<i>Batis molitor</i>	2	1	11.6 ^a
<i>Batrachostomus javensis</i>	1	1	46.3 ^a
<i>Berenicornis comatus</i>	1	1	1470 ^a
<i>Berlepschia rikeri</i>	1	1	37 ^a
<i>Bias flammulatus</i>	1	1	28 ^a
<i>Bias musicus</i>	1	2	21.9 ^a
<i>Bleda eximia</i>	1	1	35.7 ^a
<i>Bleda syndactyla</i>	1	1	46.1 ^a

<i>Blythipicus rubiginosus</i>	1	1	81.9 ^a
<i>Boissonneaua flavescens</i>	1	1	4.2 ^a
<i>Boissonneaua matthewsii</i>	1	1	7.7 ^a
<i>Bombycilla cedrorum</i>	2	3	31.6 ^a
<i>Bombycilla japonica</i>	3	3	21.3 ^a
<i>Bonasa bonasia</i>	1	1	429 ^a
<i>Bonasa umbellus</i>	1	1	530.9 ^a
<i>Bostrychia hagedash</i>	2	1	1238 ^a
<i>Bostrychia rara</i>	2	1	866 ^b
<i>Brachygalba lugubris</i>	2	1	15.9 ^a
<i>Bradypterus barratti</i>	1	2	18.8 ^a
<i>Bradypterus cinnamomeus</i>	1	1	17.4 ^a
<i>Bradypterus graueri</i>	2	1	17 ^a
<i>Bradypterus lopezi</i>	1	1	19.2 ^a
<i>Brotogeris chrysopterus</i>	1	1	54.5 ^a
<i>Brotogeris cyanoptera</i>	1	1	56 ^a
<i>Brotogeris jugularis</i>	2	1	63.2 ^a

<i>Brotoogeris sanctithomae</i>	2	1	59 ^a
<i>Brotoogeris tirica</i>	2	1	63 ^a
<i>Brotoogeris versicolurus</i>	2	1	71.5 ^a
<i>Bubo bubo</i>	2	1	2668.5 ^a
<i>Bubo nipalensis</i>	1	1	1396.4 ^a
<i>Bubo poensis</i>	1	1	654.9 ^a
<i>Bubo sumatranus</i>	1	1	1525 ^a
<i>Bubo virginianus</i>	2	1	1575.7 ^c
<i>Bubulcus ibis</i>	3	2	366 ^a
<i>Buccanodon duchaillui</i>	1	1	41.8 ^a
<i>Bucco capensis</i>	1	1	54 ^a
<i>Bucco macrodactylus</i>	1	1	25 ^a
<i>Bucco tamatia</i>	1	1	35.5 ^a
<i>Buceros bicornis</i>	1	1	2790.8 ^a
<i>Buceros rhinoceros</i>	1	1	2371.6 ^a
<i>Buceros vigil</i>	1	1	2887.6 ^a
<i>Busarellus nigricollis</i>	1	1	766.1 ^a

<i>Butastur indicus</i>	2	3	414.6 ^a
<i>Buteo albicaudatus</i>	3	1	885 ^a
<i>Buteo albonotatus</i>	1	1	745.9 ^a
<i>Buteo brachyurus</i>	1	1	496 ^a
<i>Buteo buteo</i>	3	2	759.1 ^a
<i>Buteo jamaicensis</i>	2	3	1101.2 ^a
<i>Buteo leucorrhois</i>	1	1	290 ^a
<i>Buteo lineatus</i>	1	3	603.7 ^a
<i>Buteo magnirostris</i>	1	1	269 ^a
<i>Buteo nitidus</i>	1	1	519 ^a
<i>Buteo oreophilus</i>	1	1	700 ^a
<i>Buteo platypterus</i>	2	3	453.7 ^a
<i>Buteo polyosoma</i>	2	2	782.6 ^a
<i>Buteo rufofuscus</i>	3	2	1530 ^a
<i>Buteo swainsoni</i>	3	3	946.6 ^a
<i>Buteogallus anthracinus</i>	2	1	975.1 ^a
<i>Buteogallus meridionalis</i>	3	1	808 ^a

<i>Buteogallus urubitinga</i>	2	1	1152.9 ^a
<i>Butbraupis montana</i>	1	1	87.1 ^a
<i>Butorides striatus</i>	3	1	201.5 ^a
<i>Butorides virescens</i>	3	1	201.5 ^a
<i>Bycanistes albotibialis</i>	1	1	1141 ^a
<i>Cacicus cela</i>	1	1	85.5 ^a
<i>Cacicus chrysopterus</i>	1	1	36.2 ^a
<i>Cacicus haemorrhous</i>	1	1	83.7 ^a
<i>Cacicus leucoramphus</i>	1	1	63.7 ^f
<i>Cacicus solitarius</i>	1	1	79.8 ^a
<i>Cacicus uropygialis</i>	1	1	61.2 ^a
<i>Cacomantis merulinus</i>	2	1	25.3 ^a
<i>Cacomantis sonneratii</i>	1	1	33.8 ^a
<i>Cacomantis variolosus</i>	1	1	35.8 ^a
<i>Cairina moschata</i>	3	1	2415.8 ^a
<i>Calidris alba</i>	3	3	51.7 ^a
<i>Calidris fuscicollis</i>	3	3	43 ^a

<i>Calidris melanotos</i>	3	3	79.7 ^a
<i>Calidris minutilla</i>	3	3	22.9 ^a
<i>Callipepla amethystina</i>	2	1	2.8 ^a
<i>Calorhamphus fuliginosus</i>	1	1	42.6 ^a
<i>Calyptrichla serina</i>	1	1	38.7 ^a
<i>Calyptromena viridis</i>	1	1	58.5 ^a
<i>Camaroptera brachyura</i>	1	1	9.7 ^a
<i>Camaroptera chloronota</i>	1	1	11.4 ^a
<i>Camaroptera supercilialis</i>	1	1	9.7 ^a
<i>Campephaga flava</i>	2	1	32.3 ^a
<i>Campephaga petiti</i>	1	1	31.3 ^a
<i>Campephaga phoenicea</i>	2	1	27.1 ^a
<i>Campephilus gayaquilensis</i>	1	1	241.2 ^a
<i>Campephilus guatemalensis</i>	1	1	242 ^a
<i>Campephilus haematogaster</i>	1	1	237 ^a
<i>Campephilus melanoleucos</i>	1	1	256 ^a
<i>Campephilus pollens</i>	1	1	273.3 ^f

<i>Campephilus robustus</i>	1	1	200 ^a
<i>Campephilus rubricollis</i>	1	1	210.7 ^a
<i>Campethera abingoni</i>	2	1	67.5 ^a
<i>Campethera caroli</i>	1	1	59 ^a
<i>Campethera nivosa</i>	1	1	37.8 ^a
<i>Campethera tullbergi</i>	1	1	50.5 ^a
<i>Camptostoma obsoletum</i>	2	1	8.1 ^a
<i>Campylopterus curvipennis</i>	1	1	6 ^a
<i>Campylopterus falcatus</i>	1	1	8.2 ^a
<i>Campylopterus hemileucurus</i>	1	1	9.7 ^a
<i>Campylopterus largipennis</i>	1	1	8.3 ^a
<i>Campylorhamphus falcularius</i>	1	1	42.6 ^a
<i>Campylorhamphus procurviroides</i>	1	1	33.5 ^a
<i>Campylorhamphus pusillus</i>	1	1	40.5 ^a
<i>Campylorhamphus trochilirostris</i>	1	1	40.8 ^a
<i>Campylorhynchus fasciatus</i>	3	1	30 ^a

<i>Campylorhynchus turdinus</i>	1	1	32.6 ^a
<i>Campylorhynchus zonatus</i>	1	1	34.6 ^a
<i>Capito dayi</i>	1	1	67.2 ^a
<i>Capito hypoleucus</i>	1	1	70.9 ^a
<i>Capito niger</i>	1	1	57.4 ^a
<i>Capito squamatus</i>	1	1	89.7 ^a
<i>Caprimulgus cayennensis</i>	3	2	35.5 ^a
<i>Caprimulgus europaeus</i>	2	3	67 ^a
<i>Caprimulgus indicus</i>	1	3	87 ^a
<i>Caprimulgus macrurus</i>	2	1	66.2 ^a
<i>Caprimulgus nigrescens</i>	2	1	35.7 ^a
<i>Caprimulgus pectoralis</i>	2	2	51.4 ^a
<i>Caprimulgus poliocephalus</i>	2	2	41.8 ^a
<i>Caprimulgus rufus</i>	2	1	94.2 ^a
<i>Caprimulgus vociferus</i>	1	3	53.3 ^a
<i>Capsiempis flaveola</i>	1	1	7.7 ^a
<i>Caracara plancus</i>	3	1	1078.6 ^a

<i>Cardellina rubrifrons</i>	2	3	8.1 ^a
<i>Cardinalis cardinalis</i>	2	1	42.6 ^a
<i>Carduelis barbata</i>	1	1	160.6 ^a
<i>Carduelis cannabina</i>	2	1	19.5 ^a
<i>Carduelis carduelis</i>	2	1	16 ^a
<i>Carduelis chloris</i>	2	1	26 ^a
<i>Carduelis flammea</i>	1	2	13 ^a
<i>Carduelis magellanica</i>	3	1	13.6 ^a
<i>Carduelis pinus</i>	1	1	12.7 ^a
<i>Carduelis psaltria</i>	2	1	8.8 ^a
<i>Carduelis sinica</i>	2	3	18.5 ^a
<i>Carduelis spinus</i>	1	1	13.2 ^a
<i>Carduelis tristis</i>	1	1	12.8 ^a
<i>Carduelis xanthogastra</i>	2	1	12.7 ^a
<i>Cariama cristata</i>	2	1	1400 ^a
<i>Carpococcyx renauldi</i>	1	1	550 ^b
<i>Carpodacus erythrurus</i>	2	3	24 ^a

<i>Carpodacus mexicanus</i>	2	2	21.4 ^a
<i>Carpodacus purpureus</i>	2	2	23.3 ^a
<i>Carpornis cucullata</i>	1	1	74.2 ^a
<i>Caryothraustes canadensis</i>	2	1	34.5 ^a
<i>Caryothraustes humeralis</i>	1	1	37 ^a
<i>Caryothraustes poliogaster</i>	1	1	41.8 ^a
<i>Casiornis rufa</i>	2	3	24.8 ^a
<i>Catamenia inornata</i>	3	1	13.4 ^a
<i>Cathartes aura</i>	3	2	1518.2 ^a
<i>Cathartes burrovianus</i>	3	1	935 ^a
<i>Cathartes melambrotus</i>	3	1	1373 ^a
<i>Catharus aurantirostris</i>	2	1	29.8 ^a
<i>Catharus dryas</i>	3	1	16.1 ^a
<i>Catharus fuscater</i>	1	1	33.7 ^a
<i>Catharus fuscescens</i>	1	3	31.9 ^a
<i>Catharus guttatus</i>	1	3	30.1 ^a
<i>Catharus minimus</i>	2	3	31.6 ^a

<i>Catharus mustelinus</i>	1	3	50.1 ^a
<i>Catharus ustulatus</i>	1	3	30.3 ^a
<i>Celeus brachyurus</i>	1	1	85.9 ^a
<i>Celeus castaneus</i>	1	1	85.6 ^a
<i>Celeus elegans</i>	1	1	135 ^a
<i>Celeus flavescens</i>	1	1	139 ^a
<i>Celeus flavus</i>	1	1	147.3 ^a
<i>Celeus grammicus</i>	1	1	73.4 ^a
<i>Celeus loricatus</i>	1	1	76.6 ^a
<i>Celeus torquatus</i>	1	1	134 ^a
<i>Celeus undatus</i>	1	1	64.5 ^a
<i>Centropus bengalensis</i>	2	1	148.9 ^a
<i>Centropus leucogaster</i>	1	1	307.8 ^a
<i>Centropus rectunguis</i>	1	1	238 ^a
<i>Centropus sinensis</i>	2	1	280.7 ^a
<i>Centropus superciliosus</i>	1	1	148.5 ^a
<i>Ceratogymna atrata</i>	1	1	1194.8 ^a

<i>Ceratogymna brevis</i>	1	1	1232.8 ^a
<i>Ceratogymna bucinator</i>	2	1	639.4 ^a
<i>Ceratogymna cylindricus</i>	1	1	921 ^a
<i>Ceratogymna elata</i>	1	1	1917 ^a
<i>Ceratogymna fistulator</i>	1	1	508.1 ^a
<i>Ceratogymna subcylindricus</i>	2	1	1195.4 ^a
<i>Cercococcyx montanus</i>	1	1	58.2 ^a
<i>Cercococcyx olivinus</i>	1	1	65.9 ^a
<i>Cercomacra cinerascens</i>	1	1	14.3 ^a
<i>Cercomacra laeta</i>	1	1	16 ^a
<i>Cercomacra manu</i>	1	1	18.1 ^a
<i>Cercomacra nigrescens</i>	1	1	16.5 ^a
<i>Cercomacra nigricans</i>	1	1	16.6 ^a
<i>Cercomacra serva</i>	1	1	17 ^a
<i>Cercomacra tyrannina</i>	1	1	16.3 ^a
<i>Cercotrichas hartlaubi</i>	2	1	19.2 ^a
<i>Cercotrichas quadrivirgata</i>	1	1	26.1 ^a

<i>Certhia americana</i>	1	1	8.1 ^a
<i>Certhia brachydactyla</i>	1	1	8.2 ^a
<i>Certhia familiaris</i>	1	1	9 ^a
<i>Certhiaxis cinnamomea</i>	2	1	15.2 ^a
<i>Cettia cetti</i>	2	1	13.2 ^a
<i>Cettia diphone</i>	1	3	72.1 ^a
<i>Centhmocharis aereus</i>	1	1	65.8 ^a
<i>Ceyx erithacus</i>	1	1	17.8 ^a
<i>Chaetura andrei</i>	1	3	22.2 ^a
<i>Chaetura brachyura</i>	1	1	18.3 ^a
<i>Chaetura chapmani</i>	1	2	23.8 ^a
<i>Chaetura cinereiventris</i>	1	1	13.9 ^a
<i>Chaetura egregia</i>	1	1	23.4 ^a
<i>Chaetura pelagica</i>	1	3	23.6 ^a
<i>Chaetura spinicauda</i>	1	1	15.7 ^a
<i>Chaetura vauxi</i>	1	3	17.1 ^a
<i>Chalcoparia singalensis</i>	1	1	8.2 ^a

<i>Chalcophaps indica</i>	1	1	136.5 ^a
<i>Chalybura buffonii</i>	1	1	6.8 ^a
<i>Chamaea fasciata</i>	2	1	14.7 ^a
<i>Chamaeza campanisona</i>	1	1	90.6 ^a
<i>Chamaeza nobilis</i>	1	1	123 ^a
<i>Charadrius collaris</i>	3	2	28.3 ^a
<i>Charadrius dubius</i>	3	3	38.7 ^a
<i>Chauna chavaria</i>	3	1	2557.6 ^a
<i>Chelidoptera tenebrosa</i>	3	1	35.9 ^a
<i>Chiroxiphia caudata</i>	1	1	25.6 ^a
<i>Chiroxiphia pareola</i>	1	1	16.8 ^a
<i>Chlorestes notatus</i>	3	1	4.2 ^a
<i>Chloroceryle aenea</i>	2	1	13.8 ^a
<i>Chloroceryle amazona</i>	3	1	126.4 ^a
<i>Chloroceryle americana</i>	3	1	33.7 ^a
<i>Chloroceryle inda</i>	2	1	52.1 ^a
<i>Chlorocichla flavicollis</i>	1	1	26.6 ^a

<i>Chlorocichla flaviventris</i>	1	1	39.4 ^a
<i>Chlorophanes spiza</i>	1	1	19 ^a
<i>Chlorophonia callophrys</i>	2	1	25.8 ^a
<i>Chlorophonia cyanea</i>	2	1	14 ^a
<i>Chloropipo holochlora</i>	1	1	14.9 ^a
<i>Chloropsis cochinchinensis</i>	1	1	24.5 ^a
<i>Chloropsis cyanopogon</i>	1	1	22 ^a
<i>Chloropsis sonnerati</i>	1	1	45.8 ^a
<i>Chlorornis riefferii</i>	1	1	53 ^a
<i>Chlorospingus flavigularis</i>	2	1	17 ^a
<i>Chlorospingus ophthalmicus</i>	1	1	20 ^a
<i>Chlorostilbon aureoventris</i>	2	2	3.5 ^a
<i>Chlorostilbon gibsoni</i>	2	2	2.6 ^a
<i>Chlorostilbon mellisugus</i>	2	2	2.6 ^a
<i>Chondrobierax uncinatus</i>	2	2	286.1 ^a
<i>Chordeiles acutipennis</i>	2	2	48.4 ^a
<i>Chordeiles minor</i>	3	3	79.3 ^a

<i>Chordeiles pusillus</i>	3	2	30.6 ^a
<i>Chordeiles rupestris</i>	2	2	58 ^b
<i>Chrysococcyx caprius</i>	2	1	32.8 ^a
<i>Chrysococcyx cupreus</i>	2	1	37.7 ^a
<i>Chrysococcyx klaas</i>	2	1	27.4 ^a
<i>Chrysococcyx maculatus</i>	1	1	27.4 ^a
<i>Chrysococcyx xanthorhynchus</i>	1	1	22.4 ^a
<i>Chrysocolaptes lucidus</i>	2	1	142 ^a
<i>Chrysolampis mosquitus</i>	2	3	3.9 ^a
<i>Chrysuronia oenone</i>	1	1	5 ^a
<i>Cicbicolaptes leucophrus</i>	1	1	50 ^b
<i>Ciconia stormi</i>	2	1	3086.3 ^f
<i>Cinclus cinclus</i>	3	1	61.5 ^a
<i>Cinnycerthia unirufa</i>	1	1	29.4 ^a
<i>Cinnyricinclus leucogaster</i>	2	3	44 ^a
<i>Cinnyricinclus sharpii</i>	1	2	41 ^a
<i>Circaetus gallicus</i>	3	3	1699.1 ^a

<i>Cissa chinensis</i>	1	1	126.6 ^a
<i>Cissopis leveriana</i>	2	1	76 ^a
<i>Cisticola chubbi</i>	2	1	16 ^a
<i>Cisticola erythrops</i>	3	1	14.8 ^a
<i>Cisticola juncidis</i>	3	3	6.8 ^a
<i>Clamator coromandus</i>	2	3	74.6 ^a
<i>Clamator levaillantii</i>	2	3	122 ^a
<i>Claravis pretiosa</i>	1	2	68.2 ^a
<i>Clibanornis dendrocolaptoides</i>	1	1	48.2 ^a
<i>Chypicternus oseryi</i>	1	1	137.7 ^a
<i>Clytoctantes atrogularis</i>	1	1	31 ^a
<i>Cnemoscopus rubrirostris</i>	1	1	18 ^a
<i>Cnemotriccus fuscatus</i>	2	2	13.6 ^a
<i>Cnipodectes subbrunneus</i>	1	1	23.2 ^a
<i>Coccothraustes coccothraustes</i>	1	2	56.6 ^a
<i>Coccyzus americanus</i>	1	3	64 ^a
<i>Coccyzus cinereus</i>	2	3	41 ^a

<i>Coccyzus erythrophthalmus</i>	1	3	50.9 ^a
<i>Coccyzus euleri</i>	1	3	52.3 ^a
<i>Coccyzus melacoryphus</i>	1	1	49.7 ^a
<i>Cochlearius cochlearia</i>	2	1	643.2 ^a
<i>Cochlearius cochlearius</i>	2	1	643.2 ^a
<i>Coeligena coeligena</i>	1	1	6 ^a
<i>Coeligena iris</i>	1	1	6.9 ^a
<i>Coeligena torquata</i>	1	1	4.9 ^a
<i>Coereba flaveola</i>	2	1	10 ^a
<i>Colaptes auratus</i>	2	2	131.5 ^a
<i>Colaptes campestris</i>	3	1	158 ^a
<i>Colaptes melanochloros</i>	2	1	127.3 ^a
<i>Colaptes pitius</i>	1	1	380 ^a
<i>Colaptes punctigula</i>	2	1	62 ^a
<i>Colibri delphinae</i>	1	1	6.4 ^a
<i>Colibri serrirostris</i>	3	1	6.7 ^a
<i>Colibri thalassinus</i>	2	1	5.9 ^a

<i>Colinus cristatus</i>	2	1	134.5 ^a
<i>Colinus striatus</i>	2	1	51.1 ^a
<i>Collocalia esculenta</i>	1	1	6.3 ^a
<i>Colonia colonus</i>	1	1	18.3 ^a
<i>Columba araucana</i>	1	1	24 ^b
<i>Columba arquatrix</i>	2	1	400 ^a
<i>Columba cayennensis</i>	3	1	229 ^a
<i>Columba fasciata</i>	2	1	366.3 ^a
<i>Columba livia</i>	3	1	354.2 ^a
<i>Columba maculosa</i>	2	1	347 ^a
<i>Columba nigristrois</i>	2	1	176 ^a
<i>Columba oenas</i>	3	1	290.8 ^a
<i>Columba palumbus</i>	1	1	490 ^a
<i>Columba picazuro</i>	3	1	279 ^a
<i>Columba plumbea</i>	1	1	178.8 ^a
<i>Columba speciosa</i>	1	1	258.5 ^a
<i>Columba subvinacea</i>	2	1	162.5 ^a

<i>Columba unicincta</i>	2	1	389.1 ^a
<i>Columbina cruziana</i>	3	1	50.1 ^a
<i>Columbina minuta</i>	2	1	32.6 ^a
<i>Columbina passerina</i>	2	1	35.4 ^a
<i>Columbina picui</i>	3	3	47 ^a
<i>Columbina squammata</i>	3	1	52.9 ^a
<i>Columbina talpacoti</i>	3	2	46 ^a
<i>Conioptilon mcilhennyi</i>	1	1	90 ^a
<i>Conirostrum cinereum</i>	2	1	8.8 ^a
<i>Conirostrum leucogenys</i>	2	1	7 ^a
<i>Conirostrum sitticolor</i>	1	1	11.9 ^a
<i>Conirostrum speciosum</i>	2	1	8.8 ^a
<i>Conopias albobittata</i>	1	1	24.4 ^a
<i>Conopias parva</i>	1	1	21 ^a
<i>Conopias trivirgata</i>	1	1	24.4 ^f
<i>Conopophaga aurita</i>	1	1	26.3 ^a
<i>Conopophaga lineata</i>	1	1	25.2 ^a

<i>Conopophaga melanogaster</i>	1	1	42.5 ^a
<i>Conopophaga melanops</i>	1	1	20.1 ^a
<i>Conopophaga peruviana</i>	1	1	23 ^a
<i>Contopus albogularis</i>	2	2	9.6 ^a
<i>Contopus cinereus</i>	2	2	11.6 ^a
<i>Contopus cooperi</i>	1	3	32.1 ^a
<i>Contopus fumigatus</i>	1	1	19.9 ^a
<i>Contopus nigrescens</i>	1	1	9.5 ^a
<i>Contopus pertinax</i>	2	1	27.2 ^a
<i>Contopus sordidulus</i>	1	3	13.1 ^a
<i>Contopus virens</i>	1	3	13.9 ^a
<i>Copsychus malabaricus</i>	1	1	29.2 ^a
<i>Copsychus saularis</i>	1	1	36 ^a
<i>Coracina azurea</i>	1	1	46.1 ^a
<i>Coracina caesia</i>	1	1	47.3 ^a
<i>Coracina fimbriata</i>	1	1	29.3 ^a
<i>Coracina melaschistos</i>	1	1	43 ^a

<i>Coracina striata</i>	1	1	111 ^a
<i>Coragyps atratus</i>	3	2	1881.7 ^a
<i>Corapipo gutturalis</i>	1	1	8.2 ^a
<i>Corapipo leucorrhoa</i>	1	1	11.7 ^a
<i>Corvus albicollis</i>	3	2	894 ^a
<i>Corvus albus</i>	3	2	524 ^a
<i>Corvus brachyrhynchos</i>	2	2	448.8 ^a
<i>Corvus corax</i>	3	2	928 ^a
<i>Corvus cornix</i>	3	1	499 ^a
<i>Corvus corone</i>	3	1	570 ^a
<i>Corvus enca</i>	1	1	251 ^a
<i>Corvus frugilegus</i>	3	1	452.1 ^a
<i>Corvus levaillantii</i>	2	1	513.1 ^a
<i>Corvus macrorhynchos</i>	2	1	513.1 ^a
<i>Corvus monedula</i>	3	2	246 ^a
<i>Corvus ossifragus</i>	3	1	284.6 ^a
<i>Corvus torquatus</i>	3	1	470.5 ^a

<i>Corydon sumatranus</i>	1	1	140 ^a
<i>Corybospingus cucullatus</i>	1	1	14.3 ^a
<i>Corybospingus pileatus</i>	1	1	15.3 ^a
<i>Corythaeola cristata</i>	1	1	965 ^a
<i>Corythopsis delalandi</i>	1	1	15 ^a
<i>Corythopsis torquata</i>	1	1	14.3 ^a
<i>Cossypha anomala</i>	1	1	24.5 ^a
<i>Cossypha caffra</i>	2	1	28.5 ^a
<i>Cossypha cyanocampter</i>	1	1	26.7 ^a
<i>Cossypha dichroa</i>	1	1	45.8 ^a
<i>Cossypha heuglini</i>	2	1	34.8 ^a
<i>Cossypha natalensis</i>	2	1	34.1 ^a
<i>Cossypha niveicapilla</i>	2	1	36.7 ^a
<i>Cossypha polioptera</i>	1	1	18.7 ^a
<i>Cossyphicula roberti</i>	1	1	34 ^b
<i>Cotinga cayana</i>	1	1	61.8 ^a
<i>Cotinga cotinga</i>	1	1	54 ^a

<i>Cotinga maynana</i>	1	1	69 ^a
<i>Cotinga nattererii</i>	1	1	64.1 ^f
<i>Cotinga ridgwayi</i>	1	1	57.2 ^a
<i>Coturnix chinensis</i>	2	1	40.4 ^a
<i>Cranioleuca antisensis</i>	1	1	16.9 ^a
<i>Cranioleuca erythroptis</i>	1	1	16.9 ^a
<i>Cranioleuca gutturata</i>	1	1	14.9 ^a
<i>Cranioleuca obsoleta</i>	1	1	13.4 ^a
<i>Cranioleuca pallida</i>	1	1	11.5 ^a
<i>Cranioleuca semicinerea</i>	1	1	15 ^a
<i>Cranioleuca vulpina</i>	2	1	15.7 ^a
<i>Crax alberti</i>	1	1	2800 ^b
<i>Crax alector</i>	1	1	2872 ^a
<i>Crax fasciolata</i>	1	1	2600 ^a
<i>Crax rubra</i>	1	1	4133 ^a
<i>Criniger calurus</i>	1	1	37.7 ^a
<i>Criniger chloronotus</i>	1	1	45 ^a

<i>Crotophaga ani</i>	3	1	110.1 ^a
<i>Crotophaga major</i>	1	1	148.3 ^a
<i>Crotophaga sulcirostris</i>	3	1	82 ^a
<i>Crypsirina temia</i>	1	1	157 ^b
<i>Cryptospiza jacksoni</i>	1	1	13 ^a
<i>Cryptospiza reichenovii</i>	1	1	12.4 ^a
<i>Cryptospiza salvadorii</i>	1	1	12.2 ^a
<i>Cryptospiza shelleyi</i>	1	1	17.6 ^a
<i>Crypturellus atrocapillus</i>	1	1	453 ^a
<i>Crypturellus bartletti</i>	1	1	241 ^a
<i>Crypturellus brevirostris</i>	1	1	411.5 ^f
<i>Crypturellus cinereus</i>	1	1	506.6 ^a
<i>Crypturellus cinnamomeus</i>	1	1	422 ^a
<i>Crypturellus obsoletus</i>	1	1	443.4 ^a
<i>Crypturellus parvirostris</i>	2	1	199 ^a
<i>Crypturellus soui</i>	1	1	216.2 ^a
<i>Crypturellus strigulosus</i>	1	1	430.6 ^a

<i>Crypturellus tataupa</i>	2	1	218.8 ^a
<i>Crypturellus transfasciatus</i>	1	1	16.3 ^a
<i>Crypturellus undulatus</i>	1	1	564.4 ^a
<i>Crypturellus variegatus</i>	1	1	378 ^a
<i>Cuculus canorus</i>	1	3	111.4 ^a
<i>Cuculus clamosus</i>	2	3	78.8 ^a
<i>Cuculus fugax</i>	1	1	79.7 ^a
<i>Cuculus micropterus</i>	1	1	88.8 ^a
<i>Cuculus poliocephalus</i>	1	1	79.7 ^a
<i>Cuculus saturatus</i>	2	2	245 ^a
<i>Cuculus solitarius</i>	2	2	76.7 ^a
<i>Cuculus vagans</i>	1	1	57.7 ^a
<i>Culicicapa ceylonensis</i>	1	1	7.7 ^a
<i>Curaeus curaens</i>	3	1	10.9 ^a
<i>Cyanerpes caeruleus</i>	1	1	12 ^a
<i>Cyanerpes cyaneus</i>	1	1	14 ^a
<i>Cyanerpes lucidus</i>	1	1	11.4 ^a

<i>Cyanerpes nitidus</i>	1	1	9 ^a
<i>Cyanicterus cyanicterus</i>	1	1	34 ^a
<i>Cyanocitta cristata</i>	2	1	88 ^a
<i>Cyanocitta stelleri</i>	1	1	128 ^a
<i>Cyanocompsa brissonii</i>	2	1	27.5 ^a
<i>Cyanocompsa cyanooides</i>	1	1	32.5 ^a
<i>Cyanocorax affinis</i>	1	1	212 ^a
<i>Cyanocorax chrysops</i>	1	1	166 ^a
<i>Cyanocorax cristatellus</i>	2	1	178 ^a
<i>Cyanocorax violaceus</i>	1	1	262 ^a
<i>Cyanocorax yncas</i>	2	1	78.5 ^a
<i>Cyanolyca turcosa</i>	1	1	104 ^f
<i>Cyanoptila cyanomelana</i>	1	3	22.5 ^a
<i>Cyclarhis gujanensis</i>	2	1	28.8 ^a
<i>Cygnus olor</i>	3	3	10682 ^a
<i>Cymbilaimus lineatus</i>	1	1	35.8 ^a
<i>Cymbilaimus sanctaemariae</i>	1	1	28.7 ^a

<i>Cymbirhynchus macrorhynchos</i>	2	1	59.4 ^a
<i>Cyornis banyumas</i>	1	1	14.5 ^a
<i>Cyornis caeruleatus</i>	1	1	22.5 ^b
<i>Cyornis hainanus</i>	1	2	26.8 ^b
<i>Cyornis superbus</i>	1	1	27.3 ^b
<i>Cyornis tickelliae</i>	2	1	16.2 ^a
<i>Cyornis turcosus</i>	1	1	21.2 ^b
<i>Cyornis unicolor</i>	1	1	21 ^a
<i>Cyphorhinus aradus</i>	1	1	20.9 ^a
<i>Cyphorhinus phaeocephalus</i>	1	1	24.6 ^a
<i>Cypsiurus balasiensis</i>	1	1	9.1 ^a
<i>Dacnis cayana</i>	2	1	13 ^a
<i>Dacnis egregia</i>	2	1	39 ^f
<i>Dacnis flaviventer</i>	2	1	13 ^a
<i>Dacnis lineata</i>	2	1	11 ^a
<i>Dacnis venusta</i>	2	1	16.1 ^a

<i>Damophila julie</i>	1	1	3.3 ^a
<i>Daptrius americanus</i>	1	1	624 ^a
<i>Daptrius ater</i>	2	2	351.8 ^a
<i>Deconychura longicauda</i>	1	1	23.8 ^a
<i>Deconychura stictolaema</i>	1	1	18.5 ^a
<i>Delichon dasyptus</i>	2	3	18 ^a
<i>Delichon urbica</i>	3	3	14.5 ^a
<i>Dendragapus obscurus</i>	1	1	1047.3 ^a
<i>Dendrexetastes rufigula</i>	1	1	69.6 ^a
<i>Dendrocincla anabatina</i>	1	1	34.2 ^a
<i>Dendrocincla fuliginosa</i>	1	1	38.7 ^a
<i>Dendrocincla homochroa</i>	1	1	39.4 ^a
<i>Dendrocincla merula</i>	1	1	50 ^a
<i>Dendrocincla turdina</i>	1	1	39 ^a
<i>Dendrocincla tyrannina</i>	1	1	16.3 ^a
<i>Dendrocolaptes certhia</i>	1	1	68.7 ^a
<i>Dendrocolaptes hoffmannsi</i>	1	1	83 ^a

<i>Dendrocolaptes picumnus</i>	1	1	76.6 ^a
<i>Dendrocolaptes platyrostris</i>	1	1	61.7 ^a
<i>Dendrocopos canicapillus</i>	1	1	25.8 ^a
<i>Dendrocopos kizuki</i>	2	1	143 ^a
<i>Dendrocopos leucotos</i>	1	1	116.8 ^a
<i>Dendrocopos major</i>	1	1	74.9 ^a
<i>Dendrocopos medius</i>	1	1	59 ^a
<i>Dendrocopos minor</i>	1	1	19.8 ^a
<i>Dendrocopos syriacus</i>	2	1	76.7 ^a
<i>Dendrocygna autumnalis</i>	3	2	755.3 ^a
<i>Dendrocygna bicolor</i>	3	2	756.4 ^a
<i>Dendrocygna viduata</i>	3	2	690 ^a
<i>Dendroica castanea</i>	1	3	11.8 ^a
<i>Dendroica cerulea</i>	1	3	9 ^a
<i>Dendroica coronata</i>	1	3	11.9 ^a
<i>Dendroica discolor</i>	2	3	7.6 ^a
<i>Dendroica dominica</i>	2	3	9.7 ^a

<i>Dendroica fusca</i>	1	3	9.7 ^a
<i>Dendroica magnolia</i>	1	3	8.1 ^a
<i>Dendroica nigrescens</i>	1	3	8.7 ^a
<i>Dendroica occidentalis</i>	1	3	10.4 ^a
<i>Dendroica palmarum</i>	1	3	10.3 ^a
<i>Dendroica pensylvanica</i>	2	3	9.3 ^a
<i>Dendroica petechia</i>	1	3	10.2 ^a
<i>Dendroica pinus</i>	2	3	11.8 ^a
<i>Dendroica striata</i>	1	3	11.8 ^a
<i>Dendroica tigrina</i>	1	3	10 ^a
<i>Dendroica townsendi</i>	1	3	8.8 ^a
<i>Dendroica virens</i>	1	3	8.7 ^a
<i>Dendronanthus indicus</i>	1	3	16.4 ^a
<i>Dendropicos elliotii</i>	1	1	37.2 ^a
<i>Dendropicos fuscescens</i>	2	1	26 ^a
<i>Dendropicos gabonensis</i>	2	1	27 ^a
<i>Dendropicos griseocephalus</i>	1	1	40.9 ^a

<i>Dendropicos xantholophus</i>	1	1	81.5 ^a
<i>Derophtus accipitrinus</i>	1	1	246 ^a
<i>Dicaeum agile</i>	2	1	9 ^a
<i>Dicaeum chrysorrheum</i>	1	1	9 ^a
<i>Dicaeum concolor</i>	1	1	6.2 ^a
<i>Dicaeum cruentatum</i>	1	1	5.6 ^a
<i>Dicaeum ignipectus</i>	1	1	5.7 ^a
<i>Dicaeum minullum</i>	1	1	6.2 ^a
<i>Dicaeum trigonostigma</i>	1	1	7.1 ^a
<i>Dichrozona cincta</i>	1	1	16 ^a
<i>Dicrurus adsimilis</i>	3	1	40.3 ^a
<i>Dicrurus aeneus</i>	1	1	26 ^a
<i>Dicrurus annectans</i>	2	3	44 ^a
<i>Dicrurus atripennis</i>	1	1	38.9 ^a
<i>Dicrurus hottentottus</i>	1	1	79.2 ^a
<i>Dicrurus leucophaeus</i>	1	1	37.6 ^a
<i>Dicrurus ludwigii</i>	2	1	29.4 ^a

<i>Dicrurus modestus</i>	2	1	45.6 ^a
<i>Dicrurus paradiseus</i>	1	1	85.8 ^a
<i>Dicrurus remifer</i>	1	1	43.1 ^a
<i>Diglossa albilatera</i>	2	1	9.8 ^a
<i>Diglossa humeralis</i>	2	1	13.1 ^a
<i>Diglossopsis cyanea</i>	2	1	17 ^a
<i>Dinopium javanense</i>	2	1	72.8 ^a
<i>Dinopium rafflesii</i>	1	1	102 ^a
<i>Dioptrornis chocolatinus</i>	2	1	23 ^a
<i>Discosura longicauda</i>	1	1	3.4 ^a
<i>Dives dives</i>	2	1	91.3 ^a
<i>Dives warszewiczi</i>	2	1	73.3 ^a
<i>Dolichonyx oryzivorus</i>	1	3	31.5 ^a
<i>Dolospingus fringilloides</i>	2	1	12.7 ^a
<i>Donacobius atricapillus</i>	3	1	36.8 ^a
<i>Doryfera ludovicae</i>	3	1	110.1 ^a
<i>Dromococcyx pavoninus</i>	1	1	46.4 ^a

<i>Dromococcyx phasianellus</i>	1	1	84.5 ^a
<i>Dryophila caudata</i>	1	1	46.8 ^a
<i>Dryophila devillei</i>	1	1	10.2 ^a
<i>Dryophila ferruginea</i>	1	1	10.6 ^a
<i>Dryophila malura</i>	1	1	13 ^a
<i>Dryophila ochropyga</i>	1	1	10.5 ^a
<i>Dryophila squamata</i>	1	1	10.8 ^a
<i>Dryocopus javensis</i>	1	1	271.4 ^a
<i>Dryocopus lineatus</i>	2	1	183.2 ^a
<i>Dryocopus martius</i>	1	1	321 ^a
<i>Dryocopus pileatus</i>	1	1	286.6 ^a
<i>Dryoscopus angolensis</i>	1	1	36 ^a
<i>Dryoscopus cubla</i>	2	1	26.2 ^a
<i>Dryoscopus gambensis</i>	2	1	31.9 ^a
<i>Ducula aenea</i>	1	1	545 ^a
<i>Ducula badia</i>	2	1	486 ^a
<i>Dumetella carolinensis</i>	1	3	35.3 ^a

<i>Dysithamnus mentalis</i>	1	1	14.9 ^a
<i>Dysithamnus puncticeps</i>	1	1	14.9 ^a
<i>Dysithamnus stictothorax</i>	1	1	16.2 ^f
<i>Egretta thula</i>	3	2	371 ^a
<i>Elaenia albiceps</i>	2	2	15.5 ^a
<i>Elaenia chiriquensis</i>	3	2	15.4 ^a
<i>Elaenia cristata</i>	3	1	18.2 ^a
<i>Elaenia flavogaster</i>	3	1	24.8 ^a
<i>Elaenia mesoleuca</i>	1	2	17.6 ^a
<i>Elaenia obscura</i>	1	1	23.9 ^a
<i>Elaenia parvirostris</i>	2	3	13.8 ^a
<i>Elaenia ruficeps</i>	2	1	18.7 ^a
<i>Elaenia spectabilis</i>	1	3	27.3 ^a
<i>Elanoides forficatus</i>	2	3	416.2 ^a
<i>Elanus caeruleus</i>	3	2	259.8 ^a
<i>Elanus leucurus</i>	3	2	346 ^a
<i>Electron carinatum</i>	1	1	64.9 ^a

<i>Electron platyrhynchum</i>	1	1	73 ^a
<i>Elminia albicauda</i>	2	1	8.3 ^a
<i>Elminia albiventris</i>	1	1	8.9 ^a
<i>Elminia longicauda</i>	1	1	9.9 ^a
<i>Elminia_albonotata</i>	1	1	8 ^a
<i>Elminia_nigromitrata</i>	1	1	10 ^a
<i>Elvira chionura</i>	1	1	3 ^a
<i>Emberiza calandra</i>	3	2	48.5 ^a
<i>Emberiza cia</i>	3	2	23.4 ^a
<i>Emberiza cioides</i>	2	1	16.3 ^a
<i>Emberiza cirrus</i>	2	2	25.6 ^a
<i>Emberiza citrinella</i>	3	2	29.7 ^a
<i>Emberiza rustica</i>	2	3	20.5 ^a
<i>Emberiza schoeniclus</i>	2	2	18.4 ^a
<i>Emberiza spodocephala</i>	2	1	11.3 ^a
<i>Emberiza variabilis</i>	3	2	21 ^a
<i>Emberizoides herbicola</i>	3	1	27.6 ^a

<i>Empidonax alnorum</i>	1	3	12.7 ^a
<i>Empidonax difficilis</i>	1	3	10.7 ^a
<i>Empidonax flavescens</i>	1	1	12.5 ^a
<i>Empidonax flaviventris</i>	1	3	11.8 ^a
<i>Empidonax hammondi</i>	1	3	10.4 ^a
<i>Empidonax minimus</i>	1	3	10 ^a
<i>Empidonax oberholseri</i>	2	3	10.4 ^a
<i>Empidonax traillii</i>	2	3	13.4 ^a
<i>Empidonax virescens</i>	2	3	12.6 ^a
<i>Empidonax varius</i>	2	3	27.1 ^a
<i>Enicurus leschenaulti</i>	3	1	33.8 ^a
<i>Enicurus ruficapillus</i>	2	1	28.2 ^a
<i>Enicurus schistaceus</i>	3	1	31 ^a
<i>Ensifera ensifera</i>	1	1	11.2 ^a
<i>Eophona personata</i>	2	1	18.6 ^a
<i>Epinecrophyla feldsaai</i>	1	1	8.7 ^a
<i>Eremomela badiceps</i>	1	1	8.3 ^a

<i>Eriocnemis alinae</i>	1	1	5.8 ^a
<i>Eriocnemis vestitus</i>	2	1	4.5 ^a
<i>Eribacus akabige</i>	1	1	25.9 ^a
<i>Eribacus rubecula</i>	1	3	17.7 ^a
<i>Erpornis zantholeuca</i>	1	1	11.8 ^a
<i>Erythrocercus mccallii</i>	1	1	14.1 ^a
<i>Erythrura prasina</i>	2	1	14 ^b
<i>Estrilda astrild</i>	3	1	8.3 ^a
<i>Estrilda atricapilla</i>	2	1	7.8 ^a
<i>Estrilda melanotis</i>	2	1	6.5 ^a
<i>Estrilda nonnula</i>	3	1	7.7 ^a
<i>Eubucco bourcierii</i>	1	1	37 ^a
<i>Eubucco richardsoni</i>	1	1	33.9 ^a
<i>Eubucco tucinkae</i>	1	1	41.7 ^a
<i>Eucometis penicillata</i>	1	1	27 ^a
<i>Eudynamys scolopacea</i>	2	2	194.9 ^a
<i>Eugralla paradoxa</i>	2	2	200 ^a

<i>Eumyias thalassina</i>	1	1	18.1 ^a
<i>Eumyias thalassinus</i>	1	1	18.1 ^a
<i>Eupetes macrocerus</i>	1	1	101 ^b
<i>Eupetomena macroura</i>	3	1	9 ^a
<i>Eupherusa eximia</i>	1	1	4.3 ^a
<i>Euphonia affinis</i>	2	1	10 ^a
<i>Euphonia cayennensis</i>	1	1	14.3 ^a
<i>Euphonia chlorotica</i>	2	1	11 ^a
<i>Euphonia chrysopasta</i>	2	1	14 ^a
<i>Euphonia elegantissima</i>	2	1	15 ^a
<i>Euphonia fulvicrissa</i>	2	1	11 ^a
<i>Euphonia gouldi</i>	2	1	14 ^a
<i>Euphonia hirundinacea</i>	2	1	14 ^a
<i>Euphonia imitans</i>	2	1	14 ^a
<i>Euphonia lanirostris</i>	2	1	15 ^a
<i>Euphonia luteicapilla</i>	2	1	13 ^a
<i>Euphonia minuta</i>	1	1	10 ^a

<i>Euphonia musica</i>	2	1	14.7 ^a
<i>Euphonia pectoralis</i>	1	1	14.4 ^a
<i>Euphonia plumbea</i>	2	1	8.9 ^a
<i>Euphonia rufiventris</i>	2	1	14 ^a
<i>Euphonia saturata</i>	2	1	13.3 ^f
<i>Euphonia violacea</i>	2	1	15 ^a
<i>Euphonia xanthogaster</i>	2	1	13 ^a
<i>Eurostopodus macrotis</i>	1	1	140.6 ^a
<i>Eurostopodus temminckii</i>	2	1	95.9 ^a
<i>Eurylaimus javanicus</i>	1	1	78.2 ^a
<i>Eurylaimus ochromalus</i>	1	1	33.3 ^a
<i>Eurypyga belias</i>	2	1	210 ^a
<i>Eurystomus glaucurus</i>	2	1	110 ^a
<i>Eurystomus gularis</i>	2	1	96.3 ^a
<i>Eurystomus orientalis</i>	2	3	143 ^a
<i>Euscarthmus meloryphus</i>	2	1	6.8 ^a
<i>Euschistospiza cinereovinacea</i>	2	1	12 ^a

<i>Eutoxeres aquila</i>	1	1	10.6 ^a
<i>Eutoxeres condamini</i>	1	1	10.8 ^a
<i>Falci pennis canadensis</i>	1	1	473.7 ^a
<i>Falco ardosiaceus</i>	3	1	234.2 ^a
<i>Falco columbarius</i>	3	3	188.5 ^a
<i>Falco femoralis</i>	3	2	335.8 ^a
<i>Falco peregrinus</i>	3	2	760 ^a
<i>Falco ruficularis</i>	2	1	163.6 ^a
<i>Falco sparverius</i>	3	3	114.6 ^a
<i>Falco subbuteo</i>	3	3	208.2 ^a
<i>Falco tinnunculus</i>	3	3	183.2 ^a
<i>Ficedula albicollis</i>	1	3	12.7 ^a
<i>Ficedula dumetoria</i>	1	1	9.4 ^a
<i>Ficedula hodgsonii</i>	1	1	10 ^a
<i>Ficedula hypoleuca</i>	1	3	13.8 ^a
<i>Ficedula mugimaki</i>	2	3	11.7 ^a
<i>Ficedula narcissina</i>	2	3	13.1 ^a

<i>Ficedula parva</i>	1	3	10.8 ^a
<i>Florisuga fusca</i>	2	1	8.1 ^c
<i>Florisuga mellivora</i>	1	1	7.4 ^a
<i>Fluvicola nengeta</i>	3	2	21 ^a
<i>Fluvicola pica</i>	3	2	12.3 ^a
<i>Formicarius analis</i>	1	1	62.2 ^a
<i>Formicarius colma</i>	1	1	47 ^a
<i>Formicarius rufifrons</i>	1	1	55.5 ^a
<i>Formicivora grisea</i>	2	1	10.4 ^a
<i>Formicivora serrana</i>	2	1	11.4 ^a
<i>Forpus coelestis</i>	2	1	26.1 ^a
<i>Forpus conspicillatus</i>	2	1	26.4 ^a
<i>Forpus sclateri</i>	2	1	27.2 ^a
<i>Forpus xanthopterygius</i>	2	1	31 ^a
<i>Fraseria ocreata</i>	1	1	35.5 ^a
<i>Frederickena unduligera</i>	1	1	83 ^a
<i>Frederickena viridis</i>	1	1	76 ^a

<i>Fringilla coelebs</i>	1	2	23.8 ^a
<i>Fringilla montifringilla</i>	1	3	23.2 ^a
<i>Furnarius leucopus</i>	2	1	54.8 ^a
<i>Furnarius rufus</i>	3	1	46.4 ^a
<i>Galbula albirostris</i>	1	1	22.1 ^a
<i>Galbula chalcothorax</i>	1	1	23.9 ^a
<i>Galbula cyanescens</i>	1	1	25.3 ^a
<i>Galbula cyanicollis</i>	1	1	23.2 ^a
<i>Galbula dea</i>	1	1	27.4 ^a
<i>Galbula leucogastra</i>	2	1	15.9 ^a
<i>Galbula ruficauda</i>	2	1	26.5 ^a
<i>Gallinago gallinago</i>	3	3	112.9 ^a
<i>Gallinago paraguayae</i>	3	2	109.4 ^a
<i>Gallinago stenura</i>	3	3	113 ^a
<i>Gallinula chloropus</i>	3	1	339.6 ^a
<i>Gallirallus striatus</i>	2	1	116 ^a
<i>Galloperdix spadicea</i>	2	1	359.1 ^a

<i>Gallus gallus</i>	1	1	751.7 ^a
<i>Gamponyx swainsonii</i>	2	1	92.9 ^a
<i>Garrulax chinensis</i>	2	1	78.6 ^f
<i>Garrulax leucolophus</i>	2	1	123.4 ^a
<i>Garrulax monileger</i>	1	1	83.7 ^a
<i>Garrulus glandarius</i>	1	2	159.5 ^a
<i>Gecinulus viridis</i>	1	1	69.2 ^a
<i>Geopelia striata</i>	2	1	56.6 ^a
<i>Geothlypis aequinoctialis</i>	3	1	13.1 ^c
<i>Geothlypis poliocephala</i>	2	1	14.6 ^a
<i>Geothlypis trichas</i>	1	3	9.5 ^a
<i>Geotrygon chiriquensis</i>	1	1	308 ^a
<i>Geotrygon montana</i>	1	1	133.9 ^a
<i>Geotrygon saphirina</i>	1	1	203 ^a
<i>Geotrygon veraguensis</i>	1	1	155 ^a
<i>Geotrygon violacea</i>	1	1	95.2 ^a
<i>Geranospiza caerulescens</i>	2	1	302.5 ^a

<i>Gerygone sulphurea</i>	2	1	5.6 ^a
<i>Glaucidium brasilianum</i>	2	1	75.1 ^c
<i>Glaucidium brodiei</i>	1	1	59.4 ^a
<i>Glaucidium cuculoides</i>	1	1	198.2 ^a
<i>Glaucidium barchyi</i>	1	1	58.3 ^a
<i>Glaucidium jardinii</i>	2	1	66.9 ^a
<i>Glaucidium minutissimum</i>	1	1	50 ^a
<i>Glaucidium nanum</i>	2	2	83.7 ^a
<i>Glaucidium passerinum</i>	1	2	57.9 ^a
<i>Glaucidium peruanum</i>	2	1	62.4 ^a
<i>Glaucidium sjostedti</i>	1	1	139 ^a
<i>Glaucidium tephronotum</i>	1	1	87.4 ^a
<i>Glaucis aenea</i>	1	1	4.8 ^a
<i>Glaucis hirsuta</i>	1	1	6.8 ^a
<i>Glaucis hirsutus</i>	1	1	6.8 ^a
<i>Glyphorhynchus spirurus</i>	1	1	14.6 ^a
<i>Gorsachius goisagi</i>	2	1	159 ^a

<i>Gorsachius melanolophus</i>	2	2	412.3 ^a
<i>Gracula religiosa</i>	1	1	192 ^a
<i>Grallaria dignissima</i>	1	1	110 ^a
<i>Grallaria guatemalensis</i>	1	1	94.2 ^a
<i>Grallaria ruficapilla</i>	1	1	82.7 ^a
<i>Grallaria rufula</i>	1	1	35.7 ^a
<i>Grallaria squamigera</i>	1	1	136 ^a
<i>Grallaria varia</i>	1	1	119 ^a
<i>Grallaricula flavirostris</i>	1	1	17.7 ^a
<i>Granatellus pelzelni</i>	1	1	11.2 ^a
<i>Grænia vittata</i>	1	1	15.5 ^a
<i>Graydidascalus brachyurus</i>	1	1	159 ^a
<i>Griseotyrannus aurantioatrocristatus</i>	2	3	33 ^a
<i>Grus grus</i>	3	3	5500 ^a
<i>Gubernates yetapa</i>	3	2	68.4 ^a
<i>Guirra guirra</i>	3	1	141 ^a

<i>Guiraca caerulea</i>	2	3	27.4 ^a
<i>Guttera pucherani</i>	1	1	1149 ^a
<i>Gymnobucco bonapartei</i>	1	2	63 ^a
<i>Gymnobucco calvus</i>	1	2	54.5 ^a
<i>Gymnocichla nudiceps</i>	1	1	30.5 ^a
<i>Gymnoderus foetidus</i>	2	1	275.5 ^a
<i>Gymnomystax mexicanus</i>	3	1	98 ^a
<i>Gymnopathys leucaspis</i>	1	1	27.2 ^a
<i>Gymnopathys rufigula</i>	1	1	29.1 ^a
<i>Gymnopathys salvini</i>	1	1	25.9 ^a
<i>Gymnostinops bifasciatus</i>	1	1	335.7 ^a
<i>Gymnostinops montezuma</i>	1	1	309.9 ^a
<i>Gypobierax angolensis</i>	2	1	1600 ^a
<i>Gypsositta vulturina</i>	1	1	145 ^c
<i>Habia fuscicauda</i>	2	1	36.8 ^a
<i>Habia gutturalis</i>	2	1	33.4 ^c
<i>Habia rubica</i>	1	1	32.5 ^a

<i>Haematoderus militaris</i>	2	1	227 ^a
<i>Hakyon badia</i>	1	1	57.9 ^a
<i>Hakyon coromanda</i>	1	3	77.5 ^a
<i>Hakyon leucocephala</i>	2	2	41.8 ^a
<i>Hakyon malimbica</i>	1	1	91.8 ^a
<i>Hakyon smyrnensis</i>	2	2	91.4 ^a
<i>Haliaeetus leucogaster</i>	2	1	2827 ^a
<i>Hapalopsittaca pyrrhops</i>	1	1	103 ^a
<i>Haplospiza unicolor</i>	1	2	15.3 ^a
<i>Harpactes diardii</i>	1	1	98.3 ^a
<i>Harpactes duvaucelii</i>	1	1	38.9 ^a
<i>Harpactes erythrocephalus</i>	1	1	80.3 ^a
<i>Harpactes fasciatus</i>	1	1	62 ^a
<i>Harpactes kasumba</i>	1	1	96.9 ^a
<i>Harpactes oreskios</i>	1	1	57.3 ^a
<i>Harpactes orrhophaeus</i>	1	1	52.8 ^a
<i>Harpagus bidentatus</i>	2	1	215.3 ^a

<i>Harpagus diodon</i>	2	3	200 ^a
<i>Harpia harpyja</i>	1	1	4800 ^a
<i>Heliactin bilophum</i>	2	1	4.4 ^a
<i>Heliangelus exortis</i>	1	1	4.6 ^a
<i>Heliobletus contaminatus</i>	1	1	14 ^a
<i>Heliodoxa aurescens</i>	1	1	6.4 ^a
<i>Heliodoxa jacula</i>	1	1	8.1 ^a
<i>Heliodoxa rubinoides</i>	1	1	6.7 ^a
<i>Heliodoxa schreibersii</i>	1	1	9.2 ^a
<i>Heliomaster longirostris</i>	1	1	6.6 ^a
<i>Heliomaster squamosus</i>	1	1	5.9 ^a
<i>Heliornis fulica</i>	2	1	132 ^a
<i>Heliothryx aurita</i>	1	1	5.4 ^a
<i>Heliothryx barroti</i>	1	1	5.5 ^a
<i>Hellmayrea gularis</i>	1	1	12.4 ^a
<i>Helmittheros vermivorus</i>	1	3	14.2 ^a
<i>Hemicircus canente</i>	1	1	43 ^a

<i>Hemicircus concretus</i>	1	1	29.4 ^a
<i>Hemignathus flavus</i>	1	1	12.6 ^a
<i>Hemiprocne comata</i>	1	1	20.6 ^a
<i>Hemiprocne longipennis</i>	1	1	42.8 ^a
<i>Hemipus birundinaceus</i>	1	1	10.2 ^a
<i>Hemipus picatus</i>	1	1	9 ^a
<i>Hemispingus superciliaris</i>	1	1	14 ^a
<i>Hemispingus verticalis</i>	1	1	14 ^a
<i>Hemitesia neumanni</i>	1	1	14.2 ^a
<i>Hemithraupis flavicollis</i>	1	1	12.8 ^a
<i>Hemithraupis guira</i>	1	1	12 ^a
<i>Hemithraupis ruficapilla</i>	1	1	11 ^a
<i>Hemitriccus diops</i>	1	1	10 ^a
<i>Hemitriccus granadensis</i>	1	1	7.9 ^a
<i>Hemitriccus griseipectus</i>	1	1	8.9 ^c
<i>Hemitriccus iobannis</i>	1	1	10 ^a
<i>Hemitriccus josephinae</i>	1	1	10.8 ^a

<i>Hemitriccus minimus</i>	2	1	7 ^a
<i>Hemitriccus minor</i>	1	1	7.4 ^a
<i>Hemitriccus nidipendulus</i>	1	1	7.5 ^a
<i>Hemitriccus obsoletus</i>	1	1	11.6 ^a
<i>Hemitriccus orbitatus</i>	1	1	9.7 ^a
<i>Hemitriccus striaticollis</i>	2	1	8.6 ^a
<i>Hemitriccus zosterops</i>	1	1	8.7 ^a
<i>Hemixos flava</i>	1	2	32.5 ^a
<i>Henicorbina leucophrys</i>	1	1	16.2 ^a
<i>Henicorbina leucosticta</i>	1	1	14.2 ^a
<i>Herpetotheres cachinnans</i>	2	1	623.6 ^a
<i>Herpsilochmus atricapillus</i>	1	1	9.9 ^a
<i>Herpsilochmus dorsimaculatus</i>	1	1	9.9 ^a
<i>Herpsilochmus dugandi</i>	1	1	10.3 ^a
<i>Herpsilochmus pileatus</i>	2	1	8.9 ^a
<i>Herpsilochmus rufimarginatus</i>	1	1	10.6 ^a
<i>Herpsilochmus stictocephalus</i>	1	1	8.8 ^a

<i>Herpsilochmus sticturus</i>	1	1	9.5 ^a
<i>Hesperiphona vespertina</i>	1	2	57.3 ^a
<i>Heterocercus flavivertex</i>	1	1	21.3 ^a
<i>Heterocercus lineatus</i>	1	1	21.2 ^a
<i>Heterospingus xanthopygius</i>	1	1	38.8 ^a
<i>Hieraaetus ayresii</i>	3	1	806.1 ^a
<i>Hieraaetus kienerii</i>	2	1	799 ^a
<i>Hieraaetus pennatus</i>	1	3	823.8 ^a
<i>Hierococcyx nasicolor</i>	1	3	79.7 ^a
<i>Hierococcyx sparveroides</i>	1	3	151 ^a
<i>Himantornis haematopus</i>	1	1	585.7 ^a
<i>Hippolais icterina</i>	1	3	13.2 ^a
<i>Hirundapus cochinchinensis</i>	1	3	83.8 ^a
<i>Hirundapus giganteus</i>	1	3	127 ^a
<i>Hirundinea ferruginea</i>	3	2	30.6 ^a
<i>Hirundo abyssinica</i>	3	1	13.5 ^a
<i>Hirundo angolensis</i>	3	1	17.8 ^a

<i>Hirundo daurica</i>	3	3	22.2 ^a
<i>Hirundo fuliginosa</i>	1	2	17.6 ^f
<i>Hirundo fuligula</i>	3	2	21.1 ^a
<i>Hirundo pyrrhonota</i>	3	3	21.6 ^a
<i>Hirundo rustica</i>	3	3	17.9 ^a
<i>Hirundo senegalensis</i>	3	1	45.4 ^a
<i>Hirundo tabitica</i>	3	2	17.8 ^a
<i>Hydropsalis brasiliana</i>	2	2	58.5 ^a
<i>Hydropsalis climacocerca</i>	2	1	46 ^a
<i>Hylexetastes perrotii</i>	1	1	103.1 ^a
<i>Hylexetastes stresemanni</i>	1	1	111 ^a
<i>Hylexetastes uniformis</i>	1	1	117 ^f
<i>Hylia prasina</i>	1	1	13.1 ^a
<i>Hylocharis chrysura</i>	2	1	4.5 ^a
<i>Hylocharis cyanus</i>	1	1	3.3 ^a
<i>Hylocharis eliciae</i>	2	1	3.6 ^a
<i>Hylocharis sapphirina</i>	2	1	4.4 ^a

<i>Hylocichla mustelina</i>	1	3	50.1 ^a
<i>Hylocistestes subulatus</i>	1	1	28.8 ^a
<i>Hylopezus berlepschi</i>	1	1	48 ^a
<i>Hylopezus macularius</i>	1	1	44.2 ^a
<i>Hylopezus perspicillatus</i>	1	1	43 ^a
<i>Hylophilus amaurocephalus</i>	1	1	11.6 ^f
<i>Hylophilus brunneiceps</i>	2	1	9.9 ^a
<i>Hylophilus decurtatus</i>	2	1	8.5 ^a
<i>Hylophilus flavipes</i>	2	1	11.7 ^a
<i>Hylophilus hypoxanthus</i>	1	1	17 ^a
<i>Hylophilus muscicapinus</i>	1	1	11.7 ^a
<i>Hylophilus ochraceiceps</i>	1	1	11.6 ^a
<i>Hylophilus poicilotis</i>	1	1	10.4 ^a
<i>Hylophilus semibrunneus</i>	1	1	11.4 ^a
<i>Hylophilus semicinereus</i>	1	1	13 ^a
<i>Hylophilus thoracicus</i>	1	1	13.8 ^a
<i>Hylophylax naevia</i>	1	1	14.2 ^a

<i>Hylophylax naevioides</i>	1	1	17.8 ^a
<i>Hylophylax poecilinota</i>	1	1	18.4 ^a
<i>Hylophylax punctulata</i>	1	1	12.4 ^a
<i>Hypargos niveoguttatus</i>	2	1	14.6 ^a
<i>Hypnelus ruficollis</i>	2	1	49.8 ^a
<i>Hypocnemis cantator</i>	1	1	12.3 ^a
<i>Hypocnemis hypoxantha</i>	1	1	11.5 ^a
<i>Hypocnemis striata</i>	1	1	12.3 ^a
<i>Hypocnemoides maculicauda</i>	1	1	11.8 ^a
<i>Hypocnemoides melanopogon</i>	1	1	14.1 ^a
<i>Hypoedaleus guttatus</i>	1	1	38.8 ^a
<i>Hypogramma hypogrammicum</i>	1	1	11.4 ^a
<i>Hypothymis azurea</i>	1	1	11.1 ^a
<i>Hypsipetes leucocephalus</i>	1	3	51.8 ^a
<i>Ichthyophaga humilis</i>	3	1	782 ^a
<i>Icteria virens</i>	2	1	24.9 ^a

<i>Icterus auricapillus</i>	2	1	32.2 ^a
<i>Icterus cayanensis</i>	2	1	35.4 ^a
<i>Icterus chrysater</i>	2	1	53.4 ^a
<i>Icterus galbula</i>	1	3	32.8 ^a
<i>Icterus gularis</i>	2	1	55.2 ^a
<i>Icterus mesomelas</i>	2	1	39.9 ^a
<i>Icterus nigrogularis</i>	2	1	40.2 ^a
<i>Icterus spurius</i>	2	3	19.4 ^a
<i>Ictinaetus malayensis</i>	1	1	1028 ^a
<i>Ictinia mississippiensis</i>	2	3	276 ^a
<i>Ictinia plumbea</i>	2	1	249.9 ^a
<i>Ilicura militaris</i>	1	1	12.7 ^a
<i>Illadopsis albipectus</i>	1	1	30.6 ^a
<i>Illadopsis cleaveri</i>	1	1	30.8 ^a
<i>Illadopsis fulvescens</i>	1	1	27.9 ^a
<i>Illadopsis pyrrhoptera</i>	1	1	24.8 ^a
<i>Illadopsis rufipennis</i>	1	1	22.2 ^a

<i>Indicator archipelagicus</i>	1	1	32.4 ^a
<i>Indicator conirostris</i>	1	1	31.8 ^a
<i>Indicator exilis</i>	1	1	17.9 ^a
<i>Indicator indicator</i>	2	1	50.4 ^a
<i>Indicator maculatus</i>	1	1	46.8 ^a
<i>Indicator minor</i>	2	1	28.2 ^a
<i>Indicator pumilio</i>	1	1	13.1 ^a
<i>Indicator variegatus</i>	2	1	48.8 ^a
<i>Indicator willcocksi</i>	1	1	15.4 ^a
<i>Inezia inornata</i>	2	3	12 ^a
<i>Inezia subflava</i>	1	1	8.4 ^a
<i>Iodopleura fusca</i>	1	1	15.3 ^a
<i>Iodopleura isabellae</i>	1	1	20 ^a
<i>Iole olivacea</i>	1	1	24 ^b
<i>Iole propinqua</i>	1	1	25.9 ^a
<i>Irena puella</i>	1	1	64.9 ^a
<i>Ispidina lecontei</i>	1	1	9.5 ^a

<i>Ispidina picta</i>	1	1	12.7 ^a
<i>Ixobrychus cinnamomeus</i>	1	1	126.5 ^a
<i>Ixonotus guttatus</i>	1	1	35.2 ^a
<i>Ixos malaccensis</i>	1	1	37.3 ^a
<i>Jabiru mycteria</i>	3	1	5996.3 ^a
<i>Jacamerops aureus</i>	1	1	62.9 ^a
<i>Jacana jacana</i>	3	1	106.2 ^a
<i>Junco hyemalis</i>	2	3	19.5 ^a
<i>Jynx torquilla</i>	1	3	35 ^a
<i>Kakamega poliothorax</i>	1	1	36.4 ^a
<i>Kenopia striata</i>	1	1	19.9 ^a
<i>Ketupa ketupu</i>	2	1	1293 ^a
<i>Klais guimeti</i>	1	1	2.6 ^a
<i>Knipolegus cyanirostris</i>	2	2	15.4 ^a
<i>Knipolegus lophotes</i>	3	1	31.8 ^a
<i>Knipolegus poecilocercus</i>	2	1	14.3 ^a
<i>Knipolegus poecilurus</i>	2	1	14.6 ^a

<i>Lacedo pulchella</i>	1	1	47.3 ^a
<i>Lafresnaya lafresnayi</i>	1	1	5.3 ^a
<i>Lagopus lagopus</i>	3	1	566.9 ^a
<i>Lalage nigra</i>	2	1	28.4 ^a
<i>Lampornis castaneiventris</i>	1	1	5.1 ^a
<i>Lamprosar tanagrinus</i>	1	1	117.3 ^b
<i>Lamprospiza melanolenca</i>	1	1	34 ^a
<i>Lamprotornis purpureiceps</i>	1	1	62.8 ^a
<i>Lamprotornis splendidus</i>	1	2	123.1 ^a
<i>Laniarius aethiopicus</i>	2	1	49.4 ^a
<i>Laniarius ferrugineus</i>	2	1	46.8 ^a
<i>Laniarius fuelleborni</i>	1	1	46.4 ^a
<i>Laniarius luehderi</i>	2	1	42.9 ^a
<i>Laniarius poensis</i>	1	1	44.9 ^a
<i>Laniisoma elegans</i>	1	1	47.4 ^a
<i>Lanio aurantius</i>	1	1	35 ^a
<i>Lanio fulvus</i>	1	1	24 ^a

<i>Lanio versicolor</i>	1	1	17 ^a
<i>Laniocera hypopyrra</i>	1	1	51 ^a
<i>Laniocera rufescens</i>	1	1	48.1 ^a
<i>Lanius collurio</i>	3	3	28.4 ^a
<i>Lanius cristatus</i>	3	3	32.2 ^a
<i>Lanius mackinnoni</i>	3	1	34.3 ^a
<i>Lanius tephronotus</i>	3	3	46.4 ^a
<i>Lanius tigrinus</i>	2	3	30.1 ^a
<i>Laterallus albigularis</i>	2	1	47.3 ^a
<i>Laterallus leucopyrrhus</i>	1	1	45.5 ^a
<i>Laterallus melanophaius</i>	2	1	52.1 ^a
<i>Lathrotriccus euleri</i>	1	2	11.3 ^a
<i>Lathrotriccus griseipectus</i>	1	1	17 ^a
<i>Legatus leucophaius</i>	1	1	22.2 ^a
<i>Lepidocolaptes affinis</i>	1	1	35 ^a
<i>Lepidocolaptes albolineatus</i>	1	1	20.3 ^a
<i>Lepidocolaptes angustirostris</i>	2	1	29.6 ^a

<i>Lepidocolaptes fuscus</i>	1	1	21.8 ^a
<i>Lepidocolaptes souleyetii</i>	2	1	25.7 ^a
<i>Lepidocolaptes squamatus</i>	1	1	28.1 ^a
<i>Lepidopyga goudoti</i>	2	1	4 ^a
<i>Lepidothrix coronata</i>	1	1	8.3 ^a
<i>Lepidothrix iris</i>	1	1	8 ^a
<i>Lepidothrix nattereri</i>	1	1	8.4 ^a
<i>Lepidothrix serena</i>	1	1	11.4 ^a
<i>Leptasthenura aegithaloides</i>	2	1	13.7 ^a
<i>Leptodon cayanensis</i>	2	1	474 ^a
<i>Leptopogon amaurocephalus</i>	1	1	11.7 ^a
<i>Leptopogon rufipectus</i>	1	1	13.4 ^a
<i>Leptopogon superviliaris</i>	1	1	11.7 ^a
<i>Leptosittaca branickii</i>	1	1	66.5 ^b
<i>Leptotila cassini</i>	1	1	159 ^a
<i>Leptotila ochraceiventris</i>	1	2	146 ^a
<i>Leptotila pallida</i>	1	1	632 ^a

<i>Leptotila plumbeiceps</i>	1	1	170 ^a
<i>Leptotila rufaxilla</i>	1	1	157 ^a
<i>Leptotila verreauxi</i>	1	1	146.9 ^a
<i>Leucochloris albicollis</i>	1	1	6.3 ^a
<i>Leucopternis albicollis</i>	1	1	703.7 ^a
<i>Leucopternis kubli</i>	1	1	352.1 ^a
<i>Leucopternis melanops</i>	1	1	312 ^a
<i>Leucopternis occidentalis</i>	1	1	660 ^a
<i>Leucopternis plumbea</i>	1	1	482 ^a
<i>Leucopternis princeps</i>	1	1	1000 ^a
<i>Leucopternis schistacea</i>	1	1	1000 ^a
<i>Leucopternis semiplumbea</i>	1	1	285 ^a
<i>Linurgus olivaceus</i>	1	1	20.3 ^a
<i>Liosceles thoracicus</i>	1	1	40.5 ^a
<i>Lipaugus unirufus</i>	1	1	82.1 ^a
<i>Lipaugus vociferans</i>	1	1	75.4 ^a
<i>Lochmias nematura</i>	1	1	22 ^a

<i>Locustella fluviatilis</i>	3	3	16.1 ^a
<i>Locustella luscinioides</i>	3	3	13.9 ^a
<i>Lonchura fuscans</i>	3	1	9.5 ^a
<i>Lonchura leucogastra</i>	2	1	11.4 ^a
<i>Lonchura maja</i>	3	1	12.5 ^a
<i>Lonchura malacca</i>	3	1	12.6 ^a
<i>Lonchura punctulata</i>	2	1	13.6 ^a
<i>Lonchura striata</i>	3	1	12.3 ^a
<i>Lophaetus occipitalis</i>	2	1	1067 ^a
<i>Lophornis adorabilis</i>	1	1	2.7 ^a
<i>Lophornis chalybeus</i>	1	1	3 ^a
<i>Lophornis delattrei</i>	2	1	2.8 ^a
<i>Lophotrix cristata</i>	1	1	538.7 ^a
<i>Lophotriccus galeatus</i>	2	1	6.6 ^a
<i>Lophotriccus pileatus</i>	1	1	7.5 ^a
<i>Lophotriccus vitiensis</i>	1	1	6.7 ^a
<i>Lophura diardi</i>	1	1	1088.9 ^a

<i>Lophura erythrophthalma</i>	1	1	934.3 ^a
<i>Lophura ignita</i>	1	1	1768.1 ^a
<i>Lophura nycthemera</i>	1	1	1230 ^a
<i>Loriculus galgulus</i>	1	1	28 ^a
<i>Loriculus vernalis</i>	1	1	35.7 ^a
<i>Loxia curvirostra</i>	1	2	38.3 ^a
<i>Loxia leucoptera</i>	1	3	28.7 ^a
<i>Loxia pytyopsittacus</i>	1	2	53 ^a
<i>Lullula arborea</i>	3	2	26.9 ^a
<i>Lurocalis semitorquatus</i>	1	1	75.9 ^a
<i>Luscinia cyane</i>	2	3	14.8 ^a
<i>Luscinia luscinia</i>	1	3	23.8 ^a
<i>Luscinia megarhynchos</i>	1	3	19.6 ^a
<i>Lybius bidentatus</i>	1	1	79.3 ^a
<i>Lyrurus tetrix</i>	3	1	1068.7 ^a
<i>Machaeropterus pyrocephalus</i>	1	1	9.5 ^a
<i>Machaeropterus regulus</i>	1	1	9.3 ^a

<i>Macheiramphus alcinus</i>	2	1	650 ^a
<i>Machetornis rixosus</i>	3	1	29.6 ^a
<i>Mackenziaena leachii</i>	1	1	70.2 ^a
<i>Mackenziaena severa</i>	2	1	51.8 ^a
<i>Macronous gularis</i>	2	1	11.8 ^a
<i>Macronous ptilosus</i>	1	1	18 ^a
<i>Macropygia unchall</i>	1	1	168 ^a
<i>Macrosphenus concolor</i>	1	1	13.9 ^a
<i>Macrosphenus flavicans</i>	1	1	13.4 ^a
<i>Malacocincla abbotti</i>	2	1	26.8 ^a
<i>Malacocincla malaccensis</i>	2	1	21.4 ^a
<i>Malacocincla sepiarium</i>	1	1	25.5 ^a
<i>Malaconotus lagdeni</i>	1	1	92.3 ^a
<i>Malacopteron affine</i>	1	1	17.1 ^a
<i>Malacopteron cinereum</i>	1	1	18.1 ^a
<i>Malacopteron magnirostre</i>	1	1	19.3 ^a
<i>Malacopteron magnum</i>	1	1	27.2 ^a

<i>Malacoptila fusca</i>	1	1	45.4 ^a
<i>Malacoptila mystacalis</i>	1	1	49 ^a
<i>Malacoptila panamensis</i>	1	1	42.6 ^a
<i>Malacoptila rufa</i>	1	1	47.5 ^a
<i>Malacoptila semicincta</i>	1	1	44 ^a
<i>Malacoptila striata</i>	1	1	44.1 ^a
<i>Malimbus erythrogaster</i>	1	1	46.3 ^f
<i>Malimbus malimbicus</i>	1	1	36.1 ^a
<i>Malimbus nitens</i>	1	1	37.8 ^a
<i>Malimbus racheliae</i>	1	1	28 ^b
<i>Malimbus rubricollis</i>	1	1	42.2 ^a
<i>Malimbus scutatus</i>	1	1	30 ^a
<i>Manacus aurantiacus</i>	1	1	15.5 ^a
<i>Manacus candei</i>	1	1	19.8 ^a
<i>Manacus manacus</i>	1	1	16.7 ^a
<i>Manacus vitellinus</i>	1	1	18.2 ^a
<i>Mandingoa nitidula</i>	2	1	21.4 ^a

<i>Margarornis squamiger</i>	1	1	17.3 ^a
<i>Masius chrysopterus</i>	1	1	11.5 ^a
<i>Mecocerculus calopterus</i>	1	1	7 ^a
<i>Mecocerculus leucophrys</i>	1	1	11 ^a
<i>Mecocerculus poecilocercus</i>	1	1	7.7 ^a
<i>Mecocerculus stictopterus</i>	1	1	9.4 ^a
<i>Megaceryle alcyon</i>	3	2	148 ^a
<i>Megaceryle maxima</i>	3	1	325 ^a
<i>Megaceryle torquata</i>	3	1	317 ^a
<i>Megalaima australis</i>	1	1	33.3 ^a
<i>Megalaima chrysopogon</i>	1	1	151 ^a
<i>Megalaima faiostricta</i>	1	1	93 ^f
<i>Megalaima haemacephala</i>	2	1	44.5 ^a
<i>Megalaima henricii</i>	1	1	74.1 ^a
<i>Megalaima incognita</i>	1	1	93 ^f
<i>Megalaima mystacophanos</i>	1	1	77.4 ^a
<i>Megalaima rafflesii</i>	1	1	118 ^a

<i>Megarynchus pitangua</i>	2	1	69.9 ^a
<i>Megastictus margaritatus</i>	1	1	20.2 ^a
<i>Meiglyptes jugularis</i>	1	1	53.4 ^a
<i>Meiglyptes tristis</i>	1	1	31.7 ^a
<i>Meiglyptes tukki</i>	1	1	53.1 ^a
<i>Melaenornis ardesiacus</i>	2	1	29 ^a
<i>Melanerpes aurifrons</i>	2	1	80.8 ^a
<i>Melanerpes candidus</i>	2	1	108 ^a
<i>Melanerpes carolinus</i>	1	1	69.5 ^a
<i>Melanerpes cruentatus</i>	1	1	58.1 ^a
<i>Melanerpes erythrocephalus</i>	1	1	71.6 ^a
<i>Melanerpes flavifrons</i>	1	1	57.8 ^a
<i>Melanerpes formicivorus</i>	1	1	7.8 ^a
<i>Melanerpes hoffmannii</i>	2	1	72.2 ^a
<i>Melanerpes pucherani</i>	1	1	62.4 ^a
<i>Melanerpes pulcher</i>	1	1	52.5 ^a
<i>Melanerpes rubricapillus</i>	2	1	52.3 ^a

<i>Melanochlora sultanea</i>	2	1	37.6 ^a
<i>Melanotrochilus fuscus</i>	2	1	8.1 ^c
<i>Meleagris gallopavo</i>	2	1	5791.4 ^a
<i>Melospiza georgiana</i>	2	3	16.1 ^a
<i>Melospiza lincolnii</i>	2	3	16.6 ^a
<i>Melospiza melodia</i>	2	2	21.9 ^a
<i>Merops albicollis</i>	3	2	25.9 ^a
<i>Merops gularis</i>	1	1	27.3 ^a
<i>Merops leschenaulti</i>	2	1	27.2 ^a
<i>Merops muelleri</i>	1	1	22.5 ^a
<i>Merops oreobates</i>	3	1	24 ^a
<i>Merops philippinus</i>	2	3	34 ^a
<i>Merops viridis</i>	2	1	34.8 ^a
<i>Mesembrinibis cayennensis</i>	2	1	756 ^a
<i>Metallura tyrianthina</i>	1	1	3.8 ^a
<i>Micrastur buckleyi</i>	1	1	464 ^f
<i>Micrastur gilvicollis</i>	1	1	209 ^a

<i>Micrastur mintoni</i>	1	1	209 ^a
<i>Micrastur mirandollei</i>	1	1	549 ^a
<i>Micrastur ruficollis</i>	1	1	177.6 ^a
<i>Micrastur semitorquatus</i>	1	1	621.7 ^a
<i>Microbates cinereiventris</i>	2	1	11.9 ^a
<i>Microbates collaris</i>	1	1	10.2 ^a
<i>Microcerculus bambla</i>	1	1	19.1 ^a
<i>Microcerculus marginatus</i>	1	1	18.2 ^a
<i>Microcerculus philomela</i>	1	1	17.8 ^a
<i>Microhierax fringillarius</i>	2	1	42.9 ^a
<i>Microhierax latifrons</i>	2	1	41.2 ^a
<i>Micromonacha lanceolata</i>	1	1	19.8 ^a
<i>Microrhopias quixensis</i>	1	1	7.9 ^a
<i>Milvago chimachima</i>	3	1	315.2 ^a
<i>Milvago chimango</i>	2	2	35.3 ^a
<i>Milvus migrans</i>	3	3	734.1 ^a
<i>Mimus gilvus</i>	2	1	52 ^a

<i>Mimus saturninus</i>	2	2	63.7 ^a
<i>Mionectes macconnelli</i>	1	1	13.5 ^a
<i>Mionectes oleagineus</i>	1	1	11.2 ^a
<i>Mionectes olivaceus</i>	1	1	15.3 ^a
<i>Mionectes rufiventris</i>	1	1	13.3 ^a
<i>Mionectes striaticollis</i>	1	1	15 ^a
<i>Mitrephanes phaeocercus</i>	1	1	8.6 ^a
<i>Mitu mitu</i>	1	1	2850.5 ^a
<i>Mitu salvini</i>	1	1	3100 ^a
<i>Mitu tuberosa</i>	1	1	2769.5 ^a
<i>Mniotilta varia</i>	1	3	10.9 ^a
<i>Modulatrix stictigula</i>	1	1	30.5 ^a
<i>Molothrus aeneus</i>	2	1	62.6 ^a
<i>Molothrus ater</i>	2	3	40.3 ^a
<i>Molothrus bonariensis</i>	2	1	41.5 ^a
<i>Molothrus oryzivora</i>	1	1	181.9 ^a
<i>Momotus momota</i>	1	1	115 ^a

<i>Monasa atra</i>	1	1	90.4 ^a
<i>Monasa flavirostris</i>	1	1	62.8 ^a
<i>Monasa morphoeus</i>	1	1	87.9 ^a
<i>Monasa nigrifrons</i>	1	1	80.7 ^a
<i>Monticola gularis</i>	2	3	34.6 ^a
<i>Morphnus guianensis</i>	1	1	1750 ^a
<i>Motacilla aguimp</i>	2	1	27 ^a
<i>Motacilla alba</i>	3	3	23.9 ^a
<i>Motacilla capensis</i>	3	2	20.8 ^a
<i>Motacilla cinerea</i>	3	2	17.2 ^a
<i>Motacilla clara</i>	3	1	17.9 ^a
<i>Motacilla flava</i>	3	3	17.7 ^a
<i>Mulleripicus pulverulentus</i>	1	1	450.2 ^a
<i>Muscicapa adusta</i>	2	1	10.4 ^a
<i>Muscicapa caerulescens</i>	2	1	17.6 ^a
<i>Muscicapa comitata</i>	1	1	14.1 ^a
<i>Muscicapa dauurica</i>	1	3	9.8 ^a

<i>Muscicapa infuscata</i>	2	1	17.7 ^a
<i>Muscicapa olivascens</i>	1	1	15.4 ^a
<i>Muscicapa sibirica</i>	1	3	13.2 ^a
<i>Muscicapa striata</i>	1	3	15.9 ^a
<i>Muscisaxicola fluvialis</i>	3	1	14.7 ^a
<i>Musophaga johnstoni</i>	1	1	240 ^a
<i>Musophaga porphyreolopha</i>	2	1	259 ^a
<i>Musophaga rossae</i>	2	1	361.4 ^a
<i>Myadestes melanops</i>	1	1	32.1 ^a
<i>Myadestes ralloides</i>	1	1	12.3 ^a
<i>Mycteria americana</i>	3	2	2554.5 ^a
<i>Myiarchus barbirostris</i>	1	1	13.4 ^a
<i>Myiarchus crinitus</i>	1	3	32.1 ^a
<i>Myiarchus ferox</i>	2	1	27.5 ^a
<i>Myiarchus panamensis</i>	2	1	31.7 ^a
<i>Myiarchus swainsoni</i>	2	3	25.1 ^a
<i>Myiarchus tuberculifer</i>	1	1	17.7 ^a

<i>Myiarchus tyrannulus</i>	2	2	35.5 ^a
<i>Myiobius atricaudus</i>	1	1	10 ^a
<i>Myiobius barbatulus</i>	1	1	11.9 ^c
<i>Myiobius erythrurus</i>	1	1	7.4 ^a
<i>Myiobius sulphureipygius</i>	1	1	11.6 ^a
<i>Myioborus melanocephalus</i>	1	1	11.3 ^a
<i>Myioborus miniatus</i>	1	1	9 ^a
<i>Myiodynastes chrysocephalus</i>	2	1	38.3 ^a
<i>Myiodynastes luteiventris</i>	2	3	46.9 ^a
<i>Myiodynastes maculatus</i>	2	3	43.2 ^a
<i>Myiopagis caniceps</i>	1	1	10.5 ^a
<i>Myiopagis gaimardii</i>	1	1	12 ^a
<i>Myiopagis subplacens</i>	2	1	15.9 ^a
<i>Myiopagis viridicata</i>	2	3	11.5 ^a
<i>Myioparus griseigularis</i>	1	1	12.6 ^a
<i>Myioparus plumbeus</i>	1	1	12.8 ^a
<i>Myiophobus fasciatus</i>	2	3	9.9 ^a

<i>Myiophobus flavicans</i>	1	1	13.4 ^a
<i>Myiornis atricapillus</i>	1	1	5.7 ^a
<i>Myiornis auricularis</i>	1	1	5.3 ^a
<i>Myiornis ecaudatus</i>	1	1	5.3 ^a
<i>Myiotheretes fumigatus</i>	2	1	33.4 ^a
<i>Myiotriccus ornatus</i>	1	1	15.3 ^a
<i>Myiozetetes cayanensis</i>	2	1	25.9 ^a
<i>Myiozetetes granadensis</i>	2	1	29.3 ^a
<i>Myiozetetes luteiventris</i>	2	1	19 ^a
<i>Myiozetetes similis</i>	2	2	28 ^a
<i>Myiophonus caeruleus</i>	1	1	158 ^a
<i>Myiophonus borsfieldii</i>	3	1	117 ^a
<i>Myrmeciza atrothorax</i>	1	1	16.1 ^a
<i>Myrmeciza disjuncta</i>	1	1	15 ^a
<i>Myrmeciza exsul</i>	1	1	26.5 ^a
<i>Myrmeciza ferruginea</i>	1	1	26.1 ^a
<i>Myrmeciza fortis</i>	1	1	45.9 ^a

<i>Myrmeciza goeldii</i>	1	1	42 ^a
<i>Myrmeciza hemimelaena</i>	1	1	16 ^a
<i>Myrmeciza hypertybra</i>	1	1	41 ^a
<i>Myrmeciza laemosticta</i>	1	1	25 ^a
<i>Myrmeciza longipes</i>	1	1	27.9 ^a
<i>Myrmeciza loricata</i>	1	1	29.8 ^f
<i>Myrmeciza ruficauda</i>	1	1	29.8 ^f
<i>Myrmeciza squamosa</i>	1	1	18.5 ^a
<i>Myrmecocichla formicivora</i>	3	1	41.6 ^a
<i>Myrmoborus leucophrys</i>	1	1	21.5 ^a
<i>Myrmoborus myotherinus</i>	1	1	18.8 ^a
<i>Myrmornis torquata</i>	1	1	46.7 ^a
<i>Myrmothera campanisona</i>	1	1	47 ^a
<i>Myrmotherula ambigua</i>	1	1	6.7 ^a
<i>Myrmotherula assimilis</i>	1	1	9.3 ^a
<i>Myrmotherula axillaris</i>	1	1	8.1 ^a
<i>Myrmotherula brachyura</i>	1	1	6.4 ^a

<i>Myrmotherula cherriei</i>	2	1	8.2 ^a
<i>Myrmotherula erythrura</i>	1	1	11.4 ^a
<i>Myrmotherula fulviventris</i>	1	1	10.4 ^a
<i>Myrmotherula gularis</i>	1	1	11.3 ^a
<i>Myrmotherula guttata</i>	1	1	11 ^a
<i>Myrmotherula gutturalis</i>	1	1	9.2 ^a
<i>Myrmotherula haematonota</i>	1	1	8.7 ^a
<i>Myrmotherula huxwelli</i>	1	1	10.7 ^a
<i>Myrmotherula ignota</i>	1	1	7.1 ^a
<i>Myrmotherula iberingi</i>	1	1	8 ^a
<i>Myrmotherula leucophthalma</i>	1	1	9.4 ^a
<i>Myrmotherula longipennis</i>	1	1	9.4 ^a
<i>Myrmotherula menetriesii</i>	1	1	8.6 ^a
<i>Myrmotherula multostriata</i>	2	1	8 ^a
<i>Myrmotherula ornata</i>	1	1	9.4 ^a
<i>Myrmotherula schisticolor</i>	1	1	9.6 ^a
<i>Myrmotherula sclateri</i>	1	1	8.4 ^a

<i>Myrmotherula surinamensis</i>	1	1	8.3 ^a
<i>Napothera atrigularis</i>	1	1	37 ^b
<i>Napothera macrodactyla</i>	1	1	48.7 ^a
<i>Nasica longirostris</i>	1	1	92 ^a
<i>Neafrapus cassini</i>	1	2	40 ^a
<i>Nectarinia afra</i>	2	1	11.1 ^a
<i>Nectarinia amethystina</i>	2	2	13.7 ^a
<i>Nectarinia batesi</i>	1	1	6.3 ^a
<i>Nectarinia chloropygia</i>	2	2	6.2 ^a
<i>Nectarinia cyanolaema</i>	2	1	16.8 ^a
<i>Nectarinia famosa</i>	2	1	14.4 ^a
<i>Nectarinia johannae</i>	1	1	13 ^a
<i>Nectarinia jugularis</i>	2	1	9 ^a
<i>Nectarinia kilimensis</i>	2	1	15.7 ^a
<i>Nectarinia mediocris</i>	2	1	8.2 ^a
<i>Nectarinia olivacea</i>	1	1	9.7 ^a
<i>Nectarinia purpureiventris</i>	2	1	11.5 ^a

<i>Nectarinia regia</i>	2	1	6.6 ^a
<i>Nectarinia reichenowi</i>	3	2	14.7 ^a
<i>Nectarinia rubescens</i>	2	1	10.3 ^a
<i>Nectarinia seimundi</i>	1	1	6.6 ^a
<i>Nectarinia senegalensis</i>	2	2	12.2 ^a
<i>Nectarinia superba</i>	1	1	16.8 ^a
<i>Nectarinia venusta</i>	2	1	6.6 ^a
<i>Nectarinia verticalis</i>	2	1	14.4 ^a
<i>Nemosia pileata</i>	2	1	16 ^a
<i>Neobelidon tibialis</i>	1	1	10.6 ^a
<i>Neochen jubata</i>	3	2	1250 ^a
<i>Neocossyphus fraseri</i>	1	1	35.7 ^a
<i>Neocossyphus poensis</i>	1	1	51.8 ^a
<i>Neoctantes niger</i>	1	1	32 ^a
<i>Neomorphus geoffroyi</i>	1	1	355 ^a
<i>Neopelma aurifrons</i>	1	1	14 ^a
<i>Neopelma chrysocephalum</i>	1	1	15.5 ^a

<i>Neopelma pallescens</i>	2	1	18.2 ^a
<i>Neopipo cinnamomea</i>	1	1	7.7 ^a
<i>Nesocharis ansorgei</i>	2	1	8.4 ^a
<i>Nicator chloris</i>	1	1	46.1 ^a
<i>Nicator gularis</i>	1	1	45.6 ^a
<i>Nigrita bicolor</i>	1	1	10.5 ^a
<i>Nigrita canicapilla</i>	1	1	18.7 ^a
<i>Nigrita fusconota</i>	1	1	9.1 ^a
<i>Niltava sundara</i>	1	3	21.1 ^a
<i>Ninox scutulata</i>	1	3	191 ^a
<i>Nisaetus nipalensis</i>	1	1	3005 ^a
<i>Nonnula amaurocephala</i>	1	1	15.5 ^a
<i>Nonnula brunnea</i>	1	1	14.5 ^c
<i>Nonnula rubecula</i>	1	1	18.4 ^a
<i>Nonnula ruficapilla</i>	1	1	22 ^a
<i>Notharchus hyperrhynchus</i>	2	1	95.9 ^a
<i>Notharchus macrorhynchus</i>	2	1	95.9 ^a

<i>Notharchus ordii</i>	1	1	52.8 ^a
<i>Notharchus pectoralis</i>	1	1	69.1 ^a
<i>Notharchus tectus</i>	2	1	26.8 ^a
<i>Nothocrax urumutum</i>	1	1	1700 ^a
<i>Notiochelidon cyanoleuca</i>	3	2	9.7 ^a
<i>Notiochelidon murina</i>	3	2	11.7 ^a
<i>Nucifraga caryocatactes</i>	1	2	182.5 ^a
<i>Nyctanassa violacea</i>	3	2	681.7 ^a
<i>Nyctibius aethereus</i>	1	1	440.5 ^a
<i>Nyctibius bracteatus</i>	1	1	51.7 ^a
<i>Nyctibius grandis</i>	1	1	547 ^a
<i>Nyctibius griseus</i>	1	1	172 ^a
<i>Nyctibius leucopterus</i>	1	1	86.4 ^a
<i>Nycticorax nycticorax</i>	3	3	810 ^a
<i>Nyctidromus albicollis</i>	2	1	57.8 ^a
<i>Nyctiphrynus ocellatus</i>	1	1	39.6 ^a
<i>Nyctiprogne leucopyga</i>	1	1	27.3 ^a

<i>Nyctyornis amictus</i>	1	1	71.9 ^a
<i>Nyctyornis atbertoni</i>	1	1	84.2 ^a
<i>Nystalus radiatus</i>	1	1	61 ^a
<i>Nystalus striolatus</i>	1	1	47 ^a
<i>Ochthoeca cinnamomeiventris</i>	1	1	12 ^a
<i>Ochthoeca fumicolor</i>	2	1	16.6 ^a
<i>Ochthoeca rufipectoralis</i>	2	1	10 ^a
<i>Ochthornis littoralis</i>	3	1	13.4 ^a
<i>Ocreatus underwoodii</i>	1	1	3 ^a
<i>Odontophorus capueira</i>	1	1	425.4 ^a
<i>Odontophorus gujanensis</i>	1	1	314.1 ^a
<i>Odontophorus guttatus</i>	1	1	303.8 ^a
<i>Odontophorus stellatus</i>	1	1	335.8 ^a
<i>Odontorhynchus cinereus</i>	1	1	11 ^a
<i>Oenanthe oenanthe</i>	3	3	25.4 ^a
<i>Oncostoma cinereigulare</i>	2	1	6.4 ^a
<i>Oncostoma olivaceum</i>	1	1	6.6 ^a

<i>Onychognathus fulgidus</i>	1	2	109 ^a
<i>Onychognathus morio</i>	3	1	132.9 ^a
<i>Onychognathus tenuirostris</i>	1	1	118 ^a
<i>Onychognathus walleri</i>	1	1	93 ^a
<i>Onychorhynchus coronatus</i>	1	2	14 ^a
<i>Onychorhynchus occidentalis</i>	2	1	6.3 ^b
<i>Opisthocomus hoazin</i>	2	1	696 ^a
<i>Oporornis agilis</i>	1	3	13.3 ^a
<i>Oporornis formosus</i>	1	3	14 ^a
<i>Oporornis philadelphia</i>	1	3	11.7 ^a
<i>Oporornis tolmiei</i>	1	3	10.4 ^a
<i>Orchesticus abeillei</i>	1	1	31.5 ^a
<i>Oreortyx pictus</i>	2	1	233 ^a
<i>Oriolus auratus</i>	2	1	66.2 ^a
<i>Oriolus brachyrhynchus</i>	1	1	49 ^a
<i>Oriolus chinensis</i>	2	2	82.6 ^a
<i>Oriolus chlorocephalus</i>	1	2	70.9 ^f

<i>Oriolus larvatus</i>	2	1	66.1 ^a
<i>Oriolus mellianus</i>	1	3	93 ^b
<i>Oriolus oriolus</i>	1	3	79 ^a
<i>Oriolus xanthonotus</i>	1	1	45 ^a
<i>Ornithion brunneicapillum</i>	1	1	7.1 ^a
<i>Ornithion inermis</i>	1	1	7 ^a
<i>Ornithion semiflavum</i>	1	1	6.6 ^a
<i>Ortalis araucana</i>	1	1	550 ^a
<i>Ortalis cinereiceps</i>	1	1	493 ^a
<i>Ortalis columbiana</i>	1	1	547.7 ^a
<i>Ortalis erythroptera</i>	2	3	416.2 ^a
<i>Ortalis guttata</i>	1	1	547.7 ^a
<i>Ortalis motmot</i>	1	1	520 ^a
<i>Ortalis ruficeps</i>	1	1	520 ^a
<i>Ortalis vetula</i>	1	1	562.6 ^a
<i>Orthogonys chloricterus</i>	1	1	40 ^b
<i>Orthotomus atrogularis</i>	1	1	7.7 ^a

<i>Orthotomus ruficeps</i>	1	1	8.1 ^f
<i>Orthotomus sepium</i>	1	1	8.4 ^a
<i>Orthotomus sericeus</i>	2	1	10.8 ^a
<i>Orthotomus sutorius</i>	2	1	7.5 ^a
<i>Oryzoborus angolensis</i>	2	1	13 ^a
<i>Oryzoborus crassirostris</i>	2	2	20.9 ^a
<i>Oryzoborus funereus</i>	2	1	13 ^a
<i>Otus atricapillus</i>	1	1	118.9 ^a
<i>Otus bakkamoena</i>	1	1	146.3 ^c
<i>Otus choliba</i>	2	1	132 ^a
<i>Otus lettia</i>	2	1	142.3 ^c
<i>Otus rufescens</i>	1	1	76.5 ^a
<i>Otus scops</i>	2	3	92 ^a
<i>Otus spilocephalus</i>	1	1	67.5 ^a
<i>Otus sunia</i>	1	1	78.1 ^a
<i>Otus watsonii</i>	1	1	129 ^a
<i>Otus guatemalae</i>	1	1	125 ^a

<i>Oxyruncus cristatus</i>	1	1	42 ^a
<i>Pachyphantes supervilosus</i>	2	1	21.4 ^a
<i>Pachyramphus aglaiae</i>	1	2	29.7 ^a
<i>Pachyramphus albogriseus</i>	1	1	17.3 ^a
<i>Pachyramphus castaneus</i>	1	1	19.5 ^a
<i>Pachyramphus cinnamomeus</i>	1	1	20.3 ^a
<i>Pachyramphus homochrous</i>	1	2	37.3 ^a
<i>Pachyramphus marginatus</i>	1	1	18.4 ^a
<i>Pachyramphus minor</i>	1	2	36.6 ^a
<i>Pachyramphus polychopterus</i>	2	1	20.8 ^a
<i>Pachyramphus rufus</i>	2	1	18.8 ^a
<i>Pachyramphus surinamus</i>	1	2	19.7 ^a
<i>Pachyramphus validus</i>	2	3	43 ^a
<i>Pachyramphus versicolor</i>	1	1	15.5 ^a
<i>Pachyramphus viridis</i>	2	2	21 ^a
<i>Pandion haliaetus</i>	3	3	1483.2 ^a
<i>Panyptila cayennensis</i>	1	1	21.1 ^a

<i>Pardirallus nigricans</i>	2	1	217 ^a
<i>Parkerthraustes humeralis</i>	1	1	37 ^a
<i>Parmoptila jamesoni</i>	2	2	9.5 ^a
<i>parmoptila woodhousei</i>	1	2	9.4 ^a
<i>Paroaria gularis</i>	2	1	23.6 ^a
<i>Parula pitiayumi</i>	2	1	6.8 ^a
<i>Parus ater</i>	1	1	9.2 ^a
<i>Parus atricapillus</i>	1	1	10.8 ^a
<i>Parus bicolor</i>	1	1	21.6 ^a
<i>Parus caeruleus</i>	1	1	13.3 ^c
<i>Parus cristatus</i>	1	1	11 ^a
<i>Parus fasciiventer</i>	1	1	15.2 ^a
<i>Parus hudsonicus</i>	1	1	9.8 ^a
<i>Parus major</i>	1	1	16.3 ^a
<i>Parus montanus</i>	1	1	11.1 ^a
<i>Parus niger</i>	2	1	21.5 ^a
<i>Parus palustris</i>	1	1	11.1 ^a

<i>Parus rufescens</i>	1	1	9.4 ^a
<i>Parus varius</i>	2	1	8.6 ^a
<i>Passer domesticus</i>	3	1	26.5 ^a
<i>Passer griseus</i>	3	1	27.8 ^a
<i>Passer montanus</i>	2	1	21.4 ^a
<i>Passerella iliaca</i>	1	3	33.3 ^a
<i>Passerina amoena</i>	2	3	15.5 ^a
<i>Passerina cyanea</i>	2	3	14.7 ^a
<i>Patagioenas plumbea</i>	1	1	178.8 ^a
<i>Pelargopsis capensis</i>	2	2	178.7 ^a
<i>Peliperdix lathamii</i>	1	1	269 ^a
<i>Pellorneum buettikoferi</i>	1	1	25 ^a
<i>Pellorneum capistratum</i>	1	1	25.1 ^a
<i>Pellorneum ruficeps</i>	2	1	26 ^a
<i>Penelope barbata</i>	1	1	1179.7 ^f
<i>Penelope jacquacu</i>	1	1	1487.9 ^a
<i>Penelope marail</i>	1	1	879.7 ^a

<i>Penelope obscura</i>	1	1	1770 ^a
<i>Penelope purpurascens</i>	1	1	2060 ^a
<i>Penelope superciliaris</i>	1	1	895 ^a
<i>Pernostola leucostigma</i>	1	1	25.7 ^a
<i>Pernostola lophotes</i>	1	1	28 ^a
<i>Pernostola rufifrons</i>	1	1	23.9 ^a
<i>Pernostola schistacea</i>	1	1	26.4 ^f
<i>Perdix perdix</i>	3	1	405.3 ^a
<i>Pericrocotus cantonensis</i>	1	3	16.3 ^f
<i>Pericrocotus cinnamomeus</i>	2	2	8.6 ^a
<i>Pericrocotus divaricatus</i>	2	3	24.6 ^a
<i>Pericrocotus flammeus</i>	1	1	23.3 ^a
<i>Pericrocotus igneus</i>	2	1	15 ^a
<i>Pericrocotus roseus</i>	2	1	19.2 ^a
<i>Pericrocotus speciosus</i>	1	1	23.3 ^a
<i>Periporphyrus erythromelas</i>	1	1	48 ^a
<i>Perisoreus canadensis</i>	1	1	71.6 ^a

<i>Perisoreus infaustus</i>	1	1	84.4 ^a
<i>Perissocephalus tricolor</i>	1	1	338.9 ^a
<i>Pernis apivorus</i>	1	3	754.4 ^a
<i>Pernis ptilorhynchus</i>	1	3	754.4 ^a
<i>Pernis ptilorhynchus</i>	2	1	1141.1 ^a
<i>Phacellodomus erythrophthalmus</i>	2	1	24.5 ^a
<i>Phacellodomus ferrugineigula</i>	2	1	24.5 ^a
<i>Phacellodomus rufifrons</i>	2	1	24.6 ^a
<i>Phaenicophaeus chlorophaeus</i>	2	1	51.2 ^a
<i>Phaenicophaeus curvirostris</i>	1	1	154 ^a
<i>Phaenicophaeus diardi</i>	1	1	62.1 ^a
<i>Phaenicophaeus javanicus</i>	2	1	109 ^a
<i>Phaenicophaeus sumatranus</i>	1	1	91.9 ^a
<i>Phaenicophaeus tristis</i>	1	1	117 ^a
<i>Phaenostictus mcleannani</i>	1	1	51.1 ^a
<i>Phaeochroa curvierii</i>	2	1	8.9 ^a

<i>Phaeomyias murina</i>	2	1	10 ^a
<i>Phaethornis anthophilus</i>	1	1	5.9 ^a
<i>Phaethornis atrimentalis</i>	1	1	2.6 ^a
<i>Phaethornis bourcieri</i>	1	1	4.3 ^a
<i>Phaethornis eurynome</i>	1	1	5.3 ^a
<i>Phaethornis guy</i>	1	1	6.3 ^a
<i>Phaethornis hispidus</i>	1	1	5 ^a
<i>Phaethornis longirostris</i>	1	1	6 ^a
<i>Phaethornis longuemareus</i>	1	1	3 ^a
<i>Phaethornis malaris</i>	1	1	5.9 ^a
<i>Phaethornis philippii</i>	1	1	5 ^a
<i>Phaethornis pretrei</i>	2	1	5.6 ^a
<i>Phaethornis ruber</i>	1	1	2.4 ^a
<i>Phaethornis striigularis</i>	1	1	3 ^a
<i>Phaethornis stuarti</i>	1	1	2.9 ^a
<i>Phaethornis supervilosus</i>	1	1	6.3 ^a
<i>Phaethornis syrmatophorus</i>	1	1	70.5 ^a

<i>Phaetusa simplex</i>	3	2	235 ^a
<i>Phalacrocorax brasilianus</i>	3	1	1814 ^a
<i>Phalaropus fulicarius</i>	3	3	55.6 ^a
<i>Pharomacrus auriceps</i>	1	1	180 ^a
<i>Pharomacrus pavoninus</i>	1	1	163 ^a
<i>Phasianus colchicus</i>	2	1	1120.3 ^a
<i>Phœnticus chrysogaster</i>	2	1	55.8 ^a
<i>Phœnticus ludovicianus</i>	2	3	42 ^a
<i>Phœnticus melanocephalus</i>	2	3	47.1 ^a
<i>Phibalura flavirostris</i>	2	3	46.5 ^a
<i>Philentoma pyrhopterum</i>	1	1	16.9 ^a
<i>Philentoma velatum</i>	1	1	26.1 ^a
<i>Philodice bryantae</i>	2	1	3.3 ^a
<i>Philohydor lictor</i>	2	1	25.5 ^a
<i>Philidor amaurotis</i>	1	1	19.2 ^a
<i>Philidor atricapillus</i>	1	1	22.2 ^a
<i>Philidor erythrocerus</i>	1	1	25.3 ^a

<i>Philydor erythropterus</i>	1	1	30 ^a
<i>Philydor lichtensteini</i>	1	1	21 ^a
<i>Philydor pyrrhodes</i>	1	1	30.5 ^a
<i>Philydor ruficaudatus</i>	1	1	30.1 ^a
<i>Philydor rufum</i>	1	1	25 ^a
<i>Philydor rufus</i>	1	1	25 ^a
<i>Phimosus infuscatus</i>	3	2	559 ^a
<i>Pblegopsis erythroptera</i>	1	2	54.6 ^a
<i>Pblegopsis nigromaculata</i>	1	2	45.2 ^a
<i>Pblogothraupis sanguinolenta</i>	2	1	41 ^a
<i>Phodilus badius</i>	2	1	280.2 ^a
<i>Phoebastria nigripes</i>	3	2	3300 ^a
<i>Phoenicircus carnifex</i>	1	1	82.9 ^a
<i>Phoenicircus nigricollis</i>	1	1	96.3 ^a
<i>Phoeniculus bollei</i>	1	1	54.6 ^a
<i>Phoeniculus purpureus</i>	2	1	73.2 ^a
<i>Phoenicurus phoenicurus</i>	1	3	14.6 ^a

<i>Pholidornis rusbiae</i>	1	1	6 ^a
<i>Phrygilus patagonicus</i>	3	3	14.9 ^a
<i>Phyllastrephus albigularis</i>	1	1	37.7 ^a
<i>Phyllastrephus baumanni</i>	2	1	27.5 ^a
<i>Phyllastrephus cerviniventris</i>	2	1	27 ^a
<i>Phyllastrephus fischeri</i>	1	1	27 ^a
<i>Phyllastrephus flavostriatus</i>	1	1	28.3 ^a
<i>Phyllastrephus icterinus</i>	1	1	18.9 ^a
<i>Phyllastrephus placidus</i>	1	1	26 ^a
<i>Phyllastrephus terrestris</i>	1	1	31.1 ^a
<i>Phyllastrephus xavieri</i>	1	1	22.6 ^a
<i>Phyllomyias burmeisteri</i>	1	1	11.1 ^a
<i>Phyllomyias fasciatus</i>	1	1	10.3 ^a
<i>Phyllomyias griseiceps</i>	1	1	7.2 ^a
<i>Phyllomyias zeledoni</i>	1	1	10.3 ^a
<i>Phylloscartes ceciliae</i>	1	1	7.6 ^a
<i>Phylloscartes eximius</i>	1	1	7.5 ^a

<i>Phylloscartes lanyoni</i>	2	1	8 ^a
<i>Phylloscartes paulistus</i>	1	1	7.5 ^a
<i>Phylloscartes ventralis</i>	1	1	8.3 ^a
<i>Phylloscartes virescens</i>	1	1	8.5 ^a
<i>Phylloscopus borealis</i>	1	3	10.9 ^a
<i>Phylloscopus borealoides</i>	1	3	9.3 ^a
<i>Phylloscopus claudiae</i>	1	3	7.6 ^a
<i>Phylloscopus collybita</i>	1	3	8.3 ^a
<i>Phylloscopus coronatus</i>	2	3	10.1 ^a
<i>Phylloscopus ibericus</i>	1	3	8.3 ^a
<i>Phylloscopus inornatus</i>	1	3	6.6 ^a
<i>Phylloscopus laetus</i>	1	1	9.5 ^a
<i>Phylloscopus plumbeitarsus</i>	1	3	8.1 ^a
<i>Phylloscopus ricketti</i>	2	1	10.5 ^b
<i>Phylloscopus ruficapillus</i>	1	1	7.7 ^a
<i>Phylloscopus schwarzi</i>	2	3	11.4 ^a
<i>Phylloscopus sibilatrix</i>	1	3	9.2 ^a

<i>Phylloscopus tenellipes</i>	2	3	11.8 ^a
<i>Phylloscopus trochiloides</i>	1	3	8.1 ^a
<i>Phylloscopus trochilus</i>	1	3	8.7 ^a
<i>Piaya cayana</i>	1	1	102 ^a
<i>Piaya melanogaster</i>	1	1	102.8 ^a
<i>Piaya minuta</i>	1	1	39.3 ^a
<i>Pica pica</i>	3	1	217.5 ^a
<i>Picathartes oreas</i>	1	1	220 ^a
<i>Picoides arcticus</i>	1	1	69.2 ^a
<i>Picoides lignarius</i>	2	1	6.2 ^a
<i>Picoides pubescens</i>	2	1	25.6 ^a
<i>Picoides tridactylus</i>	1	1	65.5 ^a
<i>Picoides villosus</i>	1	1	62.7 ^a
<i>Piculus aurulentus</i>	1	1	75 ^a
<i>Piculus chrysochloros</i>	1	1	88 ^a
<i>Piculus flavigula</i>	1	1	55 ^a
<i>Piculus rubiginosus</i>	2	1	55.8 ^a

<i>Piculus simplex</i>	1	1	55 ^a
<i>Picumnus albosquamatus</i>	2	1	11.9 ^a
<i>Picumnus aurifrons</i>	1	1	8.9 ^a
<i>Picumnus cirratus</i>	2	1	10.2 ^a
<i>Picumnus exilis</i>	1	1	9.3 ^a
<i>Picumnus fulvescens</i>	2	1	11.3 ^f
<i>Picumnus innominatus</i>	1	1	10.2 ^a
<i>Picumnus lafresnayi</i>	1	1	9.5 ^a
<i>Picumnus minutissimus</i>	2	1	13.3 ^a
<i>Picumnus olivaceus</i>	1	1	10.6 ^a
<i>Picumnus pumilus</i>	1	1	9.5 ^a
<i>Picumnus rufiventris</i>	1	1	21 ^a
<i>Picumnus temminckii</i>	2	1	11.5 ^a
<i>Picus avokera</i>	1	3	46.6 ^a
<i>Picus canus</i>	1	1	137 ^a
<i>Picus flavinucha</i>	1	1	176.8 ^a
<i>Picus mentalis</i>	1	1	99.7 ^a

<i>Picus mineaceus</i>	1	1	97.3 ^a
<i>Picus miniaceus</i>	1	1	97.3 ^a
<i>Picus puniceus</i>	1	1	79.1 ^a
<i>Picus viridanus</i>	1	1	111.4 ^a
<i>Picus viridis</i>	1	1	176 ^a
<i>Picus vittatus</i>	2	1	108.1 ^a
<i>Pilherodius pileatus</i>	3	1	568.6 ^a
<i>Pinicola enucleator</i>	1	2	56.4 ^a
<i>Pionites leucogaster</i>	1	1	155 ^a
<i>Pionites melanocephala</i>	1	1	148.8 ^a
<i>Pionopsitta barrabandi</i>	1	1	140 ^a
<i>Pionopsitta caica</i>	1	1	132 ^a
<i>Pionopsitta haematotis</i>	1	1	149 ^a
<i>Pionopsitta pileata</i>	1	1	119 ^a
<i>Pionopsitta pyralia</i>	1	1	132.5 ^f
<i>Pionus chalcopterus</i>	1	1	210 ^a
<i>Pionus fuscus</i>	1	1	206 ^a

<i>Pionus maximiliani</i>	1	1	293 ^a
<i>Pionus menstruus</i>	2	1	251 ^a
<i>Pionus senilis</i>	1	1	212 ^a
<i>Pipile cujubi</i>	1	1	1195.8 ^a
<i>Pipile cumanensis</i>	1	1	1239.2 ^a
<i>Pipile jacutinga</i>	1	1	1241 ^a
<i>Pipile pipile</i>	1	1	1816.6 ^a
<i>Pipilo chlorurus</i>	2	3	29.4 ^a
<i>Pipilo erythrophthalmus</i>	2	2	40 ^a
<i>Pipilo maculatus</i>	2	2	39.3 ^a
<i>Pipra chloromeros</i>	1	1	16.5 ^a
<i>Pipra erythrocephala</i>	1	1	13.4 ^a
<i>Pipra fasciicauda</i>	1	1	15.9 ^a
<i>Pipra filicauda</i>	1	1	15.4 ^a
<i>Pipra mentalis</i>	1	1	15 ^a
<i>Pipra pipra</i>	1	1	11.1 ^a
<i>Pipra rubrocapilla</i>	1	1	12 ^a

<i>Pipraeidea melanonota</i>	1	1	21 ^a
<i>Pipreola arcuata</i>	1	1	118 ^a
<i>Piprites chloris</i>	1	1	16 ^a
<i>Piranga bidentata</i>	1	2	36.1 ^a
<i>Piranga flava</i>	2	1	37.7 ^a
<i>Piranga leucoptera</i>	1	1	16 ^a
<i>Piranga ludoviciana</i>	1	3	28.1 ^a
<i>Piranga olivacea</i>	1	3	28.2 ^a
<i>Piranga rubra</i>	1	3	29.1 ^a
<i>Pitangus sulphuratus</i>	3	1	62.9 ^a
<i>Pithys albifrons</i>	1	1	19.5 ^a
<i>Pitta bandii</i>	1	1	66.5 ^c
<i>Pitta caerulea</i>	1	1	202 ^a
<i>Pitta cyanea</i>	1	1	109 ^a
<i>Pitta granatina</i>	1	1	61.5 ^c
<i>Pitta guajana</i>	1	1	81.5 ^a
<i>Pitta phayrei</i>	1	1	100 ^c

<i>Pitta reichenowi</i>	1	1	94.9 ^f
<i>Pitta sordida</i>	1	3	64.5 ^a
<i>Pitta venusta</i>	1	1	94.9 ^f
<i>Pityriasis gymnocephala</i>	1	1	128.5 ^a
<i>Platalea minor</i>	3	2	1228 ^a
<i>Platycicbla flavipes</i>	2	1	65.1 ^a
<i>Platylophus galericulatus</i>	1	1	77.9 ^a
<i>Platyrrinchus coronatus</i>	1	1	9.1 ^a
<i>Platyrrinchus leucoryphus</i>	1	1	16 ^a
<i>Platyrrinchus mystaceus</i>	1	1	9.7 ^a
<i>Platyrrinchus platyrhynchos</i>	1	1	12 ^a
<i>Platyrrinchus saturatus</i>	1	1	10.7 ^a
<i>Platysmurus leucopterus</i>	1	1	180 ^a
<i>Platysteira blissetti</i>	1	1	35.5 ^a
<i>Platysteira castanea</i>	1	1	14.1 ^a
<i>Platysteira concreta</i>	1	1	11.4 ^a
<i>Platysteira cyanea</i>	2	1	14.6 ^a

<i>Platysteira jamesoni</i>	1	1	12 ^f
<i>Platysteira peltata</i>	1	1	13.7 ^a
<i>Platysteira tonsa</i>	1	1	11.5 ^a
<i>Plegadis chibi</i>	3	2	616.9 ^a
<i>Plocens alienus</i>	1	1	22.2 ^a
<i>Plocens baglafecht</i>	2	1	31.6 ^a
<i>Plocens bicolor</i>	2	1	34.5 ^a
<i>Plocens cucullatus</i>	3	1	36.1 ^a
<i>Plocens insignis</i>	1	1	27.5 ^a
<i>Plocens melanogaster</i>	1	1	22.4 ^a
<i>Plocens nigerrimus</i>	2	1	34.3 ^a
<i>Plocens nigricollis</i>	2	1	27.1 ^a
<i>Plocens philippinus</i>	2	1	28.2 ^a
<i>Plocens tricolor</i>	1	1	36.8 ^a
<i>Pluvialis dominica</i>	3	3	151.5 ^a
<i>Pluvialis fulva</i>	3	3	134.9 ^a
<i>Podager nacunda</i>	2	1	159 ^a

<i>Podica senegalensis</i>	2	1	599 ^a
<i>Poecilatriccus capitalis</i>	1	1	8 ^a
<i>Poeoptera lugubris</i>	1	1	38.8 ^a
<i>Poeoptera stublmanni</i>	1	1	39.3 ^a
<i>Pogoniulus atroflavus</i>	1	1	18.2 ^a
<i>Pogoniulus bilineatus</i>	1	1	13.1 ^a
<i>Pogoniulus coryphaeus</i>	1	1	10.7 ^a
<i>Pogoniulus leucomystax</i>	1	1	11.8 ^a
<i>Pogoniulus pusillus</i>	2	1	9.6 ^a
<i>Pogoniulus scolopaceus</i>	1	1	14.1 ^a
<i>Pogoniulus subsulphureus</i>	1	1	10.5 ^a
<i>Pogonocichla stellata</i>	1	1	18.6 ^a
<i>Poicephalus gulielmi</i>	1	1	274.6 ^a
<i>Polioptila caerulea</i>	2	3	5.8 ^a
<i>Polioptila guianensis</i>	1	1	6 ^a
<i>Polioptila lactea</i>	2	1	6.5 ^a
<i>Polioptila paraensis</i>	1	1	6 ^a

<i>Polioptila plumbea</i>	2	1	6 ^c
<i>Polioptila schistaceigula</i>	1	1	6 ^a
<i>Polyboroides typus</i>	1	1	633.2 ^a
<i>Polyplectron malacense</i>	1	1	608 ^a
<i>Polytmus theresiae</i>	3	1	3.8 ^a
<i>Pomatorhinus borsfieldii</i>	2	1	43 ^a
<i>Pomatorhinus hypoleucos</i>	1	1	75.6 ^a
<i>Pomatorhinus montanus</i>	1	1	28.6 ^a
<i>Pomatorhinus schisticeps</i>	1	1	43 ^a
<i>Poocetes gramineus</i>	3	3	25.7 ^a
<i>Popelairia conversii</i>	1	1	3 ^a
<i>Popelairia langsdorffi</i>	1	1	2.8 ^a
<i>Popelairia popelairii</i>	1	1	2.5 ^a
<i>Porphyrio martinicus</i>	2	1	235.1 ^a
<i>Porphyrolaema porphyrolaema</i>	1	1	60 ^a
<i>Premnoplex brunnescens</i>	1	1	16.3 ^a
<i>Prinia bairdii</i>	1	1	13.4 ^a

<i>Prinia flaviventris</i>	3	1	7 ^a
<i>Prinia hypoxantha</i>	2	1	10 ^a
<i>Prinia leucopogon</i>	2	1	11.7 ^a
<i>Prinia rufescens</i>	2	1	6.5 ^a
<i>Prinia subflava</i>	2	1	8.6 ^a
<i>Prionochilus maculatus</i>	1	1	8 ^a
<i>Prionochilus percussus</i>	1	1	6 ^b
<i>Prionochilus thoracicus</i>	1	1	8.9 ^a
<i>Prionochilus xanthopygius</i>	1	1	7.6 ^a
<i>Procnias nudicollis</i>	1	1	172 ^a
<i>Procnias tricarunculata</i>	1	1	210 ^a
<i>Prodotiscus insignis</i>	1	1	10.4 ^a
<i>Progne chalybea</i>	2	1	42.9 ^c
<i>Progne subis</i>	2	1	53.8 ^a
<i>Progne tapera</i>	3	1	32 ^a
<i>Protonotaria citrea</i>	1	3	14.3 ^a
<i>Prunella modularis</i>	2	1	20.2 ^a

<i>Psolidoprocne albiceps</i>	2	3	11.1 ^a
<i>Psolidoprocne nitens</i>	1	2	9.8 ^a
<i>Psolidoprocne pristoptera</i>	2	2	10.3 ^a
<i>Psarisomus dalbousiae</i>	1	1	67 ^a
<i>Psarocolius angustifrons</i>	1	1	271.5 ^a
<i>Psarocolius decumanus</i>	1	1	206.3 ^a
<i>Psarocolius oseryi</i>	1	1	137.7 ^a
<i>Psarocolius viridis</i>	1	1	287 ^a
<i>Psarocolius wagleri</i>	1	1	155.5 ^a
<i>Pseudoalcippe abyssinica</i>	1	1	18.8 ^a
<i>Pseudocalyptomena graneri</i>	1	1	31.8 ^a
<i>Pseudocolaptes boissonneautii</i>	1	1	43.9 ^a
<i>Pseudodacnis hartlaubi</i>	2	1	5.1 ^b
<i>Pseudotriccus ruficeps</i>	1	1	11.3 ^a
<i>Psilorhamphus guttatus</i>	1	1	11.3 ^a
<i>Psilorhinus morio</i>	2	1	204 ^a
<i>Psittacula longicauda</i>	1	1	107 ^a

<i>Psittacus erithacus</i>	1	1	333 ^a
<i>Psittinus cyanurus</i>	1	1	11.8 ^b
<i>Psophia crepitans</i>	1	1	1026 ^a
<i>Psophia leucoptera</i>	1	1	1315.6 ^a
<i>Psophia viridis</i>	1	1	1071 ^a
<i>Pternistis nobilis</i>	2	1	878 ^a
<i>Pternistis squamatus</i>	1	1	449.5 ^a
<i>Pteroglossus aracari</i>	1	1	250.2 ^a
<i>Pteroglossus azara</i>	1	1	144.5 ^a
<i>Pteroglossus beaubarnaesii</i>	1	1	215.9 ^a
<i>Pteroglossus bitorquatus</i>	1	1	142 ^a
<i>Pteroglossus castanotis</i>	1	1	258.1 ^a
<i>Pteroglossus frantzii</i>	1	1	253 ^a
<i>Pteroglossus inscriptus</i>	1	1	125.8 ^a
<i>Pteroglossus mariae</i>	1	1	140 ^a
<i>Pteroglossus plurivinctus</i>	1	1	261.7 ^a
<i>Pteroglossus torquatus</i>	1	1	219.1 ^a

<i>Pteroglossus viridis</i>	1	1	135 ^a
<i>Pteronetta hartlaubii</i>	3	2	790 ^a
<i>Pteroptochos castaneus</i>	2	2	93.9 ^a
<i>Ptilinopus jambu</i>	2	1	135 ^a
<i>Ptilocichla leucogrammica</i>	1	1	33 ^b
<i>Pulsatrix koeniswaldiana</i>	1	1	481 ^a
<i>Pulsatrix perspicillata</i>	1	1	983.6 ^a
<i>Pycnonotus atriceps</i>	1	1	25.5 ^a
<i>Pycnonotus barbatus</i>	2	1	32.1 ^a
<i>Pycnonotus brunneus</i>	1	1	28.8 ^a
<i>Pycnonotus cyaniventris</i>	1	1	22 ^a
<i>Pycnonotus erythroptalmos</i>	1	1	19.2 ^a
<i>Pycnonotus eutilotus</i>	1	1	35.3 ^a
<i>Pycnonotus finlaysoni</i>	2	1	26.3 ^a
<i>Pycnonotus flaviventris</i>	2	1	29.3 ^a
<i>Pycnonotus goiavier</i>	2	1	27.8 ^a
<i>Pycnonotus melanicterus</i>	2	1	29.3 ^a

<i>Pycnonotus melanoleucos</i>	1	1	31 ^a
<i>Pycnonotus plumosus</i>	2	1	34.9 ^a
<i>Pycnonotus simplex</i>	1	1	25.1 ^a
<i>Pycnonotus squamatus</i>	1	1	22 ^a
<i>Pycnonotus tricolor</i>	2	1	39 ^a
<i>Pycnonotus zeylanicus</i>	2	1	86.3 ^a
<i>Pygarrhichas albogularis</i>	2	1	11.5 ^a
<i>Pygiptila stellaris</i>	1	1	24.1 ^a
<i>Pyrenestes ostrinus</i>	2	1	34.3 ^a
<i>Pyriglena leuconota</i>	1	1	32.3 ^a
<i>Pyriglena leucoptera</i>	1	1	28.8 ^a
<i>Pyrocephalus rubinus</i>	3	3	14.4 ^a
<i>Pyroderus scutatus</i>	1	2	357 ^a
<i>Pyrrhocomma ruficeps</i>	1	2	15.6 ^a
<i>Pyrrhomyias cinnamomea</i>	2	1	9.9 ^a
<i>Pyrrhula pyrrhula</i>	1	1	24.3 ^a
<i>Pyrrhura amazonum</i>	1	1	62.1 ^a

<i>Pyrrhura frontalis</i>	1	1	72 ^a
<i>Pyrrhura melanura</i>	1	1	70.5 ^a
<i>Pyrrhura perlata</i>	1	1	75 ^a
<i>Pyrrhura picta</i>	1	1	62.1 ^a
<i>Pyrrhura rupicola</i>	1	1	75 ^a
<i>Querula purpurata</i>	1	1	107.4 ^a
<i>Quiscalus quiscula</i>	3	1	105.2 ^a
<i>Rallus aquaticus</i>	1	2	110.7 ^a
<i>Ramphastos ambiguus</i>	1	1	651.7 ^a
<i>Ramphastos dicolorus</i>	1	1	331 ^a
<i>Ramphastos sulfuratus</i>	1	1	409.7 ^a
<i>Ramphastos swainsoni</i>	1	1	651.7 ^a
<i>Ramphastos swainsonii</i>	1	1	651.7 ^a
<i>Ramphastos toco</i>	2	1	618 ^a
<i>Ramphastos tucanus</i>	1	1	659.6 ^a
<i>Ramphastos vitellinus</i>	1	1	360.4 ^a
<i>Ramphocaenus melanurus</i>	2	1	9.7 ^a

<i>Ramphocelus bresilius</i>	2	1	32.9 ^a
<i>Ramphocelus carbo</i>	2	1	25.9 ^a
<i>Ramphocelus costaricensis</i>	2	1	31.3 ^c
<i>Ramphocelus dimidiatus</i>	2	1	28 ^a
<i>Ramphocelus flammigerus</i>	2	1	33 ^a
<i>Ramphocelus icteronotus</i>	2	1	33 ^c
<i>Ramphocelus nigrogularis</i>	2	1	31 ^a
<i>Ramphocelus passerinii</i>	2	1	32 ^a
<i>Ramphotrigon fuscicauda</i>	1	1	18.6 ^a
<i>Ramphotrigon megacephala</i>	1	1	14.2 ^a
<i>Ramphotrigon ruficauda</i>	2	1	18.4 ^a
<i>Regulus calendula</i>	1	3	6.2 ^a
<i>Regulus ignicapillus</i>	1	3	5.6 ^a
<i>Regulus regulus</i>	1	3	5.5 ^a
<i>Regulus satrapa</i>	1	3	6.2 ^a
<i>Reinwardtipicus validus</i>	1	1	149 ^a
<i>Rhaphidura leucopygialis</i>	1	1	13.6 ^a

<i>Rhaphidura sabini</i>	1	1	16 ^a
<i>Rhegmatorbina cristata</i>	1	1	35 ^b
<i>Rhegmatorbina gymnops</i>	1	1	28.5 ^a
<i>Rhegmatorbina hoffmannsi</i>	1	1	31.4 ^a
<i>Rhegmatorbina melanosticta</i>	1	1	31 ^a
<i>Rhinomyias olivacea</i>	1	1	15.7 ^a
<i>Rhinomyias umbratilis</i>	1	1	18.2 ^a
<i>Rhinoplax vigil</i>	1	1	2887.6 ^a
<i>Rhinopomastus cyanomelas</i>	2	1	27.1 ^a
<i>Rhipidura javanica</i>	2	1	12.5 ^a
<i>Rhipidura perlata</i>	1	1	13.4 ^a
<i>Rhizothera longirostris</i>	1	1	746.7 ^a
<i>Rhodospingus cruentus</i>	2	1	15 ^a
<i>Rhopocichla atriceps</i>	1	1	16.3 ^a
<i>Rhynchocyclus brevirostris</i>	1	1	24.3 ^a
<i>Rhynchocyclus fulvipectus</i>	1	1	23.1 ^a
<i>Rhynchocyclus olivaceus</i>	1	1	21.3 ^a

<i>Rhyticeros undulatus</i>	1	1	2214.6 ^a
<i>Rhytipterna holerythra</i>	1	1	36.8 ^a
<i>Rhytipterna immunda</i>	2	1	27.4 ^a
<i>Rhytipterna simplex</i>	2	1	31.8 ^a
<i>Riparia riparia</i>	3	3	12.7 ^a
<i>Rollulus rouloul</i>	1	1	216.5 ^a
<i>Rupicola rupicola</i>	1	1	194.6 ^a
<i>Rynchops niger</i>	3	2	303 ^a
<i>Sakesphornus canadensis</i>	2	1	24.4 ^a
<i>Sakesphornus luctuosus</i>	1	1	31.4 ^a
<i>Sakesphornus melanothorax</i>	1	1	31.8 ^a
<i>Saltator albicollis</i>	2	1	41.3 ^a
<i>Saltator atriceps</i>	2	1	83.8 ^a
<i>Saltator atripennis</i>	2	1	55 ^a
<i>Saltator coerulescens</i>	2	1	54.9 ^a
<i>Saltator fuliginosus</i>	1	1	50.5 ^f
<i>Saltator grossus</i>	1	1	44.2 ^a

<i>Saltator maximus</i>	2	1	47.6 ^a
<i>Saltator similis</i>	2	1	43.3 ^a
<i>Saltator striatipectus</i>	2	1	39 ^a
<i>Sarcoramphus papa</i>	3	1	3400 ^a
<i>Sarothrura elegans</i>	1	1	46 ^a
<i>Sarothrura pulchra</i>	1	1	45.2 ^a
<i>Sasia abnormis</i>	1	1	9.2 ^a
<i>Sasia africana</i>	1	1	9.4 ^a
<i>Saxicola torquata</i>	3	2	14.1 ^a
<i>Sayornis nigricans</i>	1	1	9.9 ^a
<i>Sayornis phoebe</i>	2	3	19.7 ^a
<i>Schiffornis major</i>	1	1	31 ^a
<i>Schiffornis turdina</i>	1	1	31.7 ^a
<i>Schiffornis turdinus</i>	1	1	31.7 ^a
<i>Schiffornis virescens</i>	1	1	25.6 ^a
<i>Schistochlamys melanopsis</i>	2	2	33 ^a
<i>Schistochlamys ruficapillus</i>	2	1	31.2 ^a

<i>Schistolais leucopogon</i>	2	1	11.7 ^a
<i>Schontedenapus myoptilus</i>	1	2	23.5 ^a
<i>Sclateria naevia</i>	1	1	24.6 ^a
<i>Sclerurus albigularis</i>	1	1	34.8 ^a
<i>Sclerurus candacutus</i>	1	1	36.2 ^a
<i>Sclerurus guatemalensis</i>	1	1	34.7 ^a
<i>Sclerurus mexicanus</i>	1	1	25.1 ^a
<i>Sclerurus ruficularis</i>	1	1	21.6 ^a
<i>Sclerurus scansor</i>	1	1	36.9 ^a
<i>Scolopax rusticola</i>	3	3	308.3 ^a
<i>Scytalopus indigoticus</i>	1	1	16.2 ^a
<i>Scytalopus latebricola</i>	1	1	22.7 ^c
<i>Scytalopus magellanicus</i>	2	2	73.7 ^a
<i>Seiurus omeiensis</i>	1	3	7.3 ^a
<i>Seiurus soror</i>	1	3	7.3 ^a
<i>Seiurus aurocapillus</i>	1	3	18.8 ^a
<i>Seiurus motacilla</i>	1	3	19.9 ^a

<i>Seiurus noveboracensis</i>	1	3	16.3 ^a
<i>Selasphorus rufus</i>	2	1	3.5 ^a
<i>Selasphorus scintilla</i>	2	1	2.2 ^a
<i>Selenidera culik</i>	1	1	147 ^a
<i>Selenidera gouldii</i>	1	1	180.9 ^a
<i>Selenidera maculirostris</i>	1	1	164 ^a
<i>Selenidera nattereri</i>	1	1	158.4 ^a
<i>Selenidera reinwardtii</i>	1	1	161.8 ^a
<i>Sephanoides sephaniodes</i>	3	1	20.3 ^a
<i>Serilophus lunatus</i>	1	1	34 ^a
<i>Serinus atrogularis</i>	3	1	11.4 ^a
<i>Serinus burtoni</i>	1	1	30.5 ^a
<i>Serinus canicollis</i>	2	1	15.2 ^a
<i>Serinus scotops</i>	1	1	14.2 ^a
<i>Serinus serinus</i>	2	2	11.2 ^a
<i>Serinus striolatus</i>	2	1	22.4 ^a
<i>Serpophaga subcristata</i>	2	3	6.6 ^a

<i>Setophaga ruticilla</i>	1	3	8.2 ^a
<i>Sheppardia aequatorialis</i>	1	1	16.8 ^a
<i>Sheppardia gunningi</i>	1	1	17.3 ^a
<i>Sialia mexicana</i>	3	2	26.4 ^a
<i>Sialia sialis</i>	2	2	27.5 ^a
<i>Sicalis columbiana</i>	2	1	13.4 ^a
<i>Sicalis flaveola</i>	2	1	16.9 ^a
<i>Silvicultrix diadema</i>	1	1	11.5 ^a
<i>Silvicultrix frontalis</i>	1	1	10 ^a
<i>Simoxenops ucayalae</i>	1	1	49.8 ^a
<i>Sirystes sibilator</i>	1	1	32.3 ^a
<i>Sitta canadensis</i>	1	3	9.8 ^a
<i>Sitta carolinensis</i>	1	1	21 ^a
<i>Sitta europaea</i>	1	1	20.4 ^a
<i>Sitta frontalis</i>	1	1	16.5 ^a
<i>Sittasomus griseicapillus</i>	1	1	13.1 ^a
<i>Skutchia borbae</i>	1	1	50 ^a

<i>Smithornis capensis</i>	2	1	23.2 ^a
<i>Smithornis rufolateralis</i>	1	1	19 ^a
<i>Spermestes_bicolor</i>	2	1	10 ^a
<i>Spermestes_cucullatus</i>	2	1	10 ^a
<i>Spermophaga baematina</i>	1	1	23 ^a
<i>Spermophaga ruficapilla</i>	1	1	23.6 ^a
<i>Sphyrapicus ruber</i>	2	2	54.8 ^a
<i>Sphyrapicus varius</i>	1	3	50.3 ^a
<i>Spilornis cheela</i>	1	1	597.7 ^a
<i>Spizaetus africanus</i>	1	2	1047 ^a
<i>Spizaetus alboniger</i>	1	1	830 ^a
<i>Spizaetus cirrhatus</i>	1	1	1475.1 ^a
<i>Spizaetus nanus</i>	1	1	557.8 ^a
<i>Spizaetus ornatus</i>	1	1	1197.4 ^a
<i>Spizaetus tyrannus</i>	2	1	1007.1 ^a
<i>Spizastur melanoleucus</i>	2	1	850 ^a
<i>Spizella pallida</i>	2	3	11.2 ^a

<i>Spizella passerina</i>	2	3	12.2 ^a
<i>Spizella pusilla</i>	2	2	12.5 ^a
<i>Sporophila americana</i>	2	1	12 ^a
<i>Sporophila caerulea</i>	3	3	9.7 ^a
<i>Sporophila castaneiventris</i>	2	1	7.8 ^a
<i>Sporophila corvina</i>	2	2	10.6 ^a
<i>Sporophila falcirostris</i>	1	2	10.8 ^f
<i>Sporophila frontalis</i>	1	2	10.8 ^f
<i>Sporophila intermedia</i>	3	1	12.1 ^a
<i>Sporophila lineola</i>	3	3	9.8 ^a
<i>Sporophila minuta</i>	3	1	7.8 ^a
<i>Sporophila nigricollis</i>	3	1	9.6 ^a
<i>Sporophila schistacea</i>	1	2	12.7 ^a
<i>Stachyris erythroptera</i>	1	1	12.6 ^a
<i>Stachyris leucotis</i>	1	1	23.7 ^a
<i>Stachyris maculata</i>	1	1	29.2 ^a
<i>Stachyris nigriceps</i>	1	1	15.8 ^a

<i>Stachyris nigricollis</i>	1	1	26.2 ^a
<i>Stachyris poliocephala</i>	1	1	23.3 ^a
<i>Stachyris rufifrons</i>	1	1	10.6 ^a
<i>Stactolaema leucotis</i>	1	1	54.9 ^a
<i>Stactolaema olivacea</i>	1	1	49.4 ^a
<i>Stelgidopteryx ruficollis</i>	3	1	16.1 ^a
<i>Stelgidopteryx serripennis</i>	3	3	15.7 ^a
<i>Stephanoaetus coronatus</i>	1	1	3640 ^a
<i>Stephanophorus diadematus</i>	1	2	35.4 ^a
<i>Stephanoxis lalandi</i>	1	1	4 ^a
<i>Sterna superciliosa</i>	3	1	46.4 ^a
<i>Stiphodon erythrothorax</i>	1	1	15.6 ^a
<i>Streptopelia capicola</i>	3	1	142 ^a
<i>Streptopelia chinensis</i>	3	1	159 ^a
<i>Streptopelia decaocto</i>	3	1	149 ^a
<i>Streptopelia lugens</i>	3	1	155 ^a
<i>Streptopelia orientalis</i>	2	3	414.6 ^a

<i>Streptopelia semitorquata</i>	3	1	176 ^a
<i>Streptopelia turtur</i>	2	3	132 ^a
<i>Streptoprocne rutila</i>	1	2	20.2 ^a
<i>Streptoprocne zonaris</i>	1	2	86.8 ^a
<i>Strix aluco</i>	1	1	472.5 ^a
<i>Strix bubula</i>	1	1	370 ^a
<i>Strix hylophila</i>	1	1	345.4 ^a
<i>Strix leptogrammica</i>	1	1	970 ^a
<i>Strix nigrolineata</i>	1	1	457.9 ^a
<i>Strix rufipes</i>	3	2	296 ^a
<i>Strix uralensis</i>	1	1	9 ^a
<i>Strix varia</i>	2	1	711.5 ^a
<i>Strix virgata</i>	2	1	284 ^a
<i>Strix woodfordii</i>	1	1	350 ^a
<i>Sturnella militaris</i>	3	2	48 ^a
<i>Sturnus vulgaris</i>	3	2	77.1 ^a
<i>Sublegatus modestus</i>	2	3	14 ^a

<i>Surniculus lugubris</i>	2	1	29.7 ^a
<i>Sylvia atricapilla</i>	1	3	16.7 ^a
<i>Sylvia borin</i>	1	3	18.2 ^a
<i>Sylvia communis</i>	3	3	15.1 ^a
<i>Sylvia curruca</i>	3	3	11.4 ^a
<i>Sylvia melanocephala</i>	2	2	11.7 ^a
<i>Sylvia undata</i>	2	1	10.8 ^a
<i>Sylvietta denti</i>	2	1	8.4 ^a
<i>Sylvietta leucophrys</i>	2	1	11.3 ^a
<i>Sylvietta virens</i>	2	1	8.4 ^a
<i>Sylviorthorhynchus desmursii</i>	2	1	15.5 ^a
<i>Synallaxis albescens</i>	2	1	11.2 ^a
<i>Synallaxis azarae</i>	2	1	16.9 ^a
<i>Synallaxis brachyura</i>	2	1	18.3 ^a
<i>Synallaxis cherriei</i>	1	1	16 ^a
<i>Synallaxis cinerascens</i>	1	1	13.1 ^a
<i>Synallaxis erythrothorax</i>	2	1	17.1 ^a

<i>Synallaxis frontalis</i>	2	1	14 ^a
<i>Synallaxis guianensis</i>	2	1	18.1 ^a
<i>Synallaxis infusata</i>	1	1	18.2 ^a
<i>Synallaxis ruficapilla</i>	1	1	13.8 ^a
<i>Synallaxis rutilans</i>	1	1	16.7 ^a
<i>Synallaxis spixi</i>	2	1	12.6 ^a
<i>Syndactyla ruficollis</i>	1	1	31 ^a
<i>Syndactyla rufosuperciliata</i>	1	1	25.6 ^a
<i>Syndactyla subalaris</i>	1	1	28.9 ^a
<i>Syrigma sibilatrix</i>	3	2	463 ^a
<i>Syrmaticus soemmerringii</i>	3	2	759.1 ^a
<i>Tachornis squamata</i>	1	1	11 ^a
<i>Tachybaptus ruficollis</i>	3	1	169.4 ^a
<i>Tachycineta albiventer</i>	3	1	17.7 ^a
<i>Tachycineta bicolor</i>	3	3	21.2 ^a
<i>Tachycineta meyeni</i>	2	2	15.5 ^a
<i>Tachycineta thalassina</i>	3	3	14.1 ^a

<i>Tachyphonus coronatus</i>	1	1	29.3 ^a
<i>Tachyphonus cristatus</i>	1	1	18.8 ^a
<i>Tachyphonus delatrii</i>	1	1	18 ^a
<i>Tachyphonus luctuosus</i>	1	1	13 ^a
<i>Tachyphonus phoenicius</i>	2	1	21 ^c
<i>Tachyphonus rufiventer</i>	1	1	19 ^a
<i>Tachyphonus rufus</i>	2	1	34.4 ^a
<i>Tachyphonus surinamus</i>	2	1	19 ^a
<i>Tangara arthus</i>	1	1	22 ^a
<i>Tangara callophrys</i>	1	1	22.9 ^a
<i>Tangara cayana</i>	2	1	18 ^a
<i>Tangara chilensis</i>	1	1	19.8 ^a
<i>Tangara cyanicollis</i>	2	1	17 ^a
<i>Tangara cyanocephala</i>	1	1	18 ^a
<i>Tangara cyanoventris</i>	1	1	16.5 ^a
<i>Tangara desmaresti</i>	1	1	20.4 ^a
<i>Tangara fastuosa</i>	1	1	20 ^f

<i>Tangara guttata</i>	1	1	18.4 ^a
<i>Tangara gyrola</i>	1	1	21 ^a
<i>Tangara icterocephala</i>	1	1	22 ^a
<i>Tangara inornata</i>	2	1	18 ^a
<i>Tangara larvata</i>	2	1	20 ^a
<i>Tangara mexicana</i>	2	1	20.5 ^a
<i>Tangara nigrocincta</i>	1	1	17 ^a
<i>Tangara nigroviridis</i>	1	3	30.3 ^a
<i>Tangara parzudakii</i>	1	1	15 ^a
<i>Tangara punctata</i>	1	1	15 ^a
<i>Tangara schrankii</i>	1	1	19 ^a
<i>Tangara seledon</i>	1	1	18.7 ^a
<i>Tangara varia</i>	1	1	10 ^a
<i>Tangara vassorii</i>	1	1	18 ^a
<i>Tangara velia</i>	1	1	21 ^a
<i>Tangara vitriolina</i>	2	1	23 ^a
<i>Tangara xanthocephala</i>	2	1	63.8 ^a

<i>Tangara xanthogastra</i>	1	1	15 ^a
<i>Tapera naevia</i>	3	1	48.4 ^a
<i>Taraba major</i>	2	1	59.2 ^a
<i>Tauraco corythaix</i>	1	1	308 ^a
<i>Tauraco livingstonii</i>	1	1	249 ^a
<i>Tauraco macrorhynchus</i>	1	1	243.7 ^a
<i>Tauraco schalowi</i>	1	1	206.4 ^a
<i>Tauraco schuettii</i>	1	1	235 ^a
<i>Tchagra australis</i>	2	1	32.3 ^a
<i>Telacanthura melanopygia</i>	1	1	52 ^a
<i>Telacanthura ussleri</i>	1	1	32.6 ^a
<i>Telophorus bocagei</i>	2	1	27 ^a
<i>Telophorus dohertyi</i>	2	1	35 ^a
<i>Telophorus multicolor</i>	1	1	50.9 ^a
<i>Telophorus olivaceus</i>	1	1	33.6 ^a
<i>Tephrodornis gularis</i>	1	1	37.8 ^a
<i>Tephrodornis virgatus</i>	1	1	37.8 ^a

<i>Terathopius ecaudatus</i>	2	1	2200 ^a
<i>Terenotriccus erythrurus</i>	1	1	7.4 ^a
<i>Terenura humeralis</i>	1	1	7.3 ^a
<i>Terenura maculata</i>	1	1	6.5 ^a
<i>Terenura sicki</i>	1	1	6.4 ^a
<i>Terenura spodioptila</i>	1	1	6.5 ^a
<i>Terpsiphone atrocaudata</i>	2	3	18.7 ^a
<i>Terpsiphone paradisi</i>	2	2	19.2 ^a
<i>Terpsiphone rufiventer</i>	1	1	15.1 ^a
<i>Terpsiphone rufocinerea</i>	1	1	15.4 ^a
<i>Terpsiphone viridis</i>	2	1	13.2 ^a
<i>Tersina viridis</i>	2	1	29 ^a
<i>Tetrao tetrix</i>	3	1	1068.7 ^a
<i>Tetrao urogallus</i>	2	1	2716.6 ^a
<i>Tetrastes bonasia</i>	1	1	429 ^a
<i>Thalurania colombica</i>	1	1	4.5 ^a
<i>Thalurania furcata</i>	1	1	4.2 ^a

<i>Thalurania glaucopsis</i>	1	1	4.8 ^a
<i>Thalurania watertonii</i>	1	1	4.6 ^a
<i>Thamnistes anabatinus</i>	1	1	20.7 ^a
<i>Thamnomanes ardesiacus</i>	1	1	17.7 ^a
<i>Thamnomanes caesius</i>	1	1	15.7 ^a
<i>Thamnomanes saturninus</i>	1	1	20.4 ^a
<i>Thamnomanes schistogynus</i>	1	1	17.2 ^a
<i>Thamnophilus aethiops</i>	1	1	25.7 ^a
<i>Thamnophilus amazonicus</i>	1	1	18.7 ^a
<i>Thamnophilus atrinucha</i>	1	1	23.6 ^a
<i>Thamnophilus bridgesi</i>	1	1	27 ^a
<i>Thamnophilus caerulescens</i>	1	1	21.1 ^a
<i>Thamnophilus doliatus</i>	2	1	27 ^a
<i>Thamnophilus murinus</i>	1	1	19.3 ^a
<i>Thamnophilus nigrocinereus</i>	1	1	30 ^a
<i>Thamnophilus palliatus</i>	2	1	23.3 ^c
<i>Thamnophilus punctatus</i>	1	1	19.4 ^a

<i>Thamnophilus ruficapillus</i>	2	1	20.4 ^a
<i>Thamnophilus schistaceus</i>	1	1	20.3 ^a
<i>Thamnophilus stictocephalus</i>	1	1	21.5 ^a
<i>Thamnophilus torquatus</i>	2	1	21.1 ^a
<i>Thescelocichla leucopleura</i>	1	1	62.1 ^a
<i>Thlypopsis sordida</i>	2	1	17 ^a
<i>Thraupis abbas</i>	2	1	45 ^a
<i>Thraupis cyanocephala</i>	2	1	36 ^a
<i>Thraupis cyanoptera</i>	1	1	43.3 ^a
<i>Thraupis episcopus</i>	2	1	35 ^a
<i>Thraupis ornata</i>	2	1	33 ^a
<i>Thraupis palmarum</i>	2	1	39 ^a
<i>Thraupis sayaca</i>	3	1	32.5 ^a
<i>Threnetes leucurus</i>	1	1	6 ^a
<i>Threnetes ruckeri</i>	1	1	5.2 ^a
<i>Threskiornis aethiopicus</i>	3	1	1493.2 ^a
<i>Thripadectes virgaticeps</i>	2	1	79.6 ^a

<i>Thryomanes bewickii</i>	2	2	9.9 ^a
<i>Thryothorus coraya</i>	1	1	17.2 ^a
<i>Thryothorus fasciatoventris</i>	3	1	24 ^a
<i>Thryothorus genibarbis</i>	2	1	19.2 ^a
<i>Thryothorus leucopogon</i>	2	1	19.1 ^f
<i>Thryothorus leucotis</i>	2	1	19.4 ^a
<i>Thryothorus ludovicianus</i>	2	1	19 ^a
<i>Thryothorus maculipectus</i>	2	1	15 ^a
<i>Thryothorus nigricapillus</i>	1	1	23.9 ^a
<i>Thryothorus rutilus</i>	1	1	16.5 ^a
<i>Thryothorus sclateri</i>	1	2	37.3 ^a
<i>Thryothorus semibadius</i>	2	1	17 ^a
<i>Thryothorus spadix</i>	1	1	19.1 ^f
<i>Tiaris bicolor</i>	2	1	9.8 ^a
<i>Tiaris fuliginosa</i>	2	1	13.3 ^a
<i>Tiaris obscura</i>	2	2	11.2 ^a
<i>Tiaris olivacea</i>	2	2	8.5 ^a

<i>Tigriornis leucolophus</i>	2	1	846 ^b
<i>Tigrisoma lineatum</i>	2	1	813 ^a
<i>Tinamus guttatus</i>	1	1	686.2 ^a
<i>Tinamus major</i>	1	1	1026.2 ^a
<i>Tinamus solitarius</i>	1	1	1386.4 ^a
<i>Tinamus tao</i>	1	1	1600.1 ^a
<i>Tityra cayana</i>	2	1	68.1 ^a
<i>Tityra inquisitor</i>	1	1	43.1 ^a
<i>Tityra semifasciata</i>	1	1	79.3 ^a
<i>Tockus alboterminatus</i>	1	1	223.7 ^a
<i>Tockus camurus</i>	1	1	103.8 ^a
<i>Tockus fasciatus</i>	1	1	258.8 ^a
<i>Tockus bartlaubi</i>	1	1	98.5 ^a
<i>Todiramphus leucopygius</i>	1	1	47.9 ^a
<i>Todirostrum chrysocrotaphum</i>	1	1	7 ^a
<i>Todirostrum cinereum</i>	2	1	6.3 ^a
<i>Todirostrum latirostre</i>	1	1	8.1 ^a

<i>Todirostrum maculatum</i>	1	1	7.3 ^a
<i>Todirostrum nigriceps</i>	1	1	6.5 ^a
<i>Todirostrum pictum</i>	1	1	6.8 ^a
<i>Todirostrum plumbeiceps</i>	1	1	5.7 ^a
<i>Todirostrum poliocephalum</i>	1	1	7 ^f
<i>Todirostrum senex</i>	1	1	7 ^f
<i>Todirostrum sylvia</i>	1	1	7.1 ^a
<i>Tolmomyias assimilis</i>	1	1	17 ^a
<i>Tolmomyias flaviventris</i>	1	1	12.2 ^a
<i>Tolmomyias poliocephalus</i>	1	1	10.8 ^a
<i>Tolmomyias sulphureus</i>	1	1	14.3 ^a
<i>Topaza pella</i>	1	1	12.1 ^a
<i>Topaza pyra</i>	1	1	12.1 ^a
<i>Tonit buetii</i>	1	1	60 ^a
<i>Tonit purpurata</i>	1	2	59.7 ^a
<i>Tonit surda</i>	1	2	66.5 ^f
<i>Toxostoma rufum</i>	1	2	68.8 ^a

<i>Trachyphonus purpuratus</i>	1	1	76.1 ^a
<i>Treron australis</i>	2	1	229 ^a
<i>Treron calva</i>	2	2	197.5 ^a
<i>Treron capellei</i>	2	1	401.3 ^a
<i>Treron curvirostra</i>	2	1	134.5 ^a
<i>Treron olax</i>	2	1	77 ^a
<i>Treron pompadora</i>	2	1	234 ^a
<i>Treron sieboldii</i>	2	1	264.8 ^a
<i>Treron sphenurus</i>	1	2	210 ^a
<i>Treron vernans</i>	2	1	132 ^a
<i>Trichastoma bicolor</i>	1	1	27.2 ^a
<i>Trichastoma rostratum</i>	2	1	17.8 ^b
<i>Trichixos pyrropyga</i>	1	1	40.9 ^a
<i>Tricholaema hirsuta</i>	1	1	55.2 ^a
<i>Tricholestes criniger</i>	1	1	17.1 ^a
<i>Trichothraupis melanops</i>	1	1	22.6 ^a
<i>Triclaria malachitacea</i>	1	1	90 ^a

<i>Tringa flavipes</i>	3	3	77.5 ^a
<i>Tringa glareola</i>	3	3	62.1 ^a
<i>Tringa hypoleucos</i>	3	3	48 ^a
<i>Tringa macularia</i>	3	3	40.4 ^a
<i>Tringa melanoleuca</i>	3	3	161.7 ^a
<i>Tringa nebularia</i>	3	3	187 ^a
<i>Tringa ochropus</i>	3	3	71.4 ^a
<i>Tringa solitaria</i>	3	3	48.4 ^a
<i>Tringa stagnatilis</i>	3	3	77.5 ^a
<i>Trochocercus cyanomelas</i>	1	1	10.2 ^a
<i>Trochocercus nitens</i>	1	1	11.8 ^a
<i>Troglodytes aedon</i>	3	1	10.9 ^a
<i>Troglodytes musculus</i>	3	1	10.9 ^f
<i>Troglodytes ochraceus</i>	1	1	9.4 ^a
<i>Troglodytes solstitialis</i>	1	1	11.4 ^a
<i>Troglodytes troglodytes</i>	1	3	9.7 ^a
<i>Trogon bairdii</i>	1	1	94.4 ^a

<i>Trogon collaris</i>	1	1	65.4 ^a
<i>Trogon curucui</i>	1	1	54 ^a
<i>Trogon massena</i>	1	1	141 ^a
<i>Trogon melanocephalus</i>	1	1	85.1 ^a
<i>Trogon melanurus</i>	1	1	114 ^a
<i>Trogon personatus</i>	1	1	63.4 ^a
<i>Trogon rufus</i>	1	1	53.8 ^a
<i>Trogon surrucura</i>	1	1	73.3 ^a
<i>Trogon violaceus</i>	1	1	46.5 ^a
<i>Trogon viridis</i>	1	1	89.7 ^a
<i>Tropicranus albocristatus</i>	1	1	289.4 ^a
<i>Turdus abyssinicus</i>	1	1	74 ^a
<i>Turdus albicollis</i>	1	1	54 ^a
<i>Turdus amaurochalinus</i>	2	3	57.9 ^a
<i>Turdus assimilis</i>	1	1	70.2 ^a
<i>Turdus cardis</i>	2	3	14.8 ^a
<i>Turdus chrysolaus</i>	1	3	134.1 ^a

<i>Turdus falklandii</i>	1	2	56.4 ^a
<i>Turdus feae</i>	1	3	71 ^a
<i>Turdus fumigatus</i>	1	1	75.7 ^a
<i>Turdus fuscater</i>	2	1	143 ^a
<i>Turdus grayi</i>	2	1	79.5 ^a
<i>Turdus bauxwelli</i>	1	1	69 ^a
<i>Turdus ignobilis</i>	1	2	29.1 ^a
<i>Turdus iliacus</i>	1	3	61.2 ^a
<i>Turdus lawrencii</i>	1	1	72.6 ^a
<i>Turdus leucomelas</i>	1	1	69.1 ^a
<i>Turdus maculirostris</i>	2	1	69.6 ^a
<i>Turdus merula</i>	1	1	102.7 ^a
<i>Turdus migratorius</i>	3	3	78.5 ^a
<i>Turdus naumanni</i>	1	3	58.9 ^a
<i>Turdus nigriceps</i>	1	2	52.7 ^a
<i>Turdus obscurus</i>	1	3	62.6 ^a
<i>Turdus obsoletus</i>	1	1	74.8 ^a

<i>Turdus olivaceus</i>	1	1	67.3 ^a
<i>Turdus pallidus</i>	1	3	75.5 ^a
<i>Turdus pelios</i>	2	1	65.4 ^a
<i>Turdus philomelos</i>	1	3	67.7 ^a
<i>Turdus pilaris</i>	1	3	106 ^a
<i>Turdus reevei</i>	2	1	61 ^a
<i>Turdus rufiventris</i>	1	2	69.4 ^a
<i>Turdus subalaris</i>	1	2	49.5 ^a
<i>Turdus torquatus</i>	2	3	109 ^a
<i>Turdus viscivorus</i>	2	1	117.4 ^a
<i>Turnix suscitator</i>	3	1	49.1 ^a
<i>Turtur afer</i>	2	1	65.6 ^a
<i>Turtur brebmeri</i>	2	1	116 ^a
<i>Turtur tympanistria</i>	2	1	70.6 ^a
<i>Tyrannentes stolzmanni</i>	1	1	7.2 ^a
<i>Tyrannentes virescens</i>	1	1	7.2 ^a
<i>Tyrannopsis sulphurea</i>	2	1	53.6 ^a

<i>Tyrannulus elatus</i>	2	1	7 ^a
<i>Tyrannus albogularis</i>	2	3	37.1 ^a
<i>Tyrannus melancholicus</i>	3	1	37.4 ^a
<i>Tyrannus niveigularis</i>	3	3	14.4 ^a
<i>Tyrannus savana</i>	3	3	31.9 ^a
<i>Tyrannus tyrannus</i>	2	3	39.9 ^a
<i>Tyto alba</i>	3	1	403.3 ^a
<i>Upupa epops</i>	3	3	66.9 ^a
<i>Uropsalis segmentata</i>	2	1	42.9 ^a
<i>Uropsila leucogastra</i>	2	1	9 ^a
<i>Urosphena squameiceps</i>	1	3	9.3 ^a
<i>Urotiorchis macrourus</i>	1	1	492 ^a
<i>Vanellus cayanus</i>	3	1	75.8 ^a
<i>Vanellus chilensis</i>	3	1	327 ^a
<i>Veniliornis affinis</i>	1	1	34.8 ^a
<i>Veniliornis callonotus</i>	2	1	26.5 ^a
<i>Veniliornis cassini</i>	2	1	34.6 ^a

<i>Veniliornis fumigatus</i>	1	1	36.2 ^a
<i>Veniliornis kirkii</i>	1	1	59.3 ^a
<i>Veniliornis maculifrons</i>	2	1	38.7 ^f
<i>Veniliornis passerinus</i>	2	1	32.1 ^a
<i>Veniliornis spilogaster</i>	1	1	40.3 ^a
<i>Vermivora celata</i>	1	3	9.2 ^a
<i>Vermivora chrysoptera</i>	2	3	8.7 ^a
<i>Vermivora peregrina</i>	1	3	8.9 ^a
<i>Vermivora pinus</i>	1	3	8.9 ^a
<i>Vermivora ruficapilla</i>	1	3	8.1 ^a
<i>Vidua chalybeata</i>	3	1	12.5 ^a
<i>Vidua macroura</i>	3	1	15.7 ^a
<i>Vireo altiloquus</i>	2	1	19 ^a
<i>Vireo cassinii</i>	2	3	14.7 ^a
<i>Vireo flavifrons</i>	2	3	18 ^a
<i>Vireo flavoviridis</i>	2	3	17.6 ^a
<i>Vireo gilvus</i>	1	3	12.7 ^a

<i>Vireo huttoni</i>	1	1	11.3 ^a
<i>Vireo leucophrys</i>	1	1	12.3 ^a
<i>Vireo olivaceus</i>	1	3	16.1 ^a
<i>Vireo philadelphicus</i>	1	3	11.5 ^a
<i>Vireo solitarius</i>	1	3	15.3 ^a
<i>Vireolanius excimius</i>	1	1	28.2 ^f
<i>Vireolanius leucotis</i>	1	1	26 ^a
<i>Vireolanius pulchellus</i>	1	1	24 ^a
<i>Volatinia jacarina</i>	3	1	9.9 ^a
<i>Wilsonia canadensis</i>	1	3	10 ^a
<i>Wilsonia citrina</i>	2	3	10.5 ^a
<i>Wilsonia pusilla</i>	1	3	7 ^a
<i>Xenerpestes minlosi</i>	1	1	11 ^a
<i>Xenopipo atronitens</i>	1	1	15.5 ^a
<i>Xenops milleri</i>	1	1	12.2 ^a
<i>Xenops minutus</i>	1	1	10.6 ^a
<i>Xenops rutilans</i>	1	1	11.2 ^a

<i>Xenops tenuirostris</i>	1	1	9.7 ^a
<i>Xiphocolaptes albicollis</i>	1	1	118 ^a
<i>Xiphocolaptes</i> <i>promeropirhynchus</i>	1	1	93 ^a
<i>Xipholena lamellipennis</i>	1	1	29 ^b
<i>Xipholena punicea</i>	1	1	68.1 ^a
<i>Xipborhynchus elegans</i>	1	1	26 ^d
<i>Xipborhynchus erythropygius</i>	1	1	46.8 ^a
<i>Xipborhynchus flavigaster</i>	1	1	46.2 ^a
<i>Xipborhynchus fuscus</i>	1	1	21.8 ^a
<i>Xipborhynchus guttatus</i>	1	1	59.7 ^a
<i>Xipborhynchus lachrymosus</i>	1	1	56.4 ^a
<i>Xipborhynchus necopinus</i>	1	1	45 ^c
<i>Xipborhynchus obsoletus</i>	1	1	32.6 ^a
<i>Xipborhynchus ocellatus</i>	1	1	36.7 ^a
<i>Xipborhynchus pardalotus</i>	1	1	34.6 ^a
<i>Xipborhynchus picus</i>	2	1	41.3 ^a

<i>Xipborhynchus spixii</i>	1	1	31.2 ^a
<i>Xipborhynchus triangularis</i>	1	1	93 ^a
<i>Xolmis pyrope</i>	1	1	44 ^a
<i>Yubina xantholenca</i>	1	1	11.8 ^a
<i>Zebrilus undulatus</i>	1	1	123 ^a
<i>Zenaida auriculata</i>	3	1	110.2 ^a
<i>Zenaida macroura</i>	3	1	118.9 ^a
<i>Zimmerius chrysops</i>	1	1	10.4 ^a
<i>Zimmerius gracilipes</i>	1	1	7.1 ^a
<i>Zimmerius vilissimus</i>	1	1	9.7 ^a
<i>Zimmerius viridiflavus</i>	1	1	11.5 ^a
<i>Zonotrichia albicollis</i>	2	3	24.4 ^a
<i>Zonotrichia capensis</i>	3	1	20.3 ^a
<i>Zonotrichia leucophrys</i>	2	3	28 ^a
<i>Zoothera aurea</i>	1	3	134.1 ^a
<i>Zoothera cameronensis</i>	1	1	44 ^a
<i>Zoothera citrina</i>	1	3	53.3 ^a

<i>Zoothera dauma</i>	1	3	9.7 ^a
<i>Zoothera gurneyi</i>	1	2	57.9 ^a
<i>Zoothera guttata</i>	1	2	57.5 ^a
<i>Zoothera interpres</i>	2	1	81.9 ^f
<i>Zoothera naevia</i>	1	3	77.5 ^a
<i>Zoothera piaggiae</i>	1	1	51.1 ^a
<i>Zoothera princei</i>	1	1	65.4 ^a
<i>Zoothera sibirica</i>	1	3	75.5 ^a
<i>Zosterops capensis</i>	3	1	14 ^a
<i>Zosterops erythropleurus</i>	1	3	11 ^a
<i>Zosterops everetti</i>	1	1	10 ^a
<i>Zosterops japonicus</i>	2	1	17 ^a
<i>Zosterops palpebrosus</i>	1	1	8.6 ^a
<i>Zosterops senegalensis</i>	2	1	10.9 ^a

¹Habitat codes: 1 = forest dependent, 2 = non forest.

²Migration codes: 1 = resident, 2 = migratory.

³Sources of body mass data are indicated with superscripted letters:

- (a) Dunning, J.B., (2007), CRC Handbook of avian body masses, 2nd Edition, CRC Press, Boca Raton, USA.
- (b) Marcelo (2007)
- (c) Unpublished field data (J. A. Tobias and N. Seddon)
- (d) Del Hoyo, J., Elliot, A., Sargatal, J. and Christie, D.A. (1992-2009). *The Handbook of the Birds of the World*, vols. 1–14. Lynx edicions, Barcelona, Spain.
- (e) Sekercioglu, C.H., Daily, G.C. and Ehrlich, P.R., (2004), Ecosystem consequences of bird declines, *Proceedings of the National Academy of Sciences USA*, **101**: 18042–18047.
- (f) Genus average: For these species, no direct data were available and so genus averages were used.

Appendix B: Chapter 3 Supplementary Methods and Results

Contents

Expanded materials and methods: Table B1.

Expanded results: Figs. B1–B4, Tables B2–B14.

Sensitivity analyses: Tables B15–B20.

Expanded materials and methods

Measurement protocol

Tarsus length was measured as the distance between the final scale of the acrotarsium to the middle of the ankle joint. Three beak measurements were taken at or from the anterior edge of the nostrils: (1) width, (2) length to the tip of the beak and (3) depth (as vertical height). Tail length was taken from the point at which the two central rectrices meet the skin to the tip of the longest rectrix. Wing chord was measured as the distance between the carpal joint and wing tip of the unflattened wing. We also measured the distance between the first secondary to the longest primary. We used this to calculate the hand-wing index of Claramunt et al. (2012). This is the wing chord - distance between the first secondary to longest primary divided by wing chord. All measurements were taken to the nearest 0.01 mm apart from wing chord and tail length, which were measured to the nearest mm.

Removal of rarities

As we are principally interested in changes in ecosystem functioning following land-use change, we removed rare species as they are likely to provide a negligible service to the overall system relative to other species. We did this by removing species from communities if they appeared less than three times in equivalent land-uses at each of the localities.

Habitat classification

Within the tropics, species occurring in more structurally dense habitats (e.g. forest) are expected to be less dispersive than more open-habitat specialists (e.g. grassland) (Weir et al. 2009). Empirical studies show that forest species may fail to cross even narrow breaks in habitat easily crossed by open country species (Moore et al. 2008). This is thought to occur both because of the behavioural inhibition of forest species to cross gaps (Harris and Reed 2002) and because the

adaptations required for movement amongst dense vegetation (e.g. short and rounded wings) are not favourable for long distance flight (Stratford and Robinson 2005). Species were assigned to one of two habitat categories according to whether they are primarily detected in (i) forest or (ii) non-forested habitats following Bregman et al. (2014). We defined forest as any type of evergreen or deciduous woodland lacking gaps between tree canopies.

Supplementary data

TABLE B1. Diet and trait (as Principal Components) variation across bird species identified in communities. Trait data are only calculated for frugivores and insectivores; other guilds are highlighted in grey. Trophic trait = second principal component (PC2) score for a composite of beak traits; locomotory trait = PC2 for a composite of traits associated with locomotion, including tarsus, wing and tail measurements; overall size = PC1 from a second PC analysis combining the PC1 scores for both trophic and locomotory traits (PC1 in each case is strongly associated with body size).

<i>Species</i>	Guild ¹	Trophic trait	Locomotory trait	Overall size
<i>Aburria kujubi</i>	FR	-0.2	-0.6	1.9
<i>Accipiter superciliosus</i>	VE			
<i>Amazilia fimbriata</i>	NE			
<i>Amazilia versicolor</i>	NE			
<i>Amazona amazonica</i>	FR	-1.1	-0.3	1.5
<i>Amazona farinosa</i>	SE			
<i>Amazona ochrocephala</i>	FR	-1.2	-0.3	1.8
<i>Amazonetta brasiliensis</i>	VG			
<i>Ammodramus aurifrons</i>	IN	-0.1	0.2	-0.8
<i>Ammodramus humeralis</i>	SE			
<i>Ancistrops strigilatus</i>	IN	0.0	0.1	-0.4
<i>Anbima cornuta</i>	VG			
<i>Anodorhynchus hyacinthinus</i>	SE			
<i>Anthracothorax nigricollis</i>	NE			
<i>Anthus lutescens</i>	IN	0.1	0.2	-0.9
<i>Antrostomus rufus</i>	IN	0.0	-0.5	-0.2
<i>Antrostomus sericocaudatus</i>	IN	0.0	-0.6	-0.4
<i>Ara ararauna</i>	SE			
<i>Ara chloropterus</i>	SE			
<i>Ara macao</i>	SE			
<i>Ara severus</i>	SE			
<i>Aramides cajana</i>	IN	0.5	1.6	0.6
<i>Aratinga aurea</i>	FR	-0.7	-0.8	0.5

<i>Aratinga jandaya</i>	FR	-0.9	-0.9	0.9
<i>Aratinga leucophthalma</i>	FR	-1.1	-0.9	1.3
<i>Ardea alba</i>	FI			
<i>Arremon taciturnus</i>	IN	-0.2	0.2	-0.5
<i>Atbene cunicularia</i>	VE			
<i>Atticora tibialis</i>	IN	-0.1	-0.2	-1.0
<i>Attila cinnamomeus</i>	IN	0.0	0.0	-0.3
<i>Attila spadiceus</i>	IN	0.0	0.1	-0.4
<i>Automolus ochrolaemus</i>	IN	0.2	0.0	-0.4
<i>Automolus paraensis</i>	IN	0.1	0.0	-0.4
<i>Automolus rufipileatus</i>	IN	0.1	0.0	-0.4
<i>Avocettula recurvirostris</i>	NE			
<i>Baryphthengus martii</i>	OM			
<i>Brachygalba lugubris</i>	IN	1.4	-0.2	-0.4
<i>Brotogeris chrysoptera</i>	NE			
<i>Brotogeris versicolurus</i>	SE			
<i>Bubulcus ibis</i>	IN	0.8	1.6	1.4
<i>Bucco capensis</i>	VE			
<i>Bucco tamatia</i>	IN	0.0	-0.1	-0.2
<i>Busarellus nigricollis</i>	VE			
<i>Buteo albonotatus</i>	VE			
<i>Buteo brachyurus</i>	VE			
<i>Buteo nitidus</i>	VE			
<i>Butorides striata</i>	VE			
<i>Cacicus cela</i>	FR	0.2	0.1	0.3
<i>Cacicus haemorrhous</i>	IN	0.1	0.0	0.4
<i>Cairina moschata</i>	VG			
<i>Calliphlox amethystina</i>	NE			
<i>Campephilus melanoleucus</i>	IN	0.4	0.0	0.8
<i>Campephilus rubricollis</i>	IN	0.5	0.1	0.7

<i>Camptostoma obsoletum</i>	IN	0.0	0.1	-1.1
<i>Campylopterus largipennis</i>	NE			
<i>Campylorhamphus procurvoides</i>	IN	1.9	-0.2	0.0
<i>Campylorhynchus turdinus</i>	IN	0.3	0.1	-0.4
<i>Cantorchilus leucotis</i>	IN	0.2	0.3	-0.8
<i>Capsiempis flaveola</i>	IN	-0.1	0.0	-1.0
<i>Caracara plancus</i>	SC			
<i>Caryothraustes canadensis</i>	FR	-0.4	0.0	-0.3
<i>Casiornis fuscus</i>	IN	0.0	-0.1	-0.6
<i>Cathartes aura</i>	SC			
<i>Cathartes burrovianus</i>	SC			
<i>Cathartes melambrotus</i>	SC			
<i>Celeus elegans</i>	IN	0.2	-0.2	0.3
<i>Celeus flavus</i>	IN	0.1	0.0	0.2
<i>Celeus grammicus</i>	IN	0.1	0.0	-0.2
<i>Celeus torquatus</i>	IN	0.5	-0.2	0.2
<i>Celeus undatus</i>	IN	0.1	0.0	-0.3
<i>Cercomacra cinerascens</i>	IN	0.0	0.0	-0.7
<i>Cercomacra laeta</i>	IN	0.0	0.2	-0.8
<i>Cercomacra nigrescens</i>	IN	0.0	0.2	-0.7
<i>Certhiasomus stictolaemus</i>	IN	0.2	-0.1	-0.5
<i>Chaetura brachyura</i>	IN	-0.1	0.0	-1.0
<i>Chaetura chapmanii</i>	IN	-0.1	-0.1	-1.0
<i>Chaetura spinicaudus</i>	IN	-0.1	-0.1	-1.0
<i>Chelidoptera tenebrosa</i>	IN	0.1	-0.1	-0.6
<i>Chiroxiphia pareola</i>	FR	-0.1	0.3	-0.9
<i>Chloroceryle americana</i>	FI			
<i>Chloroceryle inda</i>	FI			
<i>Chlorophanes spiza</i>	FR	0.1	0.1	-0.8
<i>Chondrobierax uncinatus</i>	IN	-0.3	-0.8	1.5

<i>Chrysolampis mosquitus</i>	NE			
<i>Cissopis leverianus</i>	FR	-0.3	-0.3	0.0
<i>Claravis pretiosa</i>	SE			
<i>Cnemotriccus fuscatus</i>	IN	0.0	0.0	-0.7
<i>Coccyzus minima</i>	IN	0.1	-0.5	-0.1
<i>Coereba flaveola</i>	NE			
<i>Colaptes melanochlorus</i>	IN	0.2	-0.1	0.1
<i>Colonia colonus</i>	IN	-0.2	-1.1	-0.4
<i>Columba livia</i>	SE			
<i>Columbina minuta</i>	SE			
<i>Columbina passerina</i>	SE			
<i>Columbina squammata</i>	SE			
<i>Columbina talpacoti</i>	SE			
<i>Conopias trivirgatus</i>	IN	0.0	0.0	-0.7
<i>Conopophaga aurita</i>	IN	-0.1	0.6	-0.7
<i>Conopophaga roberti</i>	IN	-0.1	0.5	-0.7
<i>Coragyps atratus</i>	SC			
<i>Corythopsis torquatus</i>	IN	0.0	0.4	-0.7
<i>Cotinga cayana</i>	FR	-0.1	0.0	-0.4
<i>Cotinga cotinga</i>	FR	-0.1	0.0	-0.5
<i>Crotophaga ani</i>	IN	-0.7	-0.5	0.9
<i>Crotophaga major</i>	IN	-0.3	-0.9	1.8
<i>Crypturellus cinereus</i>	FR	0.0	1.1	-0.1
<i>Crypturellus obsoletus</i>	SE			
<i>Crypturellus parvirostris</i>	SE			
<i>Crypturellus soni</i>	FR	0.0	0.7	-0.5
<i>Crypturellus strigulosus</i>	FR	0.0	1.1	-0.1
<i>Crypturellus variegatus</i>	FR	0.1	0.9	-0.2
<i>Cyanerpes caeruleus</i>	OM			
<i>Cyanerpes cyaneus</i>	OM			

<i>Cyanoloxia cyanooides</i>	SE			
<i>Cyclarhis gujanensis</i>	IN	-0.2	0.2	-0.5
<i>Cymbilaimus lineatus</i>	IN	-0.2	0.2	-0.2
<i>Cyphorhina arada</i>	IN	0.1	0.4	-0.8
<i>Cypseloides senex</i>	IN	-0.1	0.1	-0.5
<i>Dacnis cayana</i>	FR	0.0	0.1	-1.0
<i>Dacnis lineata</i>	FR	0.0	0.1	-1.0
<i>Daptrius ater</i>	SC			
<i>Deconychura longicauda</i>	IN	0.3	-0.2	-0.3
<i>Dendrexetastes rufigula</i>	IN	0.1	-0.2	0.1
<i>Dendrocincla fuliginosa</i>	IN	0.3	-0.1	-0.1
<i>Dendrocincla merula</i>	IN	0.2	0.0	-0.3
<i>Dendrocolaptes certhia</i>	IN	0.4	-0.2	0.4
<i>Dendrocolaptes picumnus</i>	IN	0.5	-0.2	0.4
<i>Dendroplex picus</i>	IN	0.5	-0.1	-0.3
<i>Derophtys accipitrinus</i>	VG			
<i>Dixiphia pipra</i>	FR	-0.1	0.2	-1.0
<i>Donacobius atricapilla</i>	IN	0.2	0.1	-0.3
<i>Dromococcyx pavoninus</i>	IN	0.2	-0.6	0.0
<i>Dromococcyx phasianellus</i>	IN	0.2	-1.0	0.4
<i>Dryocopus lineatus</i>	IN	0.4	-0.3	0.7
<i>Dysithamnus mentalis</i>	IN	0.0	0.2	-0.9
<i>Elaenia cristata</i>	IN	-0.1	0.1	-0.8
<i>Elaenia flavogaster</i>	IN	-0.1	0.0	-0.7
<i>Elanoides forficatus</i>	IN	-0.3	-1.6	1.8
<i>Elanus leucurus</i>	VE			
<i>Emberizoides herbicola</i>	IN	-0.1	0.0	-0.6
<i>Empidonomus varius</i>	IN	-0.1	-0.2	-0.5
<i>Epinecrophylia leucophthalma</i>	IN	0.1	0.1	-0.9
<i>Euphonia cayennensis</i>	FR	-0.2	0.2	-0.9

<i>Euphonia chrysopasta</i>	FR	-0.2	0.1	-0.9
<i>Euphonia minuta</i>	FR	-0.1	0.1	-1.1
<i>Euphonia rufiventris</i>	FR	-0.2	0.2	-0.9
<i>Euphonia violacea</i>	FR	-0.2	0.2	-1.0
<i>Falco femoralis</i>	VE			
<i>Falco rufigularis</i>	VE			
<i>Florisuga mellivora</i>	NE			
<i>Formicarius analis</i>	IN	0.0	0.6	-0.4
<i>Formicarius colma</i>	IN	0.0	0.5	-0.5
<i>Formicivora grisea</i>	IN	0.1	0.2	-0.9
<i>Forpus passerinus</i>	FR	-0.5	-0.1	-0.5
<i>Forpus xanthopterygius</i>	FR	-0.5	-0.1	-0.4
<i>Galbula cyanicollis</i>	IN	1.0	-0.3	-0.3
<i>Galbula dea</i>	IN	1.6	-1.0	0.3
<i>Galbula ruficauda</i>	IN	1.6	-0.6	0.0
<i>Gampsonyx swainsonii</i>	VE			
<i>Geothlypis aequinoctialis</i>	IN	0.1	0.2	-0.8
<i>Geotrygon montana</i>	SE			
<i>Geranoaetus albicaudatus</i>	VE			
<i>Glaucidium hardyi</i>	IN	-0.4	0.1	-0.5
<i>Glaucis birsutus</i>	NE			
<i>Glyphorynchus spirurus</i>	IN	0.0	-0.1	-0.7
<i>Grallaria varia</i>	IN	-0.1	1.3	0.1
<i>Granatellus pelzelni</i>	IN	-0.1	0.1	-0.9
<i>Griseotyrannus aurantioatrocristatus</i>	IN	-0.1	-0.3	-0.6
<i>Guaruba guarouba</i>	FR	-1.6	-0.9	2.2
<i>Guira guira</i>	IN	-0.1	-0.8	0.8
<i>Habia rubica</i>	IN	-0.2	0.0	-0.4
<i>Haematoderus militaris</i>	FR	-0.4	-0.4	0.9

<i>Harpagus bidentatus</i>	IN	-0.6	-0.1	0.9
<i>Helimaster longirostris</i>	NE			
<i>Heliothryx auritus</i>	NE			
<i>Hemitbrapfis guira</i>	IN	0.0	0.0	-0.9
<i>Hemitriccus minimus</i>	IN	0.0	0.1	-1.0
<i>Hemitriccus striaticollis</i>	IN	0.0	0.2	-1.0
<i>Herpetotheres cachinnans</i>	VE			
<i>Herpsilochmus rufimarginatus</i>	IN	0.0	0.2	-0.9
<i>Heterospizias meridionalis</i>	VE			
<i>Hirundo rustica</i>	IN	-0.1	-0.3	-0.8
<i>Hydropsalis albicollis</i>	IN	0.0	-0.7	-0.2
<i>Hydropsalis nigrescens</i>	IN	0.0	-0.5	-0.6
<i>Hylexetastes uniformis</i>	IN	0.2	0.0	0.4
<i>Hylocharis cyanus</i>	NE			
<i>Hylopezus berlepschi</i>	IN	0.1	1.0	-0.4
<i>Hylopezus macularius</i>	IN	0.1	0.8	-0.5
<i>Hylophilus hypoxanthus</i>	IN	0.1	0.1	-1.0
<i>Hylophilus ochraceiceps</i>	IN	0.0	0.2	-1.0
<i>Hylophilus pectoralis</i>	IN	0.0	0.2	-1.0
<i>Hylophilus semicinereus</i>	IN	0.0	0.1	-0.9
<i>Hylophylax naevius</i>	IN	0.0	0.3	-0.9
<i>Hylophylax punctulatus</i>	IN	0.0	0.3	-0.9
<i>Hypocnemis hypoxantha</i>	IN	0.0	0.2	-0.8
<i>Hypocnemis striata</i>	IN	0.0	0.2	-0.9
<i>Hypocnemoides maculicauda</i>	IN	0.1	0.3	-0.9
<i>Ibycter americanus</i>	IN	-0.3	-0.7	2.1
<i>Icterus cayanensis</i>	OM			
<i>Icterus jamaicii</i>	IN	0.1	0.1	-0.1
<i>Iodopleura isabellae</i>	IN	-0.2	0.1	-0.9
<i>Jacamerops aureus</i>	IN	0.6	-0.8	0.6

<i>Jacana jacana</i>	IN	0.3	1.4	-0.1
<i>Lamprospiza melanoleuca</i>	FR	-0.2	0.1	-0.4
<i>Lanio cristatus</i>	IN	-0.1	0.0	-0.6
<i>Lanio cucullatus</i>	IN	-0.2	0.1	-0.7
<i>Lanio luctuosus</i>	IN	-0.1	0.0	-0.8
<i>Lanio surinamus</i>	IN	0.0	-0.1	-0.6
<i>Lanio versicolor</i>	IN	-0.1	-0.2	-0.7
<i>Laniocera hypopyrra</i>	IN	-0.1	-0.1	-0.4
<i>Laterallus exilis</i>	IN	0.1	0.4	-0.9
<i>Laterallus melanophaius</i>	IN	0.1	0.6	-0.7
<i>Laterallus viridis</i>	IN	0.1	0.9	-0.6
<i>Lathrotriccus euleri</i>	IN	-0.1	0.0	-0.8
<i>Legatus leucophaius</i>	FR	-0.1	-0.1	-0.7
<i>Lepidocolaptes albolineatus</i>	IN	0.5	-0.2	-0.5
<i>Lepidothrix iris</i>	FR	-0.1	0.2	-1.0
<i>Leptotila rufaxilla</i>	SE			
<i>Leptotila verreauxi</i>	SE			
<i>Leucopternis kuhli</i>	VE			
<i>Lipaugus vociferans</i>	FR	-0.1	-0.3	-0.1
<i>Lophostrix cristata</i>	IN	-0.6	-0.1	1.5
<i>Lophotriccus galeatus</i>	IN	0.0	0.1	-1.1
<i>Lurocalis semitorquatus</i>	IN	0.0	-0.5	-0.5
<i>Machaeropterus pyrocephalus</i>	FR	-0.1	0.2	-1.1
<i>Malacoptila rufa</i>	IN	0.1	0.0	-0.3
<i>Manacus manacus</i>	FR	-0.1	0.3	-1.0
<i>Megascops torquata</i>	FI			
<i>Megarynchus pitangua</i>	IN			
<i>Megascops choliba</i>	IN	-0.3	0.3	0.1
<i>Megascops usta</i>	IN	-0.5	0.3	0.4
<i>Melanerpes candidus</i>	FR	0.4	-0.1	0.2

<i>Melanerpes cruentatus</i>	IN	0.3	0.0	-0.3
<i>Micrastur mintoni</i>	VE			
<i>Micrastur mirandollei</i>	VE			
<i>Micrastur ruficollis</i>	VE			
<i>Micrastur semitorquatus</i>	VE			
<i>Microcerculus marginatus</i>	IN	0.2	0.5	-0.9
<i>Microrhophias quixensis</i>	IN	0.1	0.0	-0.9
<i>Milvago chimachima</i>	SC			
<i>Mionectes macconnelli</i>	FR			
<i>Mionectes oleagineus</i>	FR	0.0	0.1	-0.9
<i>Molothrus bonariensis</i>	SE			
<i>Molothrus oryzivorus</i>	OM			
<i>Momotus momota</i>	IN	0.2	-1.4	1.0
<i>Monasa morphoeus</i>	IN	0.1	-0.5	0.3
<i>Monasa nigrifrons</i>	IN	0.2	-0.5	0.4
<i>Mycteria americana</i>	FI			
<i>Myiarchus ferox</i>	IN			
<i>Myiarchus tuberculifer</i>	IN	0.0	-0.1	-0.3
<i>Myiarchus tyrannulus</i>	IN	0.0	-0.1	-0.4
<i>Myiobius barbatus</i>	IN	0.0	-0.1	-0.3
<i>Myiobius barbatus</i>	IN	-0.1	0.0	-0.9
<i>Myiodynastes maculatus</i>	IN	-0.1	-0.2	0.0
<i>Myiopagis caniceps</i>	IN	0.0	0.1	-1.0
<i>Myiopagis flavivertex</i>	IN	0.0	0.0	-0.9
<i>Myiopagis gaimardii</i>	IN	0.0	0.1	-0.9
<i>Myiopagis viridicata</i>	IN	0.0	0.0	-1.0
<i>Myiophobus fasciatus</i>	IN	-0.1	0.0	-0.9
<i>Myiornis ecaudatus</i>	IN	0.0	0.2	-1.3
<i>Myiozetetes cayanensis</i>	IN	-0.1	-0.1	-0.6
<i>Myiozetetes luteiventris</i>	IN	-0.1	-0.1	-0.8
<i>Myrmeciza bemimelaena</i>	IN	0.0	0.4	-0.9

<i>Myrmoborus myiotherinus</i>	IN	0.0	0.4	-0.8
<i>Myrmothera campanisona</i>	IN	0.1	1.0	-0.4
<i>Myrmotherula axillaris</i>	IN	0.1	0.2	-1.0
<i>Myrmotherula brachyura</i>	IN	0.1	0.3	-1.1
<i>Myrmotherula hauxwelli</i>	IN	0.0	0.4	-1.0
<i>Myrmotherula longipennis</i>	IN	0.0	0.2	-0.9
<i>Myrmotherula menetriesii</i>	IN	0.1	0.2	-1.0
<i>Myrmotherula multistriata</i>	IN	0.1	0.2	-1.1
<i>Myrmotherula sclateri</i>	IN	0.1	0.3	-1.1
<i>Nemosia pileata</i>	IN	0.0	0.1	-0.8
<i>Neocrex erythrops</i>	IN	0.1	0.6	-0.5
<i>Notharchus hyperrhynchus</i>	IN	0.1	-0.2	1.0
<i>Notharchus ordii</i>	IN	-0.1	-0.2	0.0
<i>Notharchus tectus</i>	IN	-0.1	-0.1	-0.3
<i>Nyctibius aethereus</i>	IN	-0.1	-2.6	1.0
<i>Nyctibius griseus</i>	IN	0.0	-1.5	0.3
<i>Nyctibius leucopterus</i>	IN	0.0	-0.9	-0.4
<i>Nyctiphrynus ocellatus</i>	IN	0.0	-0.5	-0.6
<i>Nystalus maculatus</i>	IN	0.3	0.0	0.0
<i>Nystalus striolatus</i>	IN	0.0	-0.1	0.0
<i>Odontophorus gujanensis</i>	FR	-0.5	0.6	0.1
<i>Odontorchilus cinereus</i>	IN	0.2	0.1	-1.0
<i>Onychorhynchus coronatus</i>	IN	0.0	-0.1	-0.6
<i>Ornithion inerme</i>	IN	0.0	0.1	-1.2
<i>Ortalis motmot</i>	FR	-0.2	-0.1	1.1
<i>Ortalis supercilialis</i>	FR	-0.2	-0.3	0.6
<i>Orthopsittaca manilata</i>	FR	-1.2	-1.3	2.0
<i>Pachyramphus castus</i>	IN	-0.1	0.1	-0.7
<i>Pachyramphus marginatus</i>	IN	-0.1	0.1	-0.6
<i>Pachyramphus minor</i>	IN	-0.2	0.0	-0.3

<i>Pachyramphus polychopterus</i>	IN	-0.1	0.1	-0.6
<i>Pachyramphus rufus</i>	IN	-0.2	0.1	-0.7
<i>Pachyramphus validus</i>	IN	-0.2	0.0	-0.2
<i>Pachyramphus viridis</i>	IN	-0.2	0.1	-0.6
<i>Parkerthraustes humeralis</i>	IN	-0.4	0.0	-0.2
<i>Passer domesticus</i>	SE			
<i>Patagioenas cayennensis</i>	FR	0.1	-0.3	-0.2
<i>Patagioenas picauro</i>	VG			
<i>Patagioenas plumbea</i>	FR	0.1	-0.5	0.0
<i>Patagioenas speciosa</i>	FR	0.1	-0.2	-0.2
<i>Patagioenas subvinacea</i>	FR	0.1	-0.4	-0.3
<i>Pauxi tuberosa</i>	FR	-1.3	0.7	4.1
<i>Penelope jacquacu</i>	FR	-0.2	-0.4	2.1
<i>Penelope pileata</i>	FR	-0.3	-0.3	2.3
<i>Penelope supercilialis</i>	FR	-0.2	-0.2	1.6
<i>Periporphyrus erythromelas</i>	FR	-0.6	-0.2	0.1
<i>Phaeomyias murina</i>	IN	0.0	0.1	-1.0
<i>Phaeothlypis rivularis</i>	IN	0.0	0.2	-0.8
<i>Phaethornis aethopyga</i>	NE			
<i>Phaethornis bourcierii</i>	NE			
<i>Phaethornis ruber</i>	NE			
<i>Phaethornis rupurumii</i>	NE			
<i>Phaethornis superciliosus</i>	NE			
<i>Phaethornis unknown</i>	NE			
<i>Phaetusa simplex</i>	FI			
<i>Pheugopedius coraya</i>	IN	0.1	0.2	-0.8
<i>Pheugopedius genibarbis</i>	IN	0.1	0.3	-0.7
<i>Philohydor lictor</i>	IN	0.2	-0.1	-0.5
<i>Philohydor erythrocerum</i>	IN	0.0	0.0	-0.5
<i>Philohydor erythropterum</i>	IN	0.1	0.1	-0.5

<i>Philydor pyrrhodes</i>	IN	0.1	0.1	-0.6
<i>Philydor ruficaudatum</i>	IN	0.0	0.0	-0.6
<i>Phlegopsis nigromaculata</i>	IN	0.0	0.5	-0.5
<i>Phoenicircus carnifex</i>	FR	-0.2	0.1	-0.3
<i>Phylloscartes virescens</i>	IN	0.0	0.0	-0.9
<i>Piaya cayana</i>	IN	-0.1	-1.4	1.0
<i>Piaya melanogaster</i>	IN	0.0	-0.8	0.7
<i>Piculus chrysobloros</i>	IN	0.2	0.0	-0.2
<i>Piculus flavigula</i>	IN	0.1	0.0	-0.4
<i>Picumnus aurifrons</i>	IN	0.0	0.2	-1.1
<i>Picumnus exilis</i>	IN	0.0	0.1	-1.1
<i>Picumnus pygmaeus</i>	IN	0.1	0.1	-1.0
<i>Pionites leucogaster</i>	FR	-0.8	-0.1	0.7
<i>Pionus fuscus</i>	FR	-0.9	-0.1	0.8
<i>Pionus menstruus</i>	SE			
<i>Pipra rubrocapilla</i>	FR	-0.1	0.1	-1.0
<i>Piprites chloris</i>	IN	-0.1	0.1	-0.8
<i>Pitangus sulphuratus</i>	OM			
<i>Platalea ajaja</i>	VE			
<i>Platyrrinchus platyrhynchos</i>	IN	-0.3	0.1	-0.8
<i>Platyrrinchus saturatus</i>	IN	-0.2	0.2	-0.8
<i>Poecilotriccus fumifrons</i>	IN	0.0	0.2	-1.0
<i>Poecilotriccus sylvia</i>	IN	0.0	0.3	-0.9
<i>Polioptila paraensis</i>	IN	0.0	0.1	-1.1
<i>Polioptila plumbea</i>	IN	0.1	0.1	-1.1
<i>Polytmus theresiae</i>	NE			
<i>Porphyrio martinica</i>	VG			
<i>Progne chalybea</i>	IN	-0.1	-0.2	-0.6
<i>Psarocolius bifasciatus</i>	FR	0.5	0.1	1.6
<i>Psarocolius decumanus</i>	FR	0.2	0.1	1.1

<i>Psarocolius viridis</i>	FR	0.4	0.2	1.3
<i>Pseudastur albicollis</i>	VE			
<i>Psophia dextralis</i>	FR	0.0	3.1	1.8
<i>Psophia obscura</i>	FR	0.0	3.1	1.8
<i>Pteroglossus aracari</i>	FR	1.3	-0.2	3.2
<i>Pteroglossus bitorquatus</i>	FR	0.8	-0.2	2.1
<i>Pteroglossus inscriptus</i>	FR	0.6	-0.1	1.8
<i>Pulsatrix perspicillata</i>	VE			
<i>Pygiptila stellaris</i>	IN	0.0	0.2	-0.6
<i>Pygiptila stellaris</i>	IN	0.0	0.2	-0.6
<i>Pyriglena leuconota</i>	IN	0.0	0.4	-0.5
<i>Pyrrilia vulturina</i>	FR	-0.6	-0.2	0.3
<i>Pyrrhura amazonum</i>	FR	-0.6	-0.5	0.0
<i>Pyrrhura lepidula</i>	FR	-0.7	-0.7	0.2
<i>Querula purpurata</i>	FR	-0.2	-0.3	0.5
<i>Ramphastos tucanus</i>	FR	2.7	0.2	5.5
<i>Ramphastos vitellinus</i>	FR	1.6	0.1	4.1
<i>Ramphocaenus melanurus</i>	IN	0.5	0.2	-0.9
<i>Ramphocelus carbo</i>	FR	-0.2	0.0	-0.4
<i>Ramphotrigon megacephalum</i>	IN	-0.1	0.0	-0.8
<i>Ramphotrigon ruficauda</i>	IN	-0.1	-0.1	-0.6
<i>Rhagmatorhina gymnops</i>	IN	0.1	0.4	-0.6
<i>Rhinopteryx clamator</i>	VE			
<i>Rhynchocyclus olivaceus</i>	IN	-0.3	0.0	-0.5
<i>Rhytipterna simplex</i>	IN	0.0	-0.2	-0.3
<i>Riparia riparia</i>	IN	-0.1	-0.1	-1.0
<i>Rupornis magnirostris</i>	VE			
<i>Saltator coerulescens</i>	IN	-0.3	0.0	0.0
<i>Saltator grossus</i>	IN	-0.6	-0.1	0.1
<i>Saltator maximus</i>	IN	-0.3	0.0	-0.1

<i>Sarcoramphus papa</i>	SC			
<i>Schiffornis turdina</i>	IN	-0.1	0.1	-0.6
<i>Schistochlamys melanopis</i>	FR	-0.3	0.1	-0.4
<i>Sclateria naevia</i>	IN	0.3	0.3	-0.6
<i>Sclerurus caudacutus</i>	IN	0.3	0.1	-0.5
<i>Sclerurus mexicanus</i>	IN	0.5	0.2	-0.6
<i>Sclerurus rufigularis</i>	IN	0.3	0.1	-0.7
<i>Selenidera gouldii</i>	FR	0.2	0.1	1.7
<i>Sittasomus griseicapillus</i>	IN	0.2	-0.2	-0.7
<i>Spizaetus melanoleucus</i>	VE			
<i>Spizaetus ornatus</i>	VE			
<i>Spizaetus tyrannus</i>	VE			
<i>Sporophila americana</i>	SE			
<i>Sporophila angolensis</i>	SE			
<i>Sporophila lineola</i>	SE			
<i>Sporophila minuta</i>	SE			
<i>Sporophila nigricollis</i>	SE			
<i>Stelgidopteryx ruficollis</i>	IN	-0.1	-0.2	-0.9
<i>Strix hubula</i>	IN	-0.4	0.0	1.1
<i>Strix virgata</i>	IN	-0.5	-0.1	1.0
<i>Sturnella militaris</i>	IN	0.0	0.4	-0.4
<i>Synallaxis albescens</i>	IN	0.0	-0.1	-0.9
<i>Synallaxis gujanensis</i>	IN	0.1	0.0	-0.8
<i>Synallaxis rutilans</i>	IN	0.0	0.0	-0.8
<i>Tachornis squamata</i>	IN	-0.1	-0.4	-1.0
<i>Tachycineta albiventer</i>	IN	-0.1	-0.1	-0.9
<i>Tachyphonus rufus</i>	IN	-0.1	0.1	-0.3
<i>Taeniotriccus andrei</i>	IN	-0.1	0.0	-0.9
<i>Tangara episcopus</i>	FR	-0.2	0.1	-0.5
<i>Tangara gyrola</i>	FR	-0.1	0.1	-0.9

<i>Tangara mexicana</i>	FR	-0.1	0.1	-0.8
<i>Tangara palmarum</i>	FR	-0.1	0.0	-0.5
<i>Tangara punctata</i>	FR	-0.1	0.1	-0.9
<i>Tangara sayaca</i>	FR	-0.3	0.0	-0.5
<i>Tangara varia</i>	FR	-0.1	0.1	-1.0
<i>Tangara velia</i>	FR	-0.1	0.0	-0.8
<i>Tapera naevia</i>	IN	0.0	-0.4	-0.1
<i>Taraba major</i>	IN	-0.1	0.4	-0.1
<i>Terenotriccus erythrurus</i>	IN	0.0	0.1	-1.1
<i>Tersina viridis</i>	FR	-0.2	0.0	-0.7
<i>Thalurania furcata</i>	NE			
<i>Thamnomanes caesius</i>	IN	0.0	0.0	-0.7
<i>Thamnophilus aethiops</i>	IN	-0.1	0.1	-0.6
<i>Thamnophilus amazonicus</i>	IN	0.0	0.1	-0.6
<i>Thamnophilus palliatus</i>	IN	-0.1	0.1	-0.5
<i>Thamnophilus schistaceus</i>	IN	0.0	0.2	-0.7
<i>Thamnophilus stictocephalus</i>	IN	0.0	0.1	-0.6
<i>Theristicus caudatus</i>	IN	4.0	0.6	3.5
<i>Tinamus guttatus</i>	FR	0.0	1.1	0.2
<i>Tinamus tao</i>	FR	0.0	1.4	1.0
<i>Tityra cayana</i>	IN	-0.2	0.1	0.1
<i>Tityra inquisitor</i>	IN	-0.2	0.1	0.0
<i>Tityra semifasciata</i>	IN	-0.3	0.2	0.1
<i>Todirostrum chrysocrotaphum</i>	IN	0.0	0.2	-1.0
<i>Todirostrum cinereum</i>	IN	0.1	0.3	-1.0
<i>Tolmomyias assimilis</i>	IN	-0.1	0.1	-0.7
<i>Tolmomyias flaviventris</i>	IN	-0.1	0.1	-0.9
<i>Tolmomyias poliocephalus</i>	IN	-0.1	0.1	-0.9
<i>Tolmomyias sulphurescens</i>	IN	-0.1	0.1	-0.8
<i>Touit huetii</i>	FR	-0.5	-0.2	-0.2

<i>Tringa solitaria</i>	IN	0.9	0.4	-0.4
<i>Troglodytes musculus</i>	IN	0.2	0.2	-1.0
<i>Trogon curucui</i>	IN	-0.4	-0.7	0.0
<i>Trogon melanurus</i>	FR	-0.5	-0.9	0.3
<i>Trogon rufus</i>	IN	-0.4	-0.9	0.0
<i>Trogon violaceus</i>	FR	-0.4	-0.7	-0.1
<i>Trogon viridis</i>	FR	-0.4	-1.1	0.3
<i>Turdus albicollis</i>	IN	0.0	0.2	-0.4
<i>Turdus fumigatus</i>	IN	0.1	0.2	-0.2
<i>Turdus leucomelas</i>	IN	0.1	0.2	-0.2
<i>Tyrannentes stolzmanni</i>	IN	-0.1	0.1	-1.1
<i>Tyrannopsis sulphurea</i>	IN	-0.1	0.0	-0.3
<i>Tyrannulus elatus</i>	FR	-0.1	0.0	-1.1
<i>Tyrannus melancholicus</i>	IN	0.0	-0.3	-0.1
<i>Tyrannus savana</i>	IN	-0.1	-1.2	-0.1
<i>Urubitinga urubitinga</i>	VE			
<i>Vanellus chilensis</i>	IN	0.3	1.6	0.7
<i>Veniliornis affinis</i>	IN	0.3	0.0	-0.2
<i>Vireo olivaceus</i>	IN	0.0	0.1	-0.8
<i>Vireolanius leucotis</i>	IN	-0.1	0.2	-0.7
<i>Volatinia jacarina</i>	SE			
<i>Willisornis poecilinotus</i>	IN	0.0	0.3	-0.8
<i>Xenops minutus</i>	IN	0.1	0.0	-0.9
<i>Xiphocolaptes promeropirhynchus</i>	IN	0.8	-0.1	0.6
<i>Xipholena lamellipennis</i>	FR	-0.1	0.0	-0.4
<i>Xipborhynchus guttatus</i>	IN	0.6	-0.2	0.1
<i>Xipborhynchus obsoletus</i>	IN	0.3	-0.1	-0.4
<i>Xipborhynchus spixii</i>	IN	0.4	-0.1	-0.3
<i>Zenaida auriculata</i>	SE			
<i>Zimmerius acer</i>	FR	0.0	0.1	-1.1

¹Guild codes: IN = Insectivore, FR = Frugivore, SE = Granivore, OM = Omnivore, NE = Nectarivore, VE = Vertebrate, SC = Scavenger.

Supplementary results

TABLE B2. Wilcoxon analyses of insectivore and frugivore communities testing whether observed FD is different in disturbed habitats relative to primary forest. The test is two-tailed (i.e. testing whether the observed values are significantly different from communities in primary forest in either direction).

Group	Land use	Insectivores		Frugivores	
		V	Wilcoxon <i>P</i>	V	Wilcoxon <i>P</i>
All	PF vs DPF	3695	0.742	4148	0.082
	PF vs SF	4515	<0.001	4075	<0.001
	PF vs PA	6055	<0.001	5311	<0.001
	PF vs AG	2185	<0.001		
	DPF vs SF	3511	<0.001	2732	0.013
	DPF vs PA	4688	<0.001	3686	<0.001
	DPF vs AG	1679	<0.001		
	SF vs PA	2606	0.056	2772	<0.002
	SF vs AG	1238	<0.001		
	PA vs AG	1447	<0.001		
	PF vs DPF	3917	0.307	4029	0.171
	PF vs SF	4901	<0.001	4026	<0.001
Forest	PF vs PA	4641	<0.001	5383	<0.001
	PF vs AG				
	DPF vs SF	3654	<0.001	2684	0.023
	DPF vs PA	3540	<0.001	3805	<0.001
	DPF vs AG				
	SF vs PA	2611	<0.001	2853	<0.001
	SF vs AG				
	PA vs AG				
Non-forest	PF vs DPF	313	0.549		
	PF vs SF	404	0.846		
	PF vs PA	330	0.002		
	PF vs AG	245	0.175		
	DPF vs SF	1029	0.466		
	DPF vs PA	787	<0.001		
	DPF vs AG	613	0.048		
	SF vs PA	735	<0.001		
	SF vs AG	622	0.232		
	PA vs AG	1461	<0.001		

TABLE B3. Wilcoxon analyses of insectivore and frugivore communities testing whether observed FD is significantly different from the null expectation. The test is two-tailed (i.e. testing whether the observed values are significantly different from the null expectation in either direction). The standardised effect size (SES) is provided along with the standard error (SE). Analyses were run separately for individual trait syndromes in different land-uses.

Group	Land use	Insectivores				Frugivores			
		SES (mean + SE)	No. plots < expectation (number of communities)	V	Wilcoxon <i>P</i>	SES (mean + SE)	No. plots < expectation (number of communities)	V	Wilcoxon <i>P</i>
All	Undisturbed primary forest	0.046 (0.665)	33 (97)	2757	0.172	0.083 (0.863)	43 (97)	2717	0.221
	Disturbed primary forest	0.073 (0.567)	26 (74)	1670	0.129	0.027 (0.859)	44 (74) 41 (59)	1418	0.872
	Secondary forest	-0.162 (0.573)	34 (59)	670	0.105	-0.211 (0.809)	41 (59)	541	0.010
	Pasture	0.015 (0.589)	40 (74)	1487	0.594	-0.288 (0.645)	50 (59)	357	<0.001
	Agriculture	0.167 (0.693)	8 (23)	178	0.235				
Forest	Undisturbed primary forest	-0.043 (0.725)	55 (97)	2248	0.645	0.024 (0.796)	49 (97)	2385	0.977
	Disturbed primary forest	0.061 (0.699)	37 (74)	1636	0.182	0.100 (0.930)	37 (74)	1482	0.613
	Secondary forest	-0.195 (0.743)	37 (59)	674	0.112	-0.038 (0.868)	33 (59)	777	0.417
	Pasture	-0.274 (0.553)	35 (48)	385	0.037	-0.481 (0.465)	52 (57)	157	<0.001
	Agriculture								
Non-forest	Undisturbed primary forest	0.660 (1.446)	8 (17)	104	0.207				
	Disturbed primary forest	0.611 (1.272)	17 (41)	591	0.037	-0.431 (0.365)	7 (8)	2	0.023
	Secondary forest	-0.369 (0.637)	34 (46)	211	<0.001	-0.356 (0.814)	19 (22)	49	0.010
	Pasture	-0.061 (0.941)	43 (74)	1202	0.319	1.84 (0.789)	0 (10)	55	0.002
	Agriculture	0.244 (1.097)	12 (23)	147	0.800				

TABLE B4. Wilcoxon analyses of insectivore and frugivore communities testing whether observed FDis is significantly different from the null expectation. The test is two-tailed (i.e. testing whether the observed values are significantly different from the null expectation in either direction). The standardised effect size (SES) is provided along with the standard error (SE). Analyses were run separately for individual trait syndromes in different land-uses.

Group	Land use	Insectivores		Frugivores	
		V	WilcoxonP	V	WilcoxonP
All	PF vs DPF	3150	0.172	3613	0.942
	PF vs SF	3670	0.003	4406	<0.001
	PF vs PA	2292	<0.001	6370	<0.001
	PF vs AG	590	<0.001	1135	<0.001
	DPF vs SF	2965	<0.001	3328	<0.001
	DPF vs PA	1979	0.004	4814	<0.001
	DPF vs AG	512	0.002	864	<0.001
	SF vs PA	1008	<0.001	3308	<0.001
	SF vs AG	268	<0.001	649	<0.001
	PA vs AG	653	0.053	634	0.003
Forest	PF vs DPF	2958	0.049	3516	0.821
	PF vs SF	3148	0.296	4238	<0.001
	PF vs PA	4615	<0.001	6202	<0.001
	PF vs AG	1643	<0.001	1137	<0.001
	DPF vs SF	2649	0.035	3225	<0.001
	DPF vs PA	3579	<0.001	4659	<0.001
	DPF vs AG	1252	<0.001	862	<0.001
	SF vs PA	2711	<0.001	3241	<0.001
	SF vs AG	979	<0.001	651	<0.001
	PA vs AG	928	<0.001	613	0.004
Non-forest	PF vs DPF	812	0.001	126	0.049
	PF vs SF	622	<0.001	144	0.004
	PF vs PA	464	<0.001	138	0.036
	PF vs AG	129.5	<0.001*		
	DPF vs SF	1515	0.768	549	0.147
	DPF vs PA	1242	<0.001	461	0.758
	DPF vs AG	315	<0.001		
	SF vs PA	1048	<0.001	848	0.322
	SF vs AG	278	<0.001		
	PA vs AG	601	0.018		

TABLE B5. Principal Components Analysis for Insectivores. Eigenvalues and the proportion of variance explained are presented. The first principal components for locomotory and trophic traits were used in a secondary Principal Components Analysis to create an axis representing overall size. The second Principal Components for locomotory and trophic traits represented the shape of these syndromes and was used in further analysis.

PCA	PC	Proportion variance	PCA loadings		
Locomotory	1	0.695	Tarsus length	Tail length	Wing chord
	2	0.237	0.471	0.604	0.642
Trophic	1	0.790	0.864	-0.464	-0.197
	2	0.154	Beak length	Beak width	Beak depth
Overall size	1	0.812	0.532	0.593	0.604
			0.842	-0.446	-0.304
			Trophic PC1	Locomotory PC1	
			0.707	0.707	

TABLE B6. Principal Components Analysis for Frugivores. Eigenvalues and the proportion of variance explained are presented. The first principal components for locomotory and trophic traits were used in a secondary Principal Components Analysis to create an axis representing overall size. The second Principal Components for locomotory and trophic traits represented the shape of these syndromes and was used in further analysis.

PCA	PC	Proportion variance	PCA loadings		
			Tarsus length	Tail length	Wing chord
Locomotory	1	0.847	0.560	0.565	0.606
	2	0.119	0.726	-0.687	-0.458
			Beak length	Beak width	Beak depth
Trophic	1	0.931	0.567	0.590	0.574
	2	0.056	0.760	-0.108	-0.641
			Trophic PC1	Locomotory PC1	
Overall size	1	0.686	0.707	0.707	

TABLE B7. Wilcoxon analyses of insectivore and frugivore communities testing whether observed FD is significantly different from the null expectation. The test is two-tailed (i.e. testing whether the observed values are significantly different from the null expectation in either direction). * indicates that the observed values were significantly different from the null model. The standardised effect size (SES) is provided along with the standard error (SE). Analyses were run separately for individual trait syndromes in different land-uses.

Trait syndrome	Land use	Insectivores				Frugivores			
		Mean SES (SE)	No. plots < expectation (no. of communities)	V	WilcoxonP	SES (mean + SE)	No. plots < expectation (no. of communities)	V	WilcoxonP
Dispersal	Undisturbed primary forest	-0.195 (0.755)	44 (97)	2166	0.450	0.072 (0.752)	41 (97)	2963	0.035
	Disturbed primary forest	-0.032 (0.607)	33 (74)	1480	0.620	-0.081 (0.823)	34 (74)	1337	0.788
	Secondary forest	-0.143 (0.595)	26 (59)	746	0.296	0.071 (0.843)	23 (59)	1046	0.226
	Pasture	0.375 (0.468)	12 (74)	2394	<0.001	0.363 (0.691)	16 (59)	1336	<0.001
	Agriculture	0.611 (0.427)	2 (23)	273	<0.001				
	Undisturbed primary forest	0.189 (0.576)	43 (97)	3439	<0.001	-0.053 (0.860)	44 (97)	2222	0.580
Beak	Disturbed primary forest	-0.025 (0.531)	48 (74)	1401	0.944	0.083 (0.853)	29 (74)	1472	0.651
	Secondary forest	-0.376 (0.505)	49 (59)	328	<0.001	-0.119 (0.966)	26 (59)	766	0.371
	Pasture	-0.281 (0.276)	64 (74)	213	<0.001	-0.227 (0.892)	28 (59)	628	0.053
	Agriculture	0.301 (0.439)	7 (23)	227	0.005				
	Undisturbed primary forest	0.115 (0.597)	35 (97)	3109	0.008	0.076 (1.171)	30 (97)	3076	0.012
	Disturbed primary forest	0.089 (0.575)	34 (74)	1619	0.213	0.204 (0.753)	22 (74)	1972	0.002
Overall	Secondary forest	-0.064 (0.567)	32 (59)	815	0.600	-0.327 (1.057)	31 (59)	708	0.183
	Pasture	0.173 (0.378)	20 (74)	2131	<0.001	-0.646 (0.795)	41 (59)	297	<0.001
	Agriculture	0.207 (0.633)	7 (23)	184	0.170				
	Undisturbed primary forest	0.073 (0.544)	41 (97)	2865	0.080	0.044 (0.815)	48 (97)	2485	0.700
	Disturbed primary forest	0.062 (0.479)	32 (74)	1669	0.130	0.016 (0.895)	28 (74)	2485	0.304
	Secondary forest	-0.097 (0.556)	35 (59)	683	0.128	-0.042 (0.908)	25 (59)	913	0.836
Locomotory	Pasture	0.280 (0.487)	19 (74)	2210	<0.001	-0.581 (1.015)	38 (59)	399	<0.001
	Agriculture	0.246 (0.533)	8 (23)	195	0.086				

TABLE B8. Wilcoxon analyses of forest insectivore and frugivore communities testing whether observed FD is significantly different from the null expectation. The test is two-tailed (i.e. testing whether the observed values are significantly different from the null expectation in either direction). * indicates that the observed values were significantly different from than the null model. The standardised effect size (SES) is provided along with the standard error (SE). Analyses were run separately for individual trait syndromes in different land-uses.

Trait syndrome	Land use	Insectivores				Frugivores			
		Mean SES (SE)	No. plots < expectation (no. of communities)	V	WilcoxonP	SES (mean + SE)	No. plots < expectation (no. of communities)	V	Wilcoxon P
Dispersal	Undisturbed primary forest	-0.509 (1.120)	58 (97)	1526	0.002	0.011 (0.914)	41 (97)	2725	0.211
	Disturbed primary forest	-0.101 (0.769)	35 (74)	1339	0.796	-0.015 (0.994)	31 (74)	1575	0.314
	Secondary forest	-0.067 (0.849)	26 (49)	835	0.709	0.092 (1.009)	36 (49)	1058	0.193
	Pasture	0.545 (0.745)	18 (48)	968	<0.001	0.303 (0.747)	14 (57)	1151	0.010
	Agriculture								
Beak	Undisturbed primary forest	0.176 (0.701)	46 (97)	3219	0.002	-0.116 (0.973)	47 (97)	2038	0.224
	Disturbed primary forest	-0.025 (0.605)	47 (74)	1364	0.901	0.091 (0.815)	31 (74)	1495	0.564
	Secondary forest	-0.294 (0.619)	44 (49)	438	<0.001	0.042 (0.912)	24 (49)	943	0.664
	Pasture	0.119 (0.573)	18 (48)	726	0.160	-0.098 (0.910)	26 (57)	682	0.253
	Agriculture								
Overall	Undisturbed primary forest	-0.016 (0.812)	36 (97)	2907	0.057	0.018 (0.995)	30 (97)	2798	0.130
	Disturbed primary forest	0.068 (0.758)	26 (74)	1832	0.017	0.128 (0.742)	31 (74)	1649	0.160
	Secondary forest	0.124 (0.689)	20 (49)	1107	0.094	-0.232 (0.912)	32 (49)	669	0.104
	Pasture	-0.144 (0.690)	36 (48)	390	0.042	-0.712 (0.720)	46 (57)	165	<0.001
	Agriculture								
Locomotory	Undisturbed primary forest	0.060 (0.691)	40 (97)	2921	0.050	0.239 (0.776)	40 (97)	3126	0.007
	Disturbed primary forest	0.100 (0.603)	32 (74)	1753	0.049	-0.143 (0.854)	39 (74)	1186	0.279
	Secondary forest	-0.032 (0.638)	29 (49)	979	0.480	-0.452 (0.795)	42 (49)	398	<0.001
	Pasture	-0.512	34 (48)	259	<0.001	-0.632	47 (57)	363	<0.001

(0.803)

(1.047)

Agriculture



TABLE B9. Wilcoxon analyses of non-forest insectivore and frugivore communities testing whether observed FD is significantly different from the null expectation. The test is two-tailed (i.e. testing whether the observed values are significantly different from the null expectation in either direction). * indicates that the observed values were significantly different from than the null model. The standardised effect size (SES) is provided along with the standard error (SE). Analyses were run separately for individual trait syndromes in different land-uses.

Trait syndrome	Land use	Insectivores				Frugivores			
		Mean SES (SE)	No. plots < expectation (no. of communities)	V	WilcoxonP	SES (mean + SE)	No. plots < expectation (no. of communities)	V	WilcoxonP
Dispersal	Undisturbed primary forest	-0.304 (0.815)	14 (17)	39	0.080				
	Disturbed primary forest	-0.134 (0.934)	27 (41)	304	0.103	0.109 (0.916)	3 (8)	19	0.945
	Secondary forest	-0.210 (1.082)	27 (46)	399	0.124	-0.367 (0.941)	11 (22)	96	0.337
	Pasture	0.196 (1.204)	27 (74)	1627	0.198	0.580 (0.950)	3 (10)	46	0.064
	Agriculture	0.883 (1.030)	4 (23)	246	<0.001				
	Undisturbed primary forest	0.674 (1.536)	5 (17)	110	0.120				
Beak	Disturbed primary forest	0.105 (1.099)	19 (41)	450	0.808	-0.261 (1.131)	5 (8)	16	0.844
	Secondary forest	0.237 (1.019)	22 (46)	640	0.282	-0.193 (1.011)	14 (22)	97	0.354
	Pasture	-0.168 (0.548)	50 (74)	854	0.004	0.354 (1.022)	4 (10)	40	0.232
	Agriculture	0.236 (0.516)	9 (23)	199	0.065				
	Undisturbed primary forest	0.493 (1.084)	7 (17)	110	0.120				
Overall	Disturbed primary forest	0.389 (0.971)	17 (41)	601	0.027	0.047 (0.936)	4 (8)	18	1
	Secondary forest	-0.571 (1.018)	31 (46)	239	<0.001	-0.098 (1.004)	15 (22)	103	0.463
	Pasture	0.170 (0.847)	22 (74)	1884	0.008	0.495 (0.732)	2 (10)	48	0.037
	Agriculture	0.256 (0.918)	8 (23)	184	0.170				
	Undisturbed primary forest	-0.067 (0.995)	10 (17)	68	0.712				
Locomotory	Disturbed primary forest	-0.307 (0.766)	31 (41)	257	0.024	0.130 (0.881)	4 (8)	20	0.844
	Secondary forest	0.050 (0.925)	23 (46)	606	0.481	-0.382 (0.976)	15 (22)	75	0.098
	Pasture	0.197 (1.232)	39 (74)	1482	0.613	0.670 (0.962)	2 (10)	44	0.106
	Agriculture	0.218 (0.818)	12 (23)	150	0.731				

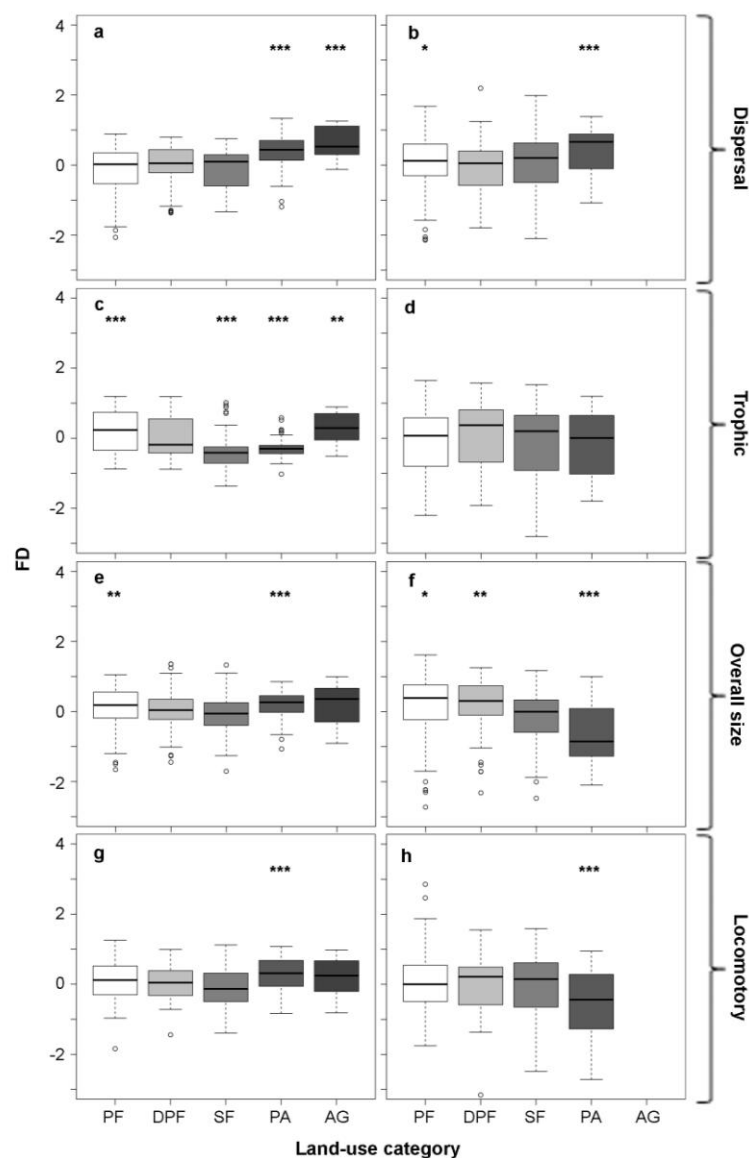


FIGURE B1. Standardised effect size of FD (Petchey and Gaston 2002) for insectivores (a,c,e,g) and frugivores (b,d,f,h) in 330 communities across five land uses: primary forest (PF), disturbed primary forest (DPF), secondary forest (SF), pasture (PF) and agriculture (AG). * indicates that observed FD was significantly different from the null expectation using a two-tailed Wilcoxon analysis (* < 0.05 , ** < 0.01 , *** < 0.001). Full Wilcoxon results and number of communities in each land use category can be found in Table B9.

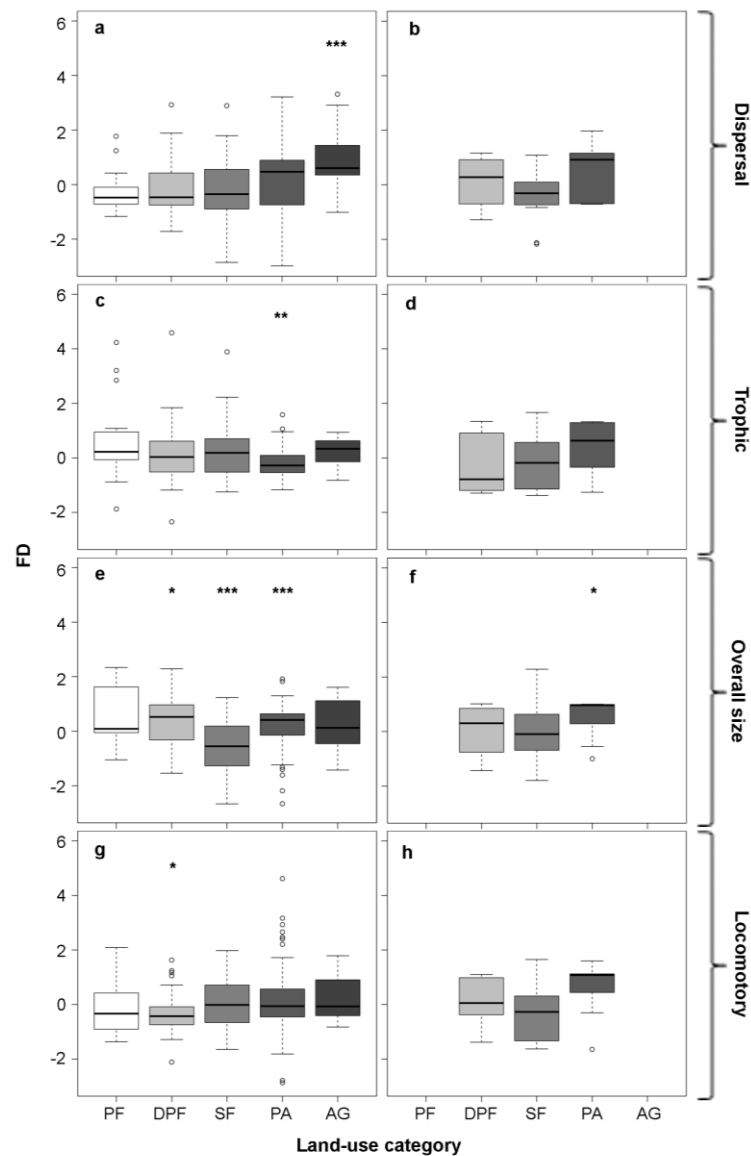


FIGURE B2. Standardised effect size of FD (Petchey and Gaston 2002) for non-forest insectivores (a,c,e,g) and frugivores (b,d,f,h) in 330 communities across five land uses: primary forest (PF), disturbed primary forest (DPF), secondary forest (SF), pasture (PF) and agriculture (AG). * indicates that observed FD was significantly different from the null expectation using a two-tailed Wilcoxon analysis (* < 0.05 , ** < 0.01 , *** < 0.001). Full Wilcoxon results and number of communities in each land use category can be found in Table B9.

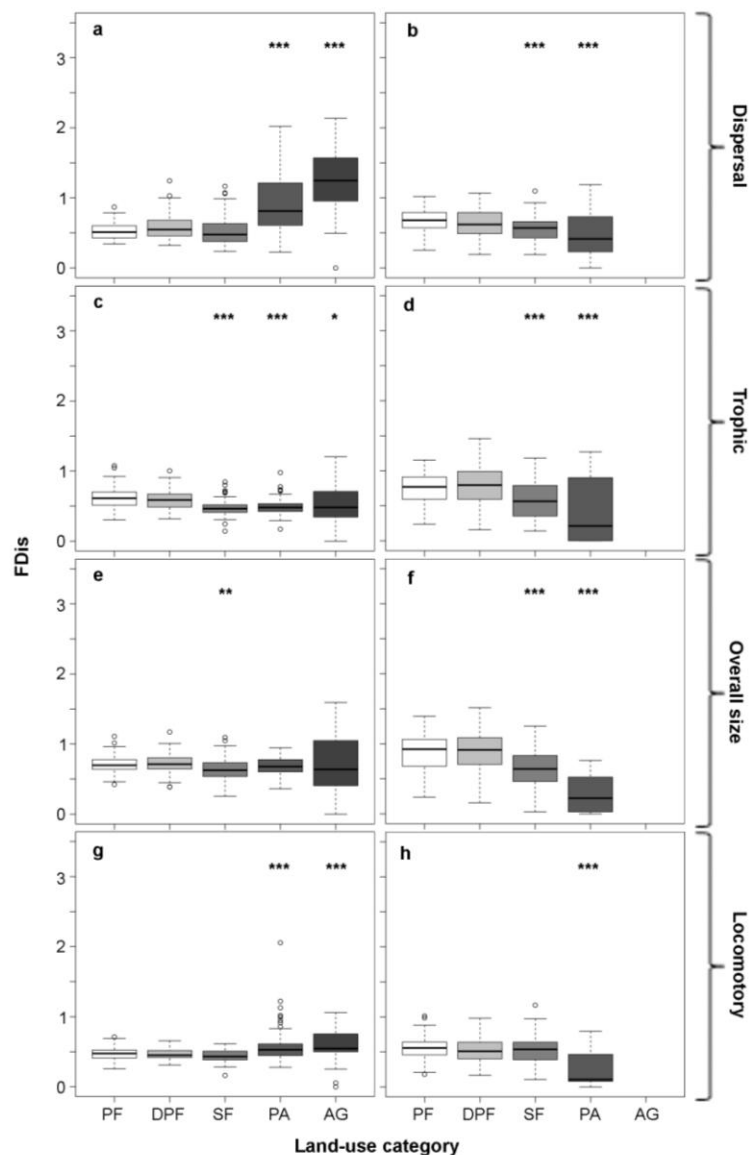


FIGURE B3. Raw FDis (Laliberte and Legendre 2010) for insectivores (a,c,e,g) and frugivores (b,d,f,h) in 330 communities across five land uses: primary forest (PF), disturbed primary forest (DPF), secondary forest (SF), pasture (PF) and agriculture (AG). * indicates that the observed FDis was significantly different from the primary forest FDis using a two-tailed Wilcoxon analysis (* < 0.05 , ** < 0.01 , *** < 0.001). Full Wilcoxon results, comparing observed FDis values between all land-use categories can be found in Tables B10.

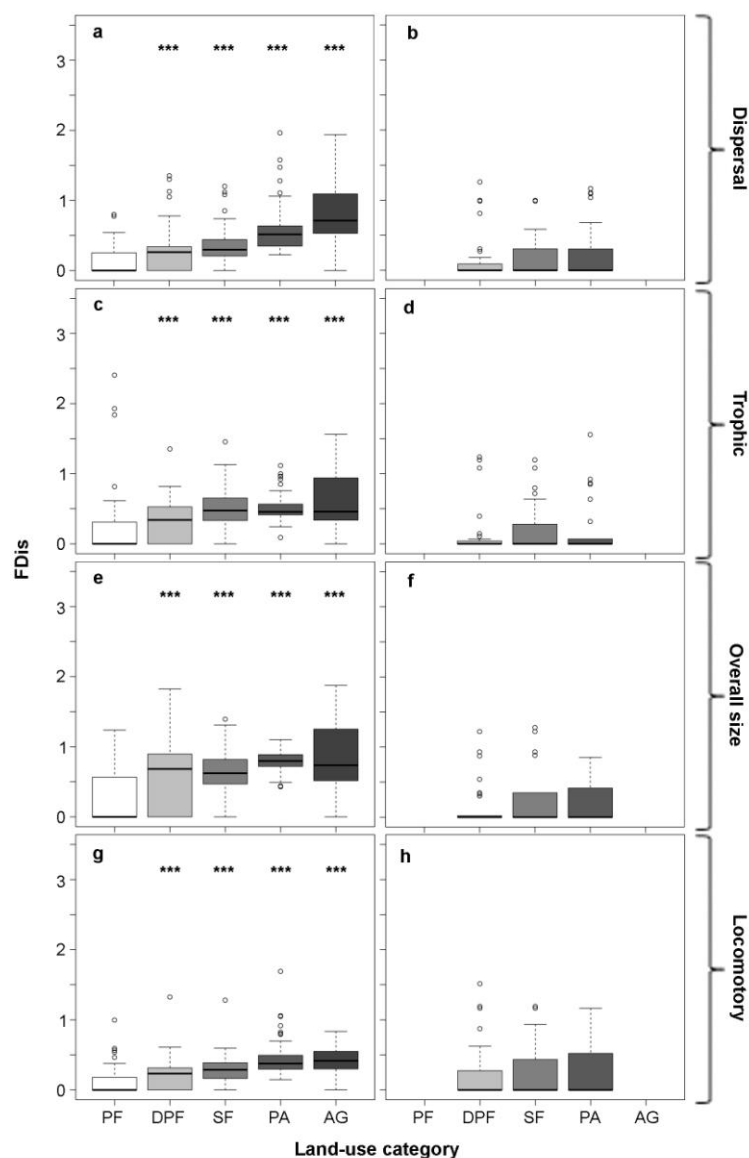


FIGURE B4. Raw FDis (Laliberte and Legendre 2010) for non-forest insectivores (a,c,e,g) and frugivores (b,d,f,h) in 330 communities across five land uses: primary forest (PF), disturbed primary forest (DPF), secondary forest (SF), pasture (PF) and agriculture (AG). * indicates that the observed FDis was significantly different from the primary forest FDis using a two-tailed Wilcoxon analysis (* < 0.05 , ** < 0.01 , *** < 0.001). Full Wilcoxon results, comparing observed FDis values between all land-use categories can be found in Table B12. Wilcoxon analyses were not undertaken for agricultural communities as there were too few species to calculate FDis.

TABLE B10. Wilcoxon analyses of insectivore and frugivore communities testing whether observed FDis is significantly different in primary forest communities relative to communities in other land-uses. The test is two-tailed (i.e. testing whether FDis is significantly different in either direction). Analyses were run separately for individual trait syndromes in different land-uses: PF (Primary forest), DPF (Disturbed primary forest), SF (Secondary forest), PA (Pasture), AG (Agriculture).

Trait syndrome	Land use comparison	Insectivores		Frugivores	
		V	Wilcoxon P	V	Wilcoxon P
Dispersal	PF vs DPF	3020	0.076	4104	0.109
	PF vs SF	3214	0.198	3909	<0.001
	PF vs PA	1130	<0.001	4641	<0.001
	PF vs AG	202	<0.001	1084	<0.001
	DPF vs SF	2653	0.033	2559	0.089
	DPF vs PA	1193	<0.001	3288	0.003
	DPF vs AG	197	<0.001	824	<0.001
	SF vs PA	860	<0.001	2413	0.071
	SF vs AG	139	<0.001	649	<0.001
	PA vs AG	520	0.002	683	<0.001
Trophic	PF vs DPF	4059	0.143	3157	0.179
	PF vs SF	4408	<0.001	3867	<0.001
	PF vs PA	5474	<0.001	4410	<0.001
	PF vs AG	1484	0.038	970	<0.001
	DPF vs SF	3189	<0.001	3143	<0.001
	DPF vs PA	3961	<0.001	3566	<0.001
	DPF vs AG	1086	0.103	754	<0.001
	SF vs PA	2006	0.424	2417	0.068
	SF vs AG	666	0.677	590	<0.001
	PA vs AG	889	0.997	610	0.009
Overall	PF vs DPF	3347	0.452	3469	0.710
	PF vs SF	3680	0.003	4131	<0.001
	PF vs PA	3913	0.313	6250	<0.001
	PF vs AG	1261	0.531	1109	<0.001
	DPF vs SF	2892	0.001	3216	<0.001
	DPF vs PA	3153	0.112	4803	<0.001
	DPF vs AG	974	0.480	853	<0.001

	SF vs PA	1775	0.065	3353	<0.001
	SF vs AG	713	0.964	638	<0.001
	PA vs AG	923	0.776	625	0.005
	PF vs DPF	3768	0.578	3979	0.225
	PF vs SF	3315	0.098	3050	0.492
	PF vs PA	2467	<0.001	5716	<0.001
	PF vs AG	641	0.001	1115	<0.001
	DPF vs SF	2464	0.204	2134	0.826
Locomotory	DPF vs PA	1745	<0.001	4188	<0.001
	DPF vs AG	454	<0.001	840	<0.001
	SF vs PA	1297	<0.001	3373	<0.001
	SF vs AG	349	<0.001	667	<0.001
	PA vs AG	780	0.375	643	0.002

TABLE B11. Wilcoxon analyses of forest insectivore and frugivore communities testing whether observed FDis is significantly different in primary forest communities relative to communities in other land-uses. The test is two-tailed (i.e. testing whether FDis is significantly different in either direction). Analyses were run separately for individual trait syndromes in different land-uses: PF (Primary forest), DPF (Disturbed primary forest), SF (Secondary forest), PA (Pasture), AG (Agriculture).

Trait syndrome	Land use comparison	Insectivores		Frugivores	
		V	WilcoxonP	V	WilcoxonP
Dispersal	PF vs DPF	2904	0.033	4210	0.053
	PF vs SF	2845	0.953	3922	<0.001
	PF vs PA	2601	0.010	5038	<0.001
	PF vs AG	1358	<0.001	1085	<0.001
	DPF vs SF	2410	0.305	2566	0.084
	DPF vs PA	2106	0.053	3553	<0.001
	DPF vs AG	1054	<0.001	819	<0.001
	SF vs PA	1885	0.395	2619	0.002
	SF vs AG	862	<0.001	644	<0.001
	PA vs AG	916	<0.001	649	<0.001
Trophic	PF vs DPF	3933	0.284	3028	0.081
	PF vs SF	4695	<0.001	3637	0.005
	PF vs PA	6554	<0.001	4091	0.005
	PF vs AG	1649	<0.001	971	<0.001
	DPF vs SF	3468	<0.001	2994	<0.001
	DPF vs PA	4956	<0.001	3348	<0.001
	DPF vs AG	1258	<0.001	756	<0.001
	SF vs PA	3404	<0.001	2333	0.082
	SF vs AG	994	<0.001	590	<0.001
	PA vs AG	946	<0.001	596	0.008
Overall	PF vs DPF	3454	0.675	3365	0.486
	PF vs SF	3632	0.005	3930	<0.001
	PF vs PA	5763	<0.001	5975	<0.001
	PF vs AG	1646	<0.001	1109	<0.001
	DPF vs SF	2776	0.007	3124	<0.001
	DPF vs PA	4401	<0.001	4634	<0.001

	DPF vs AG	1252	<0.001	857	<0.001
	SF vs PA	3299	<0.001	3218	<0.001
	SF vs AG	979	<0.001	646	<0.001
	PA vs AG	904	<0.001	615	0.003
	PF vs DPF	3492	0.764	4361	0.016
	PF vs SF	2895	0.904	3482	0.023
	PF vs PA	5147	<0.001	5768	<0.001
	PF vs AG	1649	<0.001	1112	<0.001
	DPF vs SF	2250	0.763	2364	0.415
Locomotory	DPF vs PA	3932	<0.001	4129	<0.001
	DPF vs AG	1258	<0.001	831	<0.001
	SF vs PA	3160	<0.001	3080	<0.001
	SF vs AG	1003	<0.001	645	<0.001
	PA vs AG	958	<0.001	614	0.004

TABLE B12. Wilcoxon analyses of non-forest insectivore and frugivore communities testing whether observed FDis is significantly different in primary forest communities relative to communities in other land-uses. The test is two-tailed (i.e. testing whether FDis is significantly different in either direction). Analyses were run separately for individual trait syndromes in different land-uses: PF (Primary forest), DPF (Disturbed primary forest), SF (Secondary forest), PA (Pasture), AG (Agriculture).

Trait syndrome	Land use comparison	Insectivores		Frugivores	
		V	WilcoxonP	V	WilcoxonP
Dispersal	PF vs DPF	815	0.001	126	0.049
	PF vs SF	561	<0.001	144	0.004
	PF vs PA	282	<0.001	138	0.036
	PF vs AG	84	<0.001		
	DPF vs SF	1298	<0.001	547	0.140
	DPF vs PA	763	<0.001	450	0.616
	DPF vs AG	223	<0.001		
	SF vs PA	946	<0.001	818	0.513
	SF vs AG	244	<0.001		
	PA vs AG	563	0.007		
	PF vs DPF	834	0.002	126	0.049
Trophic	PF vs SF	521	<0.001	144	0.004
	PF vs PA	452	<0.001	138	0.036
	PF vs AG	179	<0.001		
	DPF vs SF	1098	0.006	540	0.118
	DPF vs PA	1317	<0.001	460	0.745
	DPF vs AG	417	0.004		
	SF vs PA	1929	0.741	855	0.285
	SF vs AG	567	0.383		
	PA vs AG	896	0.951		
	PF vs DPF	762	<0.001	126	0.049
	PF vs SF	600	<0.001	144	0.004
Overall	PF vs PA	389	<0.001	138	0.036
	PF vs AG	170	<0.001		
	DPF vs SF	1548	0.918	546	0.137
	DPF vs PA	1475	0.002	462	0.772

Locomotory	DPF vs AG	489	0.034		
	SF vs PA	1136	<0.001	864	0.242
	SF vs AG	470	0.054		
	PA vs AG	906	0.885		
	PF vs DPF	871	0.004	126	0.049
	PF vs SF	603	<0.001	144	0.004
	PF vs PA	430	<0.001	138	0.036
	PF vs AG	168	<0.001		
	DPF vs SF	1180	0.024	553	0.161
	DPF vs PA	753	<0.001	469	0.867
	DPF vs AG	302	<0.001		
	SF vs PA	1106	<0.001	867	0.229
	SF vs AG	369	0.003		
	PA vs AG	839	0.689		

TABLE B13. Principal Components Analysis for environmental variables (mean canopy openness and mean basal area for 30 sites). Eigenvalues and the proportion of variance explained are presented.

PCA	PC	Proportion variance	PCA Loadings	
			Mean canopy openness	Mean basal area
Environmental	1	0.924	-0.707	0.707
	2	0.076	0.707	0.707

TABLE B14. The effect of disturbance (increasing canopy openness and decreasing basal area) upon functional traits in insectivores and frugivores. The initial ‘global’ model included a quadratic explanatory variable (Environmental²).

Community type	Guild	Trait syndrome	Explanatory variable	Estimate (S.E.)	t - value	P - value
Forest	IN	Dispersal	Environmental	-0.103 (0.006)	-17.762	<0.001
			Environmental ²	0.048 (0.006)	8.139	<0.001
		Trophic	Environmental	0.115 (0.006)	20.264	<0.001
			Environmental ²	-0.035 (0.006)	-5.944	<0.001
		Overall size	Environmental	0.099 (0.015)	6.384	<0.001
		Locomotory	Environmental	0.155 (0.009)	16.360	<0.001
			Environmental ²	-0.075 (0.010)	-7.840	<0.001
	FR	Dispersal	Environmental	-0.008 (0.003)	-1.058	0.010
		Trophic	Environmental	0.086 (0.010)	8.796	<0.001
		Overall size	Environmental	0.153 (0.022)	6.968	<0.001
		Locomotory	Environmental	-0.009 (0.003)	-2.710	0.007
Not Forest	IN	Dispersal	Environmental	0.191 (0.412)	1.706	0.089
			Environmental ²	0.553 (0.441)	4.600	<0.001
		Trophic	Environmental	0.037 (0.016)	2.327	0.021
		Overall	Environmental	0.237 (0.052)	4.537	<0.001
			Environmental ²	0.162 (0.056)	2.904	0.003
		Locomotory	Environmental	-0.026 (0.016)	-1.622	0.106
	FR	Dispersal	Environmental	-0.078 (0.015)	-5.238	<0.001
		Trophic	Environmental	0.043 (0.015)	2.903	0.004
		Overall	Environmental	-0.037 (0.040)	-0.928	0.356
			Environmental ²	0.135 (0.036)	3.711	<0.001
		Locomotory	Environmental	0.192 (0.025)	7.680	<0.001

Sensitivity analyses

TABLE B15. Wilcoxon analyses of insectivore communities testing whether observed FD is different in disturbed habitats relative to primary forest (Santarem and Paragominas are analysed separately). The test is two-tailed (i.e. testing whether the observed values are significantly different from communities in primary forest in either direction).

Group	Land use	Santarem		Paragominas	
		V	WilcoxonP	V	WilcoxonP
All	PF vs DPF	817	0.149	1019	0.377
	PF vs SF	1426	<0.001	820	<0.001
	PF vs PA	963	<0.001	2131	<0.001
	PF vs AG	568	<0.001	526	<0.001
	DPF vs SF	953	<0.001	723	<0.001
	DPF vs PA	670	<0.001	1877	<0.001
	DPF vs AG	392	<0.001	430	<0.001
	SF vs PA	649	0.004	533	0.774
	SF vs AG	454	<0.001	196	<0.001
	PA vs AG	243	0.002	488	<0.001
	PF vs DPF	828	0.118	1156	0.906
Forest	PF vs SF	1537	<0.001	900	<0.001
	PF vs PA	880	<0.001	1479	<0.001
	PF vs AG	132	0.004		
	DPF vs SF	1002	<0.001	738	<0.001
	DPF vs PA	616	<0.001	1204	<0.001
	DPF vs AG	93	0.005		
	SF vs PA	725	<0.001	515	<0.001
	SF vs AG	116	0.006		
	PA vs AG	38	0.376		
	PF vs DPF	1	0.534	255	0.513
	PF vs SF	6	0.321	134	0.743
Non-forest	PF vs PA	2	0.192	227	0.008
	PF vs AG	4	0.557	95	0.459
	DPF vs SF	60	0.604	285	0.480
	DPF vs PA	22	0.032	476	<0.001
	DPF vs AG	37	0.663	229	0.201

SF vs PA	172	0.004	169	<0.001
SF vs AG	214	0.352	122	0.131
PA vs AG	250	<0.001	473	<0.001

TABLE B16. Wilcoxon analyses of frugivore communities testing whether observed FD is different in disturbed habitats relative to primary forest (Santarem and Paragominas are analysed separately). The test is two-tailed (i.e. testing whether the observed values are significantly different from communities in primary forest in either direction).

Group	Land use	Santarem		Paragominas	
		V	WilcoxonP	V	WilcoxonP
All	PF vs DPF	614	0.464	1517	0.006
	PF vs SF	1008	0.174	959	<0.001
	PF vs PA	736	<0.001	2039	<0.001
	PF vs AG	131	0.005	53	0.102
	DPF vs SF	765	0.059	652	<0.001
	DPF vs PA	498	<0.001	1446	<0.001
	DPF vs AG	91	0.008	40	0.167
	SF vs PA	621	<0.001	516	0.114
	SF vs AG	114	0.007	16	0.344
	PA vs AG	44	0.056	29	0.507
	PF vs DPF	636	0.621	1426	0.035
Forest	PF vs SF	1094	0.032	967	<0.001
	PF vs PA	733	<0.001	2083	<0.001
	PF vs AG	131	0.005	53	0.102
	DPF vs SF	782	0.037	654	<0.001
	DPF vs PA	504	<0.001	1545	<0.001
	DPF vs AG	90	0.009	40	0.167
	SF vs PA	580	<0.001	589	<0.001
	SF vs AG	110	0.013	16	0.381
	PA vs AG	39	0.164	23	0.766
	PF vs DPF				
	PF vs SF				
Non-forest	PF vs PA				
	PF vs AG				
	DPF vs SF				
	DPF vs PA				
	DPF vs AG				
	SF vs PA				

SF vs AG	
PA vs AG	

TABLE B17. Wilcoxon analyses of insectivore communities testing whether observed FD is significantly different from the null expectation (Santarem and Paragominas communities were analysed separately). The test is two-tailed (i.e. testing whether the observed values are significantly different from the null expectation in either direction). The standardised effect size (SES) is provided along with the standard error (SE). Analyses were run separately for individual trait syndromes in different land-uses.

Trait syndrome	Land use	Santarem		Paragominas	
		V	WilcoxonP	V	WilcoxonP
All	Undisturbed primary forest	1030	0.610	1488	0.600
	Disturbed primary forest	475	0.944	1084	0.171
	Secondary forest	711	0.626	168	0.398
	Pasture	281	0.725	1318	0.909
	Agriculture	110	0.195	50	1
Forest	Undisturbed primary forest	899	0.569	1337	0.672
	Disturbed primary forest	444	0.615	1057	0.256
	Secondary forest	678	0.415	189	0.779
	Pasture	87	0.002	259	0.027
	Agriculture				
Non-forest	Undisturbed primary forest			160	0.235
	Disturbed primary forest	10	0.675	807	0.074
	Secondary forest	301	0.137	124	0.235
	Pasture	311	0.312	1094	0.168
	Agriculture	65	0.322	59	0.520

TABLE B18. Wilcoxon analyses of frugivore communities testing whether observed FD is significantly different from the null expectation (Santarem and Paragominas communities were analysed separately). The test is two-tailed (i.e. testing whether the observed values are significantly different from the null expectation in either direction). The standardised effect size (SES) is provided along with the standard error (SE). Analyses were run separately for individual trait syndromes in different land-uses.

Trait syndrome	Land use	Santarem		Paragominas	
		V	WilcoxonP	V	WilcoxonP
All	Undisturbed primary forest	956	0.924	1703	0.060
	Disturbed primary forest	494	0.855	973	0.678
	Secondary forest	751	0.928	98	<0.001
	Pasture	60	<0.001	511	<0.001
	Agriculture				
Forest	Undisturbed primary forest	888	0.507	1375	0.855
	Disturbed primary forest	413	0.346	1080	0.181
	Secondary forest	784	0.818	178	0.558
	Pasture	97	0.015	116	<0.001
	Agriculture				
Non-forest	Undisturbed primary forest				
	Disturbed primary forest				
	Secondary forest				
	Pasture				
	Agriculture				

TABLE B19. Wilcoxon analyses of insectivore communities testing whether observed FDis is significantly different in primary forest communities relative to communities in other land-uses (Santarem and Paragominas communities were analysed separately). The test is two-tailed (i.e. testing whether FDis is significantly different in either direction). Analyses were run separately for individual trait syndromes in different land-uses: PF (Primary forest), DPF (Disturbed primary forest), SF (Secondary forest), PA (Pasture), AG (Agriculture).

Group	Land use	Santarem		Paragominas	
		V	WilcoxonP	V	WilcoxonP
All	PF vs DPF	623	0.529	956	0.178
	PF vs SF	1031	0.012	789	<0.001
	PF vs PA	204	<0.001	954	0.010
	PF vs AG				
	DPF vs SF	761	0.065	683	<0.001
	DPF vs PA	161	<0.001	915	0.170
	DPF vs AG				
	SF vs PA	165	<0.001	163	<0.001
	SF vs AG				
	PA vs AG				
Forest	PF vs DPF	600	0.381	880	0.056
	PF vs SF	960	0.354	569	0.634
	PF vs PA	544	0.042	1860	<0.001
	PF vs AG				
	DPF vs SF	737	0.119	518	0.196
	DPF vs PA	385	0.043	1551	<0.001
	DPF vs AG				
	SF vs PA	217	<0.001	710	0.002
	SF vs AG				
	PA vs AG				
Non-forest	PF vs DPF	88	0.071	398	0.013
	PF vs SF	57	<0.001	198	0.098
	PF vs PA	0	<0.001	336	<0.001
	PF vs AG	8	<0.001	62	0.007
	DPF vs SF	158	0.011	507	0.094
	DPF vs PA	52	<0.001	780	0.025
	DPF vs AG	23	<0.001	158	0.231
	SF vs PA	232	0.007	149	<0.001
	SF vs AG	87	<0.001	44	0.021
	PA vs AG				

PA vs AG	77	0.009	254	0.992
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TABLE B20. Wilcoxon analyses of frugivore communities testing whether observed FDis is significantly different in primary forest communities relative to communities in other land-uses (Santarem and Paragominas communities were analysed separately). The test is two-tailed (i.e. testing whether FDis is significantly different in either direction). Analyses were run separately for individual trait syndromes in different land-uses: PF (Primary forest), DPF (Disturbed primary forest), SF (Secondary forest), PA (Pasture), AG (Agriculture).

Group	Land use	Santarem		Paragominas	
		V	WilcoxonP	V	WilcoxonP
All	PF vs DPF	638	0.640	1176	0.791
	PF vs SF	1350	<0.001	835	<0.001
	PF vs PA	912	<0.001	2344	<0.001
	PF vs AG				
	DPF vs SF	994	<0.001	646	0.001
	DPF vs PA	648	<0.001	1854	<0.001
	DPF vs AG				
	SF vs PA	712	<0.001	714	0.002
	SF vs AG				
	PA vs AG				
	PF vs DPF	604	0.404	1178	0.780
	PF vs SF	1294	<0.001	810	<0.001
Forest	PF vs PA	910	<0.001	2249	<0.001
	PF vs AG				
	DPF vs SF	980	<0.001	630	0.003
	DPF vs PA	648	<0.001	1768	<0.001
	DPF vs AG				
	SF vs PA	719	<0.001	681	0.002
	SF vs AG				
	PA vs AG				
	PF vs DPF				
	PF vs SF				
	PF vs PA				
	PF vs AG				
Non-forest	DPF vs SF				
	DPF vs PA				
	DPF vs AG				

SF vs PA

SF vs AG

PA vs AG

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Appendix C: Chapter 4 Supplementary Methods and Results

Contents

Expanded materials and methods: Supplementary data table.

Expanded results: Figs C1 & C2, Tables C1–C10.

Expanded materials and methods

Measurement protocol

Tarsus length was measured as the distance between the final scale of the acrotarsium to the middle of the ankle joint. Three beak measurements were taken at or from the anterior edge of the nostrils: (1) width, (2) length to the tip of the beak and (3) depth (as vertical height). Tail length was taken from the point at which the two central rectrices meet the skin to the tip of the longest rectrix. Wing chord was measured as the distance between the carpal joint and wing tip of the unflattened wing. We also measured the distance between the first secondary to the longest primary and used this to calculate the hand-wing index of Claramunt et al. (2012). This is the wing chord - distance between the first secondary to longest primary divided by wing chord. All measurements were taken to the nearest 0.01 mm apart from wing chord and tail length which were measured to the nearest mm.

Details on model selection and averaging (analysis 2a)

Traditionally, null hypotheses are tested individually, with ‘best’ models chosen based on information criteria such as Akaike’s information criterion (AIC). Whilst this method provides a means of selecting the final model it does not take into consideration model uncertainty (Whittingham et al. 2006). Recently, there has been increasing use of information theoretic (‘IT’) techniques as they allow the simultaneous testing of multiple hypotheses and generation of robust estimates for model parameters (Burnham and Anderson 2002). We follow techniques described in Grueber et al. (2011) to generate ‘top model sets’, which we then average to produce parameter estimates.

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Supplementary data

Diet and trait (as Principal Components) variation across bird species identified in communities. Trait data are only calculated for frugivores and insectivores; other guilds are highlighted in gray. Trophic trait = second principal component (PC2) score for a composite of beak traits; locomotory trait = PC2 for a composite of traits associated with locomotion, including tarsus, wing and tail measurements; overall size = PC1 from a second PC analysis combining the PC1 scores for both trophic and locomotory traits (PC1 in each case is strongly associated with body size).

Taxonomy follows Remsen et al. (2014)

Species	Guild ¹	Trophic trait	Locomotory trait	Overall size
<i>Accipiter superciliosus</i>	VE			
<i>Amazilia versicolor</i>	NE			
<i>Amazona amazonica</i>	FR	0.5	0.2	-0.5
<i>Amazona farinosa</i>	SE			
<i>Amazona kawalli</i>	FR	0.6	0.4	-0.8
<i>Amazona ochrocephala</i>	FR	0.6	0.2	-0.7
<i>Anabacerthia ruficaudatum</i>	IN	0	-0.2	0.5
<i>Anabazenops dorsalis</i>	IN	0.1	0.2	-0.2
<i>Ancistrops strigilatus</i>	IN	0	0.2	-0.1
<i>Anthracothorax nigricollis</i>	NE			
<i>Ara ararauna</i>	FR	0.8	2.1	-3
<i>Ara chloropterus</i>	FR	1.1	2	-3.6
<i>Ara macao</i>	FR	1.1	2.4	-3.5
<i>Ara severus</i>	FR	0.6	1	-1.2
<i>Arremon taciturnus</i>	IN	-0.5	0.5	0.2
<i>Attila cinnamomeus</i>	IN	-0.2	0.1	-0.2
<i>Attila spadiceus</i>	IN	-0.1	0.4	0
<i>Automolus ochrolaemus</i>	IN	0.3	0.1	0
<i>Automolus paraensis</i>	IN	0.2	0.1	0.1
<i>Automolus rufipileatus</i>	IN	0.2	0.1	-0.1
<i>Baryphthengus martii</i>	OM			
<i>Berlepschia rikeri</i>	IN	0.7	-0.1	-0.1

<i>Brotogeris chrysoptera</i>	NE			
<i>Bucco capensis</i>	OM			
<i>Buteo brachyurus</i>	VE			
<i>Buteo nitidus</i>	VE			
<i>Cacicus cela</i>	OM			
<i>Cacicus haemorrhous</i>	OM			
<i>Calliphlox amethystina</i>	NE			
<i>Campephilus melanoleucos</i>	IN	0.5	0.5	-3.2
<i>Campephilus rubricollis</i>	IN	0.8	0.8	-2.9
<i>Camptostoma obsoletum</i>	IN	0	-0.3	1.7
<i>Campylopterus largipennis</i>	NE			
<i>Campylorhamphus procurvoides</i>	IN			
<i>Campylorhynchus turdinus</i>	IN	0.6	0.4	0
<i>Cantorchilus leucotis</i>	IN	0.5	0.6	0.9
<i>Capito dayi</i>	FR	0	-0.2	0.6
<i>Capsiempis flaveola</i>	IN	-0.2	-0.2	1.4
<i>Caracara plancus</i>	SC			
<i>Cathartes melambrotus</i>	SC			
<i>Celeus elegans</i>	IN	0.2	-0.2	-1.6
<i>Celeus grammicus</i>	IN	0	-0.2	-0.4
<i>Celeus torquatus</i>	IN	0.8	-0.2	-1.4
<i>Ceratopipra rubrocapilla</i>	FR	-0.1	0	1.1
<i>Cercomacra cinerascens</i>	IN	0	0	0.6
<i>Cercomacra manu</i>	IN	-0.1	0.2	0.4
<i>Cercomacra nigrescens</i>	IN	0	0.5	0.5
<i>Certhiasomus stictolaemus</i>	IN	0.3	-0.3	0.4
<i>Chamaeza nobilis</i>	IN	-0.3	2.2	-0.9
<i>Chelidoptera tenebrosa</i>	IN	0.1	-0.7	0.4
<i>Chiroxiphia pareola</i>	FR	0	-0.2	1
<i>Chlorophanes spiza</i>	FR	-0.2	-0.1	1
<i>Cissopis leverianus</i>	FR	0.1	0	0.4
<i>Claravis pretiosa</i>	SE			

<i>Coccyzus minima</i>	IN	0	-0.7	-0.7
<i>Coereba flaveola</i>	NE			
<i>Colonia colonus</i>	IN	-0.5	-2.2	0.3
<i>Columbina talpacoti</i>	SE			
<i>Conopophaga aurita</i>	IN	-0.4	1.3	0.5
<i>Coragyps atratus</i>	SC			
<i>Corythopsis torquatus</i>	IN	-0.1	0.8	0.7
<i>Cotinga cayana</i>	FR	-0.1	0	0.7
<i>Craniola leuca gutturata</i>	IN	0.3	-0.3	1
<i>Crax fasciolata</i>	FR	0.3	-1.2	-2
<i>Crotophaga ani</i>	IN	-1.5	-0.1	-3.3
<i>Crypturellus cinereus</i>	OM			
<i>Crypturellus obsoletus</i>	OM			
<i>Crypturellus soui</i>	OM			
<i>Crypturellus strigulosus</i>	OM			
<i>Crypturellus tataupa</i>	IN	0.1	1.6	-0.4
<i>Crypturellus undulatus</i>	FR	-0.1	-0.9	0.2
<i>Crypturellus variegatus</i>	OM			
<i>Cyanerpes caeruleus</i>	NE			
<i>Cyanerpes nitidus</i>	OM			
<i>Cyanocompsa cyanooides</i>	SE			
<i>Cyclarhis gujanensis</i>	IN	-0.5	0.4	0.1
<i>Cymbilaimus lineatus</i>	IN	-0.7	0.5	-0.6
<i>Cypborhinus arada</i>	IN	0.3	0.8	0.8
<i>Dacnis cayana</i>	FR	-0.1	0	1.1
<i>Dacnis flaviventer</i>	FR	-0.2	0	1.1
<i>Dacnis lineata</i>	FR	-0.1	0	1.1
<i>Daptrius ater</i>	SC			
<i>Deconychura longicauda</i>	IN	0.4	-0.3	-0.3
<i>Dendrexetastes rufigula</i>	IN	0.1	-0.2	-1.2
<i>Dendrocincla fuliginosa</i>	IN	0.4	0	-0.7
<i>Dendrocincla merula</i>	IN	0.4	0.1	-0.2

<i>Dendrocolaptes certhia</i>	IN	0.7	-0.1	-1.8
<i>Dendrocolaptes picumnus</i>	IN	0.8	0	-1.9
<i>Dendroplex picus</i>	IN	1	-0.2	-0.3
<i>Deroptyus accipitrinus</i>	FR	0.6	0.5	-0.7
<i>Discosura langsdorffi</i>	NE			
<i>Dromococcyx pavoninus</i>	IN	0.4	-0.7	-1
<i>Dromococcyx phasianellus</i>	IN	0.3	-1.1	-1.7
<i>Dryophila devillei</i>	IN	0	-0.2	1.1
<i>Dryocopus lineatus</i>	IN	0.4	-0.2	-2.6
<i>Electron platyrhynchum</i>	IN	-0.4	-2.2	-3
<i>Empidonomus varius</i>	IN	-0.4	-0.7	0.4
<i>Epinecrophylla leucophthalma</i>	IN	0.1	0	1.3
<i>Epinecrophylla ornata</i>	IN	0	-0.1	1.3
<i>Euphonia chlorotica</i>	FR	-0.1	-0.1	1.1
<i>Euphonia chrysopasta</i>	FR	0	-0.1	1.1
<i>Euphonia minuta</i>	FR	-0.1	0	1.2
<i>Euphonia rufiventris</i>	FR	0	-0.1	1
<i>Falco rufigularis</i>	VE			
<i>Falco sparverius</i>	IN	-1.2	0.4	-2
<i>Florisuga mellivora</i>	NE			
<i>Formicarius analis</i>	IN	0	1.5	-0.1
<i>Formicarius colma</i>	IN	0	1.3	0.1
<i>Forpus modestus</i>	SE			
<i>Galbula cyanicollis</i>	IN	2.2	-1	-0.2
<i>Galbula dea</i>	IN	3.4	-2.1	-1.5
<i>Galbula ruficauda</i>	IN	3.4	-1.4	-0.7
<i>Geotrygon montana</i>	SE			
<i>Geotrygon violacea</i>	SE			
<i>Geranoospiza caerulescens</i>	VE			
<i>Glaucis birsutus</i>	NE			
<i>Glyphorhynchus spirurus</i>	IN	0	-0.4	0.8
<i>Grallaria varia</i>	IN	-0.3	3.6	-1.7

<i>Granatellus pelzelni</i>	IN	-0.2	-0.1	1.1
<i>Gymnoderus foetidus</i>	FR	0	0	0
<i>Habia rubica</i>	IN	-0.6	0	-0.2
<i>Harpagus bidentatus</i>	IN	-1.7	0.8	-3.4
<i>Helimaster longirostris</i>	NE			
<i>Heliothryx auritus</i>	NE			
<i>Hemithraupis flavicollis</i>	IN	-0.2	-0.4	1.1
<i>Hemitriccus griseipectus</i>	IN	-0.1	-0.3	1.4
<i>Hemitriccus minor</i>	IN	0	-0.1	1.5
<i>Herpetotheres cachinnans</i>	VE			
<i>Herpsilochmus rufimarginatus</i>	IN	0	0.1	1.2
<i>Hylexetastes perrotii</i>	IN	0.3	0.5	-2
<i>Hylocharis cyanus</i>	NE			
<i>Hylostistes subulatus</i>	IN	0.5	0	0.4
<i>Hylopezus berlepschi</i>	IN	0	2.6	-0.2
<i>Hylopezus whittakeri</i>	IN	0	2.2	-0.1
<i>Hylophilus hypoxanthus</i>	IN	0.1	-0.2	1.3
<i>Hylophilus ochraceiceps</i>	IN	0	0	1.3
<i>Hylophilus semicinerus</i>	IN	0	0.1	1.2
<i>Hylophylax naevius</i>	IN	-0.1	0.4	1
<i>Hypocnemis striata</i>	IN	0	0.3	1
<i>Hypocnemoides maculicauda</i>	IN	0.2	0.4	1.1
<i>Ibycter americanus</i>	IN	-1.2	0.1	-6.2
<i>Icterus cayanensis</i>	NE			
<i>Ictinia plumbea</i>	IN	-1.5	0	-3.4
<i>Jacamerops aureus</i>	IN	0.9	-1.7	-2.4
<i>Lamprospiza melanoleuca</i>	FR	0	-0.1	0.7
<i>Lanio versicolor</i>	IN	-0.2	-0.6	0.7
<i>Laniocera hypopyrra</i>	IN	-0.3	-0.3	-0.1
<i>Lathrotriccus eulerei</i>	IN	-0.3	-0.3	1.1
<i>Legatus leucophaius</i>	FR	-0.1	0.1	0.9
<i>Lepidocolaptes albolineatus</i>	IN	1	-0.6	0.2

<i>Lepidobrix nattereri</i>	FR	-0.1	0	1.2
<i>Leptodon cayanensis</i>	IN	-1.2	-0.7	-4.9
<i>Leptopogon amaurocephalus</i>	IN	0	-0.5	1.2
<i>Leptotila rufaxilla</i>	SE			
<i>Leptotila verreauxi</i>	SE			
<i>Leucopternis kuhli</i>	VE			
<i>Lipaugus vociferans</i>	FR	-0.1	0.1	0.5
<i>Lophotriccus galeatus</i>	IN	-0.1	-0.1	1.6
<i>Machaeropterus pyrocephalus</i>	FR	-0.1	-0.1	1.2
<i>Malacoptila rufa</i>	IN	0	-0.2	-0.4
<i>Manacus manacus</i>	FR	-0.1	-0.2	1.1
<i>Megarynchus pitangua</i>	IN	-0.8	-0.5	-1.7
<i>Melanerpes cruentatus</i>	OM			
<i>Micrastur mintoni</i>	VE			
<i>Micrastur mirandollei</i>	VE			
<i>Micrastur ruficollis</i>	VE			
<i>Micrastur semitorquatus</i>	VE			
<i>Microcerculus marginatus</i>	IN	0.4	0.9	1.1
<i>Microbopias quixensis</i>	IN	0.1	-0.1	1.1
<i>Mionectes oleagineus</i>	FR	-0.1	0	1.1
<i>Mitu mitu</i>	FR	0.6	-1.5	-3.1
<i>Molothrus oryzivorus</i>	OM			
<i>Momotus momota</i>	IN	0.3	-1.9	-3.2
<i>Monasa morphoeus</i>	IN	0	-0.9	-1.6
<i>Monasa nigrifrons</i>	IN	0.1	-0.8	-2
<i>Myiarchus ferox</i>	IN	-0.4	-0.2	-0.2
<i>Myiarchus tuberculifer</i>	IN	-0.3	-0.2	0.1
<i>Myiarchus tyrannulus</i>	IN	-0.4	-0.1	-0.3
<i>Myiopagis caniceps</i>	IN	-0.1	-0.1	1.3
<i>Myiopagis gaimardii</i>	IN	-0.2	0	1.2
<i>Myiornis ecaudatus</i>	IN	0	-0.2	2
<i>Myiozetetes cayanensis</i>	IN	-0.3	-0.3	0.5

<i>Myiozetetes luteiventris</i>	IN	-0.3	-0.6	0.9
<i>Myrmeciza atrothorax</i>	IN	0	0.8	0.8
<i>Myrmeciza hemimelaena</i>	IN	0	0.8	1
<i>Myrmoborus leucophrys</i>	IN	0	0.8	0.7
<i>Myrmoborus myotherinus</i>	IN	0	0.9	0.8
<i>Myrmothera campanisona</i>	IN	0	2.8	-0.3
<i>Myrmotherula axillaris</i>	IN	0.2	0	1.5
<i>Myrmotherula brachyura</i>	IN	0.2	0.3	1.6
<i>Myrmotherula bauxwelli</i>	IN	0.1	0.6	1.4
<i>Myrmotherula longipennis</i>	IN	-0.2	0	1.1
<i>Myrmotherula menetriesii</i>	IN	0.1	0	1.4
<i>Myrmotherula sclateri</i>	IN	0.2	0.2	1.6
<i>Nonnula ruficapilla</i>	IN	-0.1	-0.6	0.6
<i>Notharchus hyperrhynchus</i>	IN	-0.8	-0.4	-2.3
<i>Notharchus ordii</i>	IN	-0.5	-0.6	-1
<i>Notharchus tectus</i>	IN	-0.6	-0.6	-0.2
<i>Nystalus striolatus</i>	IN	-0.1	-0.3	-1
<i>Odontophorus gujanensis</i>	FR	0.1	-0.6	0.3
<i>Odontorchilus cinereus</i>	IN	0.3	-0.1	1.4
<i>Onychorhynchus coronatus</i>	IN	-0.2	-0.4	0.5
<i>Ornithion inermis</i>	IN	-0.1	-0.3	1.8
<i>Orthopsittaca manilata</i>	FR	0.6	0.8	-0.9
<i>Pachyramphus castaneus</i>	IN	-0.4	0	0.6
<i>Pachyramphus marginatus</i>	IN	-0.5	0.1	0.5
<i>Pachyramphus minor</i>	IN	-0.8	0	-0.3
<i>Pachyramphus polychopterus</i>	IN	-0.5	-0.1	0.4
<i>Parkerthraustes humeralis</i>	IN	-1.1	0.1	-0.6
<i>Patagioenas cayennensis</i>	FR	-0.2	0.1	0.4
<i>Patagioenas picazuro</i>	FR	-0.2	0.1	0.1
<i>Patagioenas plumbea</i>	FR	-0.2	0.3	0.3
<i>Patagioenas speciosa</i>	FR	-0.2	0.1	0.4
<i>Patagioenas subvinacea</i>	FR	-0.2	0.2	0.5

<i>Penelope jacquacu</i>	FR	0	-0.6	-1.3
<i>Phaethornis bourcieri</i>	NE			
<i>Phaethornis hispidus</i>	NE			
<i>Phaethornis ruber</i>	NE			
<i>Phaethornis superciliosus</i>	NE			
<i>Phengopedius genibarbis</i>	IN	0.2	0.5	0.7
<i>Philydor erythrocerum</i>	IN	0	-0.1	0.3
<i>Philydor erythropterum</i>	IN	0.1	0.1	0.1
<i>Philydor pyrrhodes</i>	IN	0.1	0.1	0.4
<i>Pblegopsis nigromaculata</i>	IN	0	1.2	0.1
<i>Piaya cayana</i>	IN	-0.5	-1.6	-3.1
<i>Piaya melanogaster</i>	IN	-0.3	-0.6	-2.6
<i>Piculus chrysocloros</i>	IN	0.2	-0.1	-0.6
<i>Piculus flavigula</i>	IN	0	-0.3	-0.1
<i>Picumnus aurifrons</i>	IN	0	-0.2	1.5
<i>Pionites leucogaster</i>	FR	0.3	0.2	0.1
<i>Pionus menstruus</i>	FR	0.4	0.2	-0.1
<i>Pipile cujubi</i>	FR	0	-0.2	-1.2
<i>Pipra fasciicauda</i>	FR	-0.1	-0.1	1.1
<i>Piprites chloris</i>	IN	-0.4	-0.1	1
<i>Pitangus lictor</i>	IN	0.4	-0.3	0.2
<i>Pitangus sulphuratus</i>	OM			
<i>Platyrinchus platyrhynchos</i>	IN			
<i>Platyrinchus saturatus</i>	IN	-0.9	0.2	1
<i>Poecilotriccus latirostris</i>	IN	-0.1	0.3	1.3
<i>Psarocolius bifasciatus</i>	OM			
<i>Psarocolius decumanus</i>	OM			
<i>Psittacara leucophthalmus</i>	FR	0.5	0.6	-0.3
<i>Psophia viridis</i>	FR	-0.1	-3.3	-1.2
<i>Pteroglossus beauharnaesii</i>	FR	-0.9	0	-1.3
<i>Pteroglossus bitorquatus</i>	FR	-0.6	-0.1	-0.8
<i>Pteroglossus castanotis</i>	FR	-1.2	-0.1	-1.5

<i>Pteroglossus inscriptus</i>	FR	-0.5	-0.1	-0.6
<i>Pygiptila stellaris</i>	IN	-0.2	0.3	0.2
<i>Pyriglena leuconota</i>	IN	0	1.2	0.1
<i>Pyrrhura amazonum</i>	FR	0.3	0.5	0.3
<i>Pyrrhura perlata</i>	FR	0.2	0.5	0.3
<i>Ramphastos tucanus</i>	FR	-1.7	-0.5	-3
<i>Ramphastos vitellinus</i>	FR	-1.1	-0.4	-2.1
<i>Ramphocaenus melanurus</i>	IN	1.1	0.4	1.2
<i>Ramphocelus carbo</i>	FR	0	-0.1	0.7
<i>Ramphotrigon fuscicauda</i>	IN	-0.3	-0.4	0.4
<i>Ramphotrigon megacephalum</i>	IN	-0.4	-0.3	0.9
<i>Ramphotrigon ruficauda</i>	IN	-0.5	-0.5	0.5
<i>Rhagmatorhina gymnops</i>	IN	0.1	1	0.4
<i>Rhytipterna simplex</i>	IN	-0.3	-0.3	-0.2
<i>Rupornis magnirostris</i>	VE			
<i>Sakesphorus luctuosus</i>	IN	0	0.3	0
<i>Saltator coerulescens</i>	IN	-1	0.1	-1.1
<i>Saltator grossus</i>	OM			
<i>Saltator maximus</i>	OM			
<i>Schijffornis turdina</i>	IN	-0.3	0.2	0.3
<i>Sclerurus albigularis</i>	IN	0.6	0.2	0.4
<i>Sclerurus caudacutus</i>	IN	0.6	0.2	0.3
<i>Sclerurus mexicanus</i>	IN	1	0.3	0.5
<i>Sclerurus rufigularis</i>	IN	0.5	0.2	0.7
<i>Selenidera gouldii</i>	FR	-0.3	-0.2	-0.6
<i>Sittasomus griseicapillus</i>	IN	0.2	-0.5	0.7
<i>Spizaetus ornatus</i>	VE			
<i>Spizaetus tyrannus</i>	VE			
<i>Sublegatus modestus</i>	IN	-0.3	-0.2	1.2
<i>Synallaxis cherriei</i>	IN	0.1	-0.1	1.1
<i>Synallaxis rutilans</i>	IN	0.1	-0.1	0.9
<i>Syndactyla ucayalae</i>	IN	0.2	0.4	-0.5

<i>Tachyphonus cristatus</i>	IN	-0.3	-0.1	0.3
<i>Tachyphonus luctuosus</i>	IN	-0.3	-0.3	0.9
<i>Tangara chilensis</i>	FR	-0.1	0	1
<i>Tangara cyanicollis</i>	FR	-0.1	-0.1	1
<i>Tangara gyrola</i>	FR	-0.1	0	1
<i>Tangara nigrocincta</i>	FR	0	0	1
<i>Tangara schrankii</i>	FR	-0.1	-0.1	1
<i>Tangara velia</i>	FR	-0.1	0	1
<i>Taraba major</i>	IN	-0.3	1.4	-1
<i>Terenotriccus erythrurus</i>	IN	-0.2	-0.1	1.7
<i>Thalurania furcata</i>	NE			
<i>Thamnomanes caesius</i>	IN	-0.2	-0.2	0.6
<i>Thamnomanes saturninus</i>	IN	-0.2	0.1	0.6
<i>Thamnophilus aethiops</i>	IN	-0.2	0.2	0.4
<i>Thamnophilus amazonicus</i>	IN	-0.2	0.1	0.5
<i>Thamnophilus palliatus</i>	IN	-0.3	0.3	0.2
<i>Thamnophilus schistaceus</i>	IN	-0.1	0.2	0.6
<i>Thamnophilus stictocephalus</i>	IN	-0.2	0.1	0.4
<i>Thraupis episcopus</i>	FR	0	-0.1	0.7
<i>Thraupis palmarum</i>	FR	-0.1	-0.1	0.8
<i>Threnetes leucurus</i>	NE			
<i>Tinamus major</i>	FR	-0.1	-1.3	-0.2
<i>Tinamus tao</i>	OM			
<i>Tityra inquisitor</i>	IN	-1	0	-1.1
<i>Tityra semifasciata</i>	IN	-0.9	0.5	-1.3
<i>Todirostrum chrysocrotaphum</i>	IN	-0.2	0.1	1.3
<i>Tolmomyias assimilis</i>	IN	-0.5	0	0.8
<i>Tolmomyias flaviventris</i>	IN	-0.5	-0.1	1.1
<i>Tolmomyias poliocephalus</i>	IN	-0.4	-0.2	1.2
<i>Touit huetii</i>	FR	0.1	0.2	0.6
<i>Troglodytes aedon</i>	IN	0.4	0.2	1.4
<i>Trogon collaris</i>	IN	-0.9	-1.7	-0.4

<i>Trogon curucui</i>	OM			
<i>Trogon melanurus</i>	OM			
<i>Trogon ramonianus</i>	OM			
<i>Trogon rufus</i>	IN	-1.3	-2	-0.8
<i>Trogon viridis</i>	OM			
<i>Turdus albicollis</i>	OM			
<i>Turdus baucæwelli</i>	OM			
<i>Turdus lawrencii</i>	IN	-0.1	0.2	-0.4
<i>Tyrannetes stolzmanni</i>	IN	-0.2	-0.3	1.8
<i>Tyrannopsis sulphurea</i>	IN	-0.5	-0.1	-0.4
<i>Tyrannulus elatus</i>	FR	-0.1	0	1.2
<i>Tyrannus albogularis</i>	IN	-0.3	-0.7	-0.5
<i>Tyrannus melancholicus</i>	IN	-0.5	-0.8	-0.6
<i>Tyrannus savana</i>	IN	-0.3	-2.2	-0.6
<i>Veniliornis affinis</i>	IN	0.5	-0.2	-0.6
<i>Vireo olivaceus</i>	IN	0	-0.1	1
<i>Vireolanius leucotis</i>	IN	-0.2	0.4	0.5
<i>Volatinia jacarina</i>	SE			
<i>Willisornis poecilinotus/vidua</i>	IN	0	0.7	0.8
<i>Xenops minutus</i>	IN	0.2	-0.4	1.2
<i>Xenops tenniurostris</i>	IN	0.2	-0.4	1.5
<i>Xiphocolaptes promeropirhynchus</i>	IN	1.6	1.3	-1.9
<i>Xiphorhynchus spixi/elegans</i>	IN	0.9	0	-0.4
<i>Xiphorhynchus guttatus</i>	IN	1.3	-0.1	-1.2
<i>Xiphorhynchus obsoletus</i>	IN	0.7	-0.1	-0.1
<i>Zimmerius gracilipes</i>	FR	-0.1	-0.1	1.2

¹Guild codes: IN = Insectivore, FR = Frugivore, SE = Granivore, OM = Omnivore, NE = Nectarivore, VE = Vertebrate, SC = Scavenger.

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Supplementary results

TABLE C1. Principal Components Analysis for Insectivores. Eigenvalues and the proportion of variance explained are presented. The first principal components for locomotory and trophic traits were used in a secondary Principal Components Analysis to create an axis representing overall size. The second Principal Components for locomotory and trophic traits represented the shape of these syndromes and was used in further analysis.

PCA	PC	Proportion variance	PCA loadings		
Locomotory	1	0.734	Tarsus length	Tail length	Wing chord
	2	0.198			
Trophic	1	0.785	Beak length	Beak width	Beak depth
	2	0.162			
Overall size	1	0.858	Trophic PC1	Locomotory PC1	

TABLE C2. Principal Components Analysis for Frugivores. Eigenvalues and the proportion of variance explained are presented. The first principal components for locomotory and trophic traits were used in a secondary Principal Components Analysis to create an axis representing overall size. The second Principal Components for locomotory and trophic traits represented the shape of these syndromes and was used in further analysis.

PCA	PC	Proportion variance	PCA loadings		
Locomotory	1	0.797	Tarsus length	Tail length	Wing chord
	2	0.179	0.514	0.583	0.628
Trophic	1	0.897	Beak length	Beak width	Beak depth
	2	0.070	−0.821	0.546	0.165
Overall size	1	0.804	Trophic PC1	Locomotory PC1	
			−0.707	−0.707	

TABLE C3. Phylogenetic signal of trait axes using Blomberg's K . K values close to 1 indicate a Brownian motion model of trait evolution whilst a value of 0 indicates a random model of trait evolution.

Guild	Trait	K	P
Insectivores	Locomotory	0.59	<0.001
	Trophic	0.54	<0.001
	Dispersal	0.62	<0.001
	Overall size	1.38	<0.001
Frugivores	Locomotory	1.11	<0.001
	Trophic	0.53	<0.001
	Dispersal	0.71	<0.001
	Overall size	0.47	<0.001

TABLE C4. Average fixed effect estimates derived from the top linear models, <2AICc from the best fitting model. Standardised estimates for community assembly metrics were used as the response variable and environmental variables as the fixed effects. FS – Fragment size (ha), GF – Percentage of fragment covered in good quality forest and D – Distance of fragment away from nearest forest >1000 ha.

Guild	Trait syndrome	Metric	Variable	Estimate (SE)	Z value	P value	
Insectivores	Phylogeny	MNTD	FS	−0.580 (0.143)	4.047	<0.001	
			GF	0.496 (0.381)	1.247	0.212	
	Locomotory	SDNDR	FS	0.516 (0.367)	1.345	0.179	
			D	−0.490 (0.380)	1.239	0.215	
		GF	FS	−0.357 (0.315)	1.083	0.279	
			D	−0.354 (0.315)	1.074	0.283	
		Variance	D	0.218 (0.320)	0.653	0.514	
			GF	0.564 (0.329)	1.637	0.102	
	Dispersal	SDNDR	FS	0.377 (0.339)	1.063	0.288	
			D	−0.952 (0.539)	1.700	0.089	
		Variance	FS	−0.859 (0.455)	1.827	0.067	
			D*FS	0.683 (0.621)	1.048	0.295	
		Foraging	SDNDR	D	−0.920 (0.555)	1.615	0.106
				FS	−0.678 (0.402)	1.608	0.108
	D*FS		0.68 (0.61)	1.059	0.290		
			D	−0.297 (0.432)	1.099	0.272	
	Overall size		Variance	FS	−0.852 (0.393)	2.087	0.037
				FS	0.884 (0.335)	2.533	0.011
		SDNDR	GF	0.384 (0.351)	1.042	0.297	
			Variance	FS	−0.297 (0.131)	2.271	0.031
		Frugivores	Phylogeny	MNTD	D	0.673 (0.496)	1.296
FS	1.111 (0.517)				2.083	0.037	
GF	−0.487 (0.398)				0.417	0.243	
Locomotory	SDNDR		FS*GF	−2.013 (0.802)	2.392	0.017	
			D	−0.266 (0.239)	1.063	0.288	
			FS	0.255 (0.240)	1.017	0.309	
			Variance	GF	0.060 (0.250)	0.241	0.811

Dispersal	SDNDr	D	−1.752 (0.844)	1.998	0.046	
		FS	0.577 (0.467)	1.178	0.239	
		GF	−0.900 (0.505)	1.692	0.091	
		D*FS	2.364 (0.755)	2.987	0.003	
		D*GF	2.252 (1.300)	1.644	0.100	
	Variance	D	0.816 (0.402)	1.950	0.051	
		GF	0.469 (0.431)	1.040	0.298	
	Foraging	SDNDr	D	0.879 (0.874)	0.963	0.336
			FS	1.212 (0.509)	2.271	0.023
			GF	−0.778 (0.527)	1.402	0.161
D*FS			−2.455 (0.878)	2.678	0.007	
FS*GF			−2.729 (1.193)	2.172	0.030	
Variance		FS	−1.032 (0.318)	3.113	0.002	
		GF	0.273 (0.345)	0.756	0.450	
Overall		SDNDr	D	−0.531 (0.389)	1.316	0.188
	GF		−0.494 (0.397)	1.201	0.230	
	Variance	D	−0.176 (0.340)	0.498	0.618	
		FS	0.048 (0.333)	0.346	0.890	
		GF	0.505 (0.296)	1.633	0.103	

FIGURE C1. Structuring of insectivore communities ($n = 30$) in fragments with different habitat characteristics. Regression lines are shown for significant relationships (derived from top model sets), and represent the fit of a model with fragment size as the sole predictor of the structuring metrics. Final model outputs are included in Table C4.

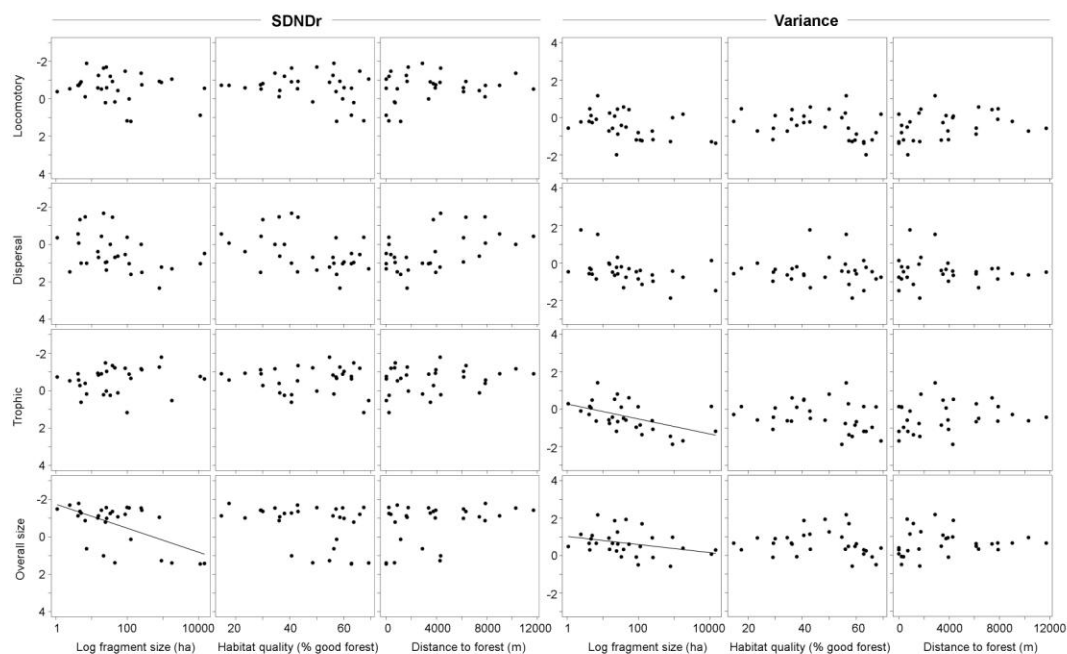


FIGURE C2. Structuring of frugivore communities ($n = 30$) in fragments with different habitat characteristics. Regression lines are shown for significant relationships (derived from top model sets), and represent the fit of a model with fragment size as the sole predictor of the structuring metrics. Final model outputs are included in Table C4.

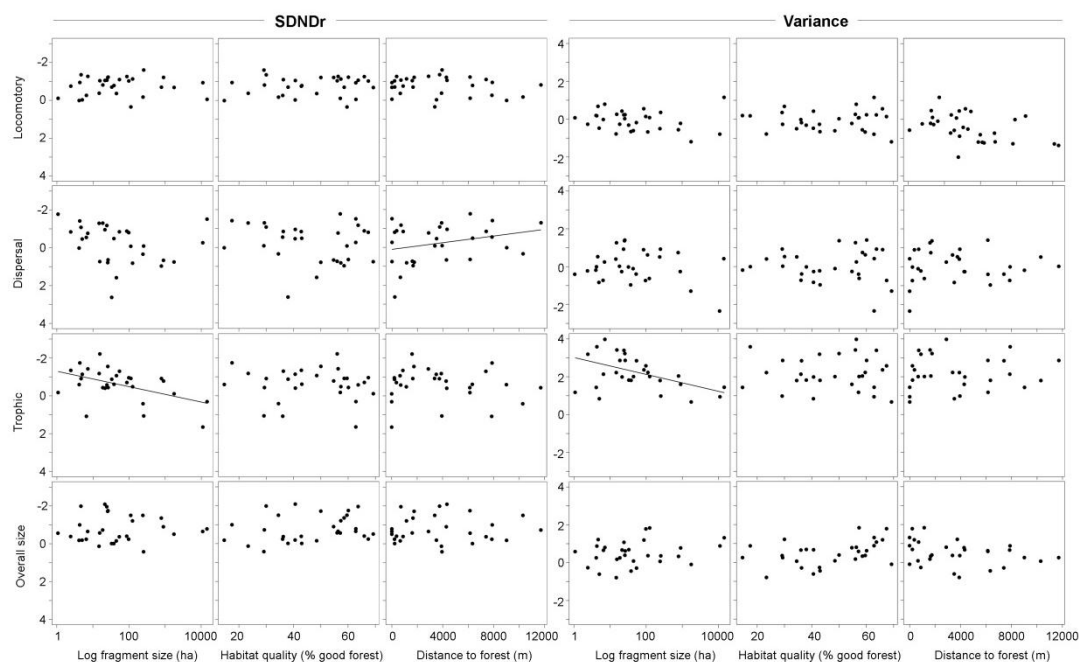


TABLE C5. Wilcoxon analyses (two-tailed) of insectivore communities ($n = 30$) testing whether observed MNTD, SDNDR and variance are significantly different the null expectation. Asterisk (*) indicates that the observed values were significantly different from the null model and provide evidence for competitive interactions. The standardised effect size (SES) is provided along with the standard error (SE). Analyses were run separately for the 15 largest and smallest fragments.

Metric	Trait syndrome	Small fragment size				Large fragment size			
		SES (mean + SE)	No. plots < expectation	V	Wilcoxon P	SES (mean + SE)	No. plots < expectation	V	Wilcoxon P
MNTD	Phylogeny	0.662 (0.938)	3	20	0.022*	-0.461 (0.663)	11	101	0.018
	Locomotory	-0.562 (0.676)	14	119	<0.001	-0.122 (1.070)	10	92	0.037
SDNDR	Feeding	-0.240 (0.702)	11	108	0.002	-0.173 (0.950)	11	102	0.008
	Dispersal	0.137 (1.007)	7	58	0.555	0.366 (0.834)	4	22	0.987
	Overall	-0.785 (0.938)	13	115	<0.001	-0.241 (0.895)	9	92	0.037
Variance	Locomotory	0.112 (0.915)	9	77	0.359	-0.163 (0.794)	12	103	0.012
	Feeding	0.270 (0.888)	8	69	0.639	-0.337 (0.814)	11	104	0.010
	Dispersal	0.228 (0.924)	12	87	0.135	-0.331 (0.644)	14	119	<0.001
	Overall	0.586 (0.610)	0	0	<0.001*	-0.041 (0.794)	5	26	0.055

TABLE C6. Wilcoxon analyses (two-tailed) of frugivore communities ($n = 30$) testing whether observed MNTD, SDNDR and variance are significantly different the null expectation. Asterisk (*) indicates that the observed values were significantly different from the null model and provide evidence for competitive interactions. The standardised effect size (SES) is provided along with the standard error (SE). Analyses were run separately for the 15 largest and smallest fragments.

Metric	Trait syndrome	Small fragment size				Large fragment size			
		SES (mean + SE)	No. plots < expectation	V	Wilcoxon P	SES (mean + SE)	No. plots < expectation	V	Wilcoxon P
MNTD	Phylogeny	-0.464 (0.972)	9	99	0.026	0.363 (0.829)	4	31	0.107
	Locomotory	-0.381 (0.585)	14	119	<0.001	-0.114 (0.696)	14	117	<0.001
SDNDR	Feeding	-0.556 (0.889)	14	110	0.001	0.302 (1.515)	11	93	0.032
	Dispersal	-0.482 (0.848)	11	109	0.002	0.177 (1.104)	8	58	0.555
	Overall	-0.608 (0.899)	14	119	<0.001	-0.063 (0.707)	14	111	0.001
	Locomotory	0.137 (0.547)	5	45	0.421	-0.226 (0.735)	10	90	0.095
Variance	Feeding	1.222 (0.886)	0	0	<0.001*	0.293 (0.794)	0	0	<0.001*
	Dispersal	0.126 (0.828)	7	45	0.421	-0.344 (1.138)	9	70	0.600
	Overall	-0.073 (0.673)	3	18	0.015*	0.015 (0.876)	3	17	0.012*

TABLE C7. Wilcoxon analyses (two-tailed) of insectivore communities ($n = 30$) testing whether observed MNTD, SDNDR and variance are significantly different the null expectation. Asterisk (*) indicates that the observed values were significantly different from the null model and provide evidence for competitive interactions. The standardised effect size (SES) is provided along with the standard error (SE). Analyses were run separately for the 15 fragments closest to forests >1000 ha and those furthest away.

Metric	Trait syndrome	Close to forest >1000 ha				Far away from forest >1000 ha			
		SES (mean + SE)	No. plots < expectation	V	Wilcoxon P	SES (mean + SE)	No. plots < expectation	V	Wilcoxon P
SDNDR	Locomotory	-0.213 (0.667)	14	118	<0.001	-0.282 (0.646)	14	119	<0.001
	Feeding	-0.158 (1.582)	13	111	0.001	-0.095 (0.987)	12	98	0.015
	Dispersal	0.175 (1.187)	8	58	0.555	-0.480 (0.730)	11	106	0.003
	Overall	-0.329 (0.748)	15	120	<0.001	-0.342 (0.952)	13	114	<0.001
Variance	Locomotory	-0.033 (0.776)	7	68	0.679	-0.056 (0.556)	8	66	0.762
	Feeding	0.821 (1.012)	0	0	<0.001*	0.694 (0.881)	0	0	<0.001*
	Dispersal	-0.071 (1.241)	7	46	0.454	-0.147 (0.746)	9	69	0.639
	Overall	0.190 (0.804)	2	7	<0.001*	-0.249 (0.690)	4	27	0.064

TABLE C8. Wilcoxon analyses (two-tailed) of frugivore communities ($n = 30$) testing whether observed MNTD, SDNDR and variance are significantly different the null expectation. Asterisk (*) indicates that the observed values were significantly different from the null model and provide evidence for competitive interactions. The standardised effect size (SES) is provided along with the standard error (SE). Analyses were run separately for the 15 fragments closest to forests >1000 ha and those furthest away.

Metric	Trait syndrome	Close to forest >1000 ha				Far away from forest >1000 ha			
		SES (mean + SE)	No. plots < expectation	V	Wilcoxon P	SES (mean + SE)	No. plots < expectation	V	Wilcoxon P
SDNDR	Locomotory	0.023 (1.246)	9	92	0.037	-0.464 (0.416)	15	120	<0.001
	Feeding	-0.020 (0.886)	10	96	0.021	-0.392 (0.734)	12	112	0.001
	Dispersal	0.661 (0.720)	2	4	1	-0.158 (0.928)	9	80	0.138
	Overall	-0.262 (0.977)	9	88	0.060	-0.765 (0.865)	13	117	<0.001
Variance	Locomotory	-0.170 (1.011)	11	96	0.041	0.119 (0.663)	10	84	0.188
	Feeding	-0.077 (1.031)	10	86	0.151	0.009 (0.763)	9	87	0.135
	Dispersal	0.175 (1.084)	11	82	0.229	-0.278 (0.390)	15	120	<0.001
	Overall	0.160 (0.974)	4	22	0.030*	0.385 (0.488)	1	1	<0.001*

TABLE C9. Wilcoxon analyses (two-tailed) of insectivore communities ($n = 30$) testing whether observed MNTD, SDNDR and variance are significantly different the null expectation. Asterisk (*) indicates that the observed values were significantly different from the null model and provide evidence for competitive interactions. The standardised effect size (SES) is provided along with the standard error (SE). Analyses were run separately for the 15 fragments with the highest proportion of good quality forest and those with the lowest proportion.

Metric	Trait syndrome	High proportion of good-quality forest				Low proportion of good quality forest			
		SES (mean + SE)	No. plots < expectation	V	Wilcoxon P	SES (mean + SE)	No. plots < expectation	V	Wilcoxon P
SDNDR	Locomotory	-0.238 (0.613)	14	119	<0.001	-0.258 (0.700)	14	117	<0.001
	Feeding	-0.222 (1.017)	12	102	0.008	-0.032 (1.557)	13	108	0.002
	Dispersal	-0.190 (1.116)	11	91	0.042	-0.115 (0.960)	8	69	0.319
	Overall	-0.127 (0.929)	13	111	0.001	-0.545 (0.714)	15	120	<0.001
Variance	Locomotory	-0.174 (0.577)	10	81	0.252	0.086 (0.736)	5	51	0.639
	Feeding	0.783 (0.859)	0	0	<0.001*	0.732 (1.034)	0	0	<0.001*
	Dispersal	-0.230 (0.598)	10	77	0.359	0.012 (1.307)	6	42	0.330
	Overall	-0.350 (0.681)	5	38	0.229	0.292 (0.735)	1	1	<0.001*

TABLE C10. Wilcoxon analyses (two-tailed) of frugivore communities ($n = 30$) testing whether observed MNTD, SDNDR and variance are significantly different the null expectation. Asterisk (*) indicates that the observed values were significantly different from the null model and provide evidence for competitive interactions. The standardised effect size (SES) is provided along with the standard error (SE). Analyses were run separately for the 15 fragments with the highest proportion of good quality forest and those with the lowest proportion.

Metric	Trait syndrome	High proportion of good-quality forest				Low proportion of good quality forest			
		SES (mean + SE)	No. plots < expectation	V	Wilcoxon P	SES (mean + SE)	No. plots < expectation	V	Wilcoxon P
SDNDR	Locomotory	-0.431 (0.540)	14	119	<0.001	-0.010 (1.211)	10	94	0.028
	Feeding	-0.240 (0.748)	11	108	0.002	-0.172 (0.914)	11	102	0.008
	Dispersal	-0.112 (0.987)	9	72	0.262	0.615 (0.693)	2	13	0.998
	Overall	-0.755 (0.959)	13	114	<0.001	-0.271 (0.891)	9	91	0.042
Variance	Locomotory	0.137 (0.576)	10	81	0.252	-0.188 (1.058)	11	96	0.041
	Feeding	0.126 (0.625)	9	84	0.188	-0.193 (1.097)	10	84	0.188
	Dispersal	-0.092 (0.820)	14	105	0.008	-0.011 (0.872)	12	96	0.041
	Overall	0.450 (0.650)	2	17	<0.001*	0.095 (0.851)	3	17	0.012*

TABLE C11. Variance Inflation Factor (VIF) for each of the continuous predictor variables used in the GLMs. A VIF threshold of >4 (or >10) is used as a rule of thumb for excessive multicollinearity in regression analyses (O'Brien 2007).

Variable	Variance Inflation Factor (VIF)
Fragment size	2.033
Habitat quality (% of good forest)	1.515
Distance to forest	2.296

TABLE C12. Mantel test for GLMs assessing spatial correlation in functional trait syndromes across our study site.

Guild	Metric	Trait syndrome	Monte-Carlo test observed value	<i>P</i> -value
Insectivore	SDNDr	Locomotory	0.072	0.245
		Trophic	−0.083	0.796
		Dispersal	−0.112	0.892
		Overall	−0.068	0.696
	Variance	Locomotory	0.074	0.222
		Trophic	−0.076	0.771
		Dispersal	−0.124	0.845
		Overall	0.028	0.367
Frugivore	SDNDr	Locomotory	−0.029	0.635
		Trophic	0.013	0.402
		Dispersal	0.192	0.077
		Overall	0.011	0.386
	Variance	Locomotory	−0.026	0.577
		Trophic	−0.011	0.524
		Dispersal	0.103	0.162
		Overall	0.013	0.542

Appendix D: Chapter 5 Supplementary Methods and Results

Contents

Expanded materials, methods and results: Fig. D1, Tables D1-D3.

Expanded materials, methods and results

Measurement protocol

Tarsus length was measured as the distance between the final scale of the acrotarsium to the middle of the ankle joint. Three beak measurements were taken at or from the anterior edge of the nostrils: (1) width, (2) length to the tip of the beak and (3) depth (as vertical height). Tail length was taken from the point at which the two central rectrices meet the skin to the tip of the longest rectrix. Wing chord was measured as the distance between the carpal joint and wing tip of the unflattened wing. We also measured the distance between the first secondary to the longest primary. We used this to calculate the hand-wing index of Claramunt et al. (2012). This is the wing chord - distance between the first secondary to longest primary divided by wing chord. All measurements were taken to the nearest 0.01 mm apart from wing chord and tail length which were measured to the nearest mm.

Estimating species probability of occurrence

Model choice

To determine which modelling framework to use for the generation of the final model we compared average AUC values of a GLMM vs a GAMM approach. These two approaches were appealing as they enabled the multiple random effects to be included in the model. The overall AUC value for the GLMM approach was 0.692 compared with 0.741 for the GAMM approach when using the Santarem dataset for training and so the latter was used for the final prediction of Amazonian communities.

Scaling-up transect probabilities

The GAMM was used to predict the probability of occurrence for each species at a transect level. Given that each transect was composed of three fixed 75 metre width point counts, the total area surveyed by each transect was:

$$\text{Area surveyed along each transect} = 3 * (\pi * 75^2) = 53,014.38 \text{ metres}^2$$

Therefore to find the number of transects needed to survey a 1 km² (1,000,000 m²) cell:

$$\text{Number of transects} = 1,000,000 \div 53,014.38 = 18.86 \text{ transects}$$

Given this information, we used the following equation to find the probability of occurrence of each species in each 1 km² cell.

$$\text{1 km}^2 \text{ resolution:} \quad P_{ci} = 1 - ((1 - P_{ii})^{18.86})$$

Where P_{ii} is the probability of occurrence for each i th species at a transect level, P_{ci} is the probability of occurrence for each species at a 1km² resolution. Then, to find the probability of occurrence at a 2500 km² resolution we averaged the probability of occurrence of each of the smaller 1 km² cells that were a part of the coarser grid cell.

Determining appropriate correlation structure for linear regression

TABLE D1. Evaluative metrics for the gape width GAMM with different correlation structures to account for spatial autocorrelation. Lower AIC and BIC values and higher logLik values indicate a better model fit.

Correlation structure	AIC	BIC	logLik
None	11080.38	11103.68	-5536.19
corRatio	4780.59	4815.54	-2384.30
corGaus	5010.71	5045.65	-2499.35
corLin	5547.69	5582.63	-2767.85
corSpher	4912.63	4947.58	-2450.32
corExp	4912.55	4947.49	-2450.27

TABLE D2. Evaluative metrics for the F_{RIC} GAMM with different correlation structures to account for spatial autocorrelation. Lower AIC and BIC values and higher logLik values indicate a better model fit.

Correlation structure	AIC	BIC	logLik
None	17105.96	17128.97	-8548.98
corRatio	11076.04	11110.57	-5532.02
corGaus	11295.61	11330.13	-5641.80
corLin	11065.46	11109.99	-5526.73
corSpher	11074.32	11110.43	-5531.71
corExp	11073.56	11110.31	-5530.63

Alternative forest cover scenarios

To provide an insight into the impact of alternative forest cover scenarios in 2050, Fig.D1 shows F_{RIC} and mean gape size of the five largest species where forest remains in 90% and 30% of the Rosa et al's (2013) forest models.

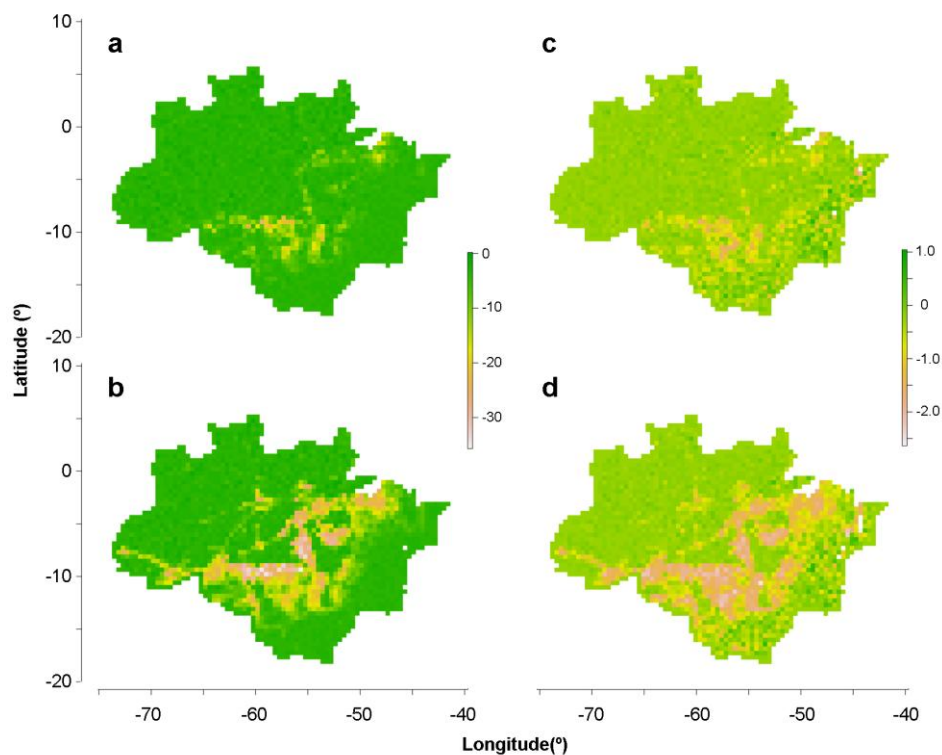


FIGURE D1. Mapping changes in an ecosystem process over time across the Brazilian Amazon in terms of predicted functional richness (F_{RIC}) of seed dispersers under (a) 90% and (b) 30% likelihood scenarios, and mean gape size (in mm) of the largest 5 species under (c) 90% and (d) 30% likelihood scenarios. Maps show the predicted future scenario in 2050 at a grain size of 2500km².

TABLE D3. Classification of frugivores into those that are predominantly seed predators (i.e. those known to destroy seeds) and seed dispersers. Classifications are provided for each genera represented in hypothetical species lists across South America.

Genus	Disperser or Predator?	Reference
<i>Aburria</i>	Disperser	2
<i>Alipiopsitta</i>	Predator	1
<i>Amazona</i>	Predator	2
<i>Ampelioides</i>	Disperser	1
<i>Ampelion</i>	Disperser	1
<i>Andigena</i>	Disperser	1
<i>Anisognathus</i>	Disperser	1
<i>Anodorhynchus</i>	Predator	1
<i>Antilophia</i>	Disperser	1
<i>Ara</i>	Predator	1
<i>Aratinga</i>	Predator	1
<i>Aulacorhynchus</i>	Disperser	1
<i>Bangsia</i>	Disperser	1
<i>Bombycilla</i>	Disperser	3
<i>Brotogeris</i>	Predator	2,4,5
<i>Buthraupis</i>	Disperser	1
<i>Cacicus</i>	Disperser	2
<i>Calochaetes</i>	Disperser	1
<i>Capito</i>	Disperser	6
<i>Carpornis</i>	Disperser	2
<i>Caryothraustes</i>	Predator	7
<i>Cephalopterus</i>	Disperser	6
<i>Chamaepetes</i>	Disperser	8
<i>Chiroxiphia</i>	Disperser	3,6
<i>Chlorochrysa</i>	Disperser	1
<i>Chlorophanes</i>	Disperser	1
<i>Chlorophonia</i>	Disperser	1
<i>Chlorornis</i>	Disperser	1
<i>Chrysothlypis</i>	Disperser	1

<i>Cichlopsis</i>	Disperser	1
<i>Cissopis</i>	Disperser	1
<i>Conioptilon</i>	Disperser	1
<i>Conirostrum</i>	Disperser	1
<i>Corapipo</i>	Disperser	3
<i>Cotinga</i>	Disperser	3, 6, 9
<i>Crax</i>	Disperser	2,10
<i>Crypturellus</i>	Predator	2,10
<i>Curaeus</i>	Disperser	1
<i>Cyanerpes</i>	Disperser	1
<i>Cyanocorax</i>	Disperser	2
<i>Dacnis</i>	Disperser	2
<i>Delothraupis</i>	Disperser	1
<i>Deroptyus</i>	Predator	1
<i>Diglossa</i>	Disperser	1
<i>Doliornis</i>	Disperser	1
<i>Dubusia</i>	Disperser	§
<i>Entomodestes</i>	Disperser	1
<i>Eubucco</i>	Disperser	3
<i>Euphonia</i>	Disperser	2, 3, 7, 11
<i>Forpus</i>	Predator	1
<i>Graydidascalus</i>	Predator	1
<i>Guaruba</i>	Predator	1
<i>Gymnoderus</i>	Disperser	6
<i>Gymnomystax</i>	Disperser	1
<i>Haematoderus</i>	Disperser	3, 9
<i>Hapalopsittaca</i>	Predator	1
<i>Heterocercus</i>	Disperser	1
<i>Heterospingus</i>	Disperser	1
<i>Hypopyrrhus</i>	Disperser	1
<i>Icterus</i>	Disperser	3, 4, 7, 11
<i>Ilicura</i>	Disperser	1

<i>Iodopleura</i>	Disperser	1
<i>Iridophanes</i>	Disperser	1
<i>Iridosornis</i>	Disperser	1
<i>Lamprospiza</i>	Disperser	1
<i>Laniisoma</i>	Disperser	1
<i>Laniocera</i>	Disperser	1
<i>Legatus</i>	Disperser	3
<i>Lepidothrix</i>	Disperser	1
<i>Lipaugus</i>	Disperser	2
<i>Machaeropterus</i>	Disperser	1
<i>Manacus</i>	Disperser	1
<i>Masius</i>	Disperser	3
<i>Melanerpes</i>	Disperser	3, 11
<i>Mimus</i>	Disperser	2, 3
<i>Mionectes</i>	Disperser	3, 12
<i>Mitrospingus</i>	Disperser	1
<i>Mitu</i>	Disperser	13
<i>Myiodynastes</i>	Disperser	2, 3
<i>Nannopsittaca</i>	Predator	1
<i>Nothocercus</i>	Predator	1
<i>Nothocrax</i>	Disperser	1
<i>Odontophorus</i>	Disperser	10
<i>Ortalis</i>	Disperser	10
<i>Orthopsittaca</i>	Predator	1
<i>Patagioenas</i>	Disperser	1
<i>Pauxi</i>	Disperser	1
<i>Penelope</i>	Disperser	2, 10
<i>Perissocephalus</i>	Disperser	6
<i>Pharomachrus</i>	Disperser	3, 6, 8
<i>Phibalura</i>	Disperser	1
<i>Phoenicircus</i>	Disperser	9
<i>Pionites</i>	Predator	1

<i>Pionopsitta</i>	Predator	1
<i>Pionus</i>	Predator	2, 3
<i>Pipile</i>	Disperser	14
<i>Pipra</i>	Disperser	3, 6
<i>Pipraeidea</i>	Disperser	1
<i>Pipreola</i>	Disperser	1
<i>Porphyrolaema</i>	Disperser	1
<i>Primolius</i>	Predator	1
<i>Procnias</i>	Disperser	1
<i>Psophia</i>	Disperser	10
<i>Pteroglossus</i>	Disperser	2, 3, 7, 11
<i>Pytilia</i>	Predator	1
<i>Pyroderus</i>	Disperser	1
<i>Pyrrhura</i>	Predator	2
<i>Querula</i>	Disperser	3, 9
<i>Ramphastos</i>	Disperser	2, 10, 11
<i>Ramphocelus</i>	Disperser	10, 11
<i>Rupicola</i>	Disperser	1
<i>Saltator</i>	Disperser	1
<i>Schistochlamys</i>	Disperser	1
<i>Selenidera</i>	Disperser	2
<i>Sericossypha</i>	Disperser	1
<i>Steatornis</i>	Disperser	1
<i>Tangara</i>	Disperser	2, 3
<i>Tersina</i>	Disperser	15
<i>Thraupis</i>	Disperser	3
<i>Tinamus</i>	Predator	2, 10
<i>Touit</i>	Predator	1
<i>Trogon</i>	Disperser	2, 10, 11
<i>Turdus</i>	Disperser	1
<i>Tyrannulus</i>	Disperser	12
<i>Xenopipo</i>	Disperser	1

<i>Xipholena</i>	Disperser	1
<i>Zaratornis</i>	Disperser	1
<i>Zimmerius</i>	Disperser	1

TABLE D4. Trait (as Principal Components) variation across bird species identified in communities. Trait data are only calculated for frugivores and insectivores; other guilds are highlighted in grey. Trophic trait = second principal component (PC2) score for a composite of beak traits; locomotory trait = PC2 for a composite of traits associated with locomotion, including tarsus, wing and tail measurements; overall size = PC1 from a second PC analysis combining the PC1 scores for both trophic and locomotory traits (PC1 in each case is strongly associated with body size).

<i>Species</i>	Trophic trait	Locomotory trait	Overall size
<i>Aburria aburri</i>	0.1	0.7	-1.7
<i>Ampelioides tshudii</i>	0.0	-0.1	0.4
<i>Ampelion rubrocristatus</i>	0.0	0.0	0.4
<i>Ampelion rufaxilla</i>	-0.1	0.1	0.5
<i>Andigena cucullata</i>	0.1	0.3	-2.0
<i>Andigena hypoglaucha</i>	-0.1	0.2	-2.2
<i>Andigena nigrirostris</i>	0.0	0.4	-2.3
<i>Anisognathus igniventris</i>	0.1	-0.1	0.5
<i>Anisognathus lacrymosus</i>	0.0	-0.1	0.5
<i>Anisognathus somptuosus</i>	0.1	0.0	0.5
<i>Antilophia galeata</i>	-0.1	0.0	0.8
<i>Aulacorhynchus coeruleicinctis</i>	-0.2	0.0	-1.6
<i>Aulacorhynchus derbianus</i>	-0.2	0.0	-1.4
<i>Aulacorhynchus haematopygus</i>	-0.5	0.2	-2.1
<i>Aulacorhynchus prasinus</i>	0.0	0.1	-1.4
<i>Bangsia melanochlamys</i>	0.2	-0.2	0.4
<i>Bombycilla cedrorum</i>	0.0	0.1	0.7
<i>Buthraupis eximia</i>	0.2	-0.2	0.2
<i>Buthraupis montana</i>	0.2	-0.2	0.1
<i>Cacicus chrysionotus</i>	-0.1	0.2	-0.1
<i>Cacicus chrysopterus</i>	-0.1	0.0	0.3
<i>Calochaetes coccineus</i>	0.1	-0.1	0.5
<i>Capito auratus</i>	0.1	-0.2	0.4
<i>Capito aurovirens</i>	0.1	-0.1	0.4
<i>Capito dayi</i>	0.1	-0.2	0.3
<i>Capito hypoleucus</i>	0.1	-0.2	0.3
<i>Capito maculicoronatus</i>	0.1	-0.2	0.4
<i>Capito niger</i>	0.0	-0.2	0.4
<i>Carpornis melanocephala</i>	0.0	0.2	0.4

<i>Cephalopterus ornatus</i>	0.3	0.1	-1.7
<i>Chamaepetes goudotii</i>	0.1	0.2	-1.3
<i>Chiroxiphia boliviana</i>	-0.1	-0.2	0.9
<i>Chiroxiphia caudata</i>	-0.1	-0.1	0.7
<i>Chiroxiphia pareola</i>	-0.1	-0.2	0.8
<i>Chlorochrysa calliparaea</i>	-0.1	-0.2	0.8
<i>Chlorophanes spiza</i>	-0.2	-0.1	0.8
<i>Chlorophonia cyanea</i>	-0.1	-0.1	0.9
<i>Chlorophonia pyrrhophrys</i>	0.0	-0.1	0.9
<i>Chlorornis riefferii</i>	0.2	-0.1	0.2
<i>Chrysobhrys salmonea</i>	-0.2	-0.1	0.9
<i>Cichlopsis leucogenys</i>	-0.1	0.2	0.4
<i>Cissopis leverianus</i>	0.2	0.3	0.1
<i>Coniopsis mclennani</i>	-0.1	0.1	0.2
<i>Conirostrum leucogenys</i>	-0.2	0.0	1.0
<i>Corapipo gutturalis</i>	-0.2	-0.2	1.0
<i>Corapipo leucorrhoa</i>	-0.1	-0.2	1.0
<i>Cotinga cayana</i>	-0.1	0.1	0.4
<i>Cotinga cotinga</i>	-0.1	0.0	0.5
<i>Cotinga maculata</i>	-0.1	0.0	0.4
<i>Cotinga maynana</i>	-0.1	0.0	0.5
<i>Cotinga nattererii</i>	0.0	0.1	0.4
<i>Crax alberti</i>	0.7	-0.3	-3.1
<i>Crax alector</i>	0.9	-0.5	-3.2
<i>Crax daubentoni</i>	0.7	-0.1	-2.8
<i>Crax fasciolata</i>	0.8	-0.2	-2.7
<i>Crax globulosa</i>	0.7	-0.3	-3.1
<i>Curaeus forbesi</i>	0.0	-0.1	0.1
<i>Cyanerpes nitidus</i>	-0.3	-0.1	1.1
<i>Cyanocorax affinis</i>	0.2	-0.1	-0.8
<i>Dacnis albiventris</i>	-0.2	-0.1	1.0
<i>Dacnis cayana</i>	-0.2	0.0	0.9
<i>Dacnis flaviventer</i>	-0.2	-0.1	0.9
<i>Dacnis hartlaubi</i>	-0.1	-0.1	0.9
<i>Dacnis lineata</i>	-0.2	-0.1	1.0
<i>Dacnis venusta</i>	-0.2	0.0	0.9
<i>Delothraupis castaneiventris</i>	0.0	-0.1	0.6
<i>Deropithecus accipitrinus</i>	1.5	0.8	-1.5
<i>Diglossa glauca</i>	-0.2	-0.1	0.9
<i>Doliornis sclateri</i>	0.0	0.0	0.4
<i>Dubusia taeniata</i>	0.0	-0.1	0.4
<i>Entomodestes leucotis</i>	-0.1	0.2	0.4
<i>Eubucco bourcierii</i>	0.0	-0.1	0.5
<i>Eubucco richardsoni</i>	0.0	-0.1	0.6
<i>Eubucco tucinkae</i>	0.1	-0.1	0.4
<i>Eubucco versicolor</i>	0.0	-0.1	0.5

<i>Euphonia cayennensis</i>	0.0	-0.2	0.9
<i>Euphonia chlorotica</i>	-0.1	-0.1	1.0
<i>Euphonia chrysopasta</i>	0.0	-0.1	0.9
<i>Euphonia concinna</i>	-0.1	-0.1	1.0
<i>Euphonia cyanocephala</i>	-0.1	-0.1	0.9
<i>Euphonia finschi</i>	-0.1	-0.2	1.0
<i>Euphonia fulvicrissa</i>	0.0	-0.2	0.9
<i>Euphonia lanirostris</i>	0.0	-0.1	0.9
<i>Euphonia mesochrysa</i>	-0.1	-0.1	0.9
<i>Euphonia minuta</i>	-0.1	-0.1	1.0
<i>Euphonia pectoralis</i>	-0.1	-0.2	0.9
<i>Euphonia plumbea</i>	-0.1	-0.1	1.0
<i>Euphonia rufiventris</i>	0.0	-0.1	0.9
<i>Euphonia trinitatis</i>	-0.1	-0.1	1.0
<i>Euphonia violacea</i>	0.0	-0.2	0.9
<i>Euphonia xanthogaster</i>	0.0	-0.2	0.9
<i>Gymnoderus foetidus</i>	0.0	0.3	-0.3
<i>Gymnomystax mexicanus</i>	-0.1	-0.1	-0.1
<i>Haematoderus militaris</i>	0.2	0.5	-0.7
<i>Heterocercus aurantiivertex</i>	-0.1	0.1	0.8
<i>Heterocercus flavivertex</i>	-0.2	0.1	0.8
<i>Heterocercus linteatus</i>	-0.1	0.1	0.8
<i>Heterospingus xanthopygius</i>	0.0	0.0	0.5
<i>Hypopyrrhus pyrohypogaster</i>	0.1	0.2	-0.3
<i>Icterus nigrogularis</i>	-0.1	0.0	0.4
<i>Ilicura militaris</i>	-0.2	-0.1	0.9
<i>Iodopleura fusca</i>	-0.1	-0.1	0.8
<i>Iridosornis jelskii</i>	0.1	-0.2	0.6
<i>Iridosornis reinhardti</i>	0.1	-0.1	0.5
<i>Iridosornis rufivertex</i>	0.1	-0.1	0.6
<i>Lamprospiza melanoleuca</i>	0.1	-0.1	0.4
<i>Laniisoma elegans</i>	-0.1	-0.1	0.5
<i>Laniocera rufescens</i>	-0.1	0.2	0.3
<i>Legatus leucophaius</i>	-0.1	0.1	0.7
<i>Lepidothrix coronata</i>	-0.2	-0.1	1.0
<i>Lepidothrix iris</i>	-0.1	-0.1	1.0
<i>Lepidothrix isidorei</i>	-0.1	-0.1	1.1
<i>Lepidothrix nattereri</i>	-0.2	0.0	1.1
<i>Lepidothrix serena</i>	-0.1	-0.2	1.0
<i>Lepidothrix suavisima</i>	-0.1	-0.2	1.0
<i>Lepidothrix vilasboasi</i>	-0.1	-0.1	1.0
<i>Lipaugus fuscocinereus</i>	0.1	0.7	-0.3
<i>Lipaugus lanioides</i>	0.0	0.3	0.0
<i>Lipaugus streptophorus</i>	0.0	0.3	0.2
<i>Lipaugus unirufus</i>	0.0	0.4	0.1
<i>Lipaugus uropygialis</i>	0.1	0.7	-0.3

<i>Lipaugus vociferans</i>	0.0	0.3	0.2
<i>Machaeropterus pyrocephalus</i>	-0.2	-0.2	1.1
<i>Machaeropterus regulus</i>	-0.2	-0.2	1.1
<i>Manacus manacus</i>	-0.1	-0.3	0.9
<i>Manacus vitellinus</i>	-0.1	-0.3	0.9
<i>Masius chrysopterus</i>	-0.1	-0.1	0.9
<i>Melanerpes flavifrons</i>	-0.3	0.1	0.3
<i>Mimus longicaudatus</i>	-0.2	-0.1	0.0
<i>Mionectes macconnelli</i>	-0.2	-0.1	0.9
<i>Mionectes oleagineus</i>	-0.2	0.0	0.9
<i>Mionectes olivaceus</i>	-0.2	0.0	0.9
<i>Mitrospingus cassinii</i>	-0.1	-0.1	0.5
<i>Mitrospingus oleagineus</i>	0.0	-0.1	0.4
<i>Mitu salvini</i>	1.2	-0.7	-3.2
<i>Mitu tomentosum</i>	1.1	-0.6	-3.0
<i>Myiodynastes luteiventris</i>	0.0	0.2	0.2
<i>Nothocrax urumutum</i>	0.7	-0.7	-1.8
<i>Odontophorus gujanensis</i>	0.3	-0.4	0.0
<i>Ortalis columbiana</i>	0.2	0.3	-1.1
<i>Ortalis guttata</i>	0.1	0.3	-0.7
<i>Ortalis motmot</i>	0.1	0.1	-0.9
<i>Ortalis ruficauda</i>	0.2	0.0	-1.0
<i>Ortalis superciliaris</i>	0.1	0.2	-0.5
<i>Patagioenas cayennensis</i>	-0.2	0.4	0.2
<i>Patagioenas picazuro</i>	-0.1	0.4	-0.2
<i>Patagioenas plumbea</i>	-0.2	0.6	0.1
<i>Patagioenas speciosa</i>	-0.2	0.3	0.2
<i>Patagioenas subvinacea</i>	-0.2	0.4	0.3
<i>Pauxi pauxi</i>	1.2	-0.5	-3.1
<i>Pauxi unicornis</i>	1.2	-0.5	-3.1
<i>Penelope argyrotis</i>	0.1	0.6	-1.2
<i>Penelope jacquacu</i>	0.2	0.4	-1.8
<i>Penelope jacucaca</i>	0.3	0.1	-1.6
<i>Penelope marail</i>	0.1	0.5	-2.1
<i>Penelope montagnii</i>	0.2	0.4	-1.2
<i>Penelope obscura</i>	0.3	0.4	-1.7
<i>Penelope ochrogaster</i>	0.2	0.4	-1.8
<i>Penelope pileata</i>	0.2	0.3	-2.0
<i>Penelope purpurascens</i>	0.2	0.8	-2.5
<i>Penelope superciliaris</i>	0.1	0.1	-1.4
<i>Perissocephalus tricolor</i>	0.2	0.0	-1.0
<i>Pharomachrus antisianus</i>	0.2	1.1	-0.2
<i>Pharomachrus auriceps</i>	0.3	1.5	-0.6
<i>Pharomachrus pavoninus</i>	0.2	1.0	-0.2
<i>Phibalura flavirostris</i>	0.0	0.3	0.4
<i>Phoenicircus carnifex</i>	0.0	-0.1	0.4

<i>Phoenicircus nigricollis</i>	0.0	-0.1	0.4
<i>Pipile cujubi</i>	0.1	0.7	-1.7
<i>Pipile cumanensis</i>	0.2	0.8	-1.7
<i>Pipile jacutinga</i>	0.2	0.8	-1.7
<i>Pipra aureola</i>	-0.1	-0.1	0.9
<i>Pipra chloromeros</i>	-0.1	-0.2	1.0
<i>Pipra cornuta</i>	-0.1	-0.1	0.8
<i>Pipra erythrocephala</i>	-0.2	-0.1	1.0
<i>Pipra fasciicauda</i>	-0.1	-0.2	0.9
<i>Pipra filicauda</i>	-0.1	-0.1	0.9
<i>Pipra pipra</i>	-0.1	-0.2	1.0
<i>Pipra rubrocapilla</i>	-0.1	-0.1	1.0
<i>Pipraeidea melanonota</i>	-0.1	0.0	0.8
<i>Pipreola aureopectus</i>	-0.1	0.0	0.6
<i>Pipreola frontalis</i>	-0.1	0.0	0.7
<i>Pipreola pulchra</i>	0.0	0.0	0.5
<i>Pipreola riefferii</i>	-0.1	-0.1	0.6
<i>Pipreola whitelyi</i>	-0.1	0.0	0.6
<i>Porphyrolaema porphyrolaema</i>	0.0	0.0	0.5
<i>Procnias albus</i>	0.0	0.0	0.2
<i>Procnias averano</i>	0.0	0.0	0.1
<i>Procnias nudicollis</i>	-0.1	-0.1	0.2
<i>Psophia crepitans</i>	0.2	-3.2	-1.7
<i>Psophia leucoptera</i>	0.3	-2.8	-1.6
<i>Psophia viridis</i>	0.2	-2.8	-1.6
<i>Pteroglossus aracari</i>	-0.3	0.2	-2.4
<i>Pteroglossus azurea</i>	-0.2	0.3	-1.7
<i>Pteroglossus bailloni</i>	0.1	0.4	-1.6
<i>Pteroglossus beaubarnaesii</i>	-0.2	0.4	-2.1
<i>Pteroglossus bitorquatus</i>	0.0	0.1	-1.5
<i>Pteroglossus castanotis</i>	-0.5	0.2	-2.4
<i>Pteroglossus inscriptus</i>	0.0	0.1	-1.3
<i>Pteroglossus pluricinctus</i>	-0.3	0.3	-2.5
<i>Pteroglossus viridis</i>	0.1	0.0	-1.6
<i>Querula purpurata</i>	0.1	0.4	-0.3
<i>Ramphastos ambiguus</i>	-0.4	-0.1	-3.9
<i>Ramphastos dicolorus</i>	-0.3	0.2	-2.9
<i>Ramphastos swainsonii</i>	-0.2	-0.1	-4.2
<i>Ramphastos toco</i>	-1.3	0.0	-4.9
<i>Ramphastos tucanus</i>	-0.6	-0.1	-4.2
<i>Ramphastos vitellinus</i>	-0.2	0.0	-3.2
<i>Ramphocelus bresilius</i>	0.0	0.1	0.4
<i>Ramphocelus carbo</i>	0.0	0.0	0.4
<i>Ramphocelus dimidiatus</i>	0.0	0.0	0.5
<i>Ramphocelus flammigerus</i>	0.1	-0.1	0.3
<i>Ramphocelus melanogaster</i>	0.1	0.0	0.4

<i>Ramphocelus nigrogularis</i>	0.0	0.1	0.4
<i>Rupicola peruvianus</i>	0.3	0.0	-0.4
<i>Rupicola rupicola</i>	0.3	-0.1	-0.4
<i>Saltator orenocensis</i>	0.3	0.0	0.2
<i>Schistochlamys melanopsis</i>	0.1	-0.1	0.4
<i>Schistochlamys ruficapillus</i>	0.2	-0.1	0.4
<i>Selenidera gouldii</i>	0.2	0.0	-1.2
<i>Selenidera maculirostris</i>	0.3	0.0	-1.1
<i>Selenidera nattereri</i>	0.1	-0.1	-1.1
<i>Selenidera piperivora</i>	-0.2	0.1	-1.2
<i>Selenidera reinwardtii</i>	0.2	0.0	-1.1
<i>Selenidera spectabilis</i>	-0.2	0.0	-1.9
<i>Sericossypha albo cristata</i>	-0.1	0.1	0.2
<i>Steatornis caripensis</i>	0.4	1.5	-1.0
<i>Tangara argyrofenges</i>	-0.1	0.0	0.9
<i>Tangara arthus</i>	-0.1	0.0	0.8
<i>Tangara callophrys</i>	-0.2	0.0	0.8
<i>Tangara cayana</i>	0.0	-0.1	0.8
<i>Tangara chilensis</i>	-0.1	0.0	0.8
<i>Tangara chrysotis</i>	0.0	-0.1	0.8
<i>Tangara cyanicollis</i>	-0.1	-0.1	0.8
<i>Tangara cyanocephala</i>	-0.1	-0.1	0.9
<i>Tangara cyanoptera</i>	0.0	0.0	0.8
<i>Tangara cyanotis</i>	-0.1	-0.1	0.9
<i>Tangara cyanoventris</i>	-0.1	-0.1	0.8
<i>Tangara guttata</i>	-0.1	-0.1	0.8
<i>Tangara gyrola</i>	-0.1	0.0	0.8
<i>Tangara heinei</i>	-0.1	0.0	0.8
<i>Tangara inornata</i>	0.0	-0.1	0.8
<i>Tangara larvata</i>	-0.1	0.0	0.8
<i>Tangara mexicana</i>	-0.1	0.0	0.8
<i>Tangara nigrocincta</i>	-0.1	0.0	0.8
<i>Tangara nigroviridis</i>	-0.1	-0.1	0.9
<i>Tangara parzudakii</i>	0.0	0.0	0.6
<i>Tangara phillipsi</i>	-0.1	0.0	0.8
<i>Tangara punctata</i>	-0.1	-0.1	0.9
<i>Tangara ruficervix</i>	-0.1	0.0	0.8
<i>Tangara schrankii</i>	-0.1	-0.1	0.8
<i>Tangara seledon</i>	0.0	0.0	0.8
<i>Tangara varia</i>	-0.1	-0.1	1.0
<i>Tangara vassorii</i>	-0.1	0.0	0.8
<i>Tangara velia</i>	-0.1	0.0	0.8
<i>Tangara viridicollis</i>	-0.1	0.0	0.8
<i>Tangara vitriolina</i>	0.0	-0.1	0.7
<i>Tangara xanthocephala</i>	-0.1	0.0	0.8
<i>Tangara xanthogastra</i>	-0.1	-0.1	0.9

<i>Tersina viridis</i>	0.0	0.0	0.7
<i>Thraupis bonariensis</i>	0.1	-0.1	0.4
<i>Thraupis cyanocephala</i>	0.0	0.0	0.5
<i>Thraupis cyanoptera</i>	0.1	0.0	0.4
<i>Thraupis episcopus</i>	0.0	0.0	0.5
<i>Thraupis ornata</i>	-0.1	0.0	0.5
<i>Thraupis palmarum</i>	0.0	0.0	0.5
<i>Thraupis sayaca</i>	0.1	0.0	0.5
<i>Trogon caligatus</i>	0.2	0.7	0.2
<i>Trogon chionurus</i>	0.3	0.8	-0.1
<i>Trogon personatus</i>	0.1	1.0	0.0
<i>Turdus flavipes</i>	-0.1	0.0	0.4
<i>Turdus fulviventeris</i>	-0.1	-0.2	0.1
<i>Turdus fuscater</i>	-0.1	-0.6	-0.5
<i>Turdus leucops</i>	-0.1	-0.1	0.3
<i>Turdus nigriceps</i>	-0.2	-0.2	0.4
<i>Turdus nudigenis</i>	-0.1	-0.2	0.3
<i>Turdus obsoletus</i>	-0.1	-0.1	0.3
<i>Turdus serranus</i>	-0.1	-0.1	0.1
<i>Tyrannulus elatus</i>	-0.1	0.0	1.1
<i>Xenopipo atronitens</i>	-0.1	0.0	0.8
<i>Xenopipo flavicapilla</i>	-0.1	0.0	0.8
<i>Xenopipo holochlora</i>	-0.2	0.0	0.9
<i>Xenopipo unicolor</i>	-0.2	0.0	0.9
<i>Xenopipo uniformis</i>	-0.1	0.0	0.8
<i>Xipholena atropurpurea</i>	-0.1	0.0	0.5
<i>Xipholena lamellipennis</i>	-0.1	0.1	0.5
<i>Xipholena punicea</i>	-0.1	0.0	0.4
<i>Zaratornis stresemanni</i>	0.1	0.0	0.3
<i>Zimmerius acer</i>	-0.2	-0.1	1.0
<i>Zimmerius bolivianus</i>	-0.2	0.0	0.9
<i>Zimmerius chrysops</i>	-0.1	-0.1	1.0
<i>Zimmerius cinereicapilla</i>	-0.1	-0.1	0.9
<i>Zimmerius gracilipes</i>	-0.2	-0.1	1.0
<i>Zimmerius viridiflavus</i>	-0.1	-0.1	1.0

References

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