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# EXTINCTION AND ITS PREVENTION IN ‘ISLAND’ SPECIES

R. W. MARTIN

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# EXTINCTION AND ITS PREVENTION IN ‘ISLAND’ SPECIES

ROBERT WILLIAM MARTIN

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## Abstract

The bird communities of the islands of Sangihe and Talaud in Indonesia provide an opportunity to examine two extinction paradigms; declining populations and small or restricted populations. On Talaud, species abundance responses to habitat modification informs relative extinction risk and future community impacts of habitat loss within an intact community with large populations. On Sangihe, identification and quantification of remnant suitable habitat for species with critically low population sizes allows the examination of options to extend their area of occupancy through targeted restoration. I constructed Distance-based density estimates for the bird community on Karakelang, Talaud Islands at eight locations along a habitat modification gradient and found species which increased, were stable, or declined in abundance along this gradient, and within those that declined individual responses occurred in both the rate of decline and the shape of the response. Forest-dependent and endemic species were far more abundant in primary forest, and as these species were large, total bird biomass declined dramatically with greater habitat modification. Several key bird species, including the Endangered Red-and-blue Lory, declined most severely where forest no longer contained primary elements. However, almost all were recorded at each step along the gradient. Currently, most species retain relatively large populations, but primary forest loss will cause substantial population reductions. For three Critically Endangered birds on Sangihe, I created habitat suitability models based on presence-absence data to identify those areas away from the current occupied range of the species with potential to support the species with minimal restoration effort. Three small areas are potential restoration sites for Cerulean Paradise-flycatcher away from the current occupied range and one area adjacent to these may be suitable for Sangihe Golden Bulbul, but more ‘nearly-suitable’ habitat exists for these species and Sangihe Whistler within the Sahendaruman crater. Restoration efforts should; focus on those areas that are most similar to currently occupied sites, enhance the abundance of lianas for Cerulean Paradise-flycatcher sites in valley habitat, and fill gaps within the ridgetop forest with native trees. Without restoration, the slow retreat of each species will continue, with Cerulean Paradise-flycatcher at greatest risk of global extinction.

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# Chapter 1: Predicting and preventing island bird extinctions caused by habitat loss

Predicting and preventing species extinction is a core aim of conservation biology (Soulé 1985). As we are currently considered to be within or entering a sixth mass extinction event (Barnosky et al. 2011; Ceballos et al. 2015) with extinctions estimated to be occurring at around 1,000 times the background rate (Pimm et al. 2014), the prediction may be relatively easy while the prevention may not. Thankfully there are now numerous successful examples of the rescue through conservation action of apparently doomed bird species (e.g. Jones & Merton 2012), and there is no longer an assumption that efforts to prevent extinction are ever in vain. Even where the species is lost, vital experience in approach and process for application to future efforts is gained (Groombridge et al. 2004; VanderWerf et al. 2006). At this, the sharp end of conservation, the primary means of species-based prioritisation for conservation action is the International Union for the Conservation of Nature's Red List (BirdLife International 2017a). All species considered Globally Threatened (those listed as Critically Endangered, Endangered or Vulnerable) are facing, at best, a high, and at worst, an extremely high, risk of extinction in the wild (IUCN 2001), with the implication that all require the current circumstances to change in order to prevent that extinction from happening.

The degradation and loss of natural habitat due to anthropogenic land use change, primarily through agricultural intensification to supply increasing consumption levels (Phalan et al. 2011; Krausmann et al. 2013), is the most significant driver of species endangerment (Vié et al. 2009; Newbold et al. 2014; Pimm et al. 2014). Declines in abundance are occurring globally across many vertebrates (Ceballos et al. 2017) and for birds species extinctions caused by habitat loss have been predicted to exceed 1,000 during the 21st Century (Pimm et al. 2006). Supporting this, absolute forest loss has been demonstrated to predict risk of bird extinction, as defined by a listing as a Globally Threatened Species, generally well (Brooks *et al.* 2002, Pimm *et al.* 2006). However, as yet, extinctions due to habitat loss have been few (Stork 2009; Loehle & Eschenbach 2012) and limited to regions with concentrations of small-ranged endemics and exceptional levels of deforestation and fragmentation (e.g. Lees & Pimm 2015). This paradox is attributed to the time lag inherent in the effect of habitat loss on bird communities (the 'relaxation' period [Diamond 1972]), resulting in an 'extinction debt' (Tilman et al. 1994; Kuussaari et al. 2009; Kitzes & Harte 2015) to be paid through the loss of species over time as a community reduces to a new, lower equilibrium as predicted by the species-area relationship for the new, smaller area of habitat (MacArthur & Wilson 1967a). This reinforces the suggestion that even where a great deal of habitat has been lost there is a 'window of conservation opportunity' where restoration and targeted conservation may prevent the extintion debt from being paid (Wearn et al. 2012). While there has been debate indicating that the use of absolute species richness overestimates extinction debt and that the endemic-species area should be used (He & Hubbell 2013), there is general consensus that there is a deterministic rate of extinction from within a community following habitat loss (Rybicki & Hanski 2013) and that extinction debts are highly prevalent across varied communities (Halley et al. 2016).

To be able to act, one needs be able to identify those species from within a community likely to be indebted, investigate their individual response to the cause of the debt and derive management options to address these (Hylander & Ehrlén 2013). At one extreme, where historic habitat loss has been very considerable those particular species should be obvious, provided the location has been adequately taxonomically sampled (Lees & Pimm 2015). They are likely to already be on the Red List and require urgent efforts to understanding their habitat requirements and demographic parameters to suggest appropriate action (Jones & Merton 2012). In extinction debt scenarios, this will almost certainly require the restoration of habitat to a suitable condition (Kuussaari et al. 2009). The emphasis then falls on identifying the simplest and fastest means to that restoration of habitat to an acceptable condition for the species in question (Miller & Hobbs 2007). Species-distribution models based on the habitat associations of the species of concern have been used to locate potentially overlooked areas in the search for species unrecorded for many years (Dunn et al. 2015) or to find new populations of rare species (Maphisa et al. 2009). As they predict relative habitat suitability, they can also been used to identify unoccupied sites that are similar to occupied locations (Guisan et al. 2013). In Europe, species-distribution models have been used to identify the suitability of release sites for Bearded Vultures in the Alps (Hirzel et al. 2004) and for White Storks in Sweden (Olsson & Rogers 2009), while in Africa potential recolonisation sites for Red-billed Oxpecker have been identified (Kalle et al. 2017). These species are not globally threatened (Bearded Vulture is Near Threatened) (BirdLife International 2017a), but species distribution models have great potential for the identification and prioritisation of restoration sites for rare species where restoration to a suitable quality may be achieved (Bracebridge et al. 2011).

The Indonesian island of Sangihe, north of Sulawesi, holds one of the greatest concentrations of birds at an extremely high risk of extinction where habitat loss is the principal threat (BirdLife International 2017a), and where securing the long-term future of this unique avifauna appears to require the restoration of currently unsuitable habitat in order to increase populations to a lower risk status (Fauzan & Bashari 2016; BirdLife International 2017b). The five Critically Endangered species on the island are all believed to solely occur within a remnant forest of less than 8 km$^2$ (Riley 2002a; BirdLife International 2017a). In Chapter 2 of this thesis, I detail the identification of potential restoration sites through building habitat suitability models for three of the Critically Endangered birds, following intensive bird and habitat surveys in locations identified as containing the best potential forest habitat across the island.

Where habitat loss is ongoing but a significant extent of habitat remains, species traits, current abundance and level of human encroachment are linked to the likelihood of endangerment (Lee & Jetz 2011; Newbold et al. 2013; Burivalova et al. 2015; Davies et al. 2015). The rapid development of remotely sensed data products has enabled global calculations of the area and rate of habitat loss (e.g. Hansen et al. 2013; Achard et al. 2014), and the application of these as surrogates for either declines in a species' population (Buchanan et al. 2008; Vetter 2009), or estimates of the area of occurrence in order to prioritise extinction risk among species (Bird et al. 2012; Tracewski et al. 2016). A key assumption acknowledged by Bird et al. (2012) and Tracewski et al. (2016a) is that a given loss of area results in an equivalent change in population size for each species assessed (Tracewski et al.

2016). To reduce the impact of violations of this assumption, species included in these analyses are limited to those classified as having ‘High’ forest dependence (BirdLife International 2017a): this is scored for each species as either ‘High’, ‘Medium’, ‘Low’ or ‘Non-forest’. This requires a level of detailed information on the ecology of each species, some of which is necessarily inferred from congeners, and assumes that a species responds identically wherever it occurs.

Species-area and endemic-area models predicting extinction debt generally assume a similar binary approach, or a constant density response to habitat modification (Brooks et al. 1999, 2002; Pimm et al. 2006; Wearn et al. 2012; He & Hubbell 2013; Rybicki & Hanski 2013). When species-abundance distributions have been incorporated into these models extinction debts are widely prevalent, but it is still assumed that this abundance distribution returns to the same distribution post habitat loss (Halley et al. 2016). At the species level, while habitat is still largely intact, linear declines in abundance have been noted (Radford et al. 2005), but there are thresholds of remaining habitat area below which abundance declines at a greater rate, and it may be that non-linear relationships are not unusual (Andrén 1994; Bender et al. 1998; Radford et al. 2005; Swift & Hannon 2010). While species richness has been demonstrated to typically increase in a linear fashion with more intact forest (Española et al. 2016), studies of abundance responses are necessary to understand the impacts of habitat loss, and few have been completed (Radford et al. 2005; Davies et al. 2015). However, when considering the application of projected land cover loss to estimates of impacts on birds, species on oceanic islands may not demonstrate the same responses as equivalent species in species-rich mainland communities. Oceanic island avifaunas have suffered the greatest number of extinctions in recent centuries, but few have been caused by habitat loss (Loehle & Eschenbach 2012). This is thought likely to change as rates of habitat loss and human impacts on islands are predicted to greatly increase during the present century and given that islands possess higher endemic richness than mainland areas (Kier et al. 2009), it may be assumed that more global extinctions will happen on islands than mainlands for an equivalent loss of habitat. But there are intrinsic differences between bird communities on mainland and oceanic islands (Stuart et al. 2012). Islands are relatively species-poor and species that do manage to colonise typically display an expanded overall niche, an ‘ecological release’ (Van Valen 1965; Diamond 1970), and achieve higher population densities in partial ‘density compensation’ (MacArthur et al. 1972a). As extinction risk is typically dependent on initial population size and habitat specialism (Purvis et al. 2000), it may be that fewer extinctions will occur on islands in a given time period in comparison to species-rich mainland communities, and may demonstrate longer relaxation periods as extinctions may be mediated through interspecific interactions which are less intense in communities with fewer species (Hylander & Ehrlén 2013; Halley et al. 2016).

In Chapter 3, I present the results of an investigation into the bird community of Karakelang, the largest and most intact of the Talaud Islands, north-eastern Indonesia, and use density estimates at points along a habitat modification gradient to directly assess species responses. Species occurring at high densities at all levels of modification cannot be thought likely to become extinct regardless of the extent of primary forest loss, while those that demonstrate very rapid declines in abundance with modification must be prioritised in terms of extinction risk. The correlation of species forest dependency codings with these responses is examined and the shape of each species modification density response assessed in conjunction with new estimates of the island population sizes for the bird community. These are used to inform discussion on the likely accuracy of predictions of species extinction from estimates of the rate of habitat loss on the island.

## 1.1 Conservation challenges in Wallacea

The scale of the current biodiversity crisis is greatest in south-east Asia (Wilcove et al. 2013), and Indonesia is the country with the greatest rise in the rate of forest loss over the past two decades (Hansen et al. 2013). The country also has the highest number of endemic bird species globally (BirdLife International 2017c) with new species becoming recognised both through investigations of new locations (Eaton et al. 2016a) and comprehensive taxonomic reappraisal of taxa globally (e.g. Collar et al. 2013; Irestedt et al. 2013; del Hoyo et al. 2014, 2016; Eaton et al. 2016b; Ng et al. 2016). The Wallacean hotspot (Myers et al. 2000) has one of the highest rates of avian endemism globally with many unique species found on single small- and medium-sized islands (Coates et al. 1997; Eaton et al. 2016b). While these islands are unsuitable for large conversion of habitat to agricultural plantation, increasing consumption and growing population size is steadily reducing primary habitat (Margono et al. 2014). As such, Indonesia is at the forefront of the biodiversity crisis but focused conservation efforts have resulted in notable success in preventing extinctions (e.g. Cahill et al. 2006), but these have focused on the enforcement of legislation to prevent exploitation of the species. Targeted habitat improvements for conservation to date appear limited to the provision of artificial nest boxes (Walker et al. 2005).

The islands of Sangihe and Talaud are located at the extreme north-east of Indonesia, lying to the north of Sulawesi at the convergence of the Moluccan Sea with the Pacific Ocean. Together they have been long recognised for their endemic and restricted range species (Stattersfield et al. 1998): both island groups contain AZE sites (Alliance for Zero Extinction 2010). Gunung Sahendaruman on Sangihe is one of the most urgent global bird conservation priorities (Ricketts et al. 2005; BirdLife International 2017d). The two islands have sharply contrasting histories of habitat loss; Sangihe is estimated to have lost in excess of 99% of original habitat (Riley 2002a), while Karakelang, largest of the three main Talaud Islands, retains a large area of intact primary forest, covering more than 30% of the total island area.

Of the ten endemic species on Sangihe, five are Critically Endangered, two are Endangered and two are Near Threatened, plus one further endemic subspecies of an Endangered species is considered recently extinct on the island (Riley 2002a; BirdLife International 2017a, 2017e). This perilous situation is due to the tiny remaining area of original native forest remaining on the island (BirdLife International 2017d), estimated at 748 hectares in 2006 (Mamengko & Mole 2006b). There is no evidence that invasive species or disease has impacted this set of species (Riley 2002a, 2002b; BirdLife International 2017c). Deforestation has occurred on the islands north of Sulawesi including Sangihe since the 17th Century. A contemporary report from a 1677 visit describes that on Sangihe the hills 'even the biggest and highest, are planted with coconut trees' (Henley 2005). Intensity of agriculture increased further around the turn of the twentieth century with the addition of nutmeg *Myristica fragrans* to the profitable copra (the dried flesh of coconut, *Cocos nucifera* agg.) export industry (Henley 2005). Human population density was already 166 km$^{-2}$ in 1930 (Henley 2005) and has actually grown little in the past 80 years and is currently 175 km$^{-2}$, with a total population of nearly 130,000 growing at an annual rate of just 0.6% (Baden Pusat Statistik 2017a). On top of the human impact, recurring violent eruptions of Gunung Awu, which effectively comprises the entire northern half of the island, have cleared surrounding habitat on five occasions between 1711 and 1966, with recent smaller eruptions occurring in 1986 and 2004 (Global Volcanism Program 2013).

Since the early 20th century, none of the high priority species mentioned above have been recorded away from a highly-restricted area of remnant forest present around the Sahendaruman crater in the south of Sangihe Besar (Whitten et al. 1987; Lambert 1999; Rozendaal & Lambert 1999; Riley 2002a; Mamengko 2006; Whitten 2006; Rosyadi 2009); but see White & Bruce 1986). Indeed, Cerulean Paradise-flycatcher was considered most likely extinct (Whitten et al. 1987), until being rediscovered in deep valleys around the Sahendaruman crater in 1998 (Riley & Wardill 2001; Whitten 2006). Given this evidence, an extinction debt would be expected to be pending or currently in the process of being paid on Sangihe. Therefore, there is an urgent need understanding of the habitat associations and likely possibility of persistence at current habitat availability is required for the species already identified as being of extremely high risk of extinction, including the potential for the restoration of previously lost habitat area.

Talaud, in contrast, retains a great deal of intact primary habitat (Riley 2003) and does not possess any Critically Endangered species, largely due to the large extent of remnant native forest and relatively large population estimates previously found for the endemic species (Riley 2003; BirdLife International 2017a). It appears that no modern extinctions have occurred on the island, hence an intact bird community is assumed to be present (Meyer & Wiglesworth 1898a, 1898b; Riley 2003). Support for this assumption comes from the presence of at least two endemic ground-dwelling rallids (Lambert 1998a, 1998b), a family over-represented in recorded island extinctions (Steadman 2006) and whose species are disproportionately likely to become extinct (McKinney & Lockwood 1999). The one

Endangered species, Red-and-blue Lory, has primarily been threatened by trapping for the pet trade (Riley 2003; BirdLife International 2017e). Rates of habitat conversion have increased with human population rises on the island in recent decades (Riley 2003; Mamengko & Mole 2006b; Margono et al. 2014) and this rate of habitat loss is expected to increase over the next century in line with other developing islands (Kier et al. 2009) prompting concern that this generally good health of the bird community at present is likely to deteriorate in the future. While on Sangihe loss of habitat is extensive and the effects are already noted, on Talaud there is the opportunity to identify individual species likely responses to future habitat degradation and loss through utilising a 'space-for-time' approach. Trait-based differences predict that endemics (Davies et al. 2015), large-bodied frugivores and mid-storey insectivores (Newbold et al. 2013; Bregman et al. 2014; Burivalova et al. 2015) are at greatest risk from habitat loss and would be expected to demonstrate the greatest abundance declines along a gradient of habitat modification. At the community level, forest bird richness would be expected to be lower in plantations than either primary or disturbed forest (e.g. Beukema *et al.* 2007; Newbold *et al.* 2015), but of particular concern is the potential for certain species to decline at a disproportionate rate to the extent of habitat impact observed. These species are those that require consideration even while populations appear secure, given the pervasive, time-lagged nature of the relaxation phase of extinction debt (Kuussaari et al. 2009; Wearn et al. 2012).

## 1.2 Previous ornithological studies on Sangihe and Talaud

Considerable exploration and collecting effort occurred on Sangihe during the latter half of the 19th Century, though less took place in the Talaud Islands (White & Bruce 1986). These observations and specimens resulted in a relatively settled avifauna for Sangihe described by the beginning of the 20th Century (Blasius 1888; Meyer & Wiglesworth 1898a) and Meyer and Wiglesworth (1898), providing a good baseline for the avifauna of the islands. Several reported species were disputed however, and several collecting trips exclusively used local hunters who may have been imprecise with the exact collecting locations among the numerous islands in the area. As an example, the following quote from Adolf Meyer hints at the potential for confusion over the certain determination of specimen origin (Meyer 1879):

> *I regret that the exact localities where I got my specimens were often destroyed, and the exact dates when I got them nearly always so. This was partly my own fault, as I had not labelled every specimen in such a way that the labels could not be torn off-partly the fault of my agent, who did not follow my instructions and keep the lots together and distinct from each other, as I had sent them. For this reason I am unable to give detailed local lists as they ought to be given, and as I hope they will be given one day. It is only from the Togian Islands, where I collected in the month of August, that I am able to give a trustworthy local list*[1]

Little further was added to these works until the latter part of the 20th century aside from a brief stop by the Crane Expedition of 1929 (Mayr and Camas 1938). Beginning with M. D. Bruce in 1978, short visits by ornithologists added some interesting sightings but typically did not obtain specimens or documentation of records, except for F.G. Rozendaal in 1985 (White & Bruce 1986; Riley & Wardill 2001).

A new round of exploration and fieldwork began in the 1990s, spurred on by the suggestion that Cerulean Paradise-flycatcher was likely extinct (Whitten et al. 1987) and other species may be threatened on the islands (ICBP 1992). Particular concern over Red-and-blue Lory (Anon 1993) resulted in a survey and status assessment (Lambert 1997) followed by an updated avifauna for the two islands in 1997 (Riley 1997a) and the formation of 'Action Sampiri' with various contributions to conservation on the islands (Riley et al. 1999; Riley & Wardill 2001; Wardill 2001; Riley 2002a, 2003). Subsequently, a series of studies was conducted as part of the Global Environment Fund (GEF) project, 'Conservation of Key Forests in the Sangihe-Talaud Islands, Indonesia'. These included surveys aimed at monitoring the population of Cerulean Paradise-flycatcher (Mamengko 2006) and fixed-point photographic monitoring of habitat extent (Mamengko & Mole 2006b). On Talaud,

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[1] This collecting period includes the period of acquisition of specimens from the Sangihe Islands in 1870-71 (Meyer 1879).

these were focused on Red-and-blue Lory, and further DISTANCE (Buckland et al. 2001)-based population estimates were made from data collected in 2003 and 2006 (Mamengko & Mole 2006b), monitoring of roost trees was carried out in 2004 and 2006 (Mamengko & Lumasuge 2006), and Wangko (2006) reported on trade monitoring. Consequently, there has been no update to the population estimates and status assessments of the other species of restricted range birds of Karakelang since 1999. Concerns over clearance and logging were raised in Riley (2003), along with suggestions that introduced rats *Rattus* spp., cats *Felis catus* and dogs *Canis lupus familiaris* may be negatively impacting the native ground-dwelling birds.

## 1.3 Island locations and geography

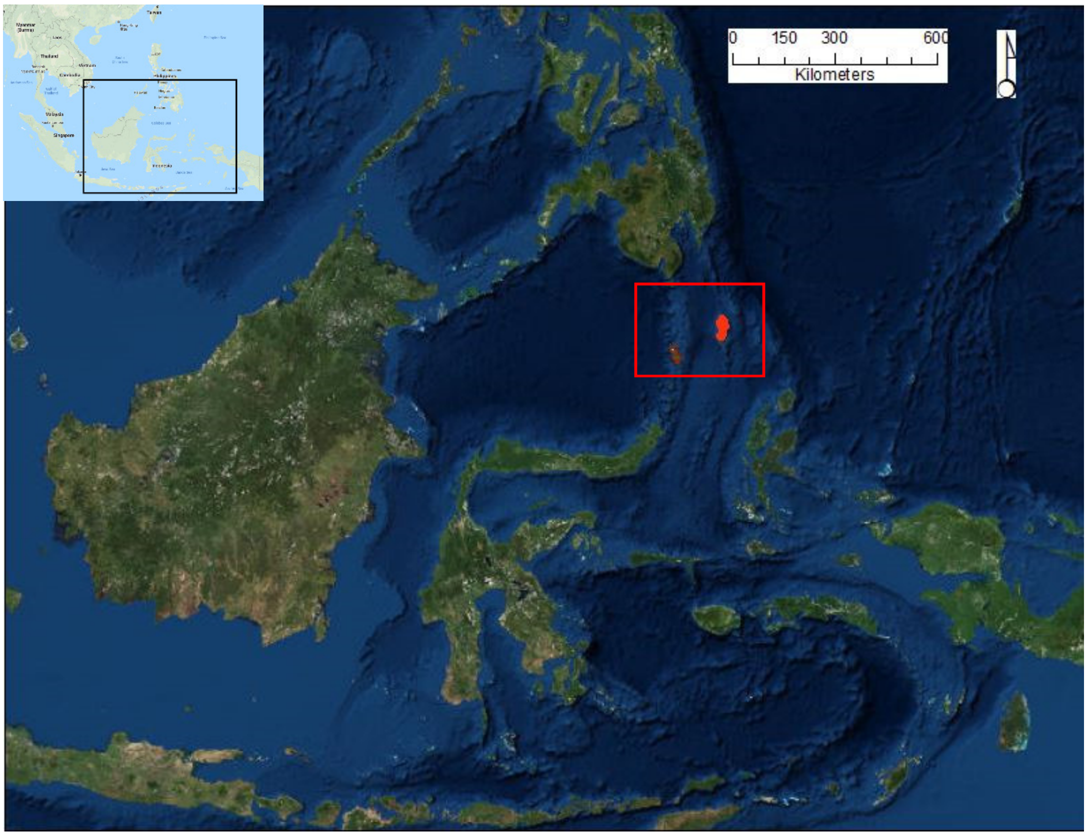


Figure 1-1. Location of the islands of Sangihe (western island in box) and Karakelang, Talaud (eastern island in box).

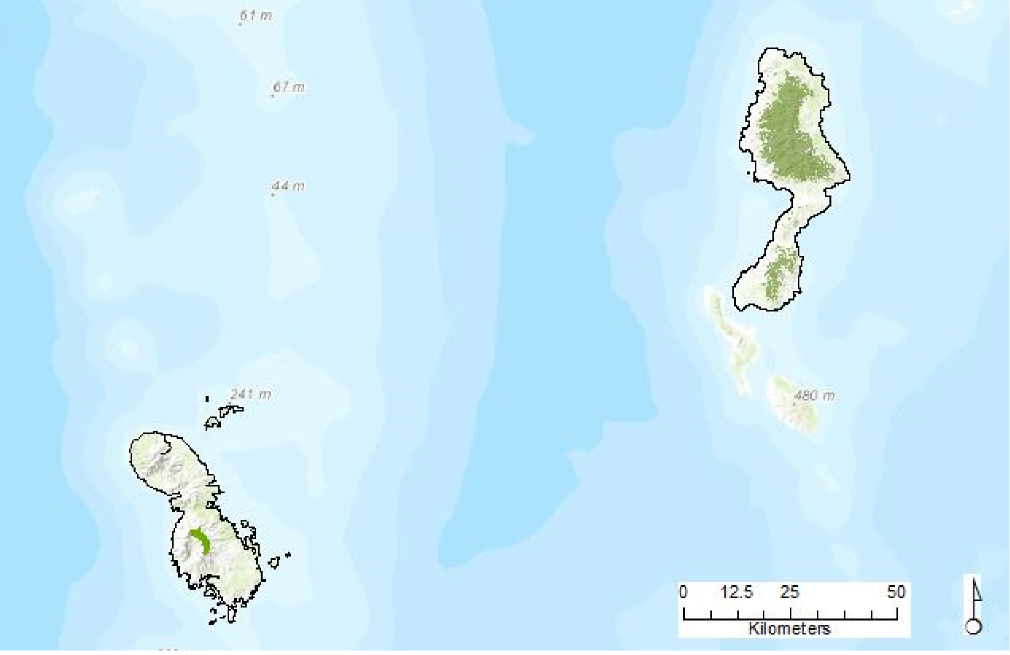


Figure 1-2. Primary forest area on the two islands: Sangihe to the south west (forest area derived from aerial imagery in conjunction with ground-truthing during the present survey), and Karakelang, Talaud to the north east (primary forest area based on Syarif [2004] and updated using the Global Forest Watch loss layer (Hansen et al. 2013).

Sangihe lies at the northern end of the chain of relatively young volcanic islands extending north from Sulawesi, approximately 220 km north of Manado (Figure 1-1). With a total land area of 737 km$^2$ (*Badan Pusat Statistik Kabupaten Kepulauan Sangihe* [Baden Pusat Statistik] 2017a), the main island is formed of three volcanoes, two of which are remnant craters. The active Gunung Awu dominates the north of island, presently measured at 1,340 m above sea level (Baden Pusat Statistik 2017a). The south is largely shaped by the remnants of the extinct Gunung Sahendaruman caldera, which rises to a horseshoe-shaped ridge of approximately 23 km circumference containing a two kilometre exit valley dividing the ridge in the southwest. The maximum elevation of the ridge is 1,046 m (Baden Pusat Statistik 2017a) at the peak of Gunung Sahengbilara on the northern point of the crater rim. The third, smaller crater is Gunung Otomata and is in the centre of the island surrounding the capital city Tahuna (population 36,782 [Baden Pusat Statistik 2017a]).

Plantations cover 215 km$^2$ of the island area, 75% coconut with the rest split between clove *Eugenia aromatica* and nutmeg. Just over 11 km$^2$ of further land is used for food crops, mostly cassava *Manihot esculenta*, sweet potato *Ipomoea* *batatas* and taro *Colocasia esculenta,* with 1 km$^2$ of paddy rice and small areas of addional crops (Baden Pusat Statistik 2017a). In addition to these official areal figures virtually all the rest of the island has been cultivated in the past, with productive forestry gardens, called *kebun*, occupying much of the remainder of the island area. There remain no strictly protected areas on the island, despite proposals raised over 20 years ago (Sujatnika. & Jepson 1995) and strongly recommended in both Lambert (1997) and Riley (2002), but suggested to likely cause strongly negative responses from the local community which may compromise the success of a protected designation (Fauzan & Bashari 2016). There are three areas of Protection Forest, (known as *Hutan Lindung*) centred on the three higher parts of the island; 42.68 km$^2$ around Gunung Sahendaruman, 48.84 km$^2$ around Gunung Awu and 11.05 km$^2$ above the city of Tahuna around Gunung Otomata (Baden Pusat Statistik 2017a).

Average rainfall for Sangihe between 2010-2014 was 3720 mm per year with rain falling on an average of 259 days in the year (Baden Pusat Statistik 2017a). Humidity averaged around 80% during 2015, being slightly lower between July and October, and temperatures are stable throughout the year around an average maximum of 31 to 33 degrees Celsius (Baden Pusat Statistik 2017a). The Sahendaruman crater is likely to have higher humidity and even higher rainfall due to its extremely steep topography, and temperatures at the ridge are several degrees cooler than in the lowlands.

The remote Talaud Islands lie at the northeast extremity of Indonesia between Halmahera and Mindanao in the Philippines, approximately 120 kilometres northeast of Sangihe (Figure 1-1). The island group is geologically relatively young, being formed of Awit sandstone and uplifted corraline limestone thrust upwards by the Sangihe Arc over-riding the Halmahera Arc less than 1 million years before present (Hall & Wilson n.d.; Moore et al. 1981). They are low-lying and non-volcanic but possess a rugged and steep interior. Karakelang, the largest of the three main islands, is approximately 60 km long and between 7 and 23 km in width with a total area of 802.43 km$^2$ (Baden Pusat Statistik 2017b). Gunung Biala (608 m) is the highest point on the island (Riley 2003). There is a long history of continual human habitation on Karakelang but the account of Hickson (1889), who visited at a time of a severe cholera outbreak that severely depleted the human population, referred to the presence of intact wild forest directly adjacent to population centres suggesting that most the island then was primary forest. Today the total human population is 55,901 at a density of nearly 62/km2, growing at 1.3% annually (Baden Pusat Statistik 2017b).

Approximately 33% of the total area of Karakelang is designated as strictly protected Wildlife Reserve (*Suaka Margasatwa*), which includes a no-take clause that prohibits hunting and timber extraction (Protected Planet 2016). The total area of 27,009 ha is split between the Northern Wildlife Reserve (23,186 hectares) and the Southern Wildlife Reserve (3,823 hectares). Surrounding much of the Wildlife Reserve and additionally in the central section of the island there is a total of 12,684 ha of *Hutan Lindung*, or Protection Forest, designated with the intent to retain forest cover for the protection of environmental services. Outside of the protected areas, much of the remaining habitat has been converted to agricultural use. This is concentrated in the coastal belt and the central plain up to 150 m altitude. Flatter areas are used predominately for coconut, clove and nutmeg plantations interspersed with small-holder mixed 'gardens' or *kebun* (Riley 2003). *Kebun* plots are also dominated by coconut, clove and/or nutmeg but incorporate additional fruiting species (banana *Musa* agg., mango *Mangifera indica*, *Citrus* spp., papaya *Carica papaya* and pineapple *Ananas comosus* in order of quantity produced) and timber trees such as *Dracontomelon dao* (Baden Pusat Statistik 2017b). These are worked rotationally, with a plot typically being used for around five years prior to about 11 years of abandonment, though conversion to coconut-clove plantations after two-three rotations (25-35 years) was anecdotally noted to be becoming more frequent. In a few areas in the central plain, there are large open fields for ground nuts *Arachis hypogaea* and cassava. Two naturally low-lying wet areas produce virtually all rice grown on the island, one north of Beo in the centre east of the island and one at Tarohan in the south east (Baden Pusat Statistik 2017b). Secondary forest, including both native and introduced tree species, and some small patches of retained primary forest are found on the steep and rocky sections that are undesirable as *kebun*, or protected for local religious traditions, though most secondary 'forest' type habitat is overgrown once or twice cleared *kebun* that will be cleared again. Occasionally, it will be several decades before an area is cleared again, depending on the circumstances and desires of the landowner and their descendents (who inherit an equal share of any land).

Talaud has a similar climate to the lowlands of Sangihe, with humidity below 80% between July and October and between 82-85% for the rest of the year, although rainfall is lower with 181 rainy days in 2015 and a total of 1,786 mm of precipitation (Baden Pusat Statistik 2017b). Temperature is marginally higher, with an average maximum of 33.4 degrees Celsius during 2015 (Baden Pusat Statistik 2017b).

## 1.4 Birds of Sangihe and Talaud

In this thesis, I follow the taxonomic checklist of BirdLife International/Handbook of the Birds of the World (del Hoyo et al. 2014, 2016). Table 1-1. Endemic, threatened and endemic subspecies recorded on Sangihe (White & Bruce 1986; Coates et al. 1997; Riley 1997a, 2002b, del Hoyo et al. 2014, 2016).and Table 1-2 list the endemic and threatened species and species represented by endemic subspecies that have previously been recorded on Sangihe and Talaud. Cerulean Paradise-flycatcher *Eutrichomyias rowleyi*, Sangihe Whistler *Coracornis sanghirensis* and Sangihe White-eye *Zosterops nehrkorni* are Critically Endangered on the IUCN Red List (BirdLife International 2017a), along with the recently recognised Sangihe Dwarf-kingfisher (del Hoyo et al. 2014) which has possibly not been recorded since the initial collection of seven specimens prior to 1878 by or on behalf of A.B. Meyer (Blasius 1888; Meyer & Wiglesworth 1898a) and was considered possibly extinct by Riley (2002). In addition, recent research has led to recognition of Sangihe Golden Bulbul *Thapsinillas platenae* as a species separate to the other members of the golden bulbul complex (Collar et al. 2013; del Hoyo et al. 2016), and has also been assessed as Critically Endangered (BirdLife International 2017a). Further taxonomic changes have seen the Sangihe Pitta *Erythropitta caeruleitorques* also recognised as a valid species (Irestedt et al. 2013; del Hoyo et al. 2016), assessed as globally Endangered (BirdLife International 2016).

Five species of bird are now considered to occur solely on the Talaud Islands; Talaud Rail *Gymnocrex talaudensis* (classified as Endangered for the IUCN Red List [BirdLife International 2017a]), Talaud Bush-hen *Amaurornis magnirostris* (VU), Talaud

Kingfisher *Todiramphus enigma* (Near Threatened), Red-and-blue Lory *Eos histrio* (Endangered), and Talaud Pitta *Erythropitta inspeculata* (VU). There are a further eight subspecies restricted to the Talaud Islands, and six more shared only with Sangihe. Of note are the number of large frugivores, with four sympatric parrots (three in the genus *Tanygnathus* plus Golden-mantled Racket-tail *Prioniturus platurus*) and four species of imperial-pigeon, *Ducula*.

The avifauna contains species shared with the Philippines, including three species occurring on Karakelang that are otherwise Philippine endemics: Chocolate Boobook; Blue-naped Parrot; and Southern Rufous Paradise-flycatcher (del Hoyo et al. 2014, 2016). Phylogenetic investigation of the relationships within the *Erythropitta* complex (Irestedt et al. 2013) revealed that Talaud Pitta was considerably more closely related to the Phillipine taxon than to either that occuring in the northern Moluccas or to any of the northern Sulwesi species, including the geographically closest taxon Sangihe Pitta *E. caeruleitorques*. Additionally, the endemic Talaud Bush-hen appears vocally and phenotypically more similar to Plain Bush-hen of the Philippines than to any Indonesian *Amaurornis* (Lambert 1998a; pers. obs.).

There are six subspecies that are shared between the two islands and nowhere else (Table 3), all of which belong to species considered globally Least Concern. Slender-billed Cuckoo-dove *Macropygia amboinensis* has recently been suggested to comprise multiple species based on an analysis of vocal differences, with the form occurring on these islands suggested as belonging to the proposed taxon M. doreya (Ng et al. 2016). However as no recordings from Talaud were included in the analysis the subspecies sanghirensis present on Sangihe and Talaud is retained as a subspecies shared between the two islands (Ng et al. 2016).

## 1.5 Approach

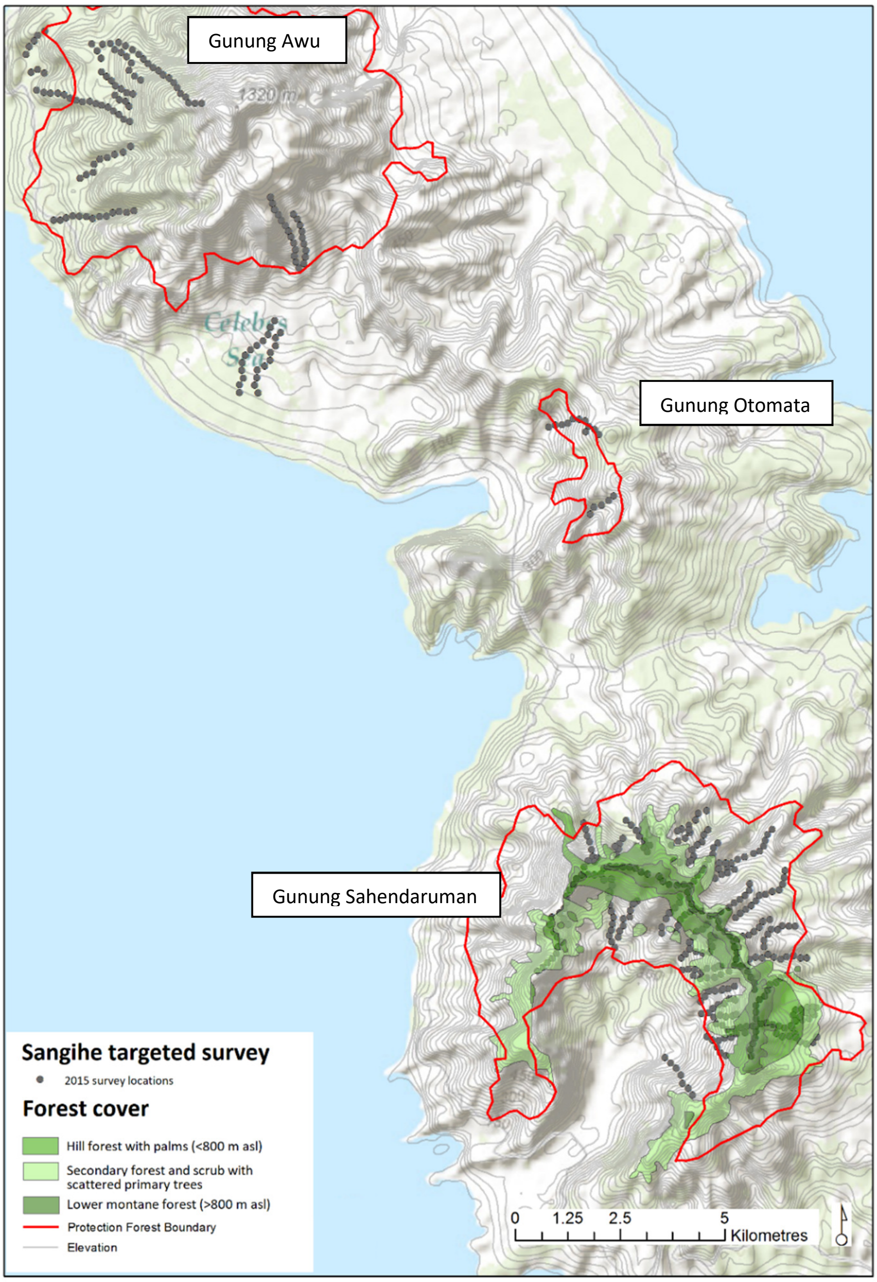


Figure 1-3. The survey approach on Sangihe was to target areas where forest was known or likely to remain in order to increase detections of threatened bird species. The forest cover for Sahendaruman is from Burung Indonesia's habitat classification (Mamengko & Mole 2006b).

Both islands required a rigorous assessment of the presence and abundance of birds coupled with their habitat requirements. On Sangihe (Figure 1-3), the small extent of forested habitat remaining led to the decision to focus on areas identified as containing forest habitat either during previous studies or via satellite imagery. Consequently a targeted survey was carried out that focused on the three areas of

Protection Forest, which hold virtually all forest habitat remaining on the island, to generate records of the Critically Endangered species and to create a detailed picture of the habitat suitability within these areas. An intensive point-count based variable-circular plot survey was used to locate records, with habitat variables recorded at all points. With additional presence locations compiled from monitoring surveys undertaken within the previous six years by Burung Indonesia (Rosyadi 2009; Bashari & Fauzan 2014; Fauzan & Bashari 2016), habitat-association models were created to identify areas that are similar to occurrence points. These represent the priority restoration sites to target to stabilise and recover these populations.

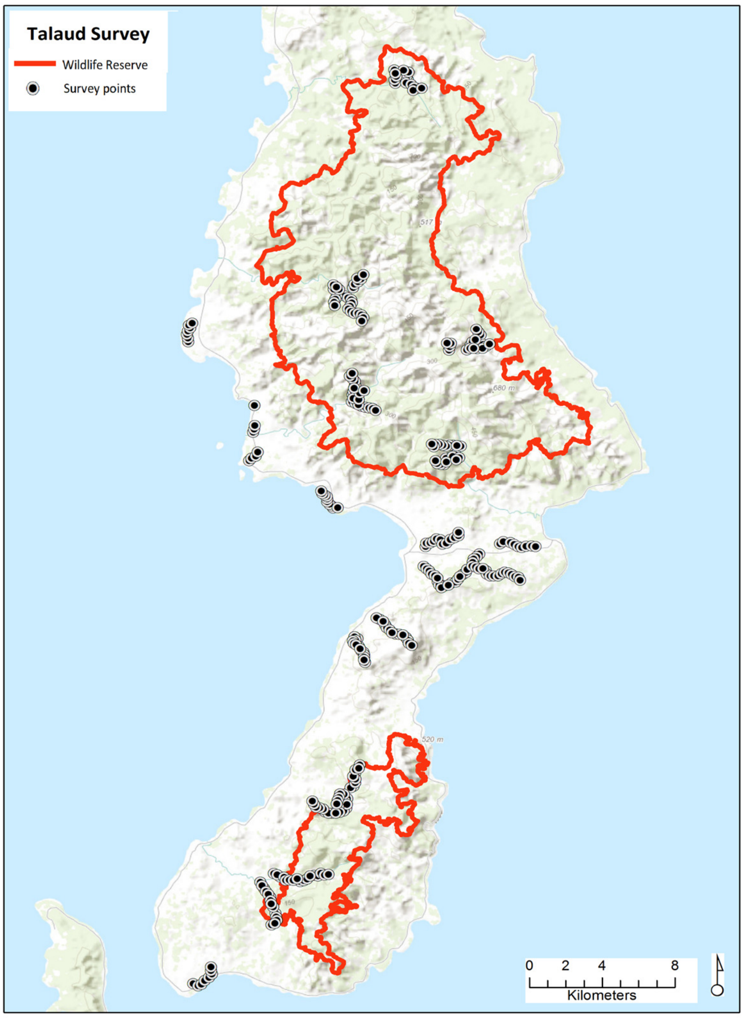


Figure 1-4. Survey approach on the island of Karakelang, Talaud, was based on sampling at representative locations across the whole island, both inside the Wildlife Reserves (which hold the majority of primary forest) and outside where a greater proportion of non-timber mixed agro-forestry plantations occur.

On Karakelang, Talaud, the intent was to quantify the impact of habitat degradation on individual species within the entire bird community by sampling across the habitat modification gradient present on the island. In common with Sangihe, a point-count based variable-circular plot survey was used but here density estimates were calculated for bird species for groups of survey points ranked along an axis of modification, derived from habitat data collected at each point. Overall population estimates were calculated using density estimates from three habitat types (corresponding to an existing land-use map of the island), and these estimates in conjunction with the density responses were used to assess the likelihood of future extinctions due to habitat modification on the island.

## 1.6 Thesis outline

Chapter 2 details the research into identifying restoration habitat for Critically Endangered birds on Sangihe, using a bird presence/absence habitat association modelling approach. Chapter 3 reports the investigations into species abundance responses to habitat modification, population sizes and relative risks of extinction within the bird community on the island of Karakelang, Talaud.

## 1.7 Tables

Table 1-1. Endemic, threatened and endemic subspecies recorded on Sangihe (White & Bruce 1986; Coates et al. 1997; Riley 1997a, 2002b, del Hoyo et al. 2014, 2016).

E, Endemic to Sangihe. ExS, endemic subspecies now considered extinct. SS, subspecies endemic to Sangihe. RR, restricted range species (Stattersfield et al. 1998); EN, Endangered; VU, Vulnerable; NT, Near Threatened; LC, Least Concern (BirdLife International 2017).

| Scientific name | English name | Distribution | Status |
|---|---|---|---|
| *Caloenas nicobaricus nicobaricus* | Nicobar Pigeon | | NT |
| *Ducula concinna* | Blue-tailed Imperial-pigeon | RR | LC |
| *Otus collari* | Sangihe Scops-owl | E | LC |
| *Ceyx sangirensis* | Sangihe Dwarf-kingfisher | E | CR |
| *Cittura sanghirensis* | Sangihe Lilac Kingfisher | E | NT |
| *Eos histrio histrio* | Red-and-blue Lory | ExS | EN |
| *Loriculus catamene* | Sangihe Hanging-parrot | E | NT |
| *Erythropitta caeruleitorques* | Sangihe Pitta | E | EN |
| *Oriolus chinensis sangirensis* | Black-naped Oriole | SS | LC |
| *Coracornis sanghirensis* | Sangihe Whistler | E | CR |
| *Edolisoma morio salvadorii* | Sulawesi Cicadabird | SS | LC |
| *Eutrichomyias rowleyi* | Cerulean Paradise-flycatcher | E | CR |
| *Thapsinillas platenae* | Sangihe Golden Bulbul | E | CR |
| *Zosterops nehrkorni* | Sangihe White-eye | E | CR |
| *Dicaeum celebicum sanghirense* | Grey-sided Flowerpecker | SS | LC |
| *Anthreptes malacensis heliosius* | Brown-throated Sunbird | SS | LC |
| *Leptocoma sericea sangirensis* | Black Sunbird | SS | LC |
| *Aethopyga duyvenbodei* | Elegant Sunbird | E | EN |

Table 1-2. Endemic and threatened species, and endemic subspecies recorded on the Talaud Islands (White & Bruce 1986; Coates et al. 1997; Riley 1997a, 2003; King 2002; del Hoyo et al. 2014, 2016).

E, Endemic to the Talaud Islands. RR (E), restricted range (now only found on Talaud). SS, subspecies endemic to Talaud. ST, Subspecies restricted to Talaud and Sangihe. RR, restricted range species (Stattersfield et al. 1998); EN, Endangered; VU, Vulnerable; NT, Near Threatened; LC, Least Concern (BirdLife International 2017a).

| Scientific name | English name | Distribution | Status |
|---|---|---|---|
| *Caloenas nicobaricus nicobaricus* | Nicobar Pigeon | | NT |
| *Ducula concinna* | Blue-tailed Imperial-pigeon | RR | LC |
| *Ducula pickeringii palmasensis* | Grey Imperial Pigeon | | VU |
| *Gymnocrex talaudensis* | Talaud Rail | E | EN |
| *Amaurornis magnirostris* | Talaud Bush-hen | E | VU |
| *Ninox randi* | Chocolate Boobook | | NT |
| *Todiramphus enigma* | Talaud Kingfisher | E | NT |
| *Eos histrio talautensis* | Red-and-blue Lory | RR (E) | EN |

| | | | |
|---|---|---|---|
| *Prioniturus platurus talautensis* | Golden-mantled Racquet-tail | SS | LC |
| *Tanygnathus lucionensis talautensis* | Blue-naped Parrot | SS | NT |
| *Erythropitta inspeculata* | Talaud Pitta | E | VU |
| *Oriolus chinensis melanisticus* | Black-naped Oriole | SS | LC |
| *Edolisoma morio talautensis* | Sulawesi Cicadabird | SS | LC |
| *Terpsiphone cinnamomea talautensis* | Southern Rufous Paradise-flycatcher | SS | LC |
| *Zosterops everetti babelo* | Everett’s White-eye | SS | LC |
| *Dicaeum celebicum talautense* | Grey-sided Flowerpecker | SS | LC |
| *Leptocoma aspasia talautensis* | Black Sunbird | SS | LC |

Table 1-3. Subspecies endemic to both Sangihe and Talaud. (White & Bruce 1986; Riley 1997a, 2002b, del Hoyo et al. 2014, 2016). All are globally Least Concern (BirdLife International 2017a).

| **Scientific name** | **English name** |
|---|---|
| *Megapodius cumingii sanghirensis* | Philippine Scrubfowl |
| *Macropygia ambionensis sanghirensis* | Slender-billed Cuckoo-dove |
| *Treron griseicauda sangirensis* | Grey-cheeked Green-pigeon |
| *Ducula aenea intermedia* | Green Imperial Pigeon |
| *Tanygnathus sumatranus sangirensis* | Blue-backed Parrot |
| *Aplonis panayensis sanghirensis* | Asian Glossy Starling |

# Chapter 2: Identification of restoration habitat for the critically endangered birds of Sangihe

## Abstract

Where populations of a species have been reduced to critical levels through habitat loss it is essential to extend the area of occupancy through targeted restoration. For a set of three Critically Endangered birds on the island of Sangihe, I created habitat suitability models based on presence-absence data to identify those areas away from the current occupied range of the species that are most similar. For Cerulean Paradise-flycatcher, three small areas on the flanks of Mount Awu are similar to occupied sites, raising the possibility that a separate population could be established in a second location. However, for this species, Sangihe Whistler and Sangihe Golden Bulbul the Sahendaruman crater remains of primary importance and there are multiple locations that may be restored with relatively little expenditure. Without restoration, the slow retreat of each species appears likely to continue, with Cerulean Paradise-flycatcher at greatest risk of global extinction.

## 2.1 Introduction

To date, islands have suffered far more bird extinctions than mainland areas, driven primarily by the devastating impact of invasive species on naiive fauna (Loehle & Eschenbach 2012; McCreless et al. 2016). Considerable success in preventing extinction on islands has occurred where invasive predators or competitors have been the principle driver (e.g. Jones & Merton 2012), but examples of successful habitat restoration to counter anticipated deterministic extinctions following habitat loss (Kuussaari et al. 2009; Wearn et al. 2012) are few to date, possibly due to the urgency in addressing impacts of invasive species and unsustainable hunting: conservation biology is a crisis discipline (Soulé 1985). There now appears to be a shift in the predominant threat to birds towards habitat loss and associated impacts beginning to cause extinctions, even in mainland areas (Szabo et al. 2012; Lees & Pimm 2015). As both responses to preventing the extinction of species threatened by invasive species have improved, allowing the rescue of species from imperiled situations (e.g. Jones & Merton 2012), and the most severely affected species have become extinct (Steadman 2006; VanderWerf et al. 2006), the next wave of extinctions are anticipated to be those currently in the relaxation phase of an extinction debt following habitat loss (Kuussaari et al. 2009; McCreless et al. 2016).

Where habitat restoration is carried out in conjunction with the removal of damaging invasive species, recovery of the target population can be greatly enhanced. The restoration of suitable habitat for the Azores Bullfinch and subsequent rapid population recovery was in part achieved through removal of invasive plant species, but the replanting of large areas of plantation habitat with native vegetation enabled recovery to a higher, secure, population level (Monticelli et al. 2010). The species was downlisted from Critically Endangered to Endangered in 2010, and to Vulnerable in 2016 (BirdLife International 2017a). Recently targeted efforts for the conservation of Taita Apalis *Apalis fuscigularis* have begun small-scale restoration within degraded habitat adjacent to existing populations (Borghesio et al. 2015).

The island of Sangihe, Indonesia, is a global priority for bird conservation as the home of a highly endangered and evolutionarily distinct avifauna (Stattersfield et al.

1998; Alliance for Zero Extinction 2010). More than 99% of original forest is estimated to have been converted to agriculture or lost through natural disasters over the past few centuries (Riley 2002b). Ongoing small scale conversion of forest is continuing within the remaining habitat (Fauzan & Bashari 2016). Ten bird species are endemic to the 737 km$^2$ island, of which five appear tolerant of some level of habitat degradation and conversion (Riley 2002a), while the remaining five are Critically Endangered (Table 1-1. Endemic, threatened and endemic subspecies recorded on Sangihe (White & Bruce 1986; Coates et al. 1997; Riley 1997a, 2002b, del Hoyo et al. 2014, 2016). [Chapter 1]). Sangihe Dwarf-kingfisher (del Hoyo et al. 2014) may not have been recorded since the initial collection of around 7 specimens prior to 1879 on behalf of A.B. Meyer (Meyer 1879; Blasius 1888; Meyer & Wiglesworth 1898a) and was considered possibly extinct by J. Riley (Riley 2002b). The remaining four species have all been recorded during the 21st century, but Sangihe White-eye on fewer than five occassions (Rasmussen et al. 2000; Rosyadi 2009; Bashari & Fauzan 2014). For these, the tiny current population size and potential range coupled with ongoing habitat loss indicates that they are at very high risk of extinction, and are incapable of using the mixed plantation agriculture that has replaced primary forest (BirdLife International 2017b). This appears to be a suite of species within that relaxation phase of extinction debt, and finding locations to extend their highly-restricted range is a conservation priority.

Since the early 20th century, none of these species have been definitively recorded away from a highly restricted area of remnant forest present around the Sahendaruman crater in the south of Sangihe Besar (Whitten et al. 1987; Riley 2002b; Mamengko 2006; Rosyadi 2009; Bashari & Fauzan 2014). Indeed, Cerulean Paradise-flycatcher was considered most likely to be extinct (Whitten et al. 1987) until its rediscovery in deep valleys below the ridgetop forest in 1998 (Riley & Wardill 2001; Whitten 2006).

No suggestion of impacts of invasive species has been suggested for these species; their assessments of endangerment are based upon their small estimated population sizes and highly-restricted ranges (BirdLife International 2017a).

### *2.1.1 Habitat associations*

Collecting appropriate and informative predictor variables that allow the creation of an informative habitat association model requires as much information as possible on the subjects (Burnham & Anderson 2002). Despite their CR status however, these species have received little study and were known from very few specimens and observations until relatively recently (Boles, W. & Sharpe 2017; del Hoyo, J., Collar, N. & Kirwan 2017). Slightly more information has been published on the Paradise-flycatcher (Riley & Wardill 2001; Bashari & Fauzan 2014). The following is a summary for these three species.

#### *2.1.1.1 Cerulean Paradise-flycatcher*

Observations of Cerulean Paradise-flycatcher following its rediscovery suggest that it is capable of utilising regenerated secondary habitat and old secondary scrub (Riley & Wardill 2001; BirdLife International 2017b), however this is always close to and/or connected to areas of primary forest (Riley & Wardill 2001). Pairs have been observed in old regenerating secondary forest following shifting cultivation (c. 40-50 years after cultivation) and in secondary scrub dominated by *Piper aduncum* and a species of Acanthaceae known as 'kupamanuk' or Kupamanu at the forest edge to the lower altitudinal range of the forest (BirdLife International 2001). Virtually all records are from the deep valleys with flowing streams between 475 m and 650 m where there is continuous broadleaf forest cover connected to the ridgetop and a canopy approximately 20 m high (Riley & Wardill 2001). There are many emergents and a sparse mid-storey but a dense herb layer and is considerably different to the ridgetop forest, with elements of the mid- to upper-storey including the trees *Myristica* cf. *fatua*, *Ficus* cf. *minhassae*, *Saurauia* spp., *Gironniera* spp. and *Syzygium* spp., and the palm *Arenga* cf. *pinnata*.(Riley & Wardill 2001).

The species predominately inhabits the subcanopy and lower storeys of this habitat (BirdLife International 2001; Riley & Wardill 2001), foraging on invertebrates gleaned from vegetation or even from the ground, or occasionally through short flycatching sallies from perches (BirdLife International 2001). Observations of the species in mixed species foraging flocks with Sulawesi Cicadabird *Coracina morio* and Sangihe Whistler occurred rarely, but a frequent association with the squirrel

*Prosciurillus* cf. *murinus* has been noted (Riley & Wardill 2001). It has been suggested that this species of squirrel may be introduced to the island, though this seems unlikely given the abundance in primary forest and apparent foraging relationship with the endemic flycatcher (Riley 2002b). However earlier researchers falied to find it on the island, but then they also failed to find the large, primary forest restricted terrestrial squirrel *Rubrisciurus rubriventer* (Riley 2002b) and no specimens of squirrels other than *Prosciurillus rosenbergi* have been located (Musser et al. 2010).

#### *2.1.1.2 Sangihe Whistler*

Known to currently occur only on the slopes of Mount Sahendaruman, the type specimens of the species colected in 1881 were lablled 'Petta', which is a village on the slopes of Gunung Awu providing circumstantial evidence that the species did previous occur away from Sahendaruman (Rozendaal & Lambert 1999). All recent records have been from the latter and have come from between *c.*575 m and 1,000 m above sea level with an apparent preference for ridgetops, although due to the exceptionally steep terrain these are disproportionately sampled in previous investigations (Lambert 1997; Rozendaal & Lambert 1999; Riley 2002a). Almost all of the habitat is primary forest, but it has been recorded in mature secondary forest over 30 years old (Riley 2002a). Rozendaal & Lambert (1999) reported that the whistler was observed in the subcanopy of trees on a boulder-strewn slope where the understorey was dominated by the very large gingers (Zingiberaceae) and in an area with a high density of large *Pandanus*, considered likely to have been subject to a fairly recent landslip.

#### *2.1.1.3 Sangihe Golden Bulbul*

Unlike virtually all other members of the golden bulbul group, with which it was previously included as a single species (Collar et al. 2013), this species is infrequently recorded and apparently highly-restricted in occurrence (Riley 1997a, 2002a; Rosyadi 2009). All records appear to be from primary forest within the higher elevation primary forest around the Sahendaruman crater, with one record adjacent to secondary forest (Riley 2002a; Rosyadi 2009). Even within this small area of habitat it appears further restricted and was only regularly recorded from three locations, despite other areas appearing suitable (Riley 2002a). It was noted that there is considerable seasonal variation in the intensity of vocalisations with the species detected easily in February 1999 (Riley 2002b). However, it was reported to be common in secondary woodland and mixed tree crop plantations during a visit in May 1986 (K.D. Bishop in Riley 1997). It seems possible that this species has become much rarer during this period. Closely related species are common in secondary and modified habitats on other island groups (Riley 2002b).

### *2.1.2 Aims*

- To model the habitat requirements of the Critically Endangered birds of Sangihe
- To quantify the area of current suitable habitat for each critically endangered species
- To identify candidate restoration areas across the island.

## 2.2 Methodology

### 2.2.1 Locations

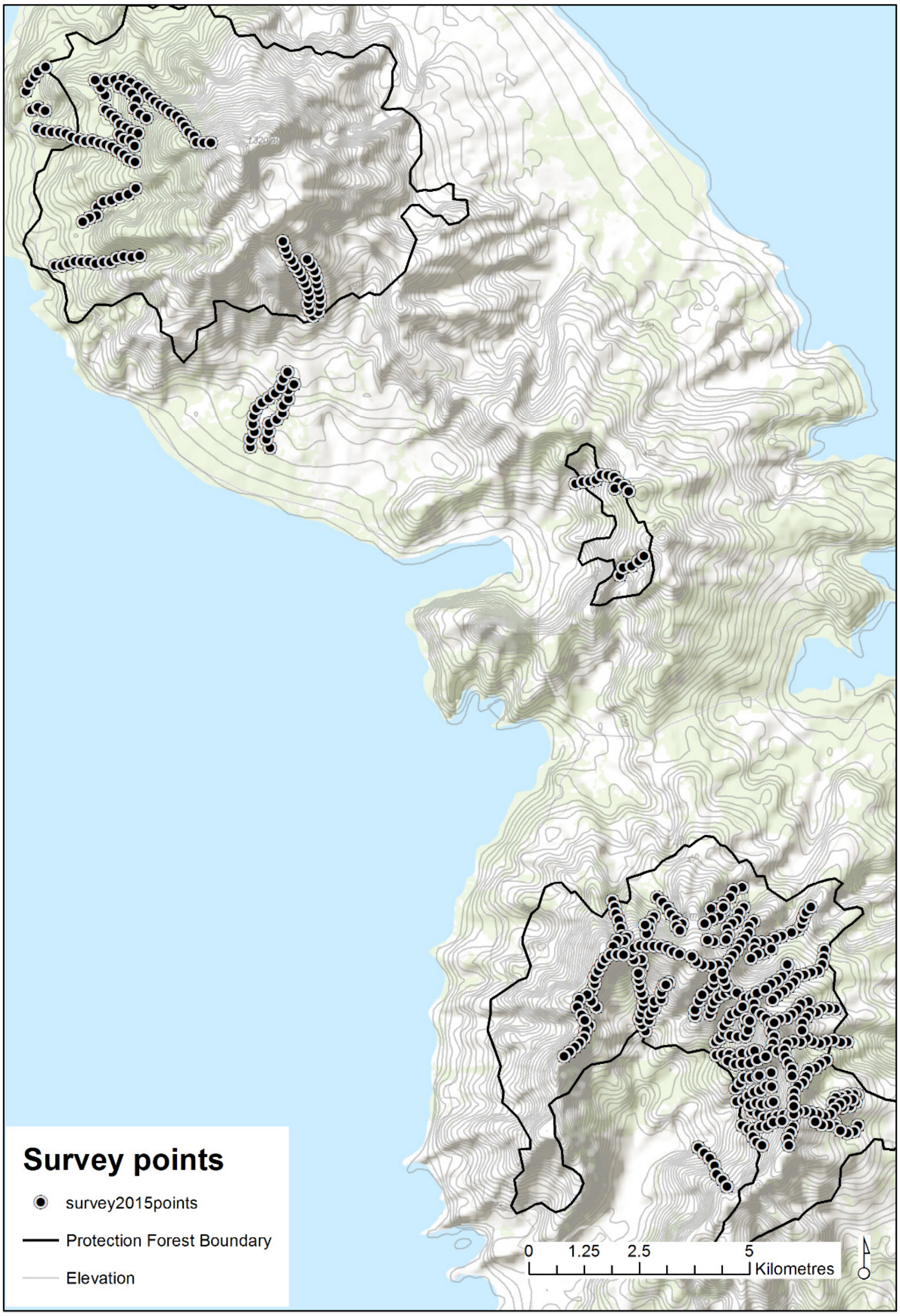


Figure 2-1. Location of survey points on Sangihe.

Detailed knowledge of the current location of remnant forest, secondary and regrowth areas from previous surveys by Burung Indonesia, combined with aerial imagery from GoogleEarth and Bing maps enabled the identification of the full extent of potentially suitable forest habitat. The very high rate of historical habitat conversion combined with volcanic activity has resulted in little original habitat remaining on the island (Whitten et al. 1987; Riley & Wardill 2001). Therefore, the intent of the survey was to establish a network of relatively closely spaced sampling points across the extent of potential habitat, which is concentrated in three main areas: Gunung Awu, Gunung Otomata and the Sahendaruman crater (Figure 2-1). The latter is the only location with recent records of any of the Critically Endangered species and as such was the focus of the survey effort with 42 transects located around the crater. Thirteen transects were located along the western flanks of Gunung Awu. Almost all of the area of the eastern flank of Awu has been converted to coconut plantations or remains bare from the effects of the last two volcanic eruptions in 1996 and 2008 (Global Volcanism Program 2013). Gunung Otomata posseses mature secondary forest patches within the boundaries of the Protection Forest and two transects were completed.

Table 2-1. Sangihe: locations surveyed, dates visited and numbers of transects and point stations

| Region | Access point | Site location (UTM) | Dates (2015) | Number of transects | Number of points |
|---|---|---|---|---|---|
| Sahendaruman | **Kentuhang** | 51N 38752N 78103E | 10-13 Feb | 4 | 39 |
| | **Kentuhang upper** | 51N 38873N 78040E | 5 May | 2 | 16 |
| | **Ulu Peliang** | 51N 38753N 78247E | 18-21 Feb, 5, 18 Mar, 28-29 Apr | 10 | 69 |
| | **Lelipang** | 51N 38593N 78243E | 25-28 Feb, 6 Mar, 7 May | 7 | 47 |
| | **Kuma** | 51N 39280N 78386E | 13-14 Mar 2015 | 4 | 29 |
| | **Kulur** | 51N 39115N 78509E | 8-9 Apr | 4 | 35 |
| | **Bukide** | 51N 38848N 78632E | 11-12 Apr | 4 | 33 |
| | **Malamenggu** | 51N 38675N 78606E | 26-27 Mar | 4 | 31 |
| | **Hiung** | 51N 39231N 78064E | 10-11 Mar | 3 | 23 |
| Gunung Awu | **Anggriss** | 51N 40256N 77146E | 21-22 Apr | 4 | 44 |
| | **Talawid** | 51N 41255N 76660E | 15-17 Apr | 5 | 40 |

|  |  |  |  |  |  |
|---|---|---|---|---|---|
|  | **Maselihe** | 51N 40539N 76778E | 18 Apr | 2 | 22 |
|  | **Kendahe** | 51N 40833N 76692E | 20 Apr | 2 | 24 |
| Gunung Otomata | **Tahuna** | 51N 39889N 77824E | 10 May | 2 | 16 |
|  |  |  | **Total** | **57** | **468** |

### *2.2.2 Bird presence data*

A point count methodology (Bibby et al. 1998) was used to survey the endemic birds of Sangihe, supplemented by additional encounters between points to provide a dataset of occupied and unoccupied locations.  Each point count lasted 5 minutes and all were audio-recorded for confirmation of uncertain identification and for documentation.  A point was included as a presence if that species was recorded during the point, or recorded within a 50-metre radius of that point.  Transects routes were planned to evenly sample at a high resolution as much of the remaining habitat within the Sahendaruman Crater as possible given the extreme terrain. Point counts were established at intervals of 150 m along the transects, which were placed to access each valley and ridge, and to incorporate sections of ridge slope.

Data was also obtained from previous surveys of Sahendaruman completed in 2009 and 2014 (Rosyadi 2009; Bashari & Fauzan 2014).  Records falling within a 50-m radius of a point count location (Euclidean distance calculated in ArcGIS 10 (ESRI 2011)) were considered to represent presence at that point.  Table 2-2. Additional presence points from survey of Rosyadi (Rosyadi 2009) and 2014 (Burung Indonesia unpub. data) included for modelling the associations of the CR species. lists the additional presence points for the three species for which sufficient data could be obtained.  The compilation of records from multiple years was done in order to increase sample size for these very rare species, an approach also taken by de Lima *et al.* (2016).  Caution is required for the purposes of extracting habitat association information from datasets including records from surveys taking place prior to the measurement of predictor variables as changes to the habitat subsequently (which may have lead to the lack of a current record) would expand the potential suitable range of each species.  As the present study's aims are to predict the relative suitability of areas which are known to not support the species at present, using these records to represent a reasonable maximum presence dataset was considered appropriate. However, while each bird species is more than capable of traveling distances greater than the 150 m between survey locations, assuming points adjacent to presence points as pseudo-presences was not appropriate (cf. Bracebridge et al. 2011), as fine scale habitat associations were the desired outcome and considerable, abrupt habitat changes are frequent due to small-scale clearance.

### *2.2.3 Sangihe habitat variables*

Previous work outlining the habitat associations of the CR species (summarised above) indicated several potential predictor variables that appear connected to their occurrence, but as these were largely anecdotal a more structured approach to habitat assessment was undertaken following the approach of Lee and Marsden (Lee & Marsden 2008b). Variables were classified as physical geography, vegetation structure or floristic and aimed to capture variables that would characterise a reasonable proportion of each of the target species niche space while minimising colinearity (Lee & Marsden 2008b). Each variable was assessed within a 10-metre radius of the point count location. A restricted subset of each variable class was then identified for each species based on the previously pubished information and personal observations of coincident abundance or use during initial assessments of occupied sites for each species.

### *2.2.4 Physical geography variables*

1; Distance to water was measured when less than 15 metres. Where it was further than this the distance was either estimated in the field, or if the closest water was not obvious it was calculated later using the relative distance from other points with estimated or measured values in ArcGIS 10 (ESRI 2011). Distance to water was log transformed for normality, and was not included in the models for the whistler and golden bulbul as the large distance from the ridgetop rendered the predictor uninformative. 2; An absolute mean slope measure was derived from the five measurements of slope taken at each point, giving an overall ‘steepness’ score for each point regardless of slope direction. 3; Altitude was measured at each point location using a hand-held GPS in the field, and subsequently values were also obtained from a 30m resolution digital elevation model to check consistency.

### *2.2.5 Vegetative structure variables*

1. Canopy cover was directly estimated as a percentage coverage of vegetation above approximately 15-metres height as viewed directly upwards, excluding emergents; 2, Mid-level cover was estimated for the vegetation below the canopy, but greater than five metres above ground level, while; 3, low level cover was an estimated percentage of vegetation cover between one and five metres and; 4, ground cover that estimated to obscure bare ground within a 10-metre radius of the point centre; 5, A canopy height variable derived by taking the mean of tree height for the second to fifth trees as ranked by height, from the estimation of the height of the ten largest trees within a 10-metre radius of the centre of the point. The first tree was excluded to reduce the potential impact of emergents; 6, the mean of the two largest trees derived from the diameter at breast height measured for the ten largest trees greater than 0.1 m in diameter at each point, to differentiate points dominated by one or two very large trees; 7, the proportion of ten trees measured with a diameter at breast height in excess of 0.6 m.

### *2.2.6 Floristic variables*

1-3. An experienced Sangirense guide made identifications of the ten largest trees, exceeding 0.1 m diameter, within a 10-m radius of the point count location. These local name identifications were cross-referenced with names assigned to specimens collected during the Action Sampiri expedition (D. Hicks *unpub. data*), then with Holthus and Lam's (1942) detailed investigation into the flora of Talaud, which shares many of the same or similar names with Sangihe. For those remaining unnamed, further works associating local names with species groups/genera were consulted to produce an idenfitication to at least family level and in most cases genus level. To identify associations of each CR species with tree species random forest modelling using the package 'caret' in R (Kuhn. et al. 2016; R Core Team 2017), with presence of the CR species as the dependent variable and a presence/absence dataset of all trees found at 15 or more points. K-fold cross-validation with ten folds and three repeats was used to evaluate the relative importance of each predictor variable and the top five most 'important' trees were retained for use in subsequent modelling.

A set of potential indicator plants were also scored at each point using an estimated abundance measure (scoring 5 for Dominant [<50% of all vegetation within the 10-m radius], 4 for Abundant [c30-50%], 3 for Frequent [10-30%], 2 for Occasional [c. 5-10%] and 1 for Rare [c.5%]). One set of indicators were compiled into a single 'crop score' variable, derived from the presence and abundance of both tree crops (clove, nutmeg, coconut, banana, durian, mango, langsat) and field crops (cassava, 'bete'). A second set were readily identifiable natively occurring plant 'types'; lianas, ginger, areca palm, tree pandanus, ground pandanus, tree ferns and a flowering high-altitude plant known as 'tompioliu'. These were retained as separate variables, but only included in models for the species for which there was a plausible connection. During the initial site assessments that at known presence locations for Cerulean Paradise-flycatcher trailing lianas from the canopy were observed to be abundant. For the whistler, the presence of 'tompioliu' was noted, along with previously reported associations with tree pandanus and large gingers (Rozendaal & Lambert 1999). The golden-bulbul was also observed in areas containing tompoliu but also higher abundances of areca palms and ground pandanus.

| **Variable** | **Cerulean Paradise Flycatcher** | **Sangihe Whistler** | **Sangihe Golden Bulbul** |
|---|---|---|---|
| **Physical geography** | | | |
| *Distance to water* | x | | |
| *Slope* | x | x | x |
| **Vegetative structure** | | | |
| *Canopy cover* | x | x | x |
| *Mid-level cover* | x | x | x |
| *Low-level cover* | x | x | x |
| *Ground cover* | x | x | x |
| *Canopy height* | x | x | x |
| *Largest trees mean girth* | x | | |
| *Proportion large trees* | | x | x |
| **Floristics** | | | |
| Cropscore | x | | |
| Lianas | x | | |
| Gingers | | x | x |
| Areca | x | x | x |
| Tree pandanus | | x | x |
| Ground pandanus | | | x |
| Tree fern | | | |
| Tompioliu | | x | x |

### *2.2.7 Modelling approach*

Generalised linear models (GLM) with a binomial error distribution and logit link function were used to investigate the relationship between each CR bird species and habitat variables. Models were fitted using the base package in 'R' (R Core Team 2017). None of these species has been recorded away from Sahendaruman in the past 20 years despite targeted search effort. There are previous reports from Gunung Awu (White & Bruce 1986), suggesting that relatively recent extinctions may have occurred, potentially through the impacts of the volcanic eruptions. The current absence of these species may not therefore reflect currently unsuitable habitat, but would be penalised in modelling. Including these as absences for the predictive modelling may overfit the model and reduce the suitability estimate for sites around Gunung Awu, when habitat may be suitable but the species concerned are unable to recolonise due to the present low population size and limited dispersal. Consequently, individual species models were fitted using only data collected around Sahendaruman (n = 327), where it was assumed that individuals of each species could select from the available habitat present. Models were then used to estimate the probability of occurrence at each of the surveyed points across the island (*n* = 468).

The current distribution of the CR species may be relictual given the evidence of previous occurrence on Gunung Awu. As such it is of importance to assess the present similarity (and assumed potential suitability) of sites outside their current, highly restricted, elevational limits. For the prediction to be unconstrained by a minimum elevation a restricted set of points was used from within those sampled at Sahendaruman based on the known elevational range, with a buffer of 40 vertical metres to account for uncertainty.

The 'dredge' function in package 'MuMIn' (Barton 2016) was subsequently used to rank models from the initial candidate set of models, with variables not present within the two AICc units removed and the process repeated for the restricted subset of variables. This final candidate set of models within two AICc units of the best model was used to generate model-averaged values in order to then predict probabilities of occurrence across the full extent of the surveyed area of the island

(Burnham & Anderson 2002). A ‘full average’ of the candidate set was employed, where all predictors are assumed present in all models with a zero parameter estimate attached where a predictor is absent, was employed as this emphasises the predictors that have the strongest effect on the response variable (Grueber et al. 2011). Model accuracy was assessed using the area under the curve (AUC) of the receiver operating characteristic (Fielding & Bell 1997): discriminatory ability was considered good above values of 0.7 (Burnham & Anderson 2002). Model-averaged logistic regression coefficients and both unconditional standard errors and standard errors conditional on the best model were then extracted; the former are considered more accurate estimates of coefficient precision as variance due to model uncertainty is included (Burnham & Anderson 2002). The ‘predicts’ function in the ‘stats’ package of ‘R’ (R Core Team 2017) was then used to obtain predicted probabilities of occurrence for the complete dataset of all 468 points sampled across the island based on the model averaged values. To visualise the results, natural neighbour interpolation was used in ArcGIS 10 (ESRI 2011) to create a probability surface for each species of the area based on weighted averages of probabilities at neighbouring points.

## 2.3 Results

### *2.3.1 Presence records of Critically Endangered species*

Table 2-2. Additional presence points from survey of Rosyadi (Rosyadi 2009) and 2014 (Burung Indonesia unpub. data) included for modelling the associations of the CR species.

| | Additional Point locations | |
|---|---|---|
| Species | 2009 | 2014 |
| Cerulean Paradise-flycatcher | 3 | 4 |
| Sangihe Whistler | 4 | 5 |
| Sangihe Golden Bulbul | 6 | 6 |

Additional point records for the CR species from the previous two surveys totalled seven for the paradise-flycatcher, four for the whistler and nine for the golden bulbul (Table 2-2). Most came from locations adjacent to presence points recorded during the current fieldwork, but there were a few previous records from places several hundred metres from the nearest 2015 records (Appendix 2-1). Notable among these is the 2014 record of Cerulean Paradise-flycatcher from an additional valley to the northwestern end of the recorded range, the only valley in which the species had previously been recorded and not found during the 2015 survey.

### *2.3.2 Cerulean Paradise-flycatcher habitat model*

#### *2.3.2.1 Summary of presence locations*

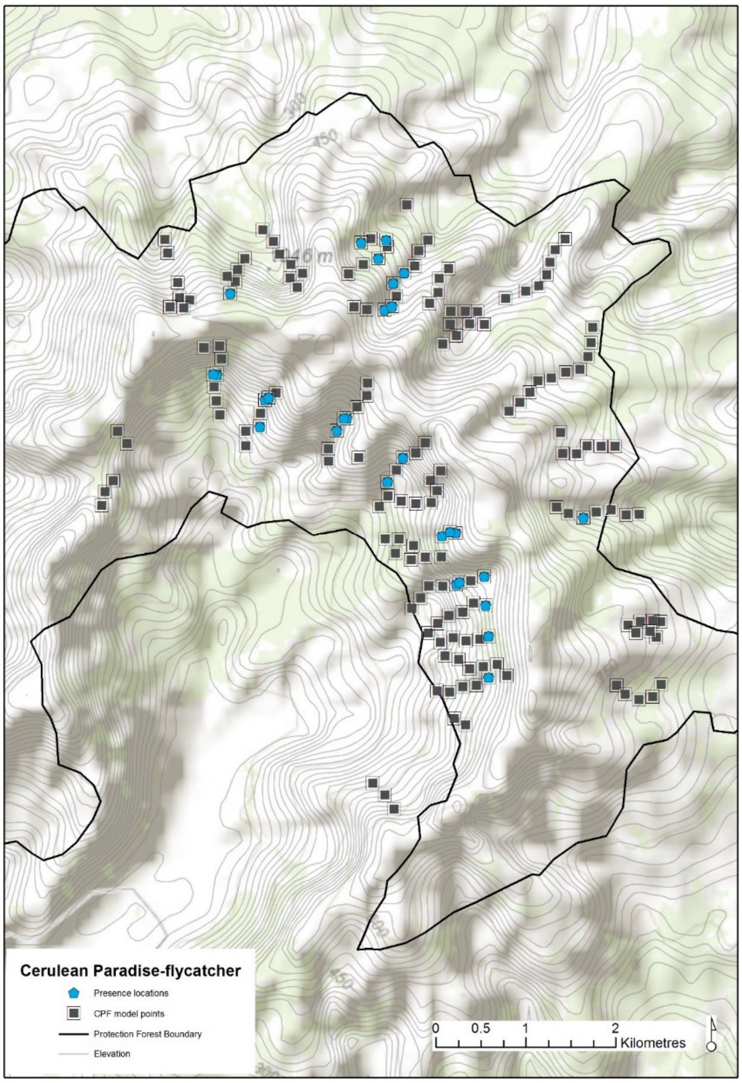


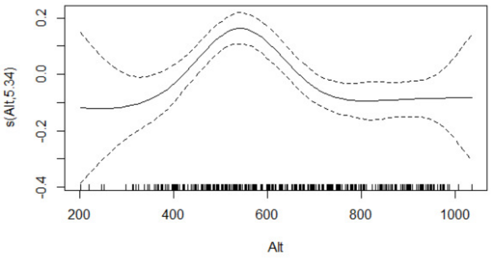
Figure 2-2. Cerulean Paradise-flycatcher presence/absence locations used for habitat suitability modelling..

Figure 2-2 presents the 28 presence locations identified for Cerulean Paradise-flycatcher. Observations were made from a total of 21 point locations during the 2015 survey, from a total of 472 islandwide survey points, with seven presence locations added from the most recent previous surveys (

Table 2-2; six from 2014 [Bashari & Fauzan 2014] and three from 2009 [Rosyadi 2009]). All records came from the Sahendaruman crater. Restricting the modelled dataset to those points within the known elevational range of the species removed to a considerable degree of the effect of altitude on the predictive model (Figure 2-3), consequently this restricted set of points ($n = 181$) was used for modelling the predicted suitability of habitat visited throughout the island, with the caveat that at present altitude is a significant correlate of occurrence for Cerulean Paradise-flycatcher.

### *2.3.2.2 Trees associated with Cerulean Paradise Flycatcher*

All five of the most important trees from the random forest model displayed significant, positive associations with the presence locations of the paradise flycatcher (univariate binomial GLM with log-link; 'Sahai' *Dendrocnide* aff. *amplissima* β = 2.32 ± 0.45, $p < 0.0001$; 'Kuriubabi' *Adinaria cf. celebica* β = 1.3 ± 0.45, $p = 0.004$; 'Lembabeka' *Endocomia macrocoma* β = 1.15 ± 0.42, $p = 0.006$; 'Bunaro' *Scolopia spinosa* β = 1.23 ± 0.49, $p = 0.01$, and 'Engahesi' *Intsia bijuga* β = 1.09 ± 0.45, $p = 0.02$. All were included in the initial full model.



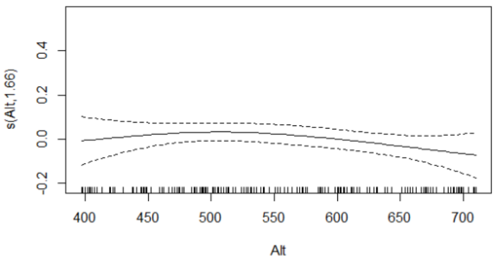


Figure 2-3 (l). Cerulean Paradise-flycatcher elevational response across all points around the Sahendaruman crater. Figure 2-4 (r). Cerulean Paradise-flycatcher elevational response within known current elevational limits.

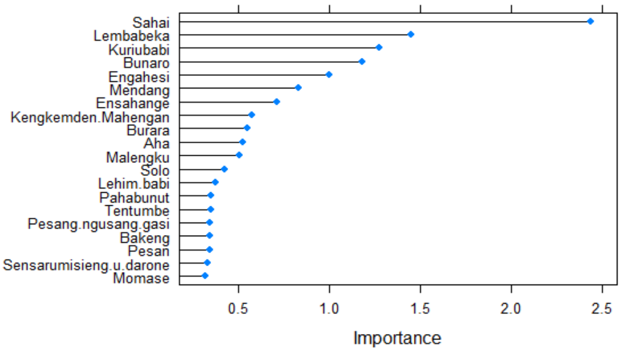


Figure 2-5. Relative importance of tree species as predictor variables from a k-fold cross-validation of a random forest model with presence/absence of Cerulean Paradise-flycatcher as the dependant variable. The top 20 ranked out of a total of 61 tree species are shown.

The best model from the restricted set of variables had an Akaike weight of 0.34, versus 0.28 for the next best model (Table 2-3. Cerulean Paradise-flycatcher: subset of best models used for the the model averaged prediction across the whole island. Models shown are those within 2 AICc units of the 'best' model. All models incorporated the two structural terms canopy cover and mid-level cover, the abundance of lianas and the presence of the trees 'Bunaro' and 'Sahai'. The models differ in the additional tree species included, suggesting considerable uncertainty in the contribution of these towards explaining the flycatcher's occurrence. That the worst of these four models still had an Akaike weight of 0.17 indicates that using the model averaged values across this set was appropriate for assessing the suitability of habitat across the island.

Table 2-3. Cerulean Paradise-flycatcher: subset of best models used for the the model averaged prediction across the whole island. Models shown are those within 2 AICc units of the 'best' model.

| ***#*** | ***model*** | ***R.$^2$*** | ***K*** | ***AICc*** | ***Δ AICc*** | ***weight*** |
|---|---|---|---|---|---|---|
| 1 | *Bunaro + CanopyCover + Engahesi + Lianas + MidCov + Sahai* | 0.35 | 7 | 93.38 | 0 | 0.34 |
| 2 | *Bunaro + CanopyCover + Engahesi + Kuriubabi + Lianas + MidCov + Sahai* | 0.35 | 8 | 93.72 | 0.35 | 0.28 |
| 3 | *Bunaro + CanopyCover + Lianas + MidCov + Sahai* | 0.34 | 6 | 94.31 | 0.93 | 0.21 |
| 4 | *Bunaro + CanopyCover + Kuriubabi + Lianas + MidCov + Sahai* | 0.34 | 7 | 94.7 | 1.32 | 0.17 |

The model-averaged coefficients showed that a strong positive association with high abundance of lianas and the presence of 'Sahai', were the strongest determinants of the presence of Cerulean Paradise-flycatcher, but a positive association with both canopy and mid-storey cover and 'Bunaro' but the high standard errors associated with the other tree species again indicates uncertainty in the influence of these predictors (Table 2-4). All significant predictors had positive coefficients, and the AUC of the ROC for the best model indicated that presence points were correctly classified 94% of the time. This high specificity is evident in the nearest neighbour interpolation of the model-averaged values (Figure 2-7), with virtually all presence points within 0.75 SD of the highest predicted probabilities. The lowest predicted suitability value (untransformed) for a presence location was 0.106, with the total area of habitat across the surveyed part of the island exceeding this value being only 10.17 km$^2$.

Table 2-4. Cerulean Paradise-flycatcher model-averaged coefficients (full average or 'zero method'(Burnham & Anderson 2002)).

| | ***Estimate*** | ***Std. Error*** | ***Adjusted SE*** | ***z value*** | ***Pr(>\|z\|)*** | |
|---|---|---|---|---|---|---|
| *(Intercept)* | -12.597 | 3.162 | 3.183 | 3.96 | <0.0001 | *** |
| *Bunaro* | 1.761 | 0.769 | 0.774 | 2.28 | 0.023 | * |
| *Canopy cover* | 0.053 | 0.024 | 0.024 | 2.18 | 0.029 | * |
| *Engahesi* | 0.730 | 0.783 | 0.786 | 0.93 | 0.35 | |
| *Lianas* | 1.257 | 0.331 | 0.333 | 3.77 | 0.0002 | *** |
| *Mid cover* | 0.063 | 0.031 | 0.031 | 2.04 | 0.042 | * |
| *Sahai* | 2.019 | 0.615 | 0.619 | 3.26 | 0.001 | ** |
| *Kuriubabi* | 0.371 | 0.577 | 0.579 | 0.64 | 0.521 | |

Table 2-5. Cerulean Paradise-flycatcher variable proportionate importance within the averaged models (< Δ2 AICc units of the lowest AICc).

| | ***Bunaro*** | ***Canopy cover*** | ***Lianas*** | ***Mid-level cover*** | ***Sahai*** | ***Engahesi*** | ***Kuriubabi*** |
|---|---|---|---|---|---|---|---|
| *Importance* | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 0.62 | 0.45 |
| *N models* | 4 | 4 | 4 | 4 | 4 | 2 | 2 |

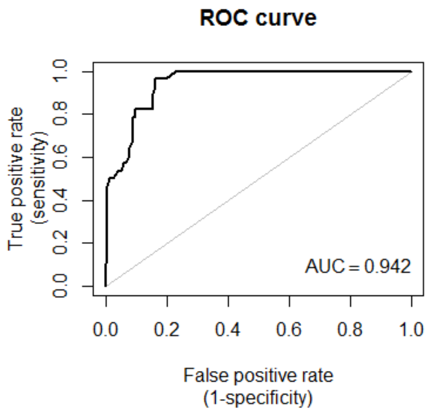


Figure 2-6. ROC of the single top ranked model by AICc for Cerulean Paradise-flycatcher

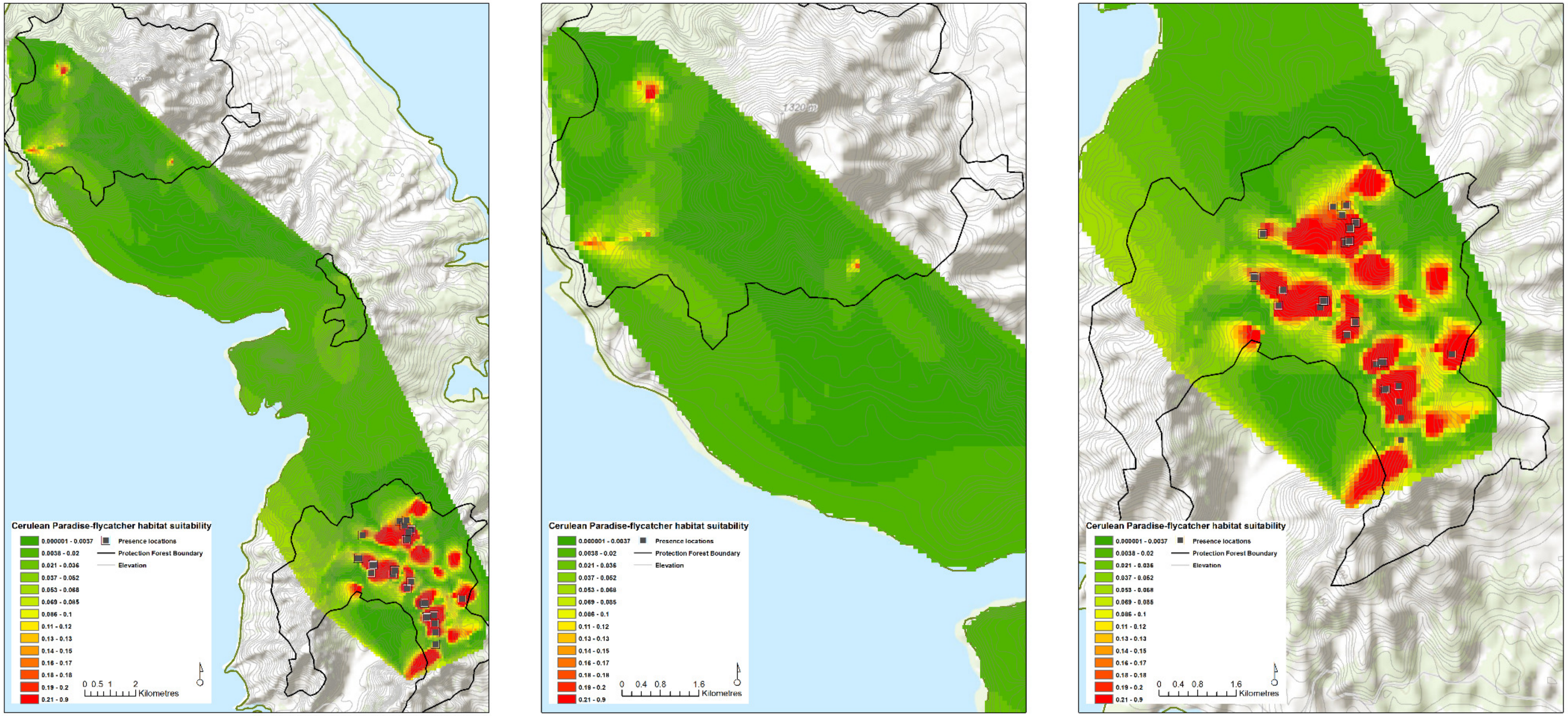


Figure 2-7. Natural neighbour interpolation of Cerulean Paradise-flycatcher predicted occurrence probabilities derived from the model-averaged values from the candidate set of GLM habitat asociation models within 2 AIC units of the best model across the surveyed locations on Sangihe. Red areas equate to the highest probability of occurrence, with a colour scale of ¼ standard deviations of occurrence, and squares represent the location of presence points.

### *2.3.1 Sangihe Whistler habitat model*

#### *2.3.1.1 Summary of presence locations*

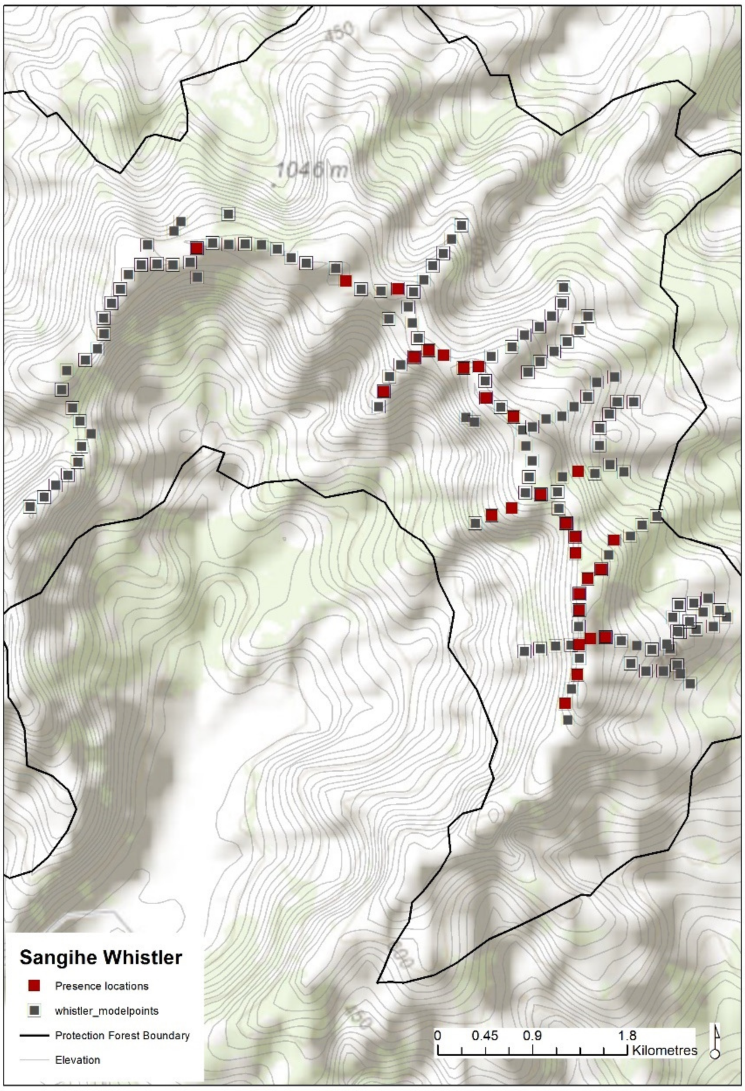


Figure 2-8. Sangihe Whistler presence/absence locations used for habitat suitability modelling.

A total of 28 presence records was obtained for Sangihe Whistler, including five locations added from 2014 data and four from the 2009 survey (Table 2-2; Figure 2-8. Sangihe Whistler presence/absence locations used for habitat suitability modelling.). All records are from the Sahendaruman Crater. Both the northwesternmost record and the most southerly records were derived from one of the two previous surveys (Appendix 2-2). The lowest altititude at which the species was recorded was 705 m above sea level, and consequently models were fitted using only points above 660 m above sea level (*n* = 172).

### *2.3.1.2 Trees associated with Sangihe Whistler*

The top three trees from the random forest model had significant, positive associations with the presence locations of the whistler (Figure 2-5) and were included in the full model (univariate GLM with log-link; 'Peka darone' *Magnolia* cf. *tsiampacca* β = 1.61 ± 0.43, p = 0.0002; 'Lilang katoan' *Schuurmansia henningsii* β = 2.06 ± 0.57, p = 0.0003; 'Lehim babi' *Elaeocarpus teysmannii* β = 1.13 ± 0.44, p = 0.009).

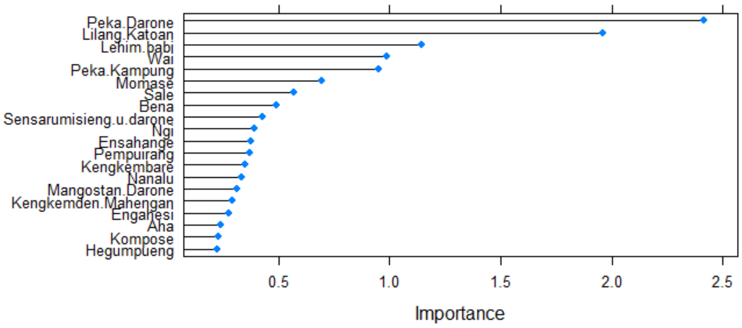


Figure 2-9. Relative importance of tree species as predictor variables from a 10-fold cross-validation of a random forest model with presence/absence of Sangihe Whistler as the dependant variable. The top 20 ranked out of a total of 61 tree species are shown.

The top ranked model from the restricted set of variables differs from the next model only in the inclusion of *Areca* palm abundance and both models have similar Akaike weights (Table 2-6). However, the third ranked model had a weight of less than half that of the top model, suggesting that the top two models were a clear improvement over the remaining three models. Model-averaged coefficients for these variables were positive (Table 2-7), with the largest effect size for the proportion of large trees. While significant, the effect size for canopy cover was very small. The only negative coefficent value was for the abundance of *Areca* palms, though the SE for this predictor was large. Two structural terms, canopy cover and the proportion of large trees were common to all models in the final set, along with the abundance of 'Tompioliu' and presence of the tree 'Lilang katoan', with a second tree species, 'Peka darone', included in four of the five final models (Table 2-8. Sangihe Whistler variable proportionate importance within the averaged models (< Δ2 AICc units of the lowest AICc).). The AUC of the ROC for the best model was 0.88 (Figure 2-10), and the habitat association model demonstrates high specificity: despite altitude not being directly included in the model predictors all areas considered suitable are restricted to the ridgetop and immediate surroundings (Figure 2-11. Nearest Neighbour Interpolation of the predicted occurrence probabilities of Sangihe Whistler derived from the model-averaged values from the candidate set of GLM habitat asociation models within 2 AIC units of the best model across the three surveyed locations on Sangihe. Red colours represent the highest probability of occurrence, with a colour scale of ¼ standard deviations in probabilities of Sangihe Whistler presence at each point., likely due to colinearity of predictors with elevation. The lowest untransformed probability value for a presence point is 0.124, and the total area predicted to contain habitat above this value is only 6 $km^2$, a considerably smaller range than the other two species.

Table 2-6. Sangihe Whistler models, restricted elevation (n = 171), within ΔAIC*c* 2 of the best model.

| | ***Model*** | ***R.2*** | ***K*** | ***AICc*** | ***ΔAICc*** | ***weight*** |
|---|---|---|---|---|---|---|
| 1 | *Areca + CanopyCov + Lilang + PekaDarone + ProportionLargeTrees + Tompoliu* | 0.27 | 7 | 112.59 | 0 | 0.32 |
| 2 | *CanopyCov + Lilang + PekaDarone + ProportionLargeTrees + Tompoliu* | 0.26 | 6 | 112.92 | 0.33 | 0.27 |
| 3 | *CanopyCov + Lehimbabi + Lilang + PekaDarone + ProportionLargeTrees + Tompoliu* | 0.27 | 7 | 114.06 | 1.47 | 0.15 |
| 4 | *Areca + CanopyCov + Lehimbabi + Lilang + PekaDarone + ProportionLargeTrees + Tompoliu* | 0.28 | 8 | 114.16 | 1.57 | 0.14 |
| 5 | *Areca + CanopyCov + Lilang + ProportionLargeTrees + Tompoliu* | 0.26 | 6 | 114.56 | 1.97 | 0.12 |

Table 2-7. Sangihe Whistler model-averaged coefficients (full average or 'zero method'(Burnham & Anderson 2002)).

| | ***Estimate*** | ***Std. Error*** | ***Adjusted SE*** | ***z value*** | ***Pr(>\|z\|)*** | |
|---|---|---|---|---|---|---|
| *(Intercept)* | -6.583 | 1.523 | 1.533 | 4.29 | <0.0001 | *** |
| *Areca* | -0.472 | 0.303 | 0.306 | 1.54 | 0.123 | |
| *Canopy cover* | 0.061 | 0.023 | 0.023 | 2.59 | 0.010 | ** |
| *Lilang* | 1.838 | 0.761 | 0.766 | 2.40 | 0.016 | * |
| *PekaDarone* | 1.087 | 0.540 | 0.544 | 2.00 | 0.046 | * |

| | | | | | | |
|---|---|---|---|---|---|---|
| *Proportion large trees* | 3.042 | 1.477 | 1.487 | 2.05 | 0.041 | * |
| *Tompoliu* | 0.827 | 0.267 | 0.269 | 3.08 | 0.002 | ** |
| *Lehimbabi* | 0.512 | 0.560 | 0.564 | 0.91 | 0.364 | |

Table 2-8. Sangihe Whistler variable proportionate importance within the averaged models (< Δ2 AICc units of the lowest AICc).

| | Canopy cov | Lilang | Prop. lrg. trees | Tompoliu | Peka | Areca | Lehimbabi |
|---|---|---|---|---|---|---|---|
| Importance | 1.00 | 1.00 | 1.00 | 1.00 | 0.88 | 0.58 | 0.30 |
| N models | 5 | 5 | 5 | 5 | 4 | 2 | 2 |

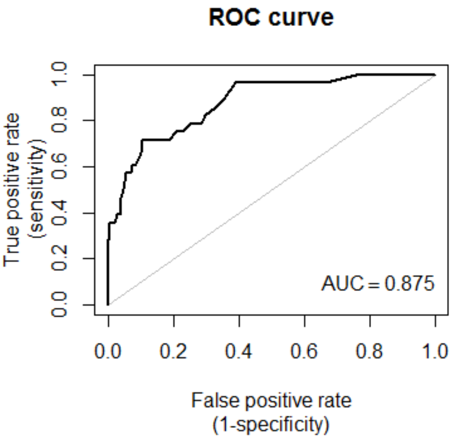


Figure 2-10. ROC of the single top ranked model by AICc for Sangihe Whistler.

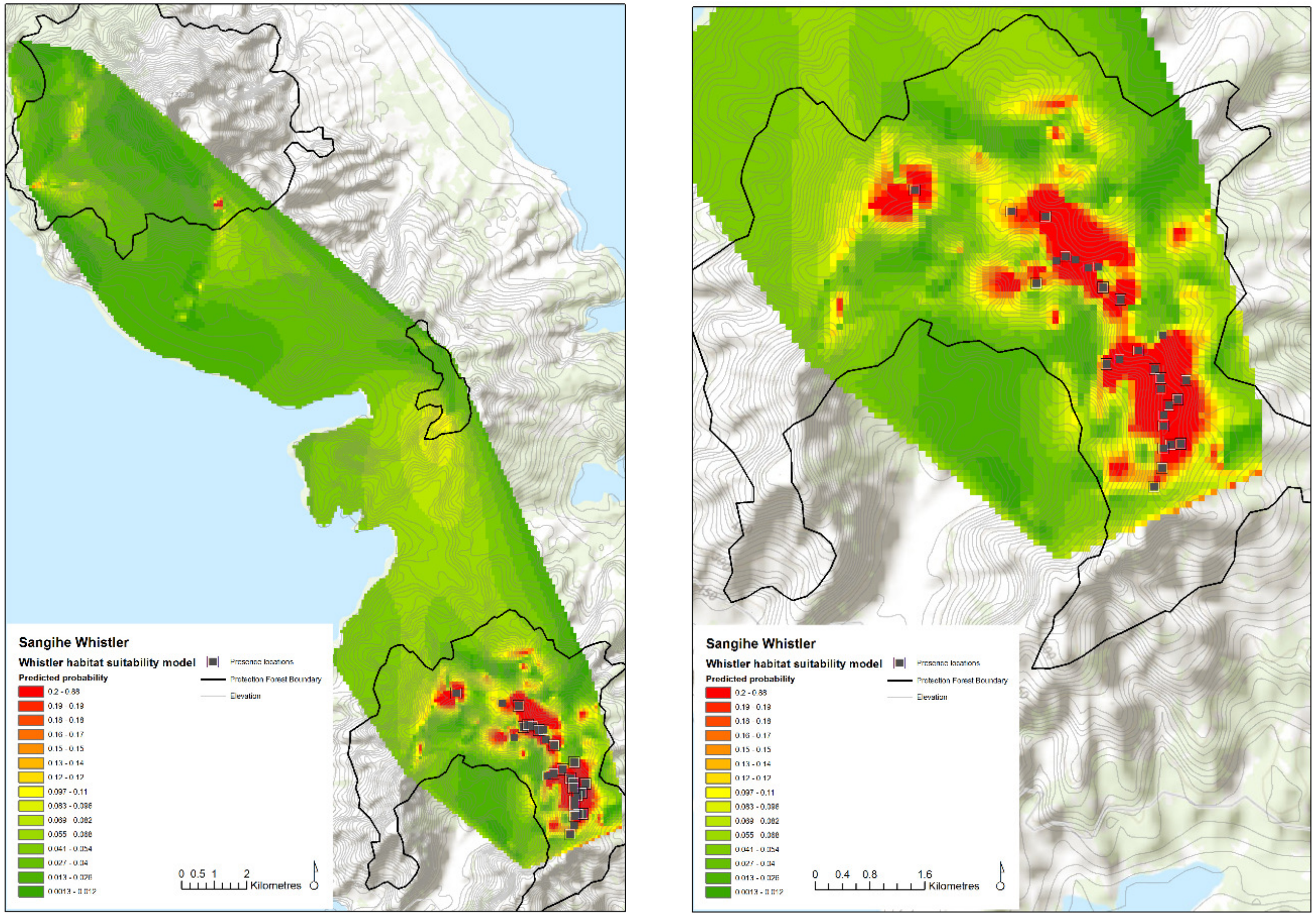


Figure 2-11. Nearest Neighbour Interpolation of the predicted occurrence probabilities of Sangihe Whistler derived from the model-averaged values from the candidate set of GLM habitat asociation models within 2 AIC units of the best model across the three surveyed locations on Sangihe. Red colours represent the highest probability of occurrence, with a colour scale of ¼ standard deviations in probabilities of Sangihe Whistler presence at each point.

### *2.3.1 Sangihe Golden Bulbul habitat suitability*

#### *2.3.1.1 Summary of presence locations*

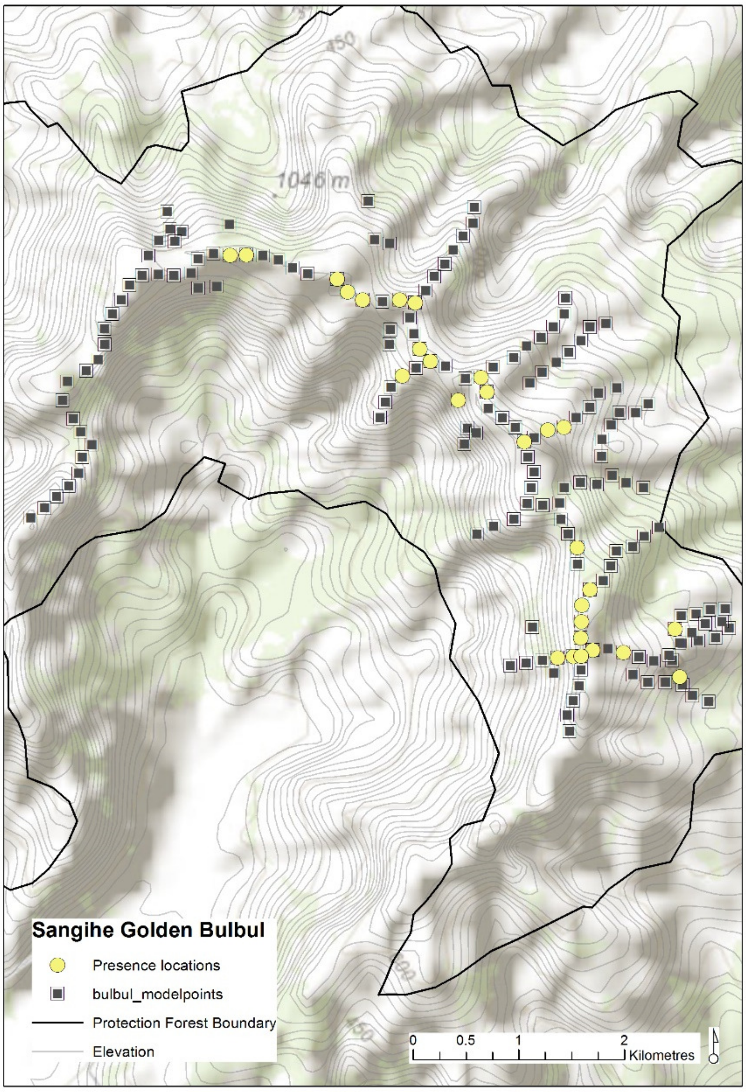


Figure 2-12. Sangihe Golden Bulbul presence/absence locations used for habitat suitability modelling.

A total of 28 presence locations were identified for Sangihe Golden Bulbul, with six locations from both the 2009 and 2014 surveys (Table 2-2. Additional presence points from survey of Rosyadi (Rosyadi 2009) and 2014 (Burung Indonesia unpub. data) included for modelling the associations of the CR species.All records wee from the Sahendaruman Crater. The additional points came from locations within the recorded extent of the range from the 2015 survey (Appendix 2-3). The lowest altitude record for the species was 660 m above sea level, hence models were fitted using points above 620 m above sea level with a 40 m buffer (*n* = 175).

### *2.3.1.2 Trees associated with Sangihe Golden Bulbul*

No tree species displayed a significant association with the occurrence of the golden bulbul, but the top three trees from the random forest model (Figure 2-13) were initially included in the full model. The three trees most associated with the occurrence of the golden bulbul were ‘Peka darone’ *Magnolia* cf. *tsiampacca* (univariate GLM with log-link, β = 0.66 ± 0.44, NS), ‘Salé’ *Zanthoxyllum integrifoliolum* (β = 0.81 ± 0.5, NS) and ‘Momase’ *Sterculia insularis* (β = 0.53 ± 0.44, NS).

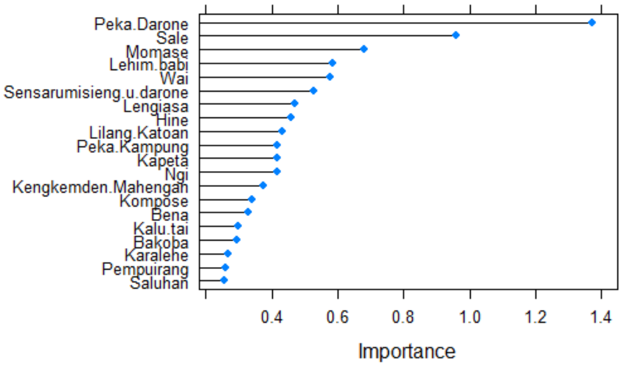


Figure 2-13. Relative importance of tree species as predictor variables from a 10-fold cross-validation of a random forest model with presence/absence of Sangihe Golden Bulbul as the dependant variable. The top 20 ranked out of a total of 61 tree species are shown.

The final candidate set contained 13 models within 2 AICc units of the best model, and all models had low weights indicating relatively low differentiation of any of the candidate models (Table 2-9. Sangihe Golden Bulbul models, restricted elevation (n = 174), within ΔAIC*c* 2 of the best model.. The only significant predictor was low-level vegetative cover and this was included in every candidate model (

Table 2-10); in fact, one model was this predictor alone. However, the largest coefficient estimate was for the proportion of large trees, although the large standard error associated with this indicates uncertainty in the influence of this predictor. With a set of similarly performing models, full averaging was appropriate for creating the habitat suitability model (Grueber et al. 2011). With this lower correspondence between the predictors and occurrence of the species the AUC of ROC is lower than in the other two models at 0.79 (Figure 2-14). As with the whistler model, despite elevation being excluded from the model, the predictions are strongly tied to high elevation sites due to the colinearity of predictor variables with altitude. The specificity of the nearest neighbour interpolation is lower, with the lowest untransformed predicted probability of occurrence for a presence location being 0.203, with 11.46 km$^2$ of habitat across the island exceeding this value (Figure 2-15).

Table 2-9. Sangihe Golden Bulbul models, restricted elevation (n = 174), within ΔAICc 2 of the best model.

| # | ***model*** | ***R.2*** | ***K*** | ***AICc*** | ***delta*** | ***weight*** |
|---|---|---|---|---|---|---|
| 1 | *LowCov + Sale + Tompoliu* | 0.08 | 4 | 141.55 | 0 | 0.11 |
| 2 | *CanopyHt + LowCov + ProportionLargeTrees + Sale + Tompoliu* | 0.1 | 6 | 141.75 | 0.2 | 0.1 |
| 3 | *LowCov + Tompoliu* | 0.06 | 3 | 141.83 | 0.28 | 0.1 |
| 4 | *CanopyHt + LowCov + ProportionLargeTrees + Tompoliu* | 0.08 | 5 | 141.93 | 0.38 | 0.09 |
| 5 | *CanopyHt + LowCov + ProportionLargeTrees + Sale* | 0.08 | 5 | 142.03 | 0.48 | 0.09 |
| 6 | *LowCov + Sale* | 0.06 | 3 | 142.28 | 0.73 | 0.08 |
| 7 | *CanopyHt + LowCov + ProportionLargeTrees* | 0.07 | 4 | 142.35 | 0.8 | 0.08 |
| 8 | *LowCov + ProportionLargeTrees + Tompoliu* | 0.07 | 4 | 142.48 | 0.93 | 0.07 |
| 9 | *CanopyHt + LowCov + Sale + Tompoliu* | 0.08 | 5 | 142.67 | 1.13 | 0.06 |
| 10 | *LowCov* | 0.05 | 2 | 142.9 | 1.36 | 0.06 |
| 11 | *LowCov + ProportionLargeTrees + Sale + Tompoliu* | 0.08 | 5 | 142.9 | 1.36 | 0.06 |
| 12 | *CanopyHt + LowCov + Sale* | 0.07 | 4 | 143.23 | 1.68 | 0.05 |
| 13 | *LowCov + ProportionLargeTrees* | 0.05 | 3 | 143.36 | 1.81 | 0.05 |

Table 2-10. Sangihe Golden Bulbul model-averaged coefficients (full average or 'zero method'(Burnham & Anderson 2002)).

| | ***Estimate*** | ***Std. Error*** | ***Adjusted SE*** | ***z value*** | ***Pr(>\|z\|)*** | |
|---|---|---|---|---|---|---|
| *(Intercept)* | -4.049 | 1.267 | 1.274 | 3.18 | 0.001 | ** |
| *Low cover* | 0.048 | 0.017 | 0.017 | 2.76 | 0.006 | ** |
| *Sale* | 0.883 | 0.548 | 0.551 | 1.60 | 0.109 | |
| *Tompoliu* | 0.821 | 0.483 | 0.486 | 1.69 | 0.092 | . |
| *Canopy height* | -0.117 | 0.079 | 0.080 | 1.46 | 0.143 | |
| *Proportion large trees* | 2.32 | 1.566 | 1.575 | 1.47 | 0.141 | |

Table 2-11. Sangihe Golden Bulbul variable proportionate importance within the averaged models (< Δ2 AICc units of the lowest AICc).

| | ***Low cover*** | ***Tompoliu*** | ***Sale*** | ***Prop. lrg. trees*** | ***Canopy height*** |
|---|---|---|---|---|---|
| *Importance* | 1.00 | 0.6 | 0.56 | 0.54 | 0.48 |
| *N models* | 13 | 7 | 7 | 7 | 6 |

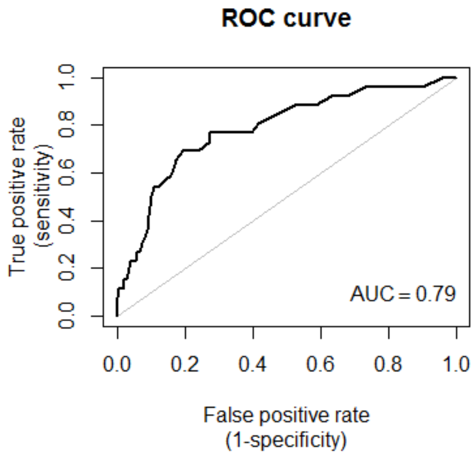


Figure 2-14. ROC of the single top ranked model by AICc for Sangihe Golden Bulbul.

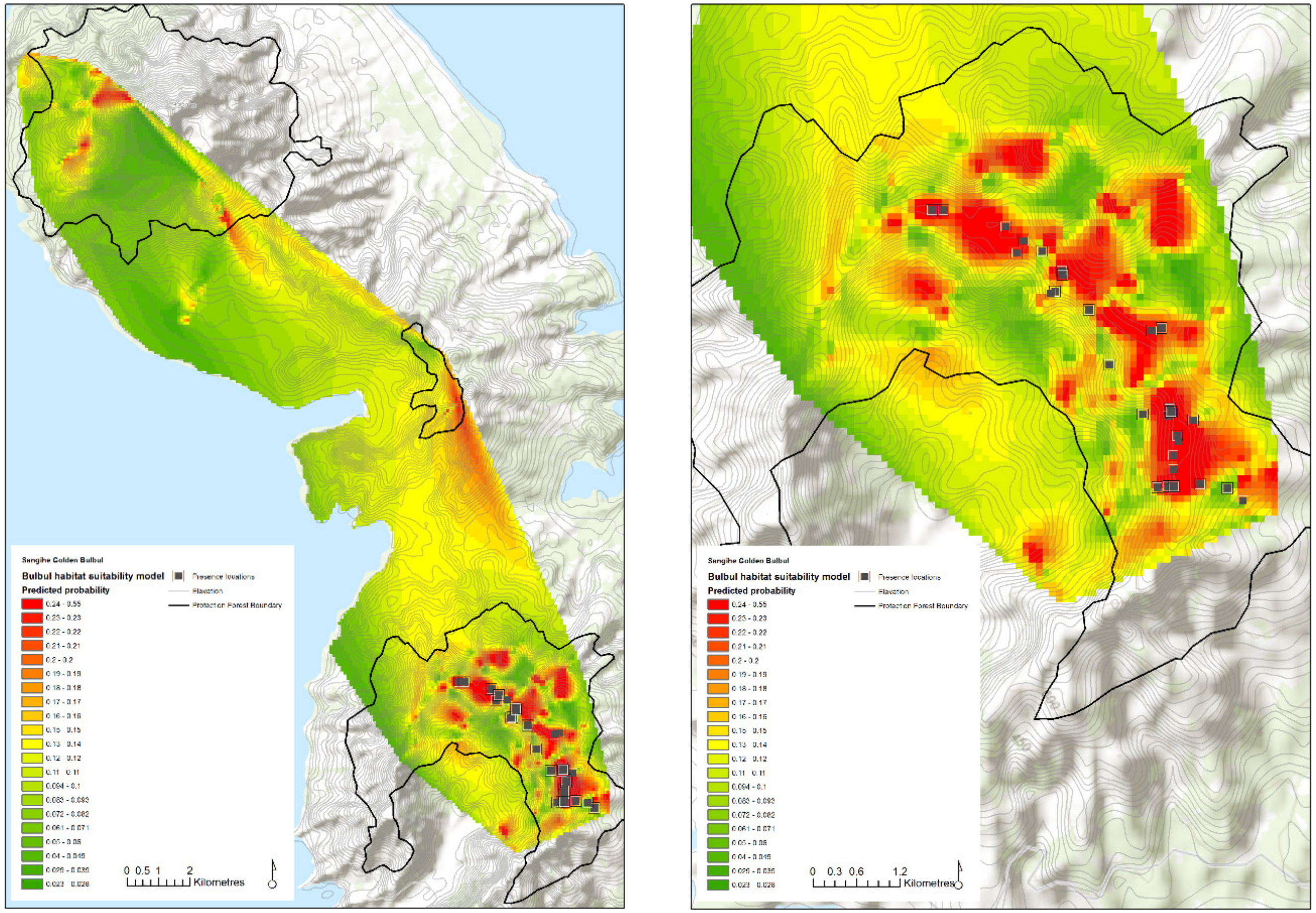


Figure 2-15. Nearest Neighbour Interpolation of the predicted occurrence probabilities of Sangihe Golden Bulbul derived from the model-averaged values from the candidate set of GLM habitat asociation models within 2 AIC units of the best model across the three surveyed locations on Sangihe. Red colours represent the highest occurrence probability, with a colour scale of ¼ standard deviations in probabilities of Sangihe Golden Bulbul presence at each point.

### *2.3.2 Predicted suitability away from Sahendaruman: potential restoration and translocation sites*

The predicted maximum extent of total suitable habitat in the three areas that hold the best remaining forest is below 12 km$^2$ for each species, and for all three over 90% of this area is around the Sahendaruman crater. Away from Sahendaruman identified habitat with a probability score equal or greater than the lowest presence point totalled only 0.31 km$^2$ in three locations around Gunung Awu for the paradise-flycatcher with one of locations at considerably lower altitude than the recorded presence points. The lower specificity of the bulbul model predicts a larger area of potentially suitable habitat for this species away from Sahendaruman, totalling 0.74 km$^2$ at Gunung Awu (mostly adjacent to the area identified for the paradise-flycatcher, but at higher elevations) and 0.27 km$^2$ at Gunung Otomata. In contrast, suitable habitat for Sangihe Whistler is entirely restricted to Sahendaruman.

## 2.4 Discussion

The results demonstrate that preserving the remaning forest around the Sahendaruman crater is essential for increasing the chance of preventing the extinction of the Critically Endangered birds that still occur on Sangihe, as almost all suitable and potentially suitable habitat is restricted to the remnant forest around the crater. Previous authors have recognised the global significance of this small area of forest (Rozendaal & Lambert 1999; Riley & Wardill 2001; Riley 2002a), but its importance for preventing multiple imminent bird extinctions cannot be overstated. There have been no records of the CR species away from this area for several decades (Whitten et al. 1987; Riley & Wardill 2001; Riley 2002a; Rosyadi 2009; Bashari & Fauzan 2014) and the results of this habitat suitability modelling for the three species that are still encountered regularly suggest that the probability of the continued presence of these species away from Sahendaruman is effectively nil.

The modelling approach used to identify potentially suitable habitat assumed that the combined positive sighting locations assembled represented a 100% detection rate for the species concerned. This naïve modelling approach, without making adjustments for imperfect detection (Mackenzie et al. 2011) is unlikely to be true, especially given that the majority of records of the whistler and bulbul were aural.

However, the present survey was not a direct repeat of the sites previously visited, for which habitat data was unavailable, hence there was no time series of presence with which to construct probabilities of occurrence. Instead, previous positive sightings from the 2009 and 2014 surveys were incorporated to maximise the range of suitable measured habitat samples, such that a maximised confirmed presence dataset was used to generate the habitat-association models. Using an occupancy modelling approach to generate probabilities of presence at sites where these exceptionally rare species were not confirmed may risk misclassifying resource use (in failing to identify a feature used in a false absence site) (MacKenzie 2006). But it also risks generalising the habitat predictors while the aim of this work was to define a narrow habitat suitability to guide opportunities for restoration management, for which the decision was taken to only use confirmed sites.

Within the Sahendaruman crater itself it appears that the four species for which modern records are confirmed have been able to persist at small population sizes without intervention for many decades, in line with theoretical predictions of an extinction debt half-life in the order of centuries to thousands of years (Kitzes & Harte 2015; Halley et al. 2016). The ability of these species to continue to persist is questionable. Time to first extinction in such systems is predicted to be relatively short (Halley et al. 2016), and the recent extinction of the endemic subspecies of Red-and-blue Lory along with the apparent recent loss of Great-billed Parrot from the island (Riley 1997b, 2002a) implies that payment of the debt is on-going. On the neighbouring island of Talaud these three parrots demonstrated the most severe decreases with increasing habitat modification and were strongly associated with primary habitats (Chapter 3). An alternative or additive possibility is that this set of rare and specialised species represent taxa in the habitat restricted, relictual final fourth stage of the ‘taxon cycle’ (Wilson 1961; Ricklefs & Bermingham 2002; Jønsson et al. 2014, 2018). These ‘stage IV’ species are at the highest risk of extinction (Ricklefs & Bermingham 2002) and represent irreplaceable evolutionary lineages. This may be an explanation for the perplexing rarity of Sangihe White-eye, which has only been seen a handful of times in recent decades (Rasmussen et al. 2000; Riley 2002a; Rosyadi 2009; Bashari & Fauzan 2014) and the only possibly record during this study was of three unidentified *Zosterops* species observed in flight only.  As such, there does not appear to be a practical conservation solution for this species aside from retaining the current extent of primary ridgetop forest and implementing efforts to extend the forested area in the hope that the species persists.  The final CR species, Sangihe Dwarf-kingfisher, has been recently considered to be extinct following the withdrawal of the last report from 1997 (Riley 1997b, 2002a), which appears to be the only report from the 20th Century.

Populations of the remaining species are very small; for Cerulean Paradise-flycatcher the most recent estimate is of 34 – 150 individuals (Fauzan & Bashari 2016), Sangihe Whistler has been estimated to number between 92-255 individuals (Rosyadi 2009) and Sangihe Golden Bulbul between 50-230 individuals (Riley 2002a) and the consistency of the actual numbers recorded with intensive repeated survey effort (Riley 2002a; Mamengko 2006; Rosyadi 2009; Fauzan & Bashari 2016; this study) implies that there are not large numbers being overlooked.  The maximum potential Area of Occurrence (AOO) for these species is equally worrying: even with estimating potential suitability outside of current observed elevational restrictions the three species possess amongst the smallest ranges of all birds, all below 12 km$^2$ and Sangihe Whistler at 6 km$^2$.  While the habitat suitability model indicates the maximum AOO for each species, additional habitat limitations on the species during parts of their lifecycle are likely to reduce the area that is utilised.  In particular, all paradise-flycatcher nests found to date have been suspended over running water within valleys (H. Bashari pers. comm., pers. obs. 2015), implying that selection for measures to guard against nest predation by terrestrial predators may be severe (Collias & Collias 1984).

Combining the highly restricted range and small population size with observations of continued small-scale habitat clearance for agriculture and selective logging (Fauzan & Bashari 2016), it is clear that the area of habitat remaining within the Sahendaruman crater needs to be secured and expanded through restoration prior to efforts to establish additional population loci on the island.  The frequency of habitat degradation and disturbance within valleys occupied by Cerulean Paradise-flycatcher shows that these areas are not seen as protected forest in the same way as the higher altitude areas: 76% (16/21) locations of Cerulean Paradise-flycatcher recorded in 2015 fall outside of the forest area defined in 2006 (Mamengko & Mole

2006b) (Figure 2-16) despite virtually all occurring in forest with canopy closure of greater than 70% and with evident primary structure (e.g. Appendix 2-6). The extent of clearance and logging in these same areas suggests that the assessment of the habitat as not protected is widespread within the local community: virtually no clearance was recorded within the 'forest' defined by Mamengko & Mole (2006b).

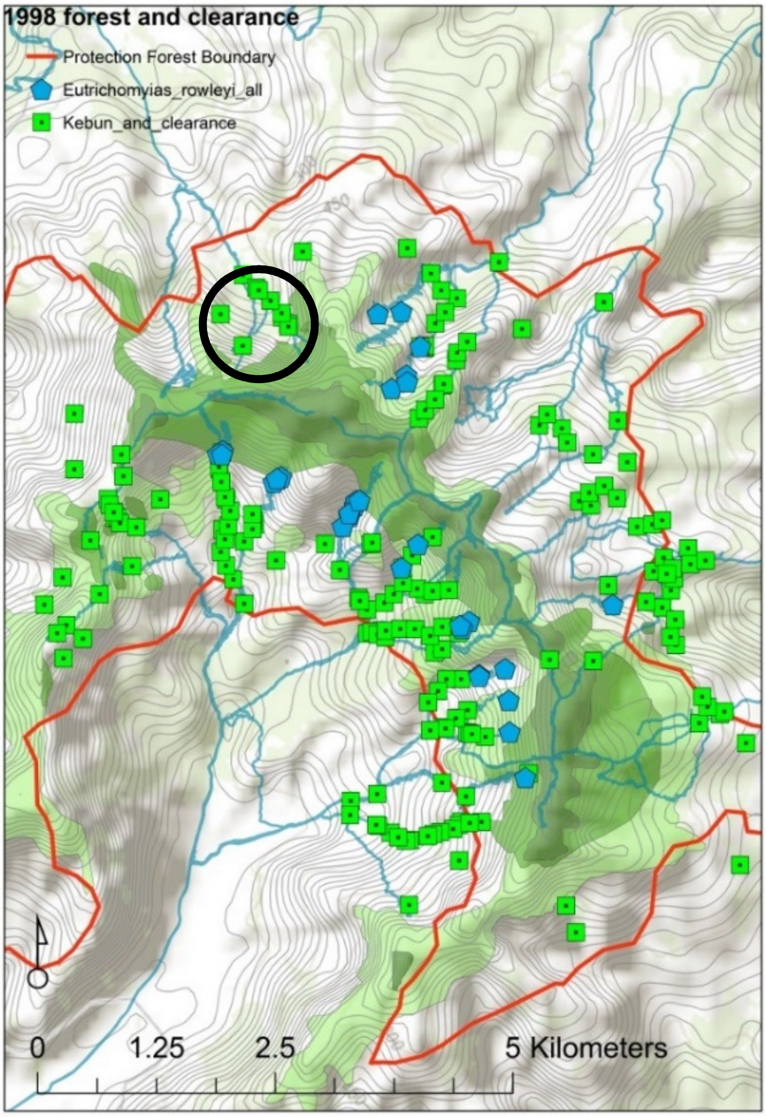


Figure 2-16. The 1998 estimate of forest habitat (Riley 2002), observations of forest clearance and records of Cerulean Paradise-flycatcher from this survey. Circled area is the valley in which the species was recorded in 2014, but could not be located in 2015.

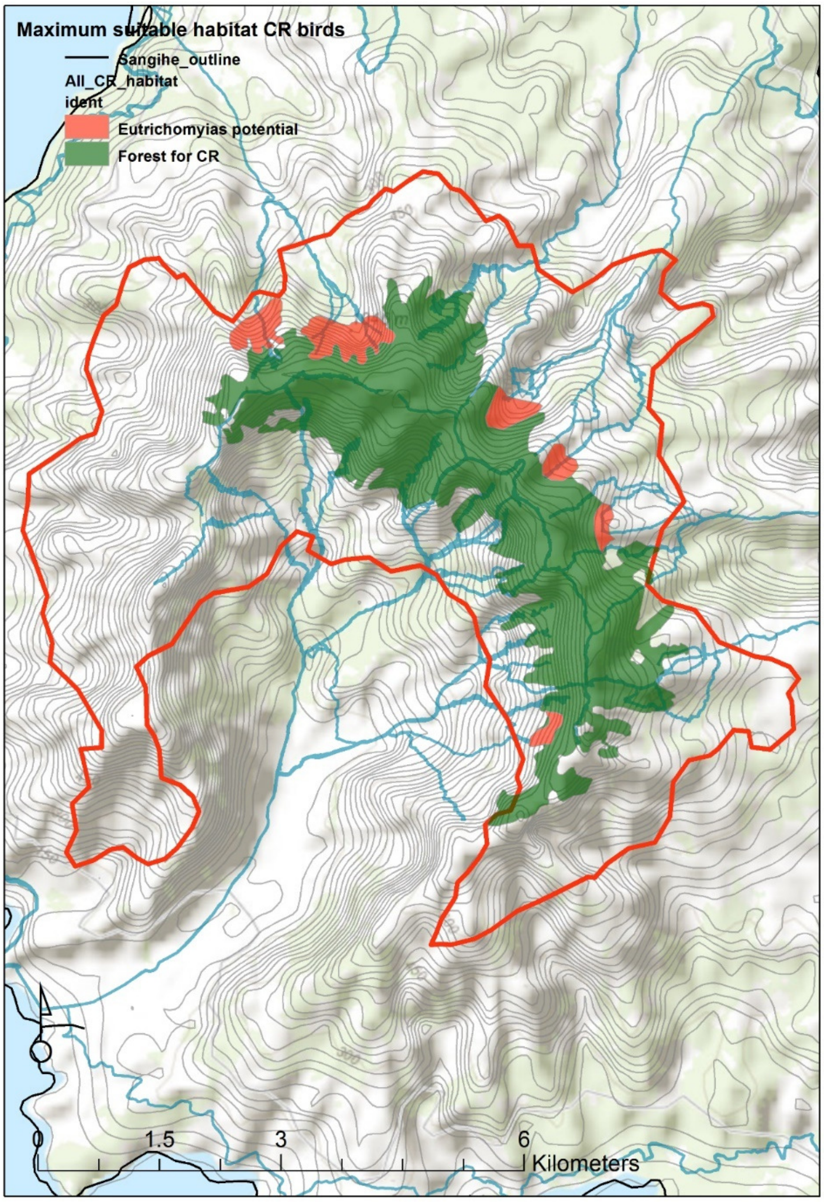


Figure 2-17. Current area remaining that possesses suitable habitat for Sangihe's four species of Critically Endangered birds

The boundary of the protection forest around the Sahendaruman Crater encloses an area of 3,549 hectares, as defined by the decree *Surat Keputusan Menhutbun Nomor 452/Kpts-II/1999*. This area is far greater than the extent of the habitat suitable for the Critically Endangered species. Most of the protection forest area is unsuitable for any of these species as it has been converted to plantation gardens. However, combining the predicted distribution of the three modelled CR species the area of forest recorded as potentially suitable for use by one or more CR bird species was 1,265 hectares (Figure 2-17. Current area remaining that possesses suitable habitat for Sangihe's four species of Critically Endangered birds), considerable larger than the previously reported extent of primary forest (Riley 2002a; Mamengko & Mole 2006b). In one location at the west side of the south-eastern edge of the crater above Lelipang district the extent of suitable habitat extends outside of the recognised protection forest boundary; this area requires protection. For these three species, habitat that appears to resemble presence locations is present across a larger area than is presently occupied within the Sahendaruman crater, and areas adjacent to existing pairs should be prioritised for restoration efforts incorporating the flora and structural features associated with each species occurrence. Initial restoration efforts should focus immediately on those areas subject to recent, even minor clearance. Clear boundaries that include Cerulean Paradise-flycatcher habitat need to be communicated to all community members that use the forest. The primary conservation target needs to be the recolonisation of these sites, which may be possible naturally for those adjacent to occupied areas but to establish additional breeding locations at more isolated sites assisted recolonisation using translocation of individuals could be required. A priority location is circled in Figure 2-16, where the paradise-flycatcher was present in 2014 but absent in 2015 with logging and small-scale clearance taking place between the two surveys. Despite this, habitat remains highly suitable and restoration may be possible with minimal expenditure through planting of 'Sahai' and 'Bunaro' saplings and the seeding of lianas into the subcanopy. The visual similarity of unoccupied and occupied sites can be seen in photographs of high potential unoccupied sites and occupied sites presented in Appendix 2-4 to 2-6.

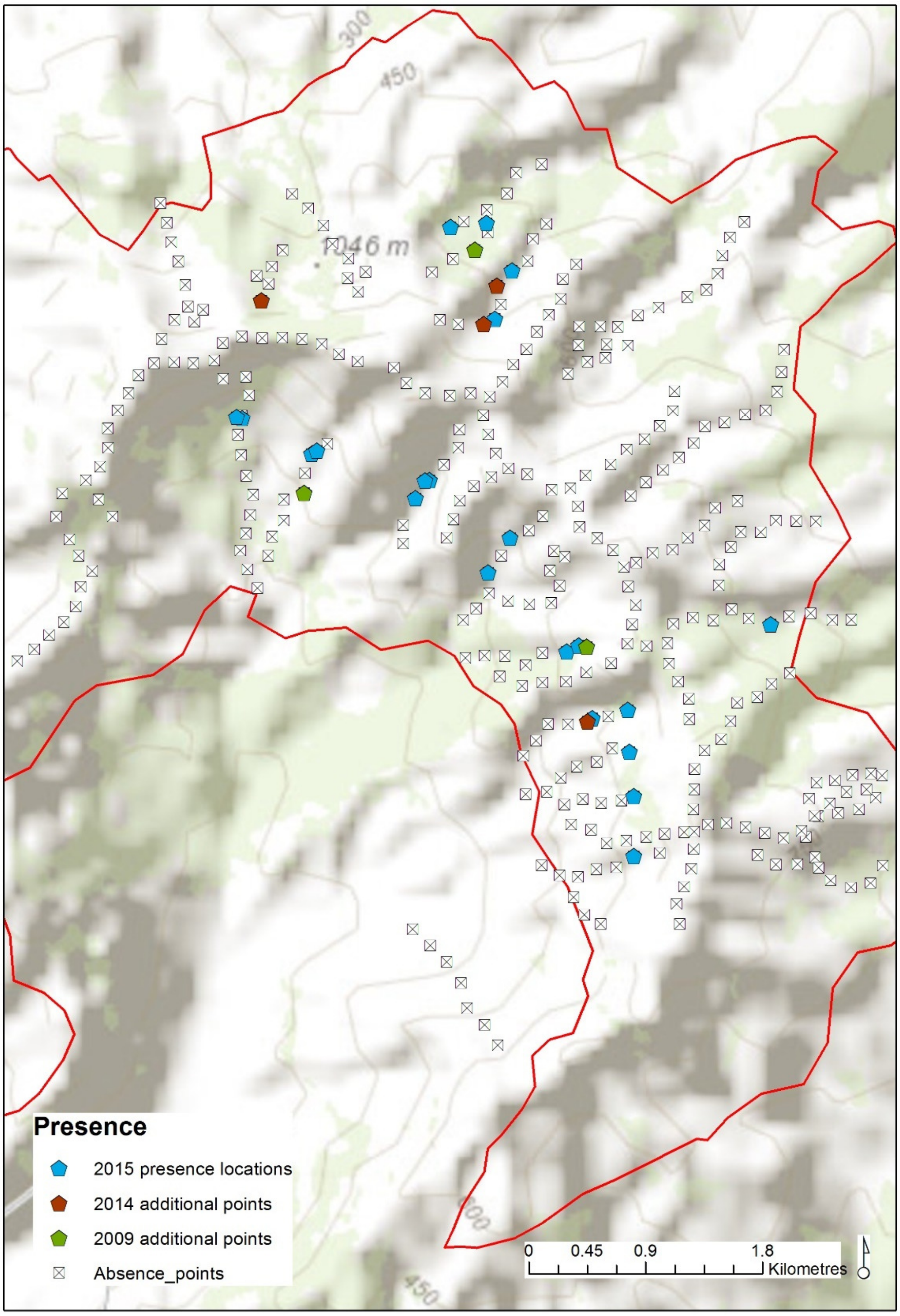
Away from Sahendaruman two small areas within the protection forest around Gunung Awu are similar to presence locations of the paradise-flycatcher within valleys in the Sahendaruman crater, including parts of a realtively long valley that lies well below the present elevational restrictions on the species (Figure 2-7). Other declining island bird species that demonstrate a retreat uphill have been demonstrated to be severely endangered through the impact of introduced diseases and their vectors, further exacerbated by climate change impacts (Benning et al. 2002; Atkinson & LaPointe 2009). No impacts of introduced species are presently suspected on Sangihe, but prior to any effort to establish supplemental populations detailed demographic studies are required to exclude such a possibility. However, those sites at Gunung Awu (Appendix 2-4) are limited in size and appear to lack some of the structure found within the the potential restoration sites within the Sahendaruman crater (Appendix 2-5). Along with the extra risk of creating a reintroduction program on the slopes of an active volcano the recommendation is to focus efforts of restoration of degraded sites within the Sahendaruman crater. Considerable basic species-specific research is needed into current breeding success and into the dispersal behaviour of post-fledging immatures in order to devise a project-specific best practice plan (Groombridge et al. 2004; Ewen 2012). With so few adult pairs remaining, any decision to remove these from the wild must be taken in possession of considerably more knowledge of the likely success of a translocation. The successful translocation of Seychelles Paradise-flycatcher (Currie et al. 2017) provides hope that translocation may be an option if required in the future. Recent advances in understanding cues used in habitat selection by individuals may also improve the likelihood of successful natural settlement of dispersing immature birds by using song cues to indicate the presence of additional conspecifics (Betts et al. 2008). Rather than capturing and moving birds, it may be preferable to start with habitat enhancement and species cues to encourage settlement in new areas, with an absolute cessation of further clearance.

Of the two high-altitude species, the wider potential habitat tolerance suggested by the model for the golden bulbul must be treated with caution, given the relatively low power of any of the constituent predictors. While the species’ closest relatives are more tolerant of secondary and open habitats (Collar et al. 2013) and previous reports have noted its highly cryptic nature when not calling (Riley 1997b) it has not been reported away from primary forest around the upper elevations of the Sahendaruman crater in recent years (Riley 2002a; Collar et al. 2013). The model for Sangihe Whistler, in contrast, indicates that there is little doubt that it is highly restricted with positive associations with higher canopy cover, a greater proportion of large trees and primary tree species of the ridgetop forest (Table 2-7). Given the lack of additional high elevation primary forest away from the Sahendaruman crater the key conservation actions are to prevent any further degradation of the ridgetop primary forest and to encourage natural restoration of damaged areas (such as those cleared for the setting of mistnets to trap bats for food [pers. obs]). The existence of an apparently unsuitable gap of habitat between the northwestern peak of Gunung Sahendaruman itself and the larger block of habitat along the eastern side of the ridge arc is concerning (Figure 2-11), potentially fragmenting this tiny range.

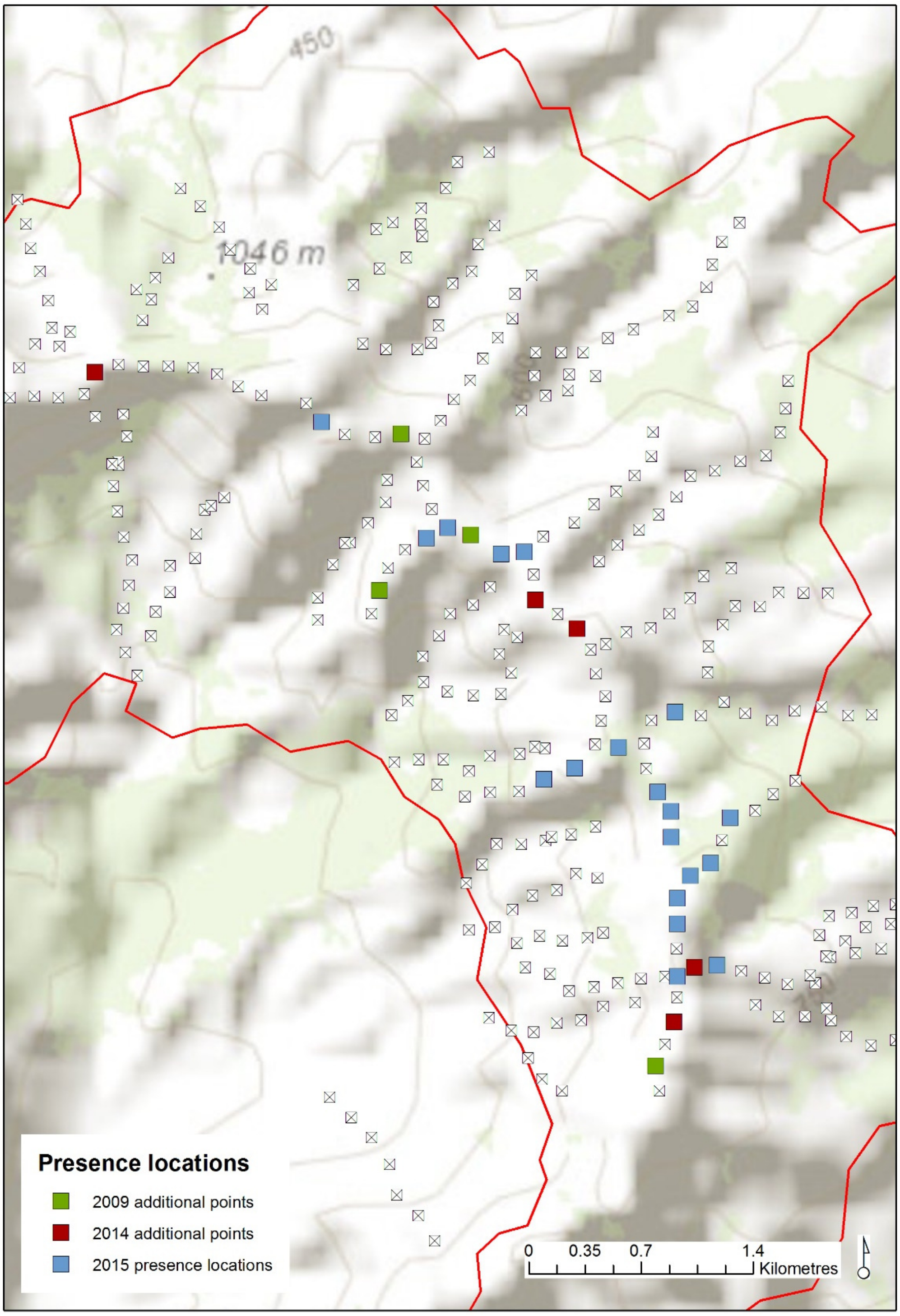
To date, this set of unique and highly endangered birds have received little conservation attention and much of the basic species information is not known; reproductive rates, phenology, intra-and interspecific interactions including predation, competition and potential disease exposure and dispersal behaviours are all unknown. Such information will increase our potential to be able to tailor conservation approaches to each species as has been successfully achieved for island endemics in locations around the world (Monticelli et al. 2010; Jones & Merton 2012; Currie et al. 2017), but action to reverse the continuing slow decline in the extent of the critical habitat identified here can and must begin immediately.

## 2.5 Appendices

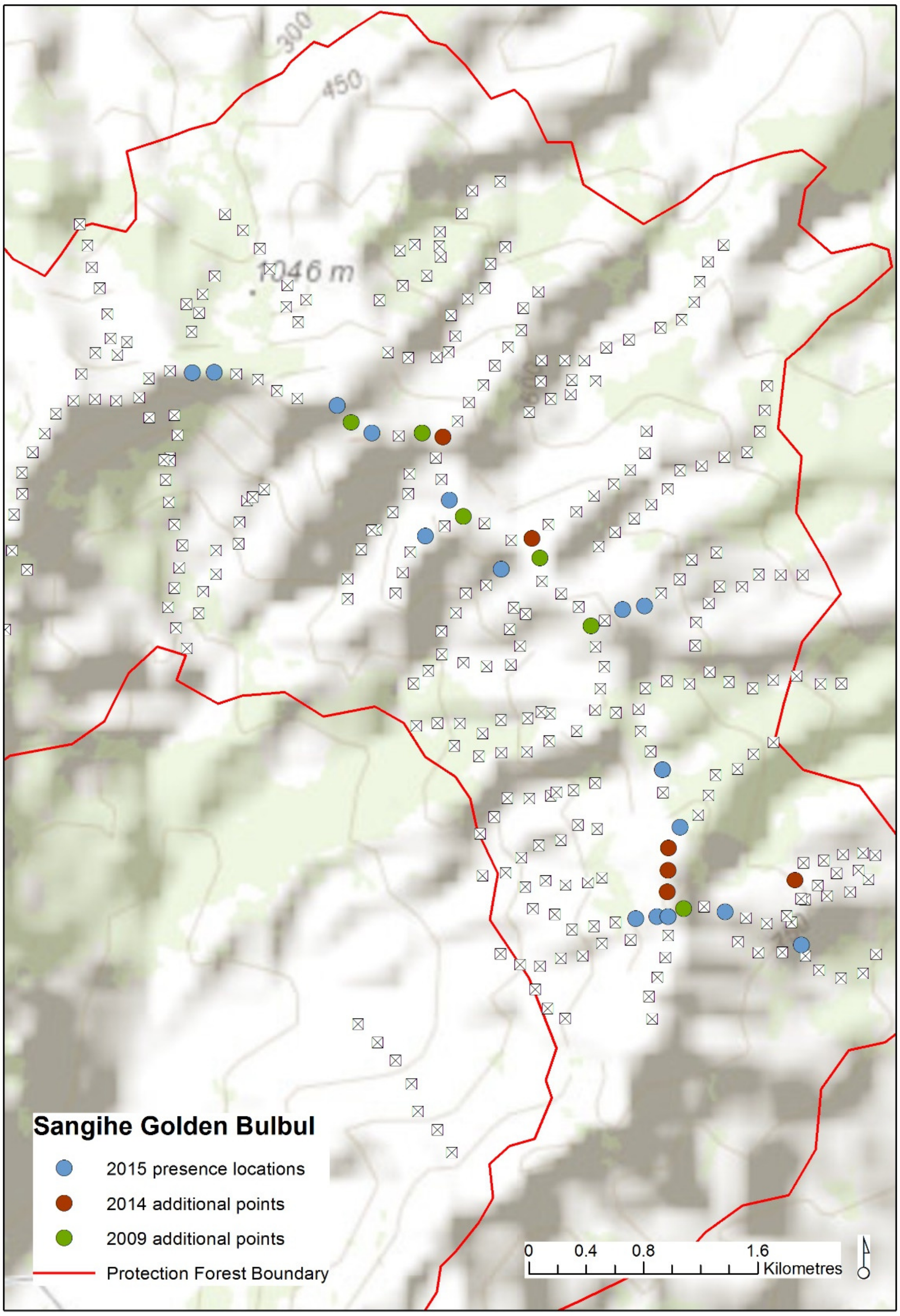
Appendix 2-1. Cerulean Paradise-flycatcher presence point locations and most recent year of observation.



Appendix 2-2. Sangihe Whistler presence point locations and most recent year of observation.



Appendix 2-3. Sangihe Golden Bulbul presence point locations and most recent year of observation.



Appendix 2-4. Gunung Awu locations with the highest predicted probabilty of suitability for Cerulean Paradise-flycatcher; l, Maselihe valley; r, Talawid Atas.

Appendix 2-5. Sahendaruman crater locations with no records of Cerulean Paradise-flycatcher, but high predicted suitability; l, Lelipang south; r, Bukide.

Appendix 2-6. Cerulean Paradise-flycatcher presence location; l, Lelipang; r, Kentuhang; lower, Ulu Peliang.

# Chapter 3: Species-specific non-linear abundance responses to habitat modification within an island bird community

## Abstract

The extent and rate of habitat loss and degradation are used as a surrogate for population declines in birds, but species-specific responses to changes in habitat quality may not be consistent even within forest-dependent species. The shape of these individual responses has been little investigated on tropical islands, despite their high endemism, high proportion of restricted-range species and accelerating rates of habitat loss. Understanding the species-level consequences of different levels of habitat modification is essential to inform the management of island ecosystems to retain biodiversity. Here I assessed the abundance of the bird community on the Indonesian island of Karakelang, in the Talaud group, at multiple points along a simple habitat modification gradient. I found there were separate groups of bird species which increased, maintained and declined in abundance along this gradient, and within those that declined individual responses displayed differences in both the rate of decline and the shape of the response. Abundance of forest species was far higher in primary forest, and as these species were large, total bird biomass declined dramatically with increasing levels of modification. All endemic species were most abundant in primary forest, and several key bird species, including the Endangered Red-and-blue Lory, declined most severely at the point on the modification gradient where forest no longer contained primary elements; however, almost all were recorded in each location along the gradient. At present these species retain relatively large and secure populations, but loss of primary forest area on the island will result in substantial population reductions for these species.

## 3.1 Introduction:

Predicting the impact of tropical forest loss and modification on bird populations in order to assess their relative extinction risk is a vital step in maintaining bird biodiversity globally (Brooks et al. 2002; Pimm et al. 2006; Lee & Jetz 2011). Absolute forest loss had already been demonstrated to predict risk of extinction in birds, as defined by the IUCN Red List (IUCN 2001) generally well (Brooks et al. 2002; Pimm et al. 2006), allowing estimates of the area and rate of habitat loss to be used as surrogates for declines either in bird populations (Buchanan et al. 2008; Vetter 2009) or in their area of occurrence (Hall et al. 2009) to prioritise extinction risk (Bird et al. 2012; Tracewski et al. 2016).

This approach is a considerable move towards addressing the severe lack of data on population trends for most bird species, but care is required to understand and address the assumptions necessary. As the models use a binary presence or absence of forest estimated from global models of land-use change (Hansen et al. 2013) they do not incorporate the impacts of habitat modification prior to loss. Consequently, sensitive species that may disappear while forest remains relatively intact (e.g. Moura et al. 2016) will be more severely impacted than predicted (Betts et al. 2017). An assumption is also made that a given loss of habitat area results in an equivalent change in population size for each species assessed (Tracewski et al. 2016), following the observation of the loss of species with known old-growth forest associations after conversion (Beukema et al. 2007), and the apparent linear abundance response for forest specialist birds (Pardini et al. 2009). However, birds are among many taxonomic groups for which a range of species-specific responses has been recorded (Radford et al. 2005; Fischer & Lindenmayer 2006; Mallari et al. 2011; Davies et al. 2015), and nonlinear abundance responses may actually be frequent (Bender et al. 1998). While these analyses are restricted to species classified as having ‘high’ (Tracewski et al. 2016) or both ‘medium’ and ‘high’ forest dependency (Bird et al. 2012), to reduce the effect of violations of this assumption, within these categories divergent abundance responses may still occur. Additionally, a species’s tolerance may vary throughout its range in response to variation in community structure and resource availability.

These assumptions leave the assessment highly dependent on the initial classification of the species's level of forest dependency. Ambiguous definitions of habitat associations may impair the identification of regional abundance trends (Fraser et al. 2017): in this instance forest dependency misclassification of a species may prevent its identification as a species at risk. Thus, understanding the impact of modification on species abundance at different locations in its range is important for the prediction of relative extinction risk.

Most studies classify habitat into a few discrete categories (primary versus secondary, intact versus degraded) (e.g. Jones et al. 1995; Waltert et al. 2005; Marsden et al. 2006; Edwards et al. 2011; Mallari et al. 2011). While these demonstrate clear reductions in density in modified habitat, few studies report the actual density response of species along a continuous gradient of modification intensity (but see Radford et al. 2005). Particularly sensitive species could be expected to demonstrate a more considerable population reduction for a modification impact than others, despite both demonstrating a reduced abundance at the coarser scale, e.g. plantation versus primary forest. Española et al. (2016) used presence-absence data to assess frugivore responses along forest quality/restoration gradients, finding that the responses of almost all Philippine frugivores were straightforward and usually linear. Similarly, occurrence data at point counts were used to model the response to a ranked set of habitat disturbance classes of 44 Amazonian bird species in two regions of Brazil, again finding typically linear responses but inter- and intraspecific variation in response magnitude (Moura et al. 2016). However, presence data may imply that a suboptimal habitat is suitable when used by a species (particularly highly mobile species) at low density, and for some species areas of degraded forest may act as population sinks (e.g. Beck et al. 2004).

Tropcal oceanic islands represent a special case of concern due to their high levels of endemism (Kier et al. 2009) and, because they hold many small and restricted-range taxa, they are expected to have a high proportion facing a high risk of extinction (Newbold et al. 2014). As such, for a given rate of habitat loss, islands may be expected to suffer more species extinctions from habitat loss in the coming decades than mainland areas, particularly given high and increasing rates of forest loss predicted to occur on islands during the 21st Century (Kier et al. 2009). However, as a result of ‘ecological release’ (Diamond 1970; MacArthur et al. 1972b; Bolnick et al. 2010), island species frequently demonstrate higher abundance than mainland congeners or even conspecific populations, with this density compensation potentially offsetting some of the additional extinction risk. Understanding the extinction-proneness of individual species within communities is necessary for anticipating the impact of habitat modification (Gardner et al. 2009): elucidating differences in these for island bird communities may identify urgent priorities.

The island of Karakelang in the Talaud Islands, at the northeast extremity of Indonesia at the merging of the Moluccan Sea and the Pacific Ocean, holds five endemic bird species, two of which are globally Endangered (BirdLife International 2017a). Small-scale illegal forest clearance and selective logging are ongoing (Mamengko & Mole 2006a). However, a considerable percentage of the island’s total area of *c.*800 km$^2$ remains as primary forest, such that a continuum of habitat modification is present (Riley 2002a) and an intact bird community is believed to remain. Here I use data from a field survey to calculate individual species abundance at multiple points along a simple modification gradient. I use these abundances to assess individual species’ responses to forest modification to assess not only density change between habitat categories but also the divergence between species in rates of abundance change, and to identify non-linear responses to modification. This allows confidence in the assessment of forest dependency and in the prediction of the relative likelihood of extinction due to future habitat loss. I also present updated estimates of population sizes across the bird community, and these together provide a more comprehensive picture of the relative extinction risk from habitat loss and degradation on the island in order to inform future management options.

## 3.2 Methods

### 3.2.1 Study site

The island of Karakelang in the Talaud Islands, Indonesia, retains a considerable proportion of primary forest, primarily in the centre of the island (Syarif 2004). Approximately 33% of the total area of the island is designated as a Wildlife Reserve/*Suaka Margasatwa* (Syarif 2004), a strict protection designation including a no-take clause (Protected Planet 2016). Moving out towards the perimeter of the island this becomes degraded, with selectively logged forest, secondary forest, forest gardens and mixed small-holder agriculture known as *kebun*, mixed plantations of several tree species and finally large plantations of one or two commercial species along with areas of field crops (Riley 2003; Syarif 2004). Essentially, there is a patchwork of differing intensity of human modification to the naturally forested island typical of many of the small to medium islands in Wallacea and Melanesia (Woinarski 2010; Davies et al. 2015). The area of different habitat types present on the island was estimated using remotely sensed data by Syarif (2004), and subsequently the area of annual loss can be obtained from the detailed Global Forest Watch dataset (Hansen et al. 2013).

The bird community is considered intact, as there is no evidence for any extinctions of bird species from the Talaud Islands when comparing avifaunal lists compiled at the turn of the 20th with those compiled at the turn of 21st centuries (Meyer & Wiglesworth 1898a, 1898b, Riley 1997a, 2003). However there has not been a detailed search for subfossil remains on the island, and there remains the possibility that as yet unknown species did occur on the island in historic times, although the island is not comparable to Pacific Islands that suffered myriad post-colonisation extinctions in their bird fauna (Steadman 2006) due to a longer period of human habitation. The bird community is species-poor in comparison to the three closest large islands, Sulawesi, Halmahera and Mindanao, as expected given the isolation and relatively small size of the islands (MacArthur & Wilson 1967b). Table 3-1 lists the endemic, threatened and restricted-range species of the Talaud Islands, with subspecies endemic to Talaud and the Sangihe and Talaud Endemic Bird Area. Two of the five species now considered to occur solely on the Talaud Islands, Talaud Rail

*Gymnocrex talaudensis* and Red-and-blue Lory *Eos histrio*, have been identified as Alliance for Zero Extinction (AZE) trigger species (Alliance for Zero Extinction 2010). Of note are the multiple species of large frugivores, with five sympatric parrots and four species of imperial-pigeon *Ducula.*

Several potentially invasive species have been introduced, including Sulawesi Pig *Sus celebensis*, but none appears to have had a major impact on the bird species present (Riley 2003). Pigs are now rare, but those that remain occur in primary forest and are the main focus of hunting on the island along with bats (Riley 2002b). Trapping is considered a major threat to one bird species on the island, the Red-and-blue Lory (BirdLife International 2017e).

Table 3-1. Endemic and threatened species, and endemic subspecies recorded on the Talaud Islands, (White & Bruce 1986; Coates et al. 1997; Riley 1997a, 2003; King 2002; del Hoyo et al. 2014, 2016). RR (E), restricted range (now only found on Talaud). E, Endemic to the Talaud Islands. SS, subspecies endemic to Talaud. ST, Subspecies restricted to Talaud and Sangihe. RR, restricted-range species (Stattersfield et al. 1998); EN, Endangered; VU, Vulnerable; NT, Near Threatened; LC, Least Concern (BirdLife International 2017a). Species are listed in order of threat status, then increasing range size.

| **Scientific name** | **English name** | **Dist** | **RL** |
|---|---|---|---|
| ***Gymnocrex talaudensis*** | Talaud Rail | E | EN |
| ***Eos histrio talautensis*** | Red-and-blue Lory | RR (E) | EN |
| ***Amaurornis magnirostris*** | Talaud Bush-hen | E | VU |
| ***Erythropitta inspeculata*** | Talaud Pitta | E | VU |
| ***Ducula pickeringii palmasensis*** | Grey Imperial Pigeon | | VU |
| ***Todiramphus enigma*** | Talaud Kingfisher | E | NT |
| ***Caloenas nicobaricus nicobaricus*** | Nicobar Pigeon | | NT |
| ***Ninox randi*** | Chocolate Boobook | | NT |
| ***Tanygnathus lucionensis talautensis*** | Blue-naped Parrot | SS | NT |
| ***Ducula concinna*** | Blue-tailed Imperial-pigeon | RR | LC |
| ***Coracina morio talautensis*** | Sulawesi Cicadabird | SS | LC |
| ***Dicaeum celebicum talautense*** | Grey-sided Flowerpecker | SS | LC |
| ***Leptocoma aspasia talautensis*** | Black Sunbird | SS | LC |
| ***Oriolus chinensis melanisticus*** | Black-naped Oriole | SS | LC |
| ***Prioniturus platurus talautensis*** | Golden-mantled Racquet-tail | SS | LC |
| ***Terpsiphone cinnamomea talautensis*** | Southern Rufous Paradise-flycatcher | SS | LC |
| ***Zosterops everetti babelo*** | Everett's White-eye | SS | LC |
| ***Aplonis panayensis sanghirensis*** | Asian Glossy Starling | ST | LC |
| ***Ducula aenea intermedia*** | Green Imperial Pigeon | ST | LC |
| ***Macropygia ambionensis*** | Slender-billed Cuckoo-dove | ST | LC |
| ***Megapodius cumingii talautensis*** | Philippine Scrubfowl | ST | LC |
| ***Tanygnathus sumatranus sangirensis*** | Blue-backed Parrot | ST | LC |
| ***Treron griseicauda sangirensis*** | Grey-cheeked Green-pigeon | ST | LC |

### 3.2.2 *Bird survey*

Birds were surveyed using a variable circular plot method (Bibby et al. 1998; Buckland et al. 2008; Thomas et al. 2010). Access points were identified in consultation with local guides and, from these, transects were established, either on existing trails or through minimal cutting for access. Within the Wildlife Reserve a camping location was identified and used as a base. Multiple transects were established radiating from the camp, although only one was surveyed per day.

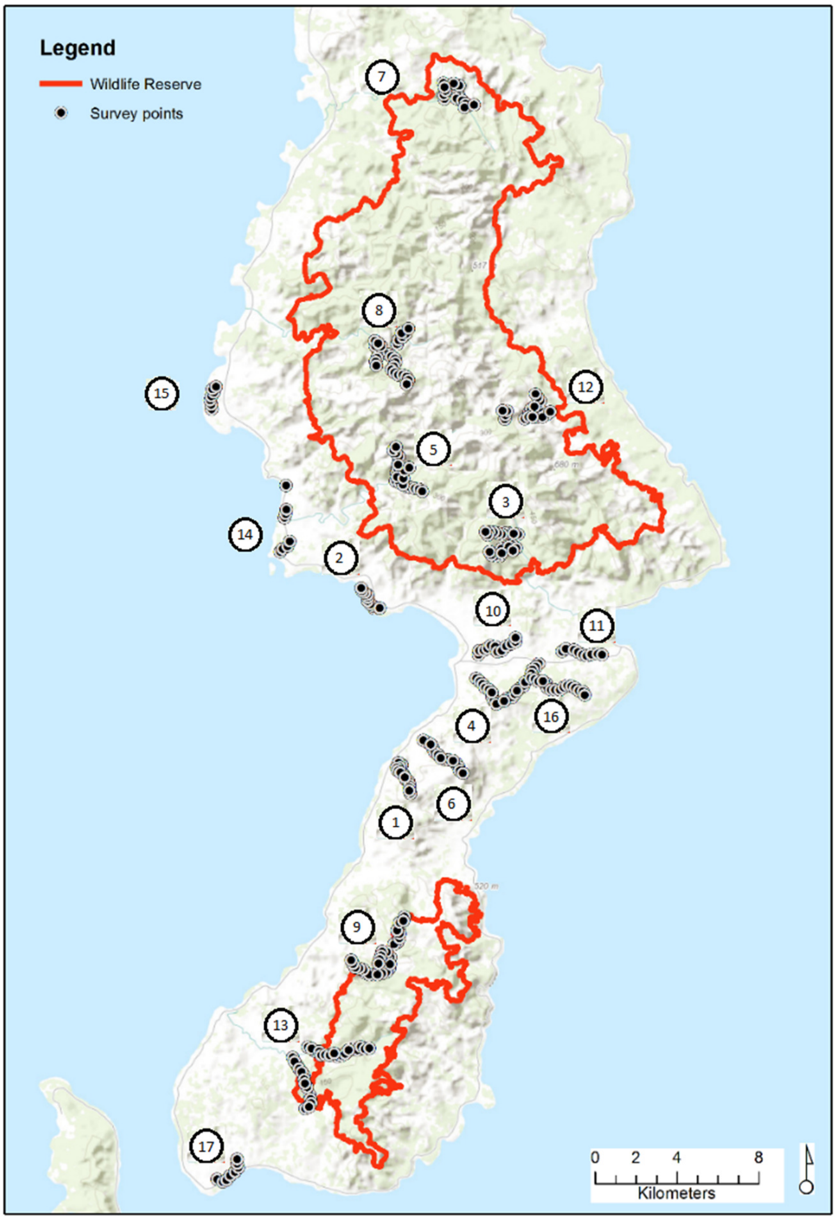


Figure 3-1. Survey locations in 2014 on the island of Karakelang. Numbers refer to the locations listed in Table 3.3.

Sites outside the protected area were typically accessed on a single day with one or two transects surveyed. Transects were established at 17 locations around the island (**Error! Reference source not found.**). Bird data were obtained from a total of 288 point counts, with 123 points in primary forest, 60 points in secondary forest and 105 points in plantation and cleared habitat (Table 3-2).

Table 3-2. Location of survey areas, transects and points completed (see also figure 3.1).

| # | Site name | Site location (UTM) | Dates | Nº transects | Nº of points |
|---|---|---|---|---|---|
| **1** | Matahit | 462825N 251768E | 18 Oct 2014 | 1 | 14 |
| **2** | Makatara | 470157N 250844E | 19 Oct 2014 | 1 | 10 |
| **3** | Bantane | 473242N 257895E | 23–25 Oct 2014 | 3 | 21 |
| **4** | Beo | 466769N 255552E | 27 Oct 2014 | 1 | 18 |
| **5** | Rae | 476353N 251816E | 30 Oct–1 Nov 2014 | 3 | 21 |
| **6** | Tarohan | 463893N 253094E | 3 Nov 2014 | 1 | 14 |
| **7** | Lalue | 494739N 254333E | 6–8 Nov 2014 | 3 | 16 |
| **8** | Ensem | 482815N 251446E | 13–15 Nov 2014 | 3 | 27 |
| **9** | Niampak | 453383N 251573E | 20–22 Nov 2014 | 3 | 30 |
| **10** | Beo road | 468710N 257734E | 24 Nov 2014 | 1 | 12 |
| **11** | Rainis road | 468710N 259285E | 24 Nov 2014 | 1 | 9 |
| **12** | Binalang | 479248N 258037E | 27–29 Nov2014 | 3 | 20 |
| **13** | Ambela | 449238N 246208E | 3–4 Dec 2014 | 2 | 32 |
| **14** | Lobbo | 472916N 246233E | 9 Dec 2014 | 1 | 8 |
| **15** | Sambuara | 482615N 243750E | 9 Dec 2014 | 1 | 7 |
| **16** | Rainis | 466276N 259282E | 10 Dec 2014 | 1 | 20 |
| **17** | Melonguane | 442609N 243712E | 12 Dec 2014 | 1 | 8 |

Points were established a minimum of 200 m apart and GPS waymarked. Following the recommendations of Lee and Marsden (2008), on Karakelang the reduction of double-counting was prioritised, with each point count restricted to 5 minutes in duration and divided into two equal halves to assess the impact on the density estimates of birds moving into the vicinity of the point during the count. No 'settling down' period was allowed—the count began upon arrival at the location (Marsden et al. 2006; Lee & Marsden 2008a), allowing any birds flushed at the point of arrival to have their estimated starting locations recorded at the start of the count (Jones et al. 1995). The observer learned the calls of the expected species during the two months prior to commencing fieldwork using material available through xeno-canto (www.xeno-canto.org). Several species or subspecies occurring on Talaud were not represented on xeno-canto, so an audio recording of each count period at each point was made. This was used to check the identifications of uncertain or similar-sounding species noted during the count. Where an identification was uncertain, this was followed up in the audio recording for the point to ensure accurate identification. One surveyor carried out all bird point counts accompanied by a local botanist and guide, ensuring consistency in procedure, bird identification and distance estimation.

Following the count, where possible distances to located birds or groups of birds were checked by laser rangefinder or by walking towards their noted locations (Buckland et al. 2008). Where groups of birds recorded during the count remained after completion, an attempt was made to accurately count the number of individuals present in the group to improve estimates of cluster size (Buckland et al. 2008). Perched, cryptic species were also searched for within the immediate vicinity of the point in the five minutes following completion (Buckland et al. 2008). Points were not carried out during rain or in wind speeds that adversely affected the observers' ability to detect birds. Records of birds in flight, except where they were flushed from the point, were excluded from the analyses, following Marsden (1999).

The timing of this study was based on the little information indicating that several species are breeding during October, including Red-and-blue Lory and the *Tanygnathus* parrots (Riley 1995; Lambert 1997), although Red-and-blue Lory may have two breeding periods within the year (Riley 1995, 2003). Several active nests of these species during the fieldwork suggests that, as much as there is a coordinated breeding season on the island, the survey was conducted within this period. Additionally, weather data indicated that this is the marginally drier season, potentially reducing the number of days unsuitable for data collection (Baden Pusat Statistik 2017b).

### *3.2.3 Habitat surveys*

During the survey, each point was assigned to one of the three large habitat categories—primary forest, secondary forest and mixed agriculture/plantations (Table 3-3)—for which an area measure had been calculated by Syarif (2004). This was based on an assessment of the canopy cover, tree branching structure, and evidence of clearance and/or crop species, cross-checked with a local guide familiar with the history and habitat of the island.

Table 3-3. Habitat type and area on Karakelang (Syarif 2004).

| Habitat type | Area (ha) | Percentage (%) |
|---|---|---|
| **Primary forest** | 26,044 | 32.5 |
| **Secondary forest** | 12,263 | 15.3 |
| **Mixed agriculture** | 23,279 | 29.0 |
| **Open land** | 15,197 | 18.9 |
| **Built land** | 821 | 1.0 |
| **No data** | 2,640 | 3.3 |
| Total | **80,243** | |

Subsequently, more detailed habitat information was collected at each survey point based on a 15-m radius area centred on the point location.  Variables measured were specifically chosen to relate to human modification, and were divided into three categories: geographical structure, vegetation structure and disturbance correlates. Geographical structure variables were: 1. altitude, measured using a handheld GPS unit in the field and later cross-referenced with the Shuttle Radar Topography Mission (SRTM) 30 m resolution Digital Elevation Model (DEM), and 2. a measure of slope, which was a mean of three gradient measurements taken with a clinometer at the centre and 5 metres each side at 90 degrees from each point.

Vegetation structure variables, measured within a 15 m radius of the point, were: 1. Canopy cover, the percentage vegetative cover above *c.* 15 m height, measured as the percentage obscured through a fixed diameter tube; 2. Ground cover, the percentage vegetative cover at ground level estimated as the percentage of bare ground obscured as viewed from standing height; 3. Mean girth of the three largest trees; 4. Mean of the estimated height of the three largest trees by girth; 5. Mean of the score assigned to an assessment of tree structure (Jones et al. 1995), whereby trees with the first branch inception above half the height of the 5. Mean of the score assigned to an assessment of tree structure tree (indicating that the tree had grown within a closed canopy) were scored as 1, trees branching above half height but with scars below (indicating growth in a closing gap or open secondary forest) scored 2, trees with multiple stems from low down (implying regrowth from clearance) scored 3, trees with the first branch inception below half the tree height were scored 4 and Coconut were scored as 5; and 6. Number of woody stems > 0.2 m diameter.

Disturbance correlates were: 1. Trail width, measured at the closest location of the trail to the point location; 2. Crop score derived from presence and abundance of coconut *Cocos nucifera* agg., banana *Musa* agg., clove *Eugenia aromatica*, nutmeg *Myristica fragrans*, other fruit trees (e.g. durian *Durio zibethinus*, *Lansium* spp., mango *Mangifera indica*) and field crops (cassava *Manihot esculenta*, groundnut *Arachis hypogaea*); and 3. Distance to a road, derived from a Geographical Information Services (GIS) road layer created from Geographical Positioning Service (GPS) tracks and through digitising satellite imagery in ArcGIS 10.0 (ESRI 2011).

### *3.2.4 Species richness and abundance distributions*

Species accumulation curves were created in EstimateS v.9.0 (Colwell 2013) to calculate estimated total species richness of each of the three broad forest habitat types using the Incidence Coverage-Estimator (ICE) (Chao et al. 2000) and produced sample based rarefaction curves to compare richness across habitats (Gotelli & Colwell 2001). Non-parametric tests (Kruskal-Wallis with post-hoc Dunn tests) in R (version 3.2.4: R Core Team 2017) were used to examine the difference between species richness between habitats. Rank Abundance plots (Whittaker 1965) were also created to examine the difference in evenness in the communities between the different habitats.

### *3.2.5 Density estimates and population sizes*

Density estimates were calculated for species with greater than 30 observations using Distance 7 Release 1 (Thomas et al. 2010). For each species, data were pooled across the three habitat types to create a single detection function for the species. Uniform, half-normal, hazard rate and negative exponential functions were assessed for each species, adjusted by varying binned data intervals and right truncating the data as deemed necessary through visual inspection of the frequency distribution (Thomas et al. 2010). The model with the lowest Akaike Information Criterion, as adjusted for small sample sizes, AICc (Burnham & Anderson 2002), was selected as the best fit (Thomas et al. 2010). The estimate was post-stratified by the three habitat types (primary, secondary and mixed agriculture/plantation); if the separate detection functions for each habitat resulted in lower AICc values than the pooled data then these were considered more reliable and used for generating density estimates for that species (Buckland et al. 2008).

For species groups considered likely to be similarly detectable throughout the island, a pooled detection function was used to reduce error in the density estimates. A pooled function was created for the three species of larger parrots in the genus *Tanygnathus*, for the three species of ground-dwelling rallids and for the two congeneric kingfishers, *Todiramphus chloris* and *T. enigma*. Using data from each set of species, a generic detection function was created through the conventional distance sampling engine in Distance 7.0. (Thomas et al. 2010). Data were pooled into intervals manually where this improved model fit (minimum AICc). The probability of observing one of the composite species of each set within this area, parameter *p*, and the standard error and degrees of freedom associated with *p*, were then entered as a multiplier for a subsequent density analysis for each species.

Separate population density estimates were calculated for the three habitat classes to increase the precision of the estimated total population size of each species. Densities are reported as individuals km$^{-2}$, with the coefficient of variance (CV: the standard error expressed as a proportion of the density estimate) and 95% confidence intervals (CI). Population estimates are given as the 95% CI range of number of individuals from the summation of the estimated population in each habitat, derived from the mean density estimate of the species in each habitat category multiplied by the area of that habitat as given in Table 3-3.

### *3.2.6 Habitat analysis*

A Principal Components Analysis of the habitat disturbance variables was carried out using package ‘vegan’ (Oksanen et al. 2015) in R (R Core Team 2017), with the eigenvalues used to rank each point along the first principal component axis (PC1). The points were then grouped into eight equal-sized ‘modification bands’ (see Fig. 3-5) of 36 points. Correlations of the collected habitat variables were investigated using Spearman’s rank correlation.

### *3.2.7 Density estimates along the modification gradient*

The same species detection functions derived for the population estimates were used to generate post-stratified density estimates at each of the eight modification bands. All estimates thus assumed detectability for each species was the same throughout the modification gradient. The density for each species is presented as the mean number of individuals $km^{-2}$ ± SE at each modification band, allowing a direct representation of a species's response to the composite modification gradient.

The package 'mgcv' (Wood 2011) was used to fit Generalised Additive Models (GAMs) with increasing splines to assess the linearity of the abundance response to the modification gradient. Splitting the modification gradient into eight groups permits an assessment of nonlinearity in the abundance response by testing for improved fit through adding more splines to the model.

The percentage density change between each modification band was calculated for each of the three groups of species to identify the point along the gradient at which the greatest density changes were occurring.

## 3.3 Results

### 3.3.1 *Bird community richness*

A total of 2,863 records of perched birds of 54 species were recorded at the 288 point counts. Of these 86% were detected by sound only, and many of the visual records were also heard. Method of detection was therefore not considered to significantly alter the detection function of the species for which density estimates were calculated. There was also little consistent variation in the number of detections recorded with time from sunrise (**Error! Reference source not found.**), consequently all points and all records of perched birds were included in subsequent analyses. A total of 140 points were within the protected areas, with 106 points within the Northern Wildlife Reserve and 34 in the Southern Wildlife Reserve.

There were minor but significant differences in mean bird species richness per point between habitats (Kruskall-Wallis, $H = 51.9$, $n = 288$, $df = 2$, $p < 0.001$), with higher species richness per point in both primary (mean = 8.2, sd = 2, n = 123; Dunn's test $Z = 7.2$, $p < 0.001$) and secondary forest (mean = 7.3, sd = 2.2, n = 60; $Z = 3.76$, $p < 0.001$ ) than in plantations (mean = 5.7, sd = 2.7, n = 105) (Figure 3-2). In contrast, the observed overall species richness was higher in plantation habitat (44 species) than both secondary (38 species) and primary forest (35 species), with the species accumulation curves appearing to be close to saturation in each habitat and this finding repeated by both the ICE and Chao2 richness estimator (Appendix 3.3).

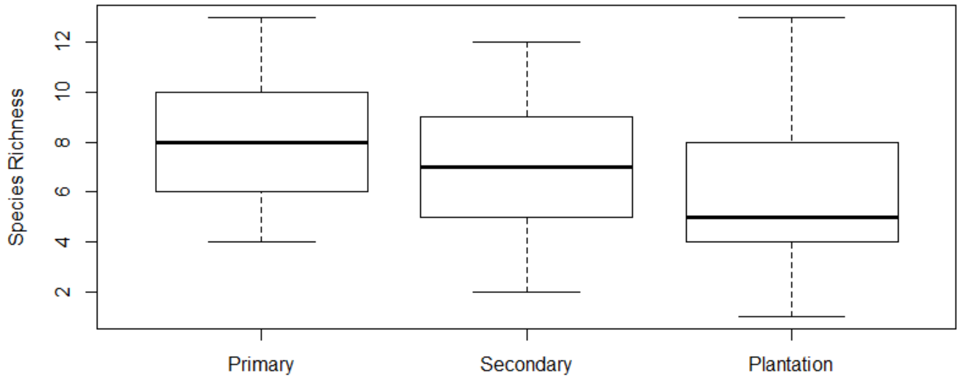


Figure 3-2. Mean species richness per point in the three habitat categories

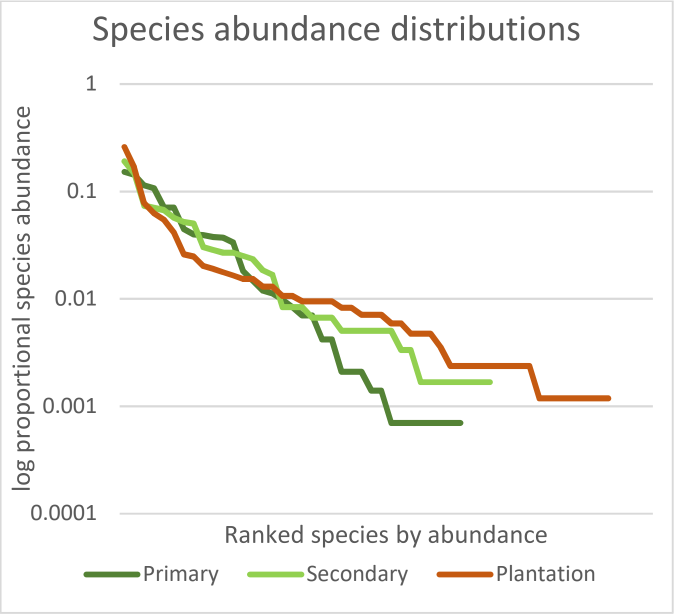


Figure 3-3. Proportional abundance versus rank order plot for all species recorded in the three habitat classes.

Species abundance distributions in the three habitats differed between the three habitat classes (Figure 3-3). A larger 'tail' of species occurring at low abundance is evident in plantation and secondary in comparison to primary habitat, where abundance has a far more equitable distribution.

### 3.3.2 *Density and population estimates for key species on Karakelang*

Density estimates for 21 species are presented in Table 3-4. Density estimates for the bird community on Karakelang in 2014 (individuals $km^{-2}$ ± SE, 95% CI below. * = species for which a pooled detection function based on similar species was used to generate the density estimates.. Percentage CV values were below 30% for 30 of the total of 63 values, and exceeded 50% for eight. The pooled detection function enabled a density estimate for Great-billed Parrot and Collared Kingfisher, both species with fewer than 30 records and for which density would otherwise not have been estimated. However, estimates were not generated for Rufous-vented Bush-hen or Talaud Rail due to the tiny number of records of those species. CV values for the remaining species contributing to the pooled detection function were reduced, hence these values were used for the density estimates for these species.

Table 3-4. Density estimates for the bird community on Karakelang in 2014 (individuals $km^{-2}$ ± SE, 95% CI below. * = species for which a pooled detection function based on similar species was used to generate the density estimates.

| | Primary (123) | Secondary (60) | Mixed agriculture /plantation (106) |
|---|---|---|---|
| **Columbidae** | | | |
| Green Imperial-pigeon | 166 ± 33 (20%) | 129 ± 48 (37%) | 8 ± 4.9 (64%) |
| | 113–245 | 63–262 | 2–24 |
| Blue-tailed Imperial-pigeon | 215 ± 16 (7%) | 78 ± 15 (18%) | 27 ± 8.4 (31%) |
| | 186–248 | 55–114 | 15–49 |
| Grey Imperial-pigeon | 9 ± 3.3 (36%) | 33 ± 11 (33%) | 20 ± 6.6 (34%) |
| | 5–18 | 18–63 | 10–37 |
| Black-naped Fruit-dove | 311 ± 25 (8%) | 372 ± 38 (10%) | 218 ± 24 (11%) |
| | 265–365 | 305–455 | 176–272 |
| **Cuculidae** | | | |
| Lesser Coucal | 6.6 ± 2.9 (45%) | 11 ± 5 (48%) | 28 ± 9 (33%) |
| | 2.8–15 | 4.6–28 | 15–54 |
| Western Koel | 36 ± 6 (18%) | 56 ± 10 (19%) | 55 ± 9 (17%) |
| | 24–49 | 39–80 | 39–76 |
| **Rallidae** | | | |
| Talaud Bush-hen* | 32 ± 12 (37%) | 14 ± 10 (75%) | 2 ± 2 (104%) |
| | 16–64 | 3.6–52 | 0.4–11 |
| **Alcedinidae** | | | |
| Collared Kingfisher* | – | 7.7 ± 3.9 (51%) | 21 ± 7 (36%) |
| | – | 3–20 | 10– 41 |
| Talaud Kingfisher* | 18 ± 5.7 (32%) | 17 ± 6 (35%) | 8.8 ± 4 (42%) |

|  |  |  |  |
|---|---|---|---|
| *Todiramphus enigma* | 10–33 | 8–34 | 4–19 |
| Red-and-blue Lory | 102 ± 21 (20%) | 47 ± 16 (34%) | 9 ± 3.5 (39%) |
|  | 69–151 | 24–91 | 4–19 |
| Golden-mantled Racquet-tail | 56 ± 13 (22%) | 25 ± 7.6 (30%) | 9 ± 3.9 (41%) |
|  | 36–86 | 14–45 | 4–21 |
| Great-billed Parrot* | 3 ± 1 (32%) | 0.4 ± 0.4 (101%) | 0.2 ± 0.2 (101%) |
|  | 1.8–6 | 0.08–2.2 | 0.05–1.3 |
| Blue-naped Parrot* | 16 ± 5 (29%) | 32 ± 8 (26%) | 12 ± 5 (38%) |
|  | 9–29 | 19–52 | 6–26 |
| Blue-backed Parrot* | 7 ± 1.7 (24%) | 2 ± 1.1 (54%) | 2 ± 0.7 (39%) |
|  | 5–11 | 1–6 | 1–4 |
| **Pittidae** |  |  |  |
| Talaud Pitta | 74 ± 10 (14%) | 51 ± 11 (22%) | 10 ± 3 (31%) |
|  | 57–97 | 33–79 | 6–19 |
| **Oriolidae** |  |  |  |
| Black-naped Oriole | 120 ± 13 (11%) | 49 ± 9 (18%) | 11 ± 4 (35%) |
|  | 98–148 | 34–70 | 6–22 |
| **Monarchidae** |  |  |  |
| Island Monarch | 129 ± 21 (16%) | 53 ± (33%) | 2 ± 2 (100%) |
|  | 95–176 | 28–101 | 0.4–10.4 |
| Rufous Paradise-flycatcher | 159 ± 25 (16%) | 157 ± 28 (18%) | 113 ± 22 (19%) |
|  | 117–216 | 111–223 | 77–164 |
| **Locustellidae** |  |  |  |
| Gray's Grasshopper-warbler | 11 ± 8 (72%) | 188 ± 53 (28%) | 230 ± 48 (21%) |
|  | 3–40 | 108–327 | 154–345 |
| **Dicaeidae** |  |  |  |
| Grey-sided Flowerpecker | 426 ± 96 (22%) | 683 ± 193 (28%) | 542 ± 135 (25%) |
|  | 275–660 | 395–1182 | 334–878 |
| **Nectariniidae** |  |  |  |
| Black Sunbird | 1166 ± 122 (11%) | 1433 ± 170 (12%) | 1361 ± 117 (9%) |
|  | 948–1432 | 1134–1811 | 1150–1612 |

Density estimates for nine of the 21 species exceeded 100 individuals km$^{-2}$ in primary forest (Table 3-4. Density estimates for the bird community on Karakelang in 2014 (individuals km$^{-2}$ ± SE, 95% CI below. * = species for which a pooled detection function based on similar species was used to generate the density estimates.. The two nectivorous passerines, the sunbird and flowerpecker, were the most abundant species pair in all habitats, with more half of all birds on the island estimated to be Black Sunbirds. The endemic species all reached their highest densities in primary forest habitat. Four of the five parrot species, including the Endangered Red-and-blue Lory, and two of the three imperial-pigeons were most abundant in primary forest. Only three Least Concern species achieved their highest density in the mixed agriculture/plantation habitat, yet all species were recorded in this habitat class. Six species attained their highest density in secondary forest, including three frugivores. Two of the three *Tanygnathus* parrots had the lowest overall population estimates of the species for which estimates could be made. Great-billed Parrot *T. megalorynchos* was the rarest, with fewer than 2,000 individuals estimated. Population estimates could not be created for one Endangered and two Near Threatened species due to the small number of encounters. No raptor species was recorded sufficiently frequently to calculate a population estimate, and this group appeared virtually absent from primary forest. Overall, the island is characterised by large populations of frugivorous and nectarivorous species, especially in primary forest habitat.

Table 3-5. Population estimates for 21 bird species of Karakelang, calculated with the revised area figures presented in Table 5. Bold figures are derived from estimates <30% CV. Figures in brackets are derived from estimates >60% CV and are excluded from the total population estimates.

| | **Primary** | **Secondary** | **Kebun** | **Total** |
|---|---|---|---|---|
| Green Imperial-pigeon | **28,360–61,500** | 7,750–32,230 | (464–5,592) | 36,111–93,730 |
| Blue-tailed Imperial-pigeon | **46,690–62,250** | **6,770–14,020** | 3,500–11,420 | 56,590–87,690 |
| Grey Imperial-pigeon | 1,255–4,520 | 2,214–7,750 | 2,330–8,620 | 5,799–20,890 |
| Black-naped Fruit-dove | **66,520–91,620** | **37,520–55,970** | **41,010–63,380** | **145,050–210,970** |
| Lesser Coucal | 700–3,770 | 570–3,440 | 3,495–12,580 | 4,765–19,790 |
| Western Koel | **6,020–12,300** | **4,800–9,840** | **9,090–17,710** | **19,910–39,850** |
| Talaud Bush-hen* | **4,020–16,100** | (440–6,400) | (90–2,560) | 4,020–16,100 |
| Collared Kingfisher* | – | 370–2,460 | **2,330–9,550** | 2,700–12,010 |
| Talaud Kingfisher | 2,510–8,280 | 980–4,180 | 930–4,430 | 4,420–16,890 |
| Red-and-blue Lory | **17,320–37,900** | 2,950–11,193 | 930–4,890 | 21,200–53,983 |
| Golden-mantled Racquet-tail | **9,040–21,590** | **1,720–5,540** | 932–4,890 | 11,692–32,020 |
| Blue-naped Parrot | **2,260–7,280** | **2,340–6,400** | 1,400–6,060 | 6,000–19,740 |
| Great-billed Parrot* | **450–1,506** | (10–246) | (12–300) | 450–1,506 |
| Blue-backed Parrot | **1,260–2,760** | 120–740 | 230–930 | 1,610–4,430 |
| Talaud Pitta | **14,300–24,350** | **4,060–9,720** | 1,400–4,430 | 19,760–38,500 |
| Black-naped Oriole | **24,600–37,150** | **4,182–8,610** | 1,400–5,130 | 30,182–50,890 |
| Island Monarch | **23,850–44,180** | 3,440–12,420 | (90–2,330) | 27,290–56,420 |
| Rufous Paradise-flycatcher | **29,370–54,220** | **13,650–27,430** | **17,940–38,210** | 60,960–119,860 |
| Gray's Grasshopper-warbler | (750–10,040) | **13,280–27,430** | **35,880–80,390** | 49,160–107,820 |
| Grey-sided Flowerpecker | **69,030–165,660** | **48,560–145,390** | **77,820–204,570** | **195,410–515,620** |
| Black Sunbird | **237,950–359,430** | **139,480–222,750** | **267,950–375,600** | **645,380–957,780** |

### *3.3.3 Modification gradient derived from ordination of habitat variables*

The first principal components axis was clearly associated with a gradient of disturbance, with points grouped into the three habitat classes (primary, secondary and mixed agriculture/plantation) assigned in the field (Figure 3-4). This axis explained 47% of the variation within the habitat variables, and high values of isolation (distance from road), tree height, girth and number of stems contrast with high values for tree structure (primary structural form scored lowest, coconut highest) and agricultural intensity (Table 3-6). The second principal component axis is less simple, relating partly to topography, with the highest loadings for low values of altitude. Accordingly, the first axis alone was used to rank points by their disturbance score.

Figure 3-5 provides a summary of the gradient, with photographs taken at point locations ranked by PCA1 into the eight groups described. This first PCA axis represented a compound variable of habitat modification, with each variable measured changing in the direction expected from greatest to least modification (Figure 3-6a-j). Most variables displayed a near-linear response along the gradient but clear steps are evident in the direct measures of agriculture and primary structure, which were correlated (Spearman's rank, $r^2$ 0.68; Figure 3-7). Bands 6 to 8 are effectively primary forest (Figure 3-5), with mean canopy cover exceeding 70% (Figure 3-6c) and the majority of trees characterised as having primary structure Figure 3-6g).

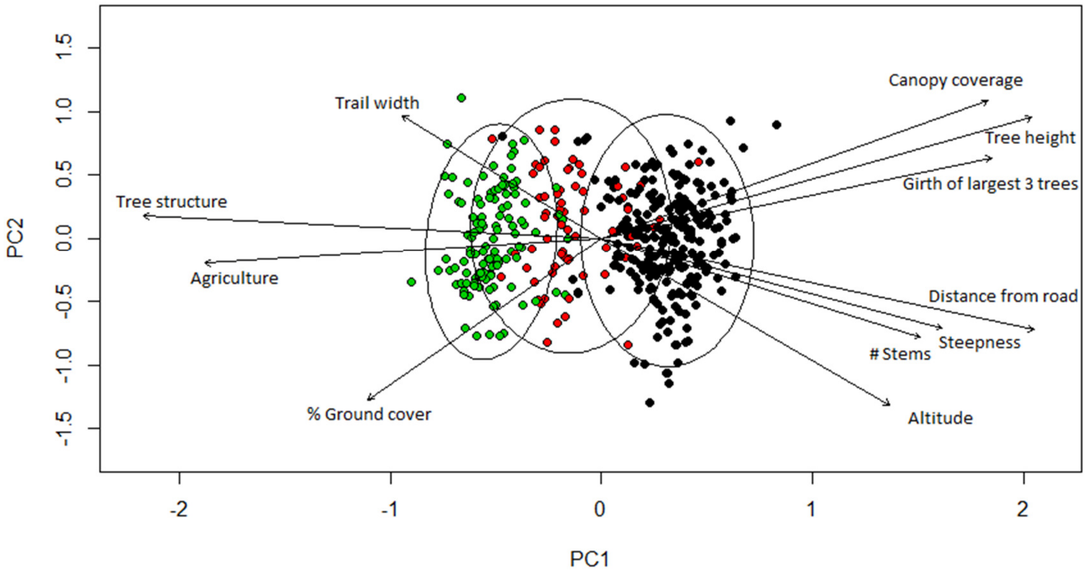


Figure 3-4. Biplot of points along PC1 and PC2 axes. PC1 explained 46.9% of the variance, PC2 12.4%. Black points are those recorded in the field as primary forest, red points secondary and green points were mixed agriculture/plantation, with ellipses containing a central 95% of points determined to belong in each category.

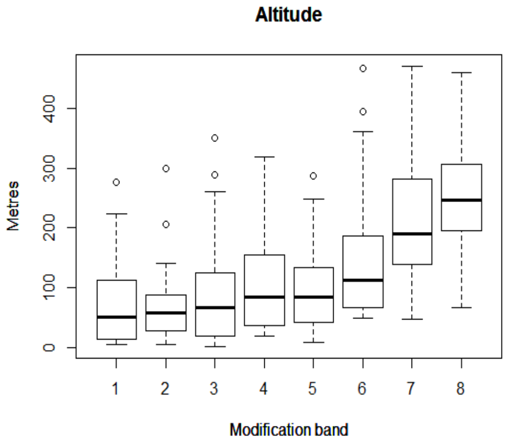
Table 3-6. Loadings of variables from the first two PC axes, ordered by eigenvalues of PC1. Numbers in bold exceed the square root of 1/number of variables.

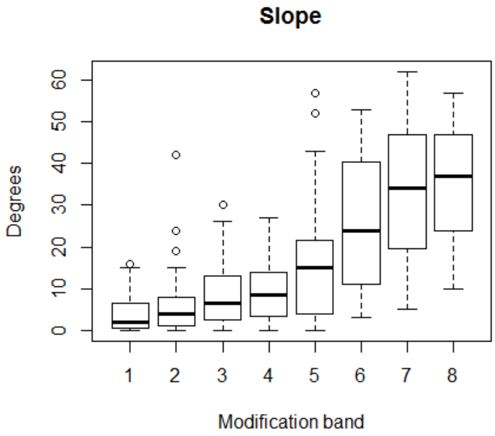
| **Variable** | **PC1** | **PC2** |
|---|---|---|
| Tree height | **0.36** | 0.26 |
| Distance from road | **0.33** | -0.29 |
| Girth of 3 largest trees | **0.33** | 0.30 |
| # Stems | **0.32** | 0.13 |
| Canopy coverage | 0.31 | 0.26 |
| Steepness | 0.29 | **-0.37** |
| Altitude | 0.28 | **-0.49** |
| Ground cover | -0.08 | **-0.52** |
| Trail width | -0.18 | 0.15 |
| Agriculture | **-0.34** | -0.05 |
| Tree structure | **-0.37** | 0.01 |

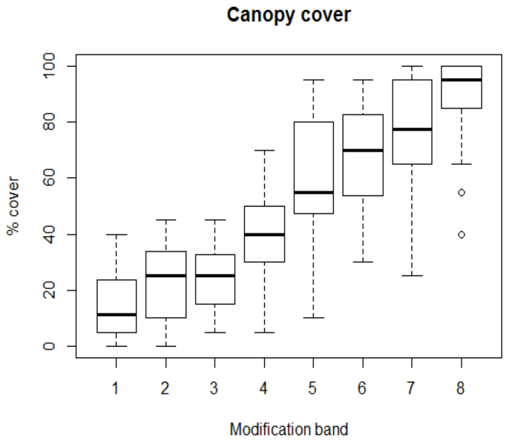
¦----------1----------¦------------2----------¦-----------3-----------¦------------4-----------¦-----------5-----------¦-----------6------------¦-----------7------------¦--------8---------¦

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Band 1. Plantation agriculture with previously cleared and burnt understorey. | Band 2. Mixed plantation agriculture. Several crop species, few remnant native trees: often the ground will have been burnt following clearance. | Band 3. Garden agriculture or *kebun* with retained forest trees and mature fruit trees. Small areas are cleared but individual tall trees and patches of mature vegetation remain immediately adjacent. Some trees may be remnants from primary forest, others are mature secondary growth. | Band 4: Secondary regrowth with no or very few large stems; most trees are pioneer species. Broken, low canopy of small trees. Dense ground layer with grass and ferns, often gingers. Often adjacent to plantation or *kebun* areas, and frequently with crop species planted among the regrowing trees. Some areas dominated by planted timber species, e.g. *Albizia* spp. | Band 5. Mature secondary forest or part-cleared primary forest. Cleared or mostly cleared >20 years previously, trees now mature but often with very dense lower strata and broken canopy, large gaps between large stems. Often the trees left are native fruit trees. Banana is occasionally present, but usually there are no planted crop trees. | Band 6. Degraded primary forest with some canopy gaps from logging or landslips; dense understorey, often climbers in midstorey, high abundance of rattan and ginger and occasional bananas. Numerous trees exceeding 20 m in height, and clear upper-, mid- and lower strata present. | Band 7: Primary forest with selective logging, here *Diaspyros* spp. Some canopy gaps, where patches of gingers and rattan are denser. | Band 8. Primary forest, with some very large-girthed and tall trees. Frequent rattan and ginger but sparse understorey and low percentage ground cover. Many lianas. |

Figure 3-5. Photographs of representative point count locations classified in each band of the modification gradient, to demonstrate the degree of habitat change that is represented along the gradient,


Altitude
Metres
0
100
200
300
400
1
2
3
4
5
6
7
8
Modification band


Slope
Degrees
0
10
20
30
40
50
60
1
2
3
4
5
6
7
8
Modification band


Canopy cover
% cover
0
20
40
60
80
100
1
2
3
4
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6
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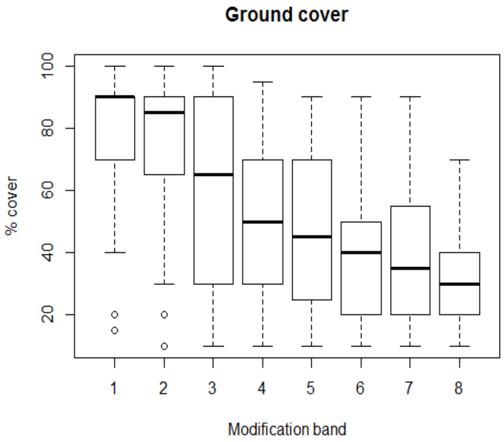
Modification band


Ground cover
% cover
20
40
60
80
100
1
2
3
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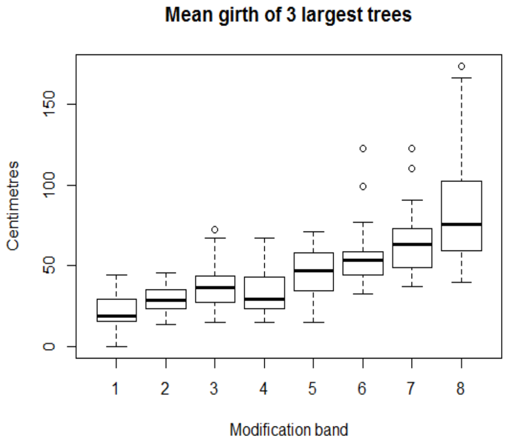
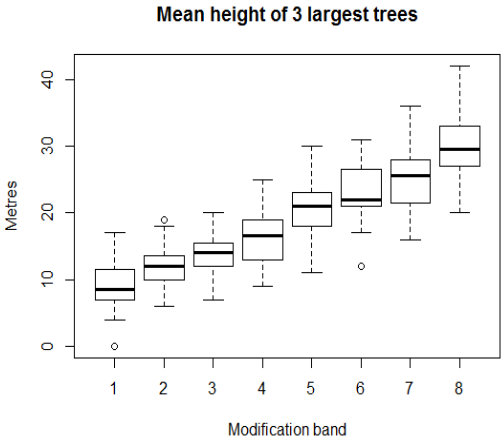
7
8
Modification band


Mean girth of 3 largest trees
Centimetres
0
50
100
150
1
2
3
4
5
6
7
8
Modification band


Mean height of 3 largest trees
Metres
0
10
20
30
40
1
2
3
4
5
6
7
8
Modification band

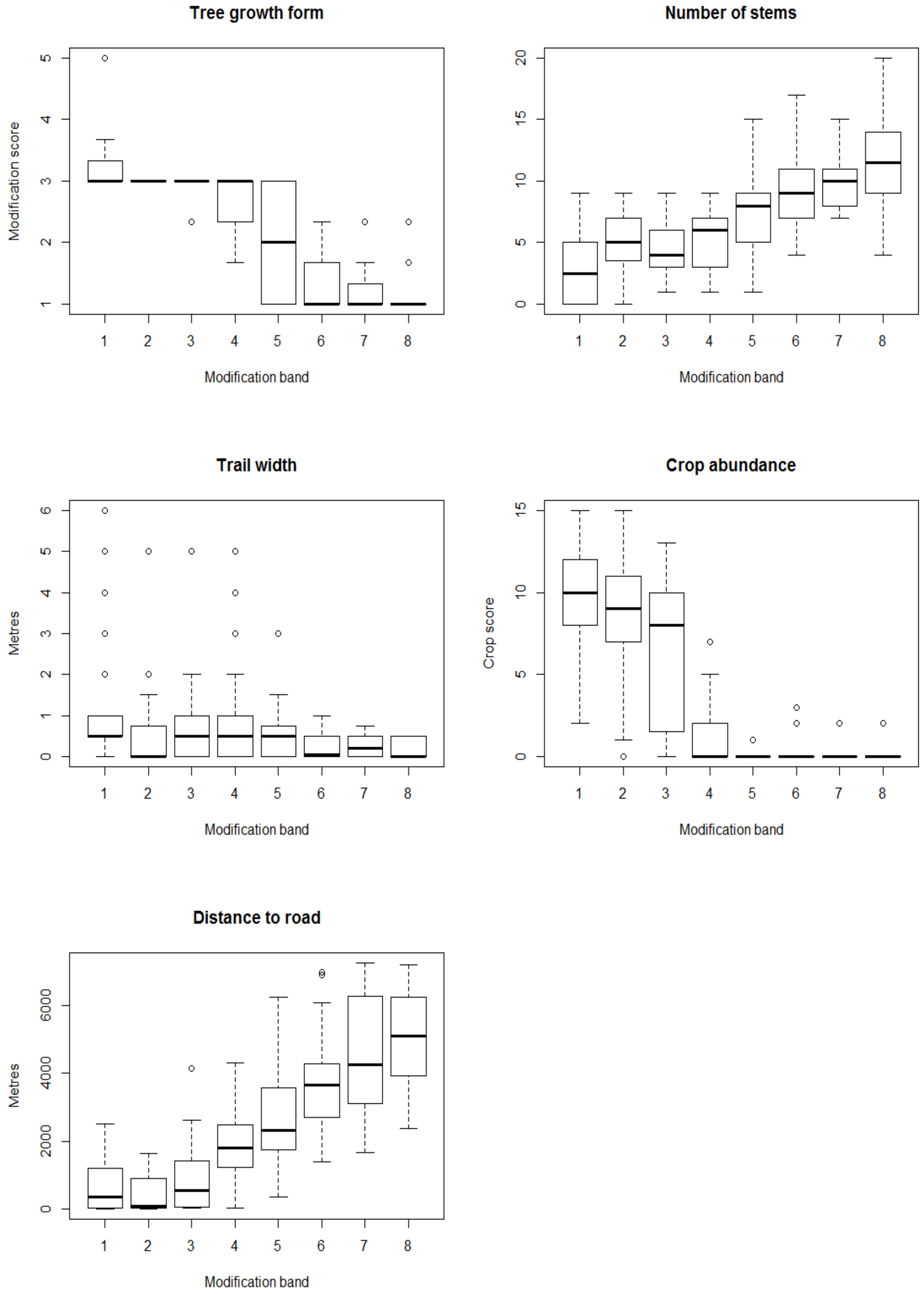


Figure 3-6. Habitat variables at each modification band, moving from most (band 1) to least (band 8) disturbed; a) Altitude (metres above sea level), b) Slope (absolute gradient mean in degrees), c) Mean estimated percentage canopy cover, d) Mean estimated percentage ground cover, e) Mean girth (diameter at breast height), e) Mean estimated tree height (metres), f) Mean tree structure form score, g) Mean count of tree stems, h) Mean width of trail (metres), i) Mean crop abundance score, j) Distance to road (isolation).

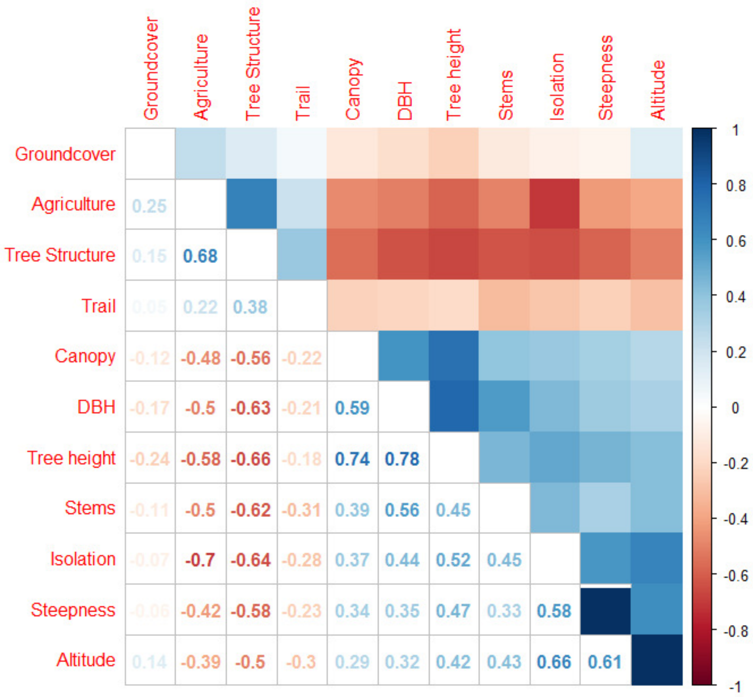


Figure 3-7. Spearman’s rank correlation of habitat variables, negative correlations in red, positive correlations in blue, with greater colour intensity for stronger correlations.

Points in each group were well distributed through the island (Appendix 3-5): no significant spatial autocorrelation was found at the island scale (Moran’s I, inverse distance squared, z = 0.31, p = 0.75).

### 3.3.4 *Species responses to the habitat modification gradient*

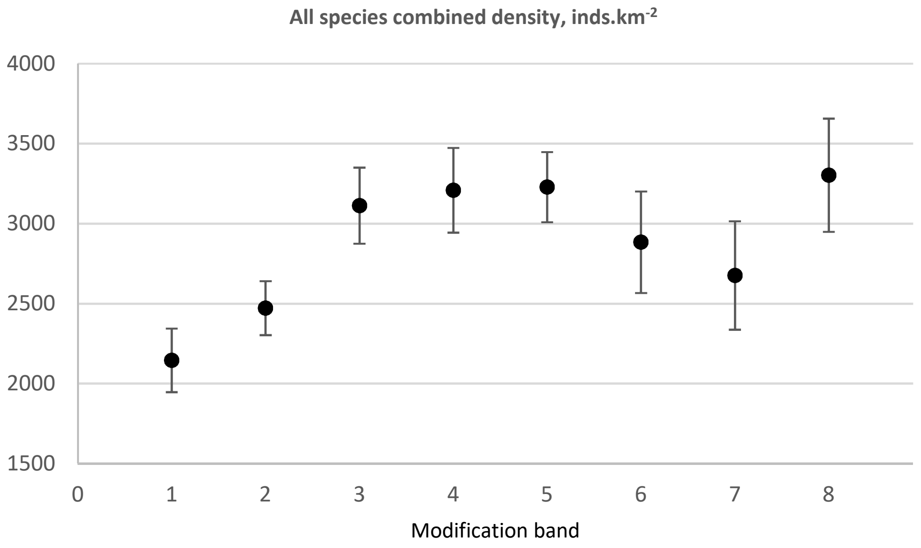


Figure 3-8. Overall density of birds across the modification gradient.

The total density of birds on Karakelang, all species combined, increases towards an intermediate peak of abundance along the modification gradient, but the intact primary forest band eight exceeds this with the highest value of 3,303 individuals km$^{-2}$ (Figure 3-8). The two most disturbed bands support the fewest overall numbers of birds. While density remains roughly similar, the actual mass represented by the numbers of birds in the more intact bands far exceeds that supported by the other bands. The species in the group adversely affected by modification are larger and heavier species, to the extent that more than 500 kg of bird biomass is present within one square kilometre of intact primary forest.

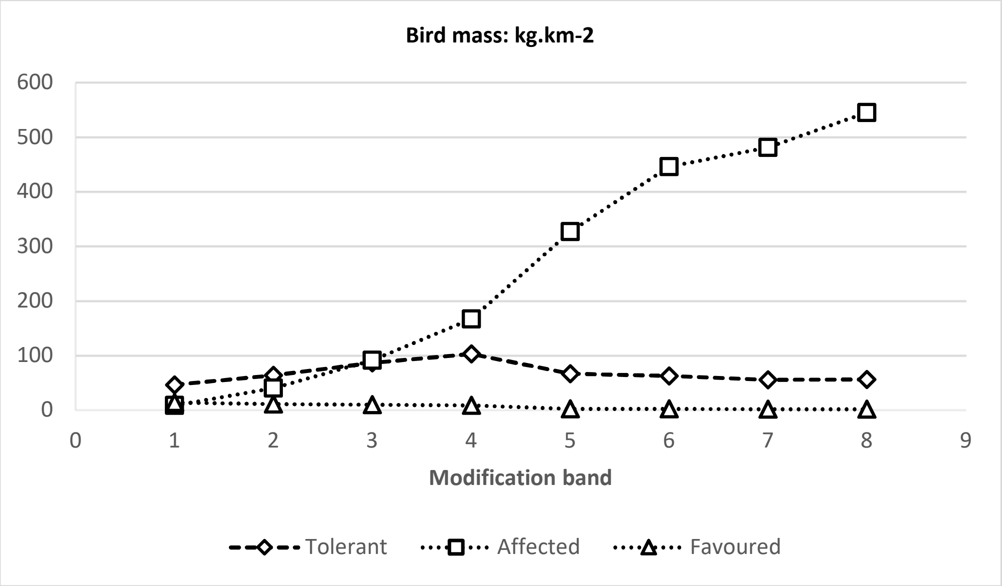


Figure 3-9. Summed total of absolute bird mass present at each modifcation band, split by the three species groups. Species mass is mean mass value from Wilman et al. (2014) multiplied by the mean density of that species at each modification band.

Table 3-7. Summed mass values (kilograms km$^{-2}$) for species at each modification band.

| *Band* | *Unaffected* | *Positively affected* | *Negatively affected* | *Total biomass* |
|---|---|---|---|---|
| ***1*** | 46 | 13 | 8 | 67 |
| ***2*** | 63 | 10 | 40 | 113 |
| ***3*** | 86 | 9 | 91 | 186 |
| ***4*** | 102 | 8 | 166 | 277 |
| ***5*** | 66 | 1 | 327 | 394 |
| ***6*** | 62 | 1 | 446 | 509 |
| ***7*** | 55 | 1 | 481 | 537 |
| ***8*** | 56 | 1 | 545 | 602 |

Of the 21 species for which densities were calculated, eleven demonstrated a negative response to modification (Figure 3-10 to Figure 3-20). This ‘negatively affected’ group includes three of Talaud’s five endemic species, three endemic subspecies and four of the island’s five parrot species. Most of these showed very high adjusted $R^2$ values (> 0.85; Table 3-8), suggesting that a simple linear response explained much of the variation in their abundance along the modification gradient. However, for 18 of the 21 species, a non-linear term improved the model fit and the density of one additional species is significantly related to to higher values of intact forest (Southern Rufous Paradise-flycatcher, Figure 3-20). Across the full gradient measured, five species demonstrated a sigmoidal abundance response (Blue-tailed Imperial-pigeon [Figure 3-12], Great-billed Parrot [Figure 3-16], Talaud Pitta [Figure 3-17], Black-naped Oriole [Figure 3-18] and Island Monarch [Figure 3-19]), while three species appeared to decline at a rate greater than linear, including the endemic Talaud Bush-hen (Figure 3-10) and two parrot species (Golden-mantled Racquet-tail [Figure 3-14] and Blue-backed Parrot [Figure 3-15]). For these two latter species, density declined even with minor selective logging. Most of the ‘unaffected species’ actually demonstrated a density peak at intermediate levels of modification: Black-naped Fruit-dove (Figure 3-24) had a large abundance spike in band 4.

Table 3-8. Linear regression of mean density against modification band for the 21 species. $^{V}$ = significant, with a non-linear term, modification-favoured species. ^ = significant with a non-linear term, modification-affected species.

| | Slope (SE) | Adjusted | F (1,6) | p |
|---|---|---|---|---|
| **Species increasing in density with more intact habitat = “Negatively affected species”** | | | | |
| *Great-billed Parrot* | 5.15 (1.89) | 0.48 | 7.44 | 0.034* |
| *Monarch* | 4.62 (0.92) | 0.77 | 25 | 0.003** |
| *Ducula aenea* | 4.44 (0.55) | 0.90 | 66.14 | <0.001*** |
| *Tanygnathus sumatranus* | 4.08 (1.17) | 0.61 | 12.11 | 0.013* |
| *Ducula concinna* | 3.98 (0.49) | 0.90 | 65.02 | <0.001*** |
| *Oriolus chinensis* | 3.97 (0.30) | 0.96 | 173.2 | <0.001*** |
| *Prioniturus platurus* | 3.95 (0.59) | 0.86 | 45.13 | <0.001*** |
| *Amaurornis magnirostris* | 3.87 (1.23) | 0.56 | 9.961 | 0.020* |
| *Erythropitta inspeculata* | 3.47 (0.52) | 0.86 | 44.29 | <0.001*** |
| *Eos histrio* | 3.35 (0.85) | 0.67 | 15.34 | 0.008** |
| | | | | |
| **Species decreasing in density with more intact habitat = “Positively affected species”** | | | | |

| | | | | |
|---|---|---|---|---|
| *Locustella fasciolata* | -4.31 (0.89) | 0.76 | 23.26 | 0.003** |
| *Centropus bengalensis* | -3.07 (1.05) | 0.52 | 8.636 | 0.026* |
| *Todiramphus chloris* | -6.88 (2.76) | 0.43 | 6.219 | 0.047* |
| | | | | |
| **Species showing no significant trend along the modification gradient = "Unaffected species"** | | | | |
| *Ducula pickeringii* | -1.26 (1.93) | -0.09 | 0.43 | 0.539 |
| *Eudynamys scolopaceus* | -0.99 (0.52) | 0.27 | 3.61 | 0.106 |
| *Leptocoma aspasia* | -0.63 (0.29) | 0.35 | 4.69 | 0.07$^{V}$ |
| *Dicaeum celebicum* | -0.44 (0.83) | -0.11 | 0.28 | 0.616 |
| *Tanygnathus lucionensis* | -0.19 (0.77) | -0.16 | 0.03 | 0.863 |
| *Ptilinopus melanospilus* | 0.55 (0.47) | 0.05 | 1.38 | 0.285 |
| *Terpsiphone cinnamomea* | 0.75 (0.32) | 0.39 | 5.51 | 0.057^ |
| *Todiramphus enigma* | 0.99 (0.71) | 0.12 | 1.93 | 0.214 |

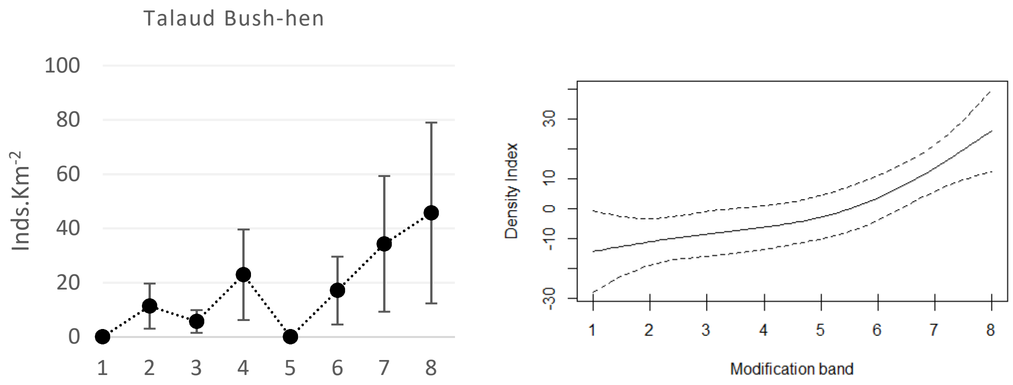


Figure 3-10. Talaud Bush-hen: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 4, R$^2$ (adj) = 0.68; F$_{(2.08, 2.47)}$ = 6.71, p = 0.039.

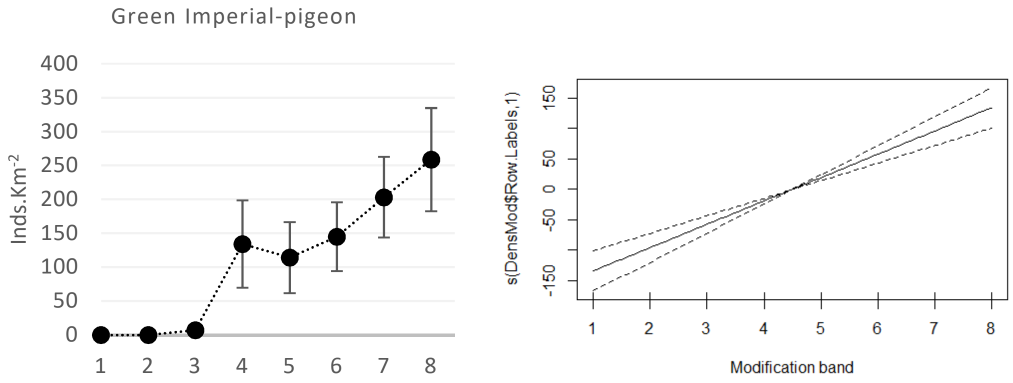


Figure 3-11. Green Imperial-pigeon: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, R$^2$ (adj) = 0.9; F$_{(1, 1)}$ = 66.1, p <0.001.

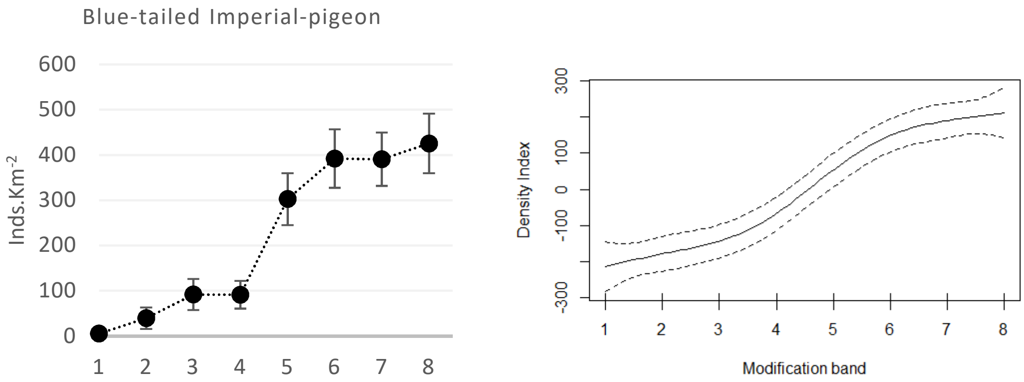


Figure 3-12: Blue-tailed Imperial-pigeon: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 6, $R^2$ (adj) = 0.95; $F_{(3.45,\ 4.09)}$ = 31.3, p = 0.003.

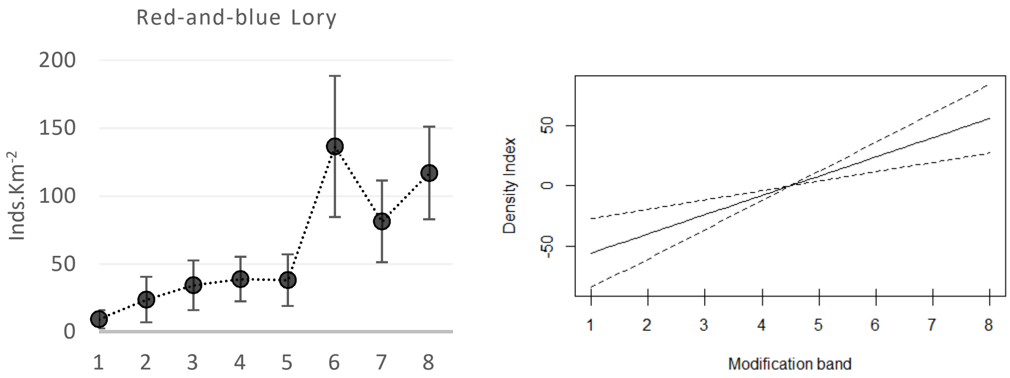


Figure 3-13. Red-and-blue Lory: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, $R^2$ (adj) = 0.67; $F_{(1,1)}$ = 15.3, p = 0.007.

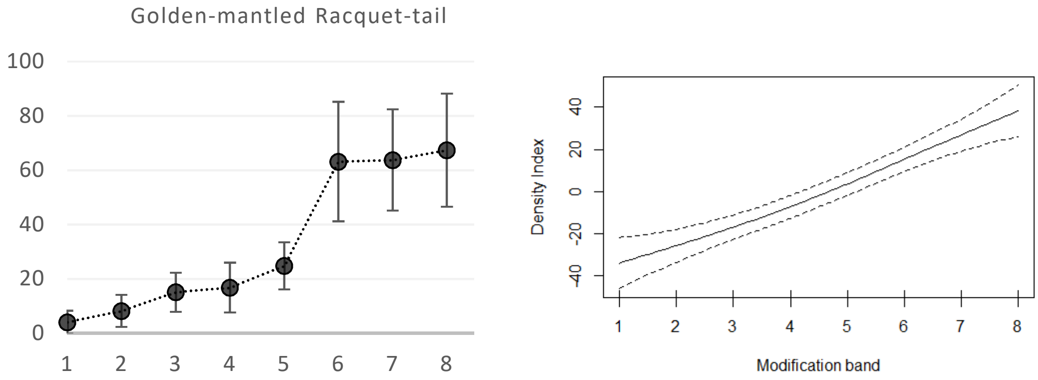


Figure 3-14. Golden-mantled Racquet-tail: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 6, $R^2$ (adj) = 0.88; $F_{(1.49,1.82)}$ = 26.18, p < 0.001.

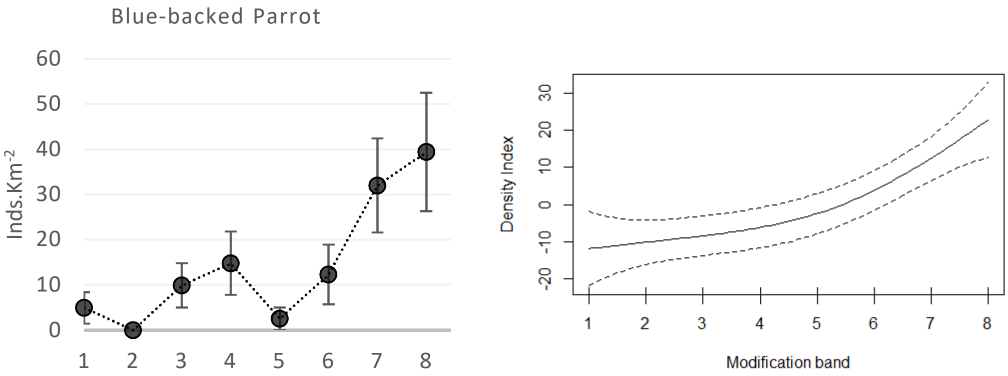


Figure 3-15. Blue-backed Parrot: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, $R^2$ (adj) = 0.67; $F_{(1,1)} = 15.3$, p = 0.007.

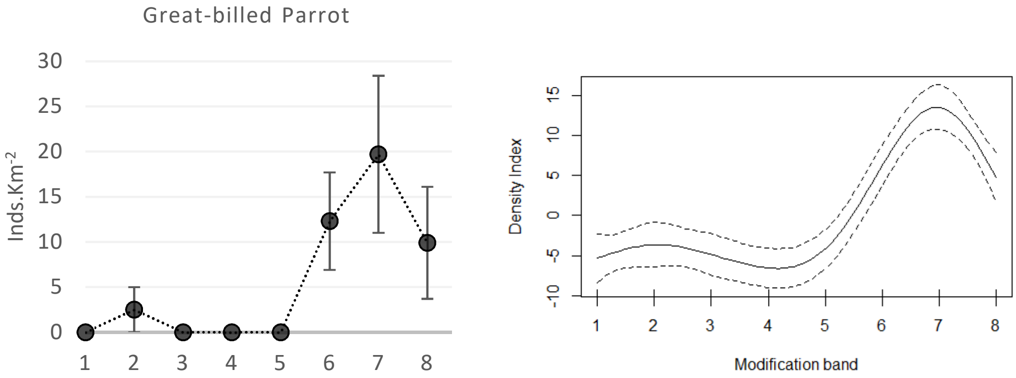


Figure 3-16. Great-billed Parrot: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 7, $R^2$ (adj) = 0.95; $F_{(5.35,\ 5.83)} = 23.2$, p = 0.06.

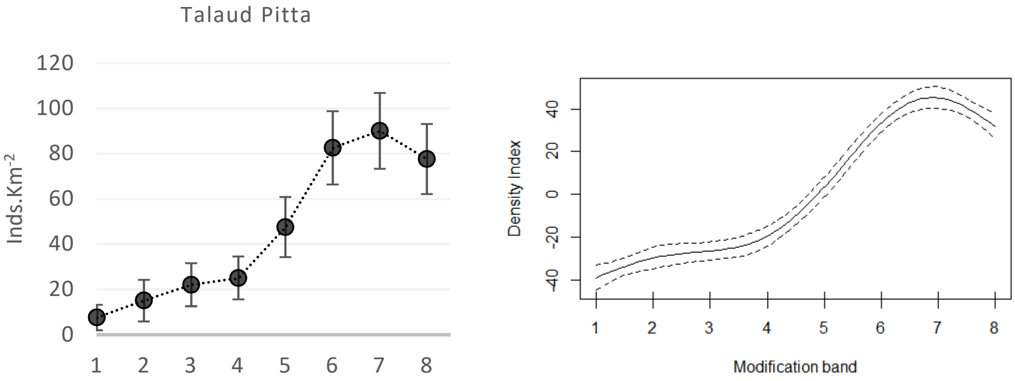


Figure 3-17. Talaud Pitta: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 6, $R^2$ (adj) = 0.99; $F_{(4.66,\ 4.943)} = 142.9$, p <0.001.

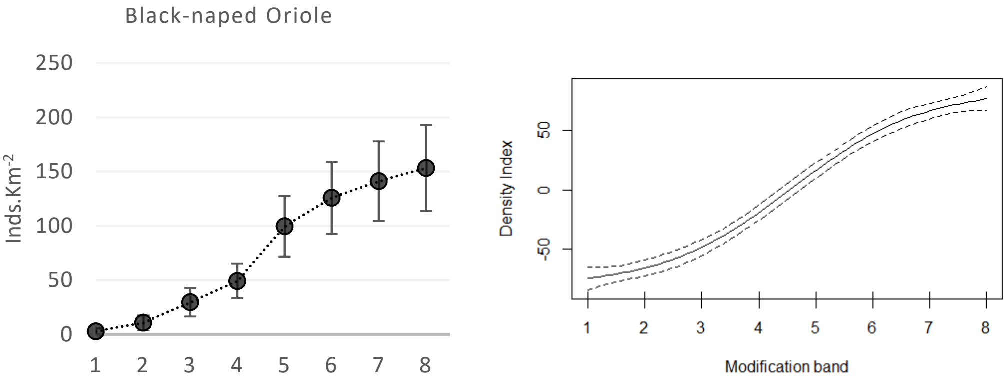


Figure 3-18. Black-naped Oriole: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 5, R$^2$ (adj) = 0.99; $F_{(3.45,\ 3.83)}$ = 197.8, p <0.0001.

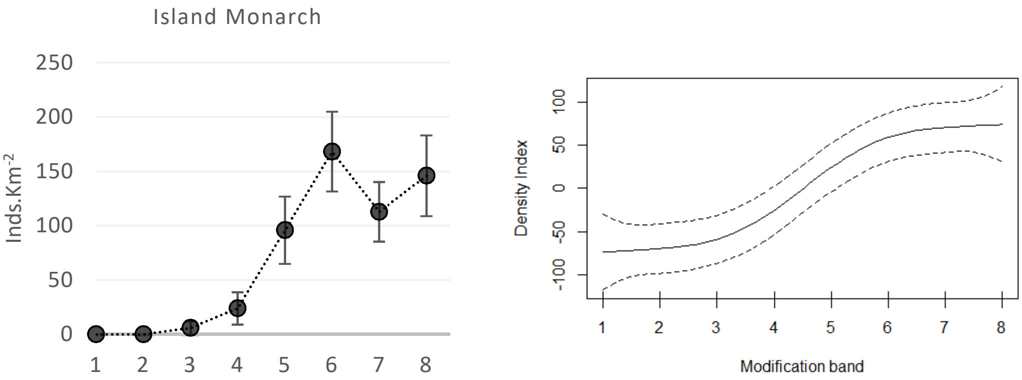


Figure 3-19. Island Monarch: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 5, R$^2$ (adj) = 0.99; $F_{(3.45,\ 3.83)}$ = 197.8, p <0.0001.

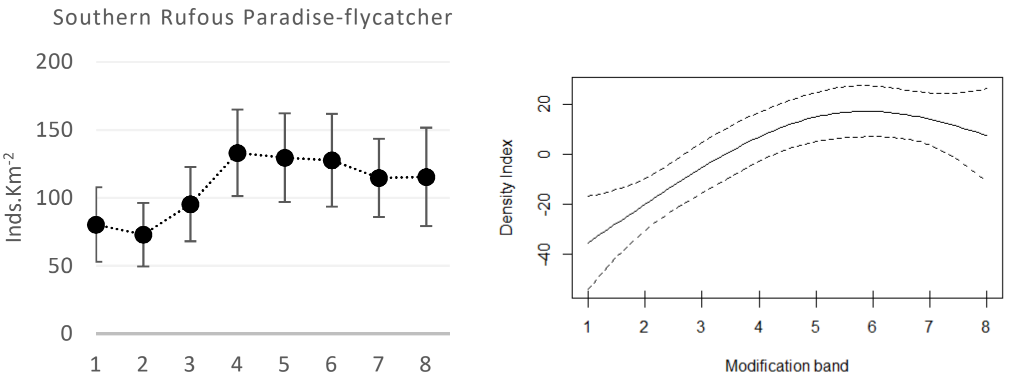


Figure 3-20. Southern Rufous Paradise-flycatcher: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 4, R$^2$ (adj) = 0.71; $F_{(2.08,\ 2.47)}$ = 7.29, p = 0.03.

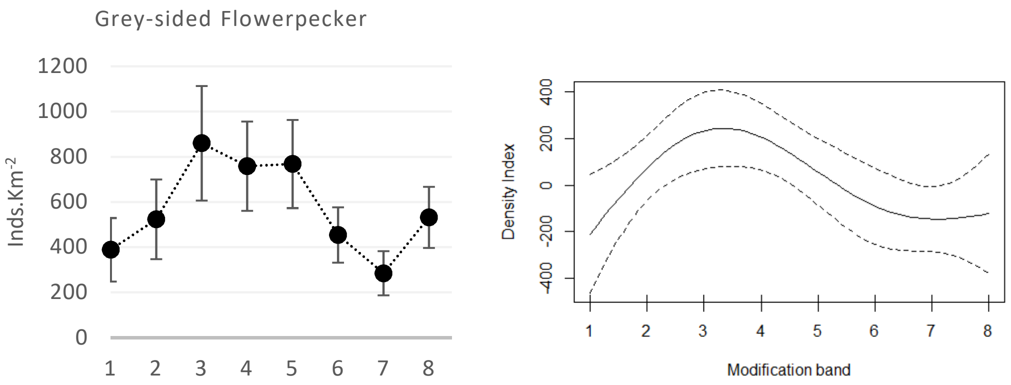


Figure 3-21. Grey-sided Flowerpecker: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 4, $R^2$ (adj) = 0.52; $F_{(2.74,\ 2.95)}$ = 2.58, p = 0.15.

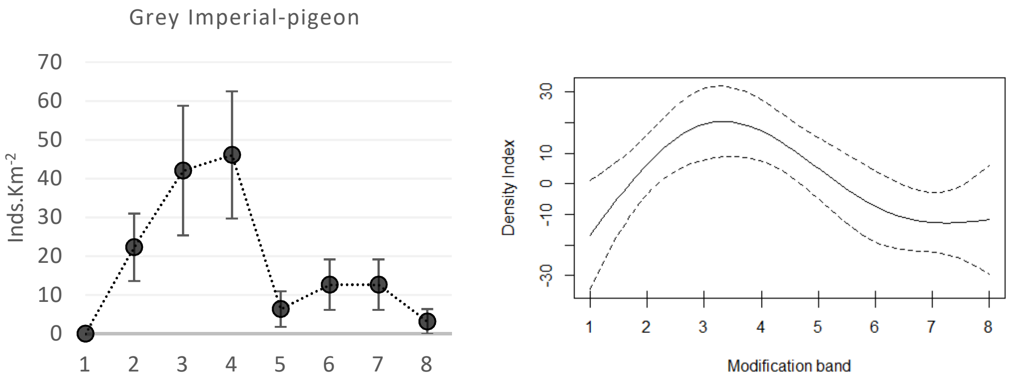


Figure 3-22. Grey Imperial-pigeon: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 4, $R^2$ (adj) = 0.62; $F_{(2.79,\ 2.97)}$ = 3.82, p = 0.09.

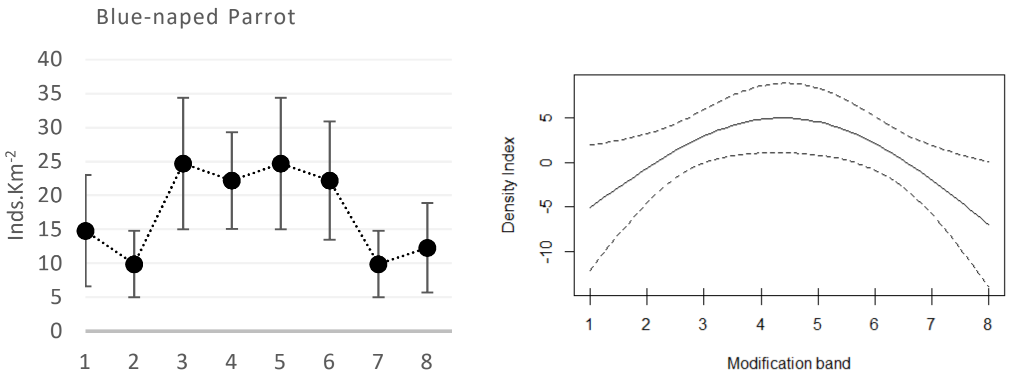


Figure 3-23. Blue-naped Parrot: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 3, $R^2$ (adj) = 0.45; $F_{(1.87,\ 1.98)}$ = 3.22, p = 0.12.

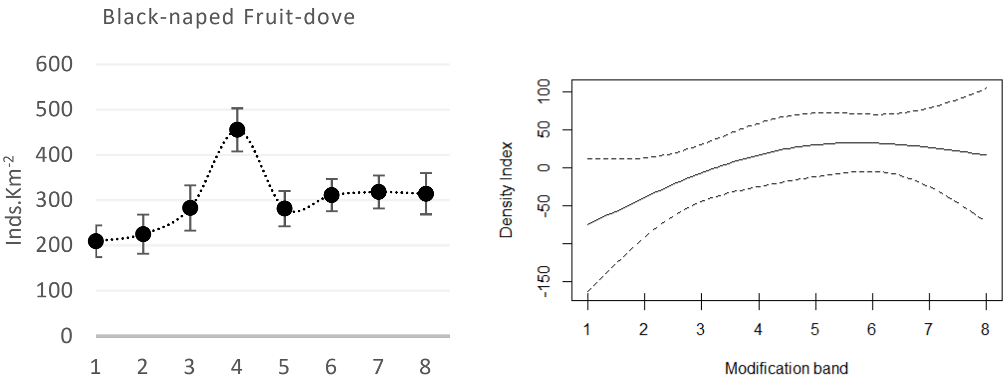


Figure 3-24. Black-naped Fruit-dove: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 3, $R^2$ (adj) = 0.22; $F_{(1.57, 1.81)}$ = 1.97, p = 0.3.

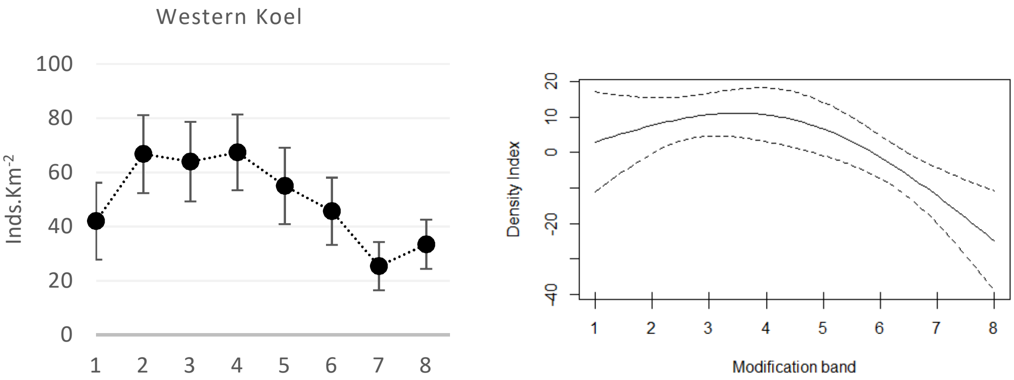


Figure 3-25. Western Koel: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 3, $R^2$ (adj) = 0.62; $F_{(1.85, 1.98)}$ = 5.52, p = 0.04.

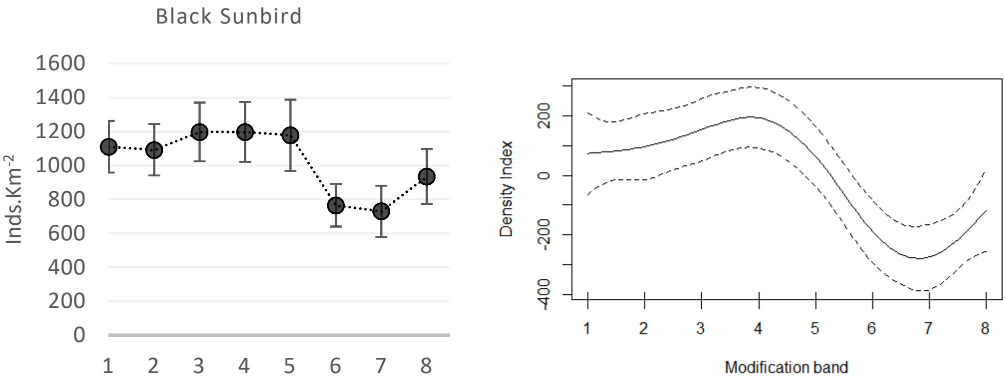


Figure 3-26. Black Sunbird: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 5, $R^2$ (adj) = 0.89; $F_{(3.89, 3.99)}$ = 15.04, p = 0.023.

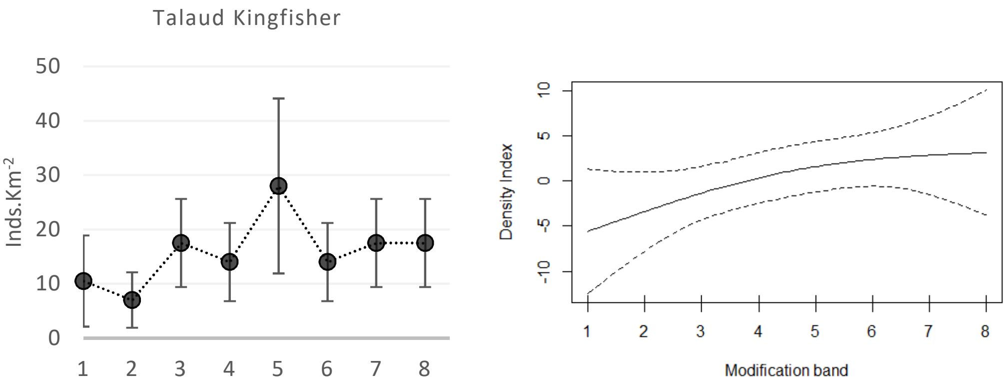


Figure 3-27. Talaud Kingfisher: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 3, $R^2$ (adj) = 0.19; $F_{(1.34,\ 1.57)}$ = 2.00, p = 0.29.

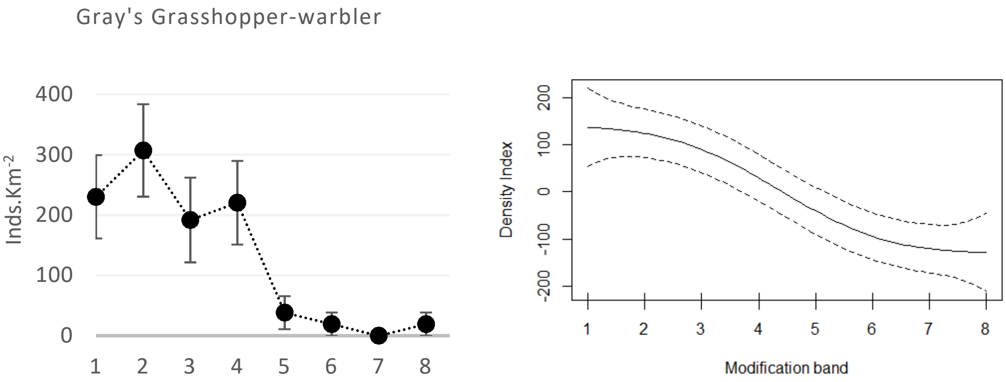


Figure 3-28. Gray’s Grasshopper-warbler: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, k = 6, $R^2$ (adj) = 0.81; $F_{(2.64,\ 3.22)}$ = 9.96, p = 0.019.

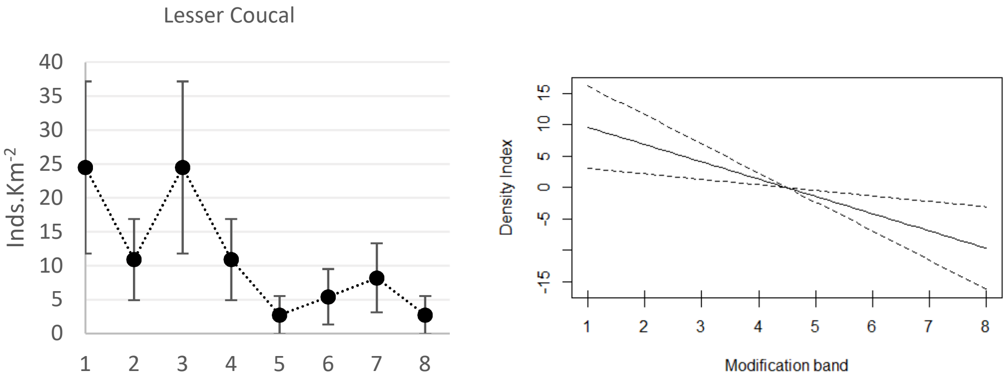


Figure 3-29. Lesser Coucal: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, $R^2$ (adj) = 0.52; $F_{(1,\ 1)}$ = 8.64, p = 0.025.

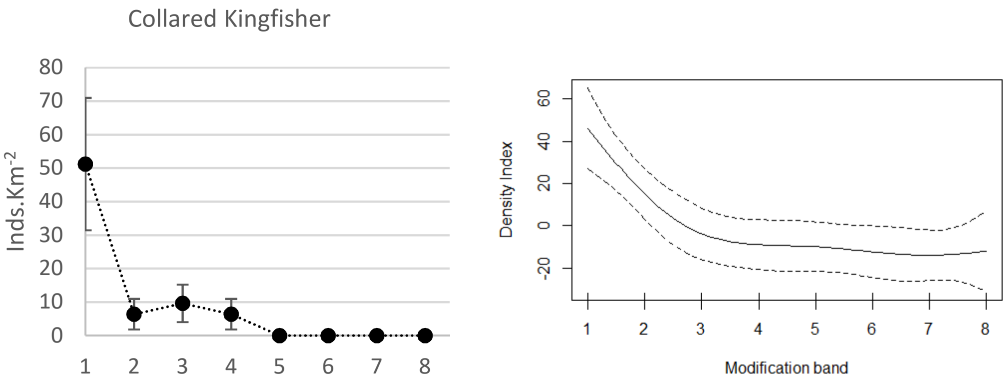


Figure 3-30. Collared Kingfisher: a) density estimate at each modification band (inds.km$^{-2}$ ±SE); b) GAM, $R^2$ (adj) = 0.77; $F_{(3.04,\ 3.53)}$ = 6.58, p = 0.046.

Three apparent significant community change points occurred along the modification gradient (Figure 3-31); favoured species showed a 23% decline in abundance from the first to the second group of points; this same group virtually disappeared from the community between group four and five; and the largest increase in abundance of the modification affected species occurred between bands five and six. These affected species increased in abundance at every step toward intact habitat. All these species were considerably more abundant in the three least-modified groups, *i.e.* those that retain some primary forest, but almost all were recorded across the entire modification gradient.

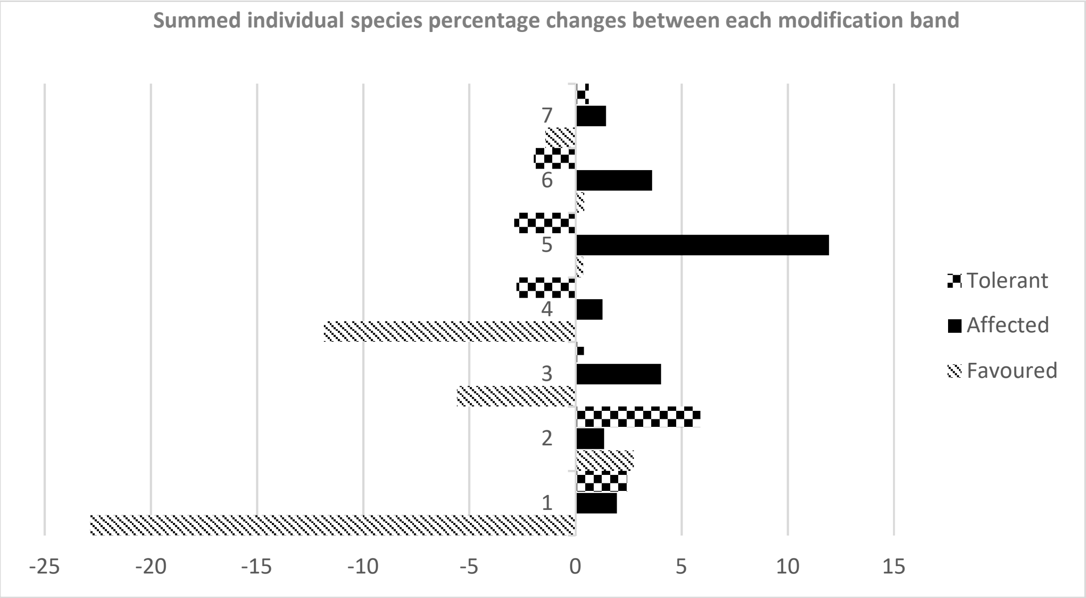


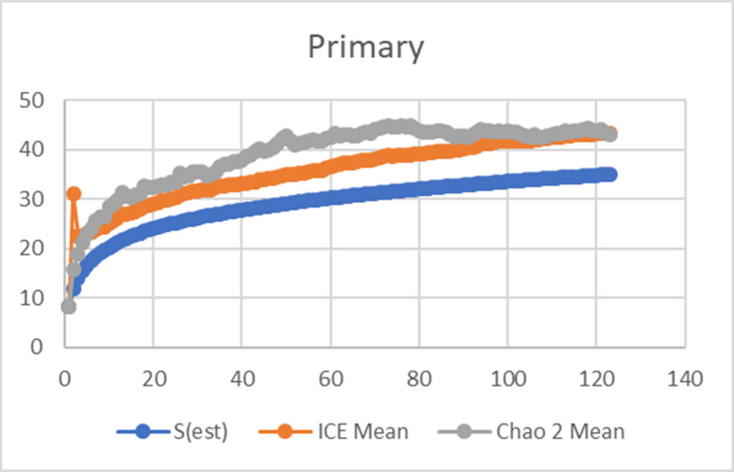
Figure 3-31. Percentage changes in abundance for the three species groups along the modification gradient.

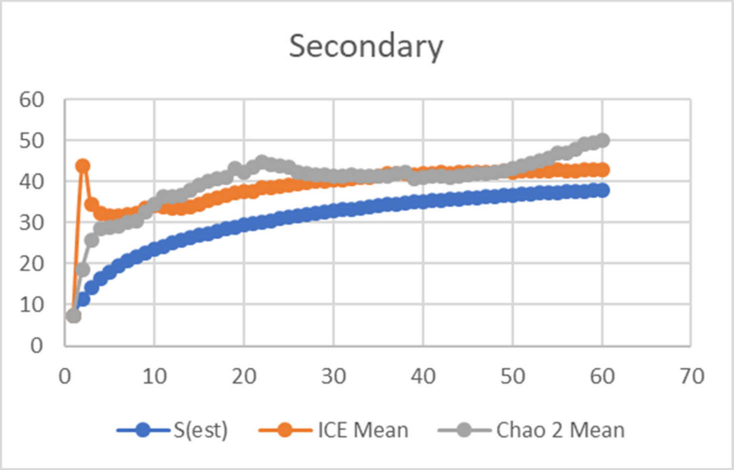
Close to 40% of the abundance of the species favoured by modification was found in the first, most heavily modified band, while less than 1% of the abundance of the group of species affected by modification was found in this band.

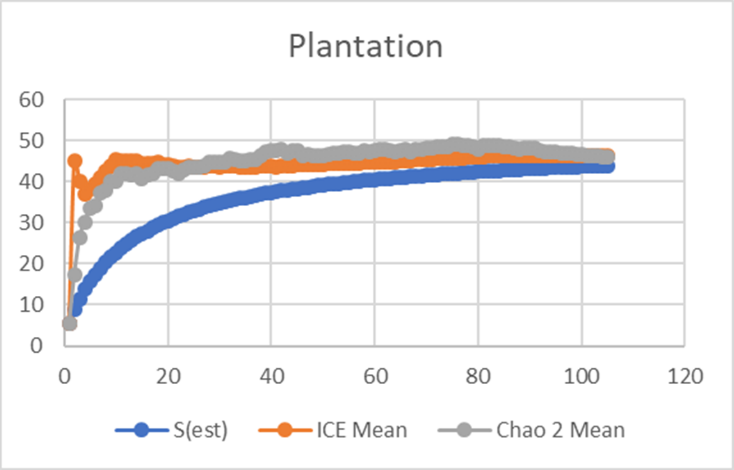
## 3.4 Discussion

The impact of forest degradation on bird abundance on the island of Karakalang is strongly species-specific, but more than half of the species declined with increasing habitat modification, and endemic taxa are disproportionately represented in this modification affected group. This result mirrors that of numerous studies observing abundance reduction for some species between discrete habitat classes (Jones et al. 1995; Edwards et al. 2011; Mallari et al. 2011; Gibson et al. 2011; Española et al. 2013; Davies et al. 2015), including the earlier study of the Karakelang avifauna by Riley (2003). The use of evenly distributed bands along a habitat modification gradient, rather than discrete categories, here allows a novel comparison of the form of the species' abundance responses to modification. Studies using occurrence data to investigate these responses have recorded mostly linear responses (Moura et al. 2016; Española et al. 2016), even if these may vary in slope between species and even in different locations for the same species. Bird occurrence and incidence response patterns resembling those identified on Karakelang occurred in response to varying tree cover percentage in a mixed woodland and grassland landscape in North America (Cunningham & Johnson 2012). As on Talaud, few species were wholly absent even at low percentages of tree cover (Cunningham & Johnson 2012). On Karakelang, for most species for which density estimates could be created part of the response was a linear slope: but this persisted for only a part of the gradient, and most species' response was sigmoidal in shape. The point at which a species that was affected by habitat modification began to decline in abundance varied slightly, but the majority declined most rapidly at the transition between degraded primary and secondary habitat, providing further evidence for the importance of primary habitat (Gibson et al. 2011). The degree of non-linearity was small considering the detail of the gradient, and only two species demonstrated greater than linear changes in abundance. Clearly for these species, Golden-mantled Racquet-tail and Blue-backed Parrot, the prediction that loss of habitat will produce a disproportionate loss of population is concerning: both appear to demonstrate a threshold response below which densities are much reduced (Swift & Hannon 2010). Of these two, the density estimates in conjunction with the response curve demonstrate clearly that Blue-backed Parrot is of higher conservation priority, occurring almost entirely in primary habitat and with abundance increasing more steeply and in a more strongly non-linear fashion than the racquet-tail (Figure 3-14 and Figure 3-15).

A considerable variation in abundance change is observed between those species considered to be equally forest-dependent (BirdLife International 2017a; Figure 3-47). Clearly therefore this assessment of forest dependence for the bird community of Karakelang is inaccurate, but these same species may show differing responses in other parts of their range (Moura et al. 2016). For endemic and restricted-range species, studies that directly investigate the density responses of species should be used to make these assessments. The two 'high' forest-dependent species, Talaud Kingfisher and Talaud Bush-hen, are by no means those most negatively affected by habitat modification. It is important to note here that a degree of the species' responses to this gradient will represent the influence of landscape factors operating within the surveyed area rather than solely anthropomorphic modification (e.g. Grainger et al. 2011). In that study the issue centred on restoration habitat and colonisation processess rather than the habitat loss from currently intact habitat in this study (Grainger et al. 2011). The relatively homogenous forested landscape on Talaud coupled with the low diversity within the bird community lowers potential impacts from landscape across the wide sample of the island during this work, and indeed there appears relatively little differentiation in bird abundance or species composition in similar habitat throughout (Appendix 3-2.Species-accumulation curves for the three habitat classes, displaying the analytical estimate of S (the estimated number of species from the pooled samples), the Incidence Coverage-based Estimator (ICE), and the Chao2 richness estimator; a) primary (n = 123), b) secondary (n = 60), c) plantation (n = 105).







Appendix 3-3; Appendix 3-5). However, the endemic kingfisher is strongly associated with the rivers that emerge from primary forest and are used as main access routes by people (pers. obs.), which leads to frequent encounters adjacent to cleared land and small-scale plantations. Talaud Kingfishers may still be highly forest dependent, and require this habitat for breeding or accessing key resources but the close adjacency between primary forest and cleared agriculture alongside the rivers may have resulted in an underestimation of the level of forest dependency.

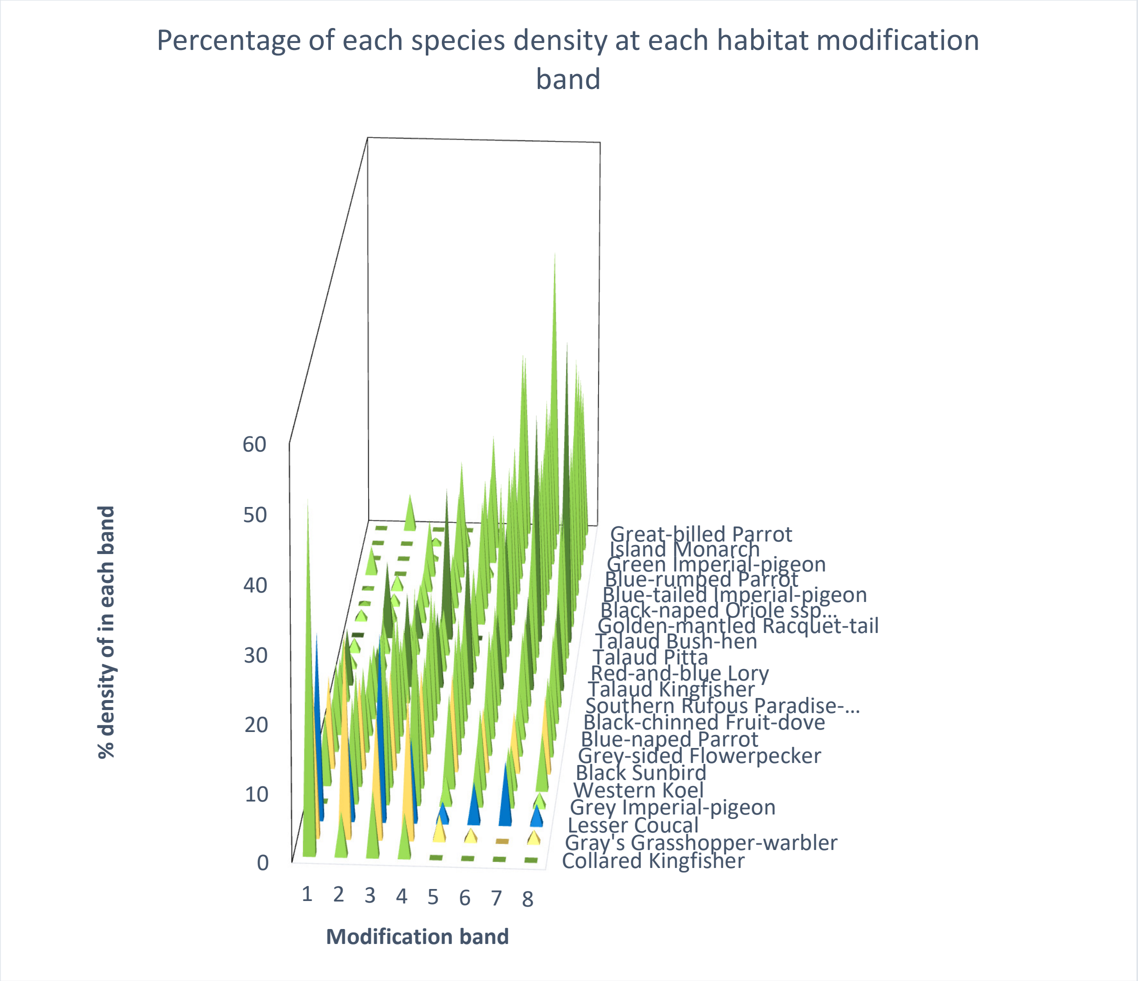


Figure 3-32. Proportion of density at each modification band, coloured per BirdLife forest-dependency coding (BirdLife International 2017a); ‘high’ dependency species are dark green, ‘medium’ are light green, ‘low’ are yellow and the single ‘non-forest’ species is blue.

The abundance of birds on Karakelang falls close to the mean value for tropical regions that was used for an assessment of the global abundance of birds (Gaston & Blackburn 1997).  Estimates in the most intact band (3,303 inds.km$^{-2}$) is strikingly close to the tropical mean of 3,382 inds.km$^{-2}$, although there was considerable variation in the studies used to construct this mean, with the highest value reported from small plots in New Guinea (Bell 1982; Terborgh et al. 1990).  In common with New Guinea there are very few mammals and no primates on Karakelang, which may allow birds to achieve such high densities (Terborgh et al. 1990).  A similarly high abundance has been reported by several studies in the Wallacean and New Guinea region (Jones et al. 1995; Marsden et al. 1997; Marsden 1998; Riley 2002a). Several species, particularly frugivores, demonstrate an abundance peak at intermediate levels of disturbance in other small-scale mixed-agriculture in forested landscapes (Marsden et al. 2006). Intriguingly this study appears to reveal a humped abundance distribution with a peak at intermediate modification levels except for the final, most intact habitat (Figure 3-8): were the final three categories to be combined as 'primary', this pattern would be masked.

Population density is generally expected to scale negatively with body size, although exceptions can occur (Gaston & Blackburn 2008). On Karakelang modification exacerbates this negative pattern, as the dramatic decline in abundance of larger-bodied species (Blue-tailed and Green Imperial-pigeons, Black-naped Oriole and *Tanygnathus* parrots) in more modified habitat is not compensated, increasing the contrast in relative abundance between these and the smaller-bodied species (e.g. Black Sunbird and Grey-sided Flowerpecker). As a result, the total biomass present in primary habitat is far greater than in modified habitat, with a maximum value of 602 kg.km$^{-2}$, considerably more than estimated by Terborgh et al. (1990) of 190 kg.km$^{-2}$ in lowland Amazonia. Even accounting for the inclusion of non-breeding and migrant individuals in this study, excluded in Terborgh et al. (1990), the bird biomass is very considerable on Karakelang.

The primary forests of Karakelang are gratifyingly full of birds. Overall, populations of key bird species appear to be large and secure, with most species recorded at higher densities than those reported in 1999 (Riley 2003; Table 3-8).

Table 3-9. Densities of key species in primary forest reported in Riley (2003) and found during the present study.

| Species | Riley 2003 | | This study | |
|---|---|---|---|---|
| | *n/K* | *D* Primary | *n/K* | *D* Primary |
| Talaud Bush-hen | 0.11 | 7.05 (3.56 – 13.9)<br>n = 24 | 0.13 | 32 (16–64)<br>n = 16 |
| Talaud Rail | 0.01 | –<br>n = 2 | 0.02 | –<br>n = 2 |
| Blue-tailed Imperial-pigeon | 1.79 | 36.71 (32.2–41.84)<br>n = 400 | 1.67 | 215 (186–248)<br>n = 205 |
| Grey Imperial-pigeon | 0.02 | –<br>n = 4 | 0.08 | 9 (5–18)<br>n = 10 |
| Red-and-blue Lory | 0.27 | 20.68 (15.2 – 28.09)<br>n = 61 | 0.46 | 102 (69 – 151)<br>n = 57 |
| Blue-naped Parrot | 0.32 | 11.29 (7.09 – 17.97)<br>n = 71 | 0.12 | 16 (9 – 29)<br>n = 15 |

| | | | | |
|---|---|---|---|---|
| Talaud Kingfisher | 0.86 | 15.56 (12.99 – 18.64)<br>n = 191 | 0.15 | 18 (10 – 33)<br>n = 19 |

Of the two species on the island currently assessed as Endangered (BirdLife International 2017a), Red-and-blue Lory has received much the greater degree of scrutiny owing to the sudden appearance of the species in the cagebird trade during the 1990s, leading to fears of unsustainable exploitation and rapid population decline (Anon. 1993).

In 1996 the population was precautionarily estimated at 9,400–24,150 individuals on the basis of point count distance sampling, while assuming 1) constant underestimation of distances by fieldworkers; 2) data obtained during the first two field sites were unsuitable, hence discarded; 3) only 70% of the area in mixed agriculture was suitable due to an infestation of coconut by *Sexava* spp. (Orthoptera); and 4) that using the mean rather than the maximum for the upper range of the population estimate provides a suitable precautionary approach (Lambert 1997). Without these modifications, using all data and the whole island area values, a non-precautionary population estimate derived from the data presented in Table 4 of Lambert (1997) would be 19,868–54,078 individuals, whereas if the data chosen for the estimate are used but with maximum estimates (rather than means) as the upper limit, the estimate becomes 9,404–40,852 individuals.

Subsequently, a second population of 8,230–21,400 birds was estimated from data collected in 1999 (Riley 2003), based on the minimum and maximum range but also using a larger area measure for both primary and 'secondary' habitat (350 km$^2$ and 600 km$^2$, versus 220 km$^2$ and 388 km$^2$). This was considered to represent a slight decline, but using the same area measure as Lambert (1997) this would be a considerable population reduction in a short time, and one which coincided with high trapping pressure with an estimate of up to 1,000 individuals exported in 1997 alone (Riley 1997b).

Fortunately, the present survey has returned similar estimates to the first survey: if a roughly equivalent binary habitat classification (combining secondary forest and mixed agriculture by summing the population estimate in each in proportion to their areas for use as a general secondary habitat estimate) and area measurements are used, the minimum and maximum estimate is 19,409–50,227 individuals, virtually identical to the estimate using all data from Lambert (1997).

There is the possibility of systematic underestimation of distance during the present survey, but particular attention was paid to checking distances to groups of this species by pacing the distance to an occupied tree immediately following the point count (Buckland et al. 2008). Given that encounter rates for Red-and-blue Lory were almost double those reported by Riley (2003), the population does appear to have recovered from a low point around the turn of the century. Recognition as a protected species under CITES in 1995 is likely to be a driver of this increase, but only after the awareness-raising efforts of the Action Sampiri project (Riley 1997b) led to local recognition of the problem and relatively strong enforcement action against some traders (Anon 2008). The subsequent Global Environment Fund project on the island to 2006 (Syarif 2004; Mamengko & Lumasuge 2006; Mamengko & Mole 2006a; Wangko 2006) reinforced this ongoing process and countering export of the species has become the main work of the non-govermental organisation KOMPAK (Komunitas Pencinta Alam Karakelang) on Karakelang. Several confiscations of birds being prepared for shipment have been seized (Doaly 2016, pers. obs. 2014), the largest of which was 111 individuals in November 2013 bound for the Philippines (Doaly 2013). Continued enforcement action against traders may have contributed to the improved fortunes of this species.

Table 3-10. Distribution of survey points at each modification band inside and outside protected areas.

| | Suaka Margasatwa N | | Suaka Margasatwa S | | Hutan Lindung | | Outside SM | |
|---|---|---|---|---|---|---|---|---|
| Band | # points (106) | % | # points (34) | % | # points (52) | % | # points (96) | % |
| 1 | 0 | 0 | 0 | 0 | 5 | 9.6 | 31 | 32.3 |
| 2 | 0 | 0 | 3 | 8.8 | 4 | 7.7 | 29 | 30.2 |
| 3 | 2 | 1.9 | 2 | 5.9 | 12 | 23.1 | 20 | 20.8 |
| 4 | 5 | 4.7 | 4 | 11.8 | 17 | 32.7 | 10 | 10.4 |
| 5 | 14 | 13.2 | 10 | 29.4 | 8 | 15.4 | 4 | 4.2 |
| 6 | 24 | 22.6 | 9 | 26.5 | 2 | 3.8 | 1 | 1 |
| 7 | 29 | 27.4 | 5 | 14.7 | 1 | 2 | 1 | 1 |
| 8 | 32 | 30.2 | 1 | 2.9 | 3 | 5.8 | 0 | 0 |

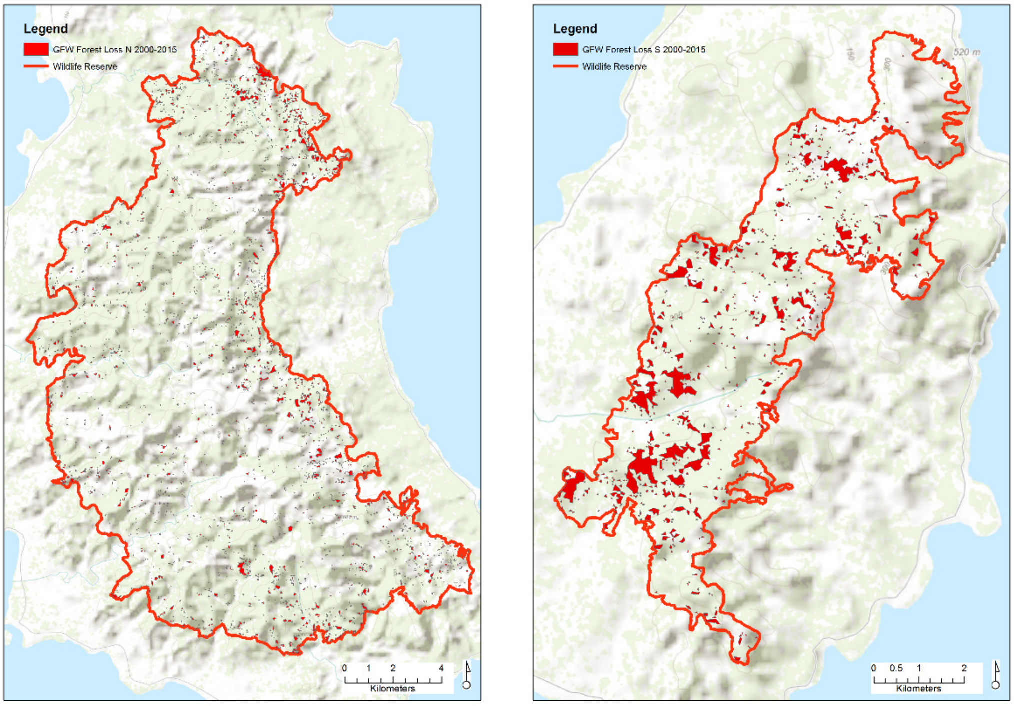


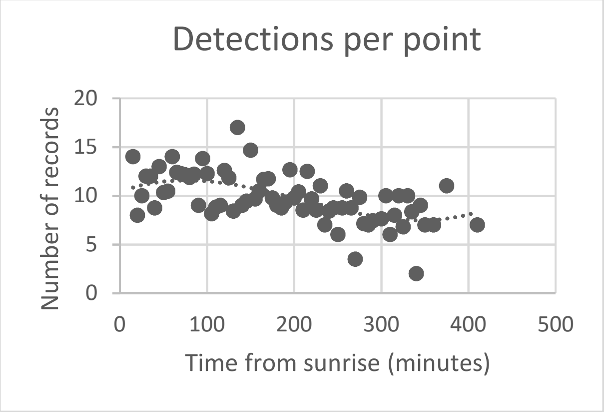
Figure 3-33. Area of habitat lost within the Wildlife Reserves (data from Hansen et al. 2013); left, northern; right, southern. Note the different scales.

Much of the ongoing degradation of the remaining primary forest, which the present study has confirmed to be the highest priority for bird conservation on Karakelang, results from selective logging for construction timber used on the island. The discovery of selectively logged trees far inside the Wildlife Reserve indicates that there is weak enforcement of the regulations pertaining to the protected area designation on the island. This is unsurprising, however, given that there is virtually no supply of timber from outside the protected area boundaries and the isolation of the island makes current supplies of ‘legal’ timber virtually non-existent; hence permission for the small-scale take of large trees is implicitly granted, allowing the expansion of an informal market. The situation appears to be an example of a perverse outcome from a conservation action due to a failure to consider the market extent at designation (Lim *et al.* 2017). While the area lost per tree is relatively small and the impact on bird populations of the logging (noted as the change between habitat bands 8 and 6) is minimal, repeated incursions have clearly led to clearances of additional land and the establishment of smallholdings within the protected areas.

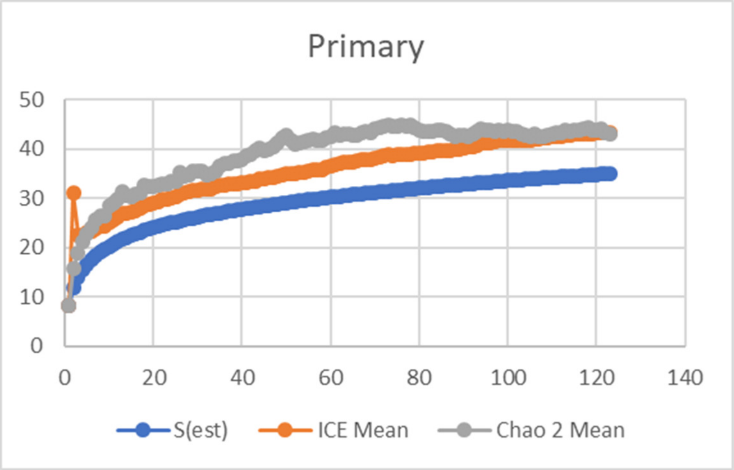
Given the potential dramatic reductions in densities of large frugivores for a relatively minor level of further habitat modification, the remaining intact primary areas need full protection.  To encourage local support for the protection of primary forest habitat there must be an improved supply to the formal market (*sensu* Lim *et al.* 2017), which may require the establishment of a limited timber concession in areas already cleared for agriculture but situated inside the Wildlife Reserve.  The present area of the Wildlife Reserves is too large for current levels of enforcement, and without clear boundaries recognised by all stakeholders the degradation of primary habitat will continue throughout.  Previous attempts to enforce the protected area restrictions through one-off arrests have caused considerable conflict and an overall weakening of respect for the the protected area boundaries (J. Loronusa [head of the Ensem community] pers. comm., M. Wangko pers. comm.).  A better defined and secured Wildlife Reserve boundary, even if smaller than present, with consideration of the supply of resources currently obtained from the protected area for the island economy, is likely to improve both local attitudes to forest protection and the prospect of long-term retention of undisturbed habitat.

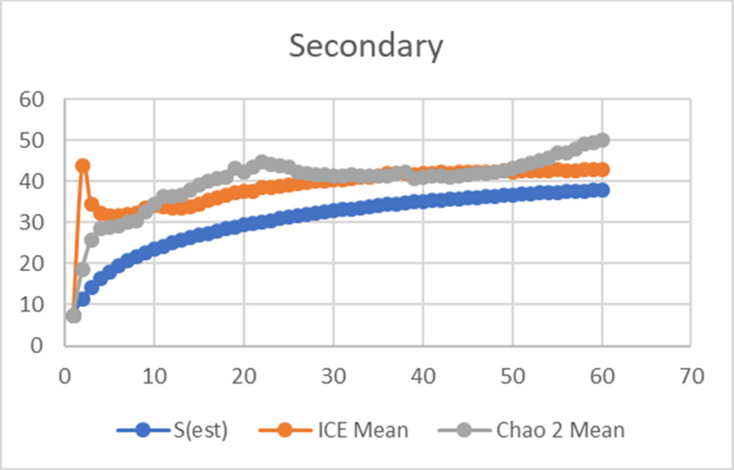
## 3.5 Appendices

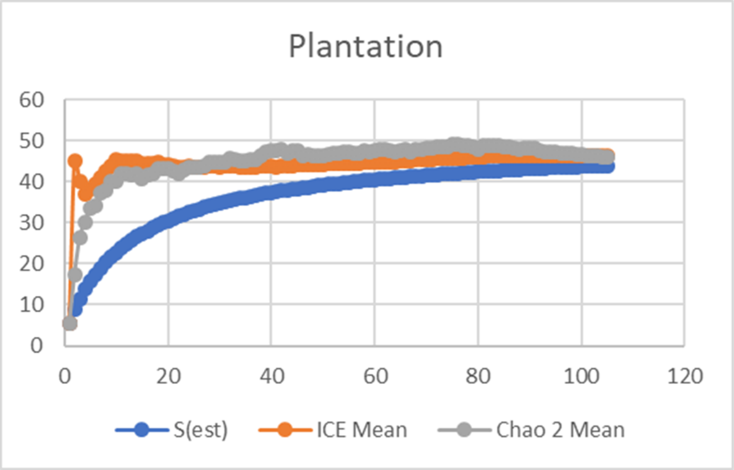
Appendix 3-1. Number of individual bird detections per point plotted against time from sunrise.



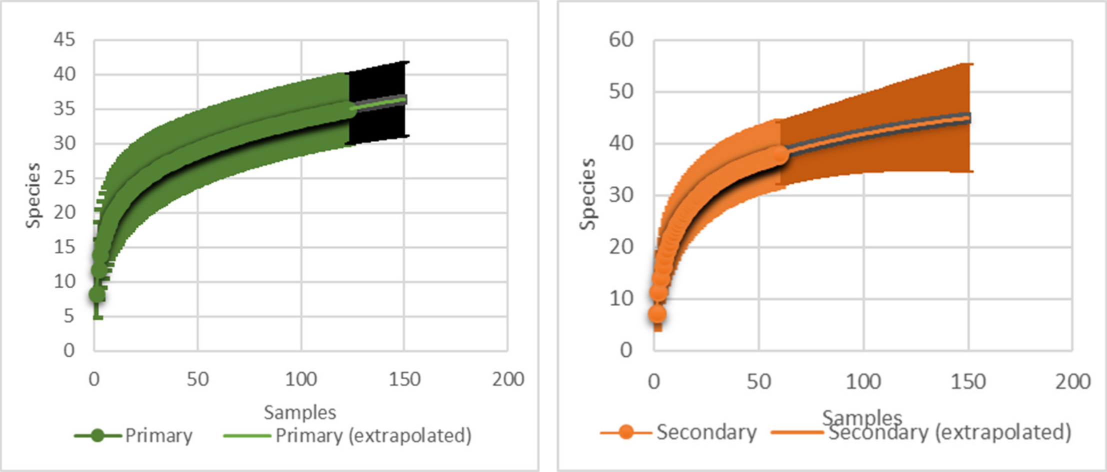
Appendix 3-2.Species-accumulation curves for the three habitat classes, displaying the analytical estimate of S (the estimated number of species from the pooled samples), the Incidence Coverage-based Estimator (ICE), and the Chao2 richness estimator; a) primary (n = 123), b) secondary (n = 60), c) plantation (n = 105).

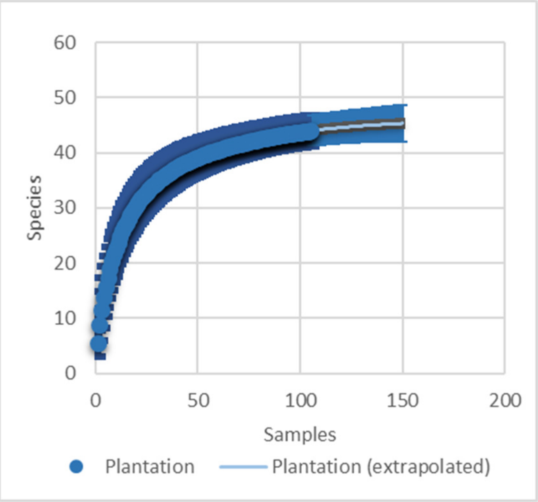






Appendix 3-3. Rarefaction curves for each of the broad habitat groups, with 100 randomisation runs and extrapolation to 150 samples, using the methods of (Colwell et al. 2012) in EstimateS v.9 (Colwell 2013).





Appendix 3-4. Encounter rates of species during point counts on Karakelang. Top row: numbers of birds/points completed (percentage Coefficient of Variance). Bottom row: lower and upper 95% confidence interval (number of observations). Density estimates were calculated for the species in bold, which were recorded > 30 times or for which a pooled detection function was used.

| | **Primary (123)** | **Secondary (60)** | **Kebun (105)** |
|---|---|---|---|
| **Megapodiidae** | | | |
| Philippine Scrubfowl | 0.08 (30%) | 0.05 (57%) | 0.01 (100%) |
| | 0.05–0.15 (n = 10) | 0.02–0.15 (n = 3) | 0.002–0.05 (n = 1) |
| **Columbidae** | | | |
| Slender-billed Cuckoo-dove | 0.008 (100%) | 0.05 (57%) | 0.1 (31%) |
| | 0.002–0.04 (n = 1) | 0.02–0.14 (n = 3) | 0.06–0.19 (n = 11) |
| Grey-capped Emerald Dove | 0.02 (57%) | 0.03 (70%) | 0.06 (40%) |
| | 0.009–0.07 (n = 3) | 0.009–0.12 (n = 2) | 0.009–0.12 (n = 6) |
| **Green Imperial-pigeon** | **0.52 (15%)** | **0.27 (33%)** | **0.04 (61%)** |
| | **0.38–0.70 (n = 64)** | **0.14–0.51 (n = 16)** | **0.01–0.12 (n = 4)** |
| **Blue-tailed Imperial-pigeon** | **1.67 (5%)** | **0.73 (17%)** | **0.15 (28%)** |
| | **1.5–1.85 (n = 205)** | **0.5–1 (n = 44)** | **0.09–0.26 (n = 16)** |
| **Grey Imperial-pigeon** | **0.08 (30%)** | **0.27 (27%)** | **0.14 (28%)** |
| | **0.05–0.15 (n = 10)** | **0.16–0.45 (n = 16)** | **0.08–0.24 (n = 15)** |
| Pied Imperial-pigeon | 0.02 (57%) | 0.05 (57%) | 0.02 (70%) |
| | 0.008–0.07 (n = 3) | 0.02–0.14 (n = 3) | 0.005–0.07 (n = 2) |
| **Black-naped Fruit-dove** | **1.8 (6%)** | **1.9 (8%)** | **1.3 (9%)** |
| | **1.6–2 (n = 217)** | **1.6–2.2 (n = 114)** | **1.1–1.6 (n = 138)** |
| **Cuculidae** | | | |
| **Lesser Coucal** | **0.05 (40%)** | **0.08 (43%)** | **0.21 (26%)** |
| | **0.02–0.1 (n = 6)** | **0.04–0.2 (n = 5)** | **0.13–0.35 (n = 22)** |
| **Western Koel** | **0.42 (14%)** | **0.67 (14%)** | **0.62 (11%)** |
| | **0.32–0.55 (n = 52)** | **0.50–0.88 (n = 40)** | **0.49–0.78 (n = 65)** |
| **Rallidae** | | | |
| Talaud Rail | 0.02 (70%) | 0.02 (100%) | - |
| | 0.005–0.06 (n = 2) | 0.003–0.09 (n = 1) | - |
| **Talaud Bush-hen*** | **0.13 (23%)** | **0.05 (57%)** | **0.01 (100%)** |
| | **0.08–0.21 (n = 16)** | **0.01–0.12 (n = 2)** | **0.002–0.05 (n = 1)** |
| **Alcedinidae** | | | |
| **Collared Kingfisher*** | - | **0.07 (49%)** | **0.17 (31%)** |
| | - | **0.03–0.17 (n = 4)** | **0.09–0.3 (n = 17)** |
| **Talaud Kingfisher** | **0.15 (28%)** | **0.15 (31%)** | **0.07 (37%)** |
| | **0.09–0.27 (n = 19)** | **0.08–0.27 (n = 9)** | **0.03–0.13 (n = 7)** |
| **Psittacidae** | | | |
| **Red-and-blue Lory** | **0.46 (12%)** | **0.18 (27%)** | **0.09 (36%)** |
| | **0.36–0.59 (n = 57)** | **0.1–0.31 (n = 11)** | **0.04–0.17 (n = 9)** |
| **Golden-mantled Racquet-tail** | **0.39 (14%)** | **0.23 (24%)** | **0.09 (36%)** |
| | **0.29–0.52 (n = 48)** | **0.15–0.37 (n = 14)** | **0.004–0.17 (n = 9)** |
| **Blue-naped Parrot** | **0.12 (24%)** | **0.28 (21%)** | **0.1 (33%)** |
| | **0.08–0.2 (n = 15)** | **0.19–0.43 (n = 17)** | **0.05–0.18 (n = 10)** |
| **Great-billed Parrot*** | **0.11 (27%)** | **0.02 (100%)** | **0.01 (100%)** |
| | **0.07–0.19 (n = 14)** | **0.003–0.09 (n = 1)** | **0.002–0.05 (n = 1)** |

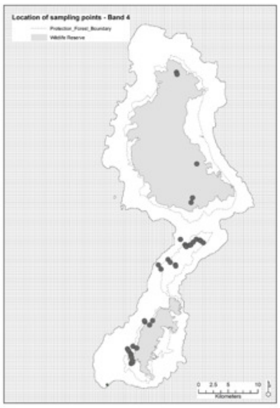
| | | | |
|---|---|---|---|
| Blue-backed Parrot | **0.21 (18%)** | **0.07 (49%)** | **0.07 (37%)** |
| | **0.15–0.3 (n = 26)** | **0.03–0.17 (n = 4)** | **0.03–0.13 (n = 7)** |
| **Pittidae** | | | |
| Talaud Pitta | **0.82 (10%)** | **0.57 (20%)** | **0.11 (30%)** |
| | **0.67–1.01 (n = 101)** | **0.38–0.85 (n = 34)** | **0.06–0.20 (n = 12)** |
| **Oriolidae** | | | |
| Black-naped Oriole | **1.22 (7%)** | **0.52 (16%)** | **0.12 (34%)** |
| | **1.07–1.39 (n = 150)** | **0.37–0.71 (n = 31)** | **0.06–0.24 (n = 13)** |
| **Camphegidae** | | | |
| Sulawesi Cicadabird | 0.1 (32%) | 0.08 (43%) | 0.1 (53%) |
| | 0.05–0.18 (n = 12) | 0.04–0.19 (n = 5) | 0.04–0.28 (n = 11) |
| **Monarchidae** | | | |
| Island Monarch | **0.56 (12%)** | **0.23 (31%)** | **0.01 (100%)** |
| | **0.44–0.71 (n = 69)** | **0.13–0.43 (n = 14)** | **0.002–0.05 (n = 1)** |
| **Southern Rufous Paradise-flycatcher** | **0.46 (13%)** | **0.5 (15%)** | **0.43 (17%)** |
| | **0.35–0.59 (n = 56)** | **0.37–0.68 (n = 30)** | **0.31–0.61 (n = 46)** |
| **Locustellidae** | | | |
| Gray's Grasshopper-warbler | **0.02 (70%)** | **0.25 (24%)** | **0.33 (16%)** |
| | **0.005–0.06 (n = 2)** | **0.15–0.41 (n = 15)** | **0.25–0.45 (n = 35)** |
| **Zosteropidae** | | | |
| Everett's White-eye | 0.008 (100%) | 0.07 (49%) | 0.16 (22%) |
| | 0.002–0.04 (n = 1) | 0.03–0.17 (n = 4) | 0.10–0.25 (n = 17) |
| **Sturnidae** | | | |
| Asian Glossy Starling | 0.008 (100%) | 0.05 (57%) | 0.05 (44%) |
| | 0.002–0.04 (n = 1) | 0.009–0.12 (n = 3) | 0.02–0.11 (n = 5) |
| **Dicaeidae** | | | |
| Grey-sided Flowerpecker | **0.32 (15%)** | **0.52 (23%)** | **0.36 (18%)** |
| | **0.24–0.43 (n = 39)** | **0.33–0.81 (n = 31)** | **0.26–0.51 (n = 38)** |
| **Nectariniidae** | | | |
| Black Sunbird | **1 (8%)** | **1.1 (10%)** | **1.54 (6%)** |
| | **0.85–1.19 (n = 124)** | **0.91–1.33 (n = 66)** | **1.36–1.75 (n = 162)** |

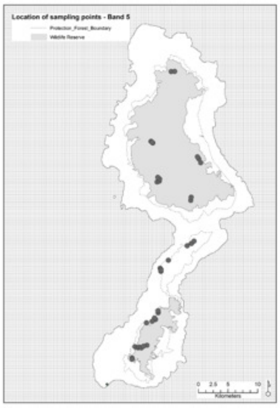
Appendix 3-5. Spatial distribution of survey points in each of the eight modification bands.

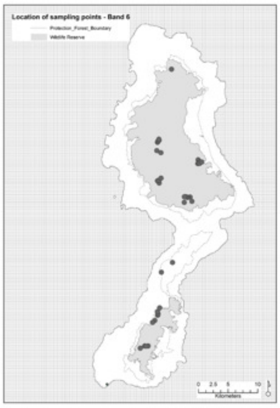

Location of sampling points - Band 1
Protection_Forest_Boundary
Wildlife Reserve
0 2.5 5 10
Kilometers

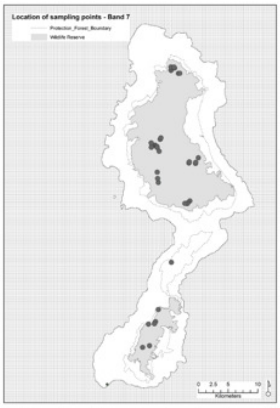

Location of sampling points - Band 2
Protection_Forest_Boundary
Wildlife Reserve
0 2.5 5 10
Kilometers

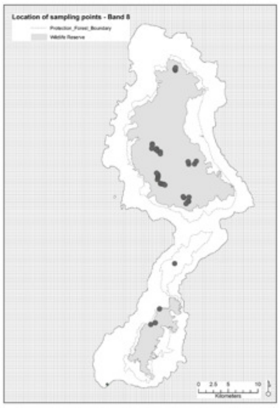

Location of sampling points - Band 3
Protection_Forest_Boundary
Wildlife Reserve
0 2.5 5 10
Kilometers


Location of sampling points - Band 4
Protection_Forest_Boundary
Wildlife Reserve
0 2.5 5 10
Kilometers

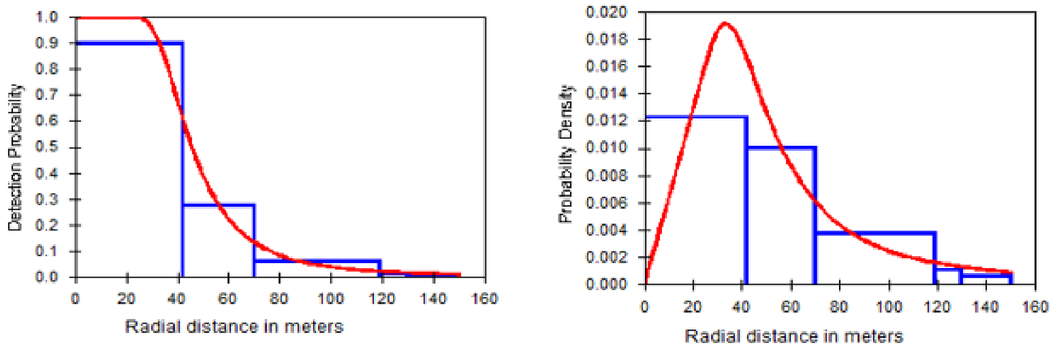

Location of sampling points - Band 5
Protection_Forest_Boundary
Wildlife Reserve
0 2.5 5 10
Kilometers


Location of sampling points - Band 6
Protection_Forest_Boundary
Wildlife Reserve
0 2.5 5 10
Kilometers

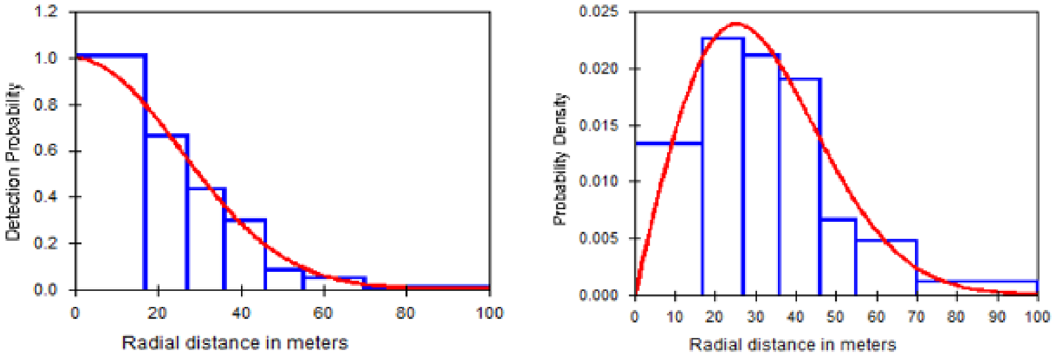

Location of sampling points - Band 7
Protection_Forest_Boundary
Wildlife Reserve
0 2.5 5 10
Kilometers


Location of sampling points - Band 8
Protection_Forest_Boundary
Wildlife Reserve
0 2.5 5 10
Kilometers

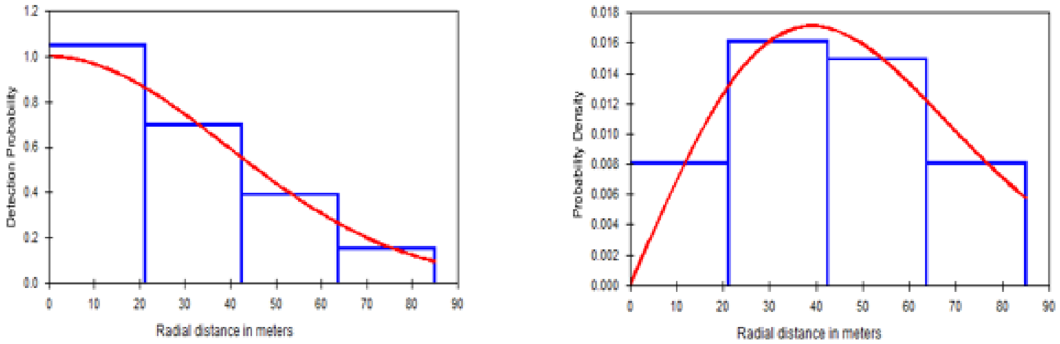
Appendix 3-6. Modelled detection functions (left) and probability densities from DISTANCE (Thomas et al. 2010) the species for which density estimates were calculated directly.



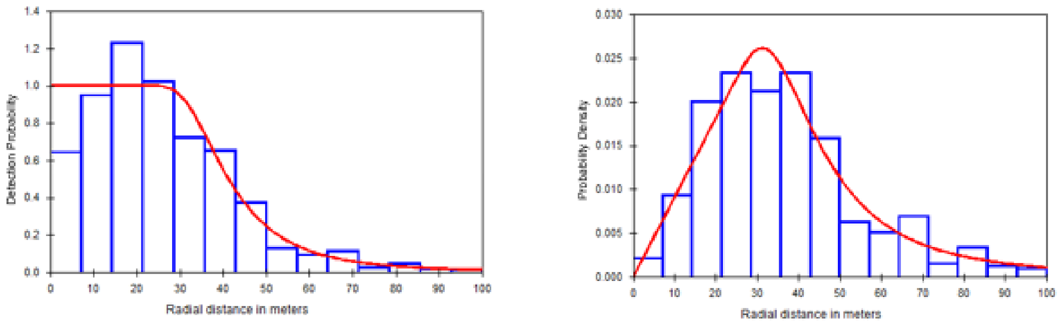
a) Green Imperial-pigeon



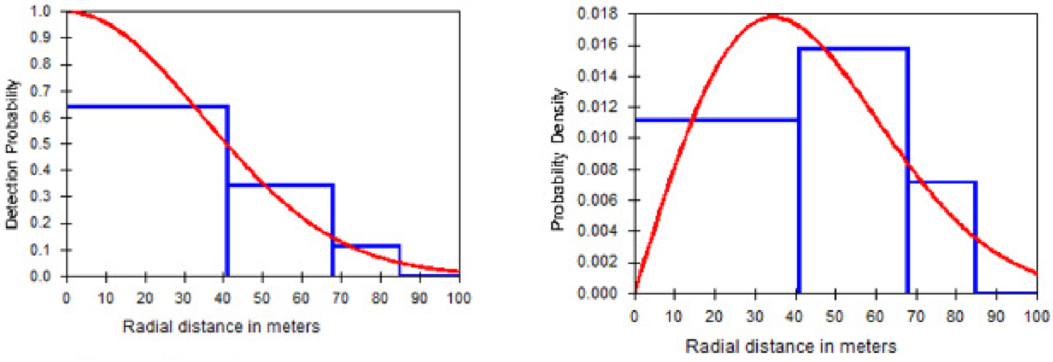
b) Blue-tailed Imperial-pigeon



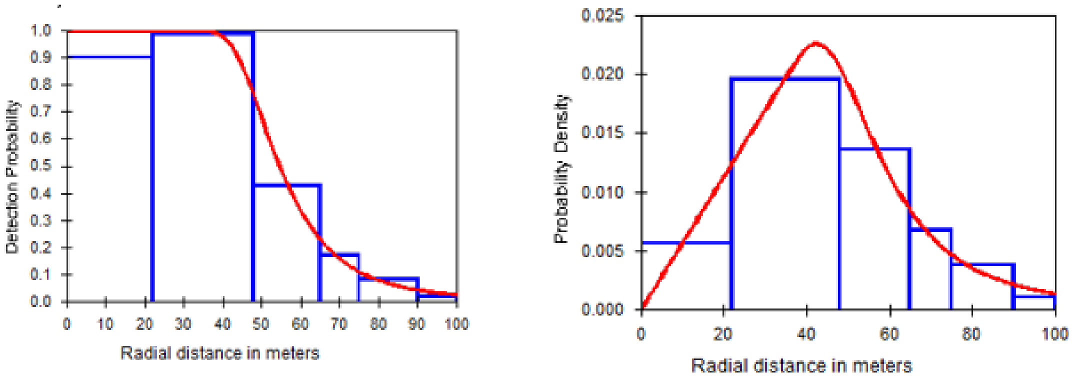
c) Grey Imperial-pigeon



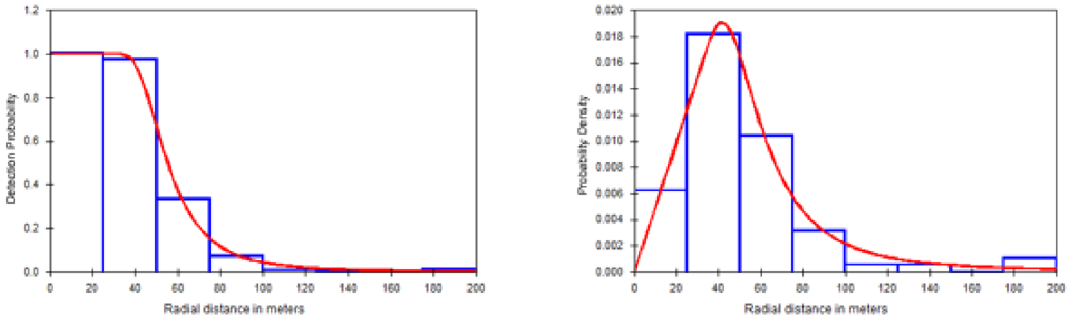
d) Black-naped Fruit-dove



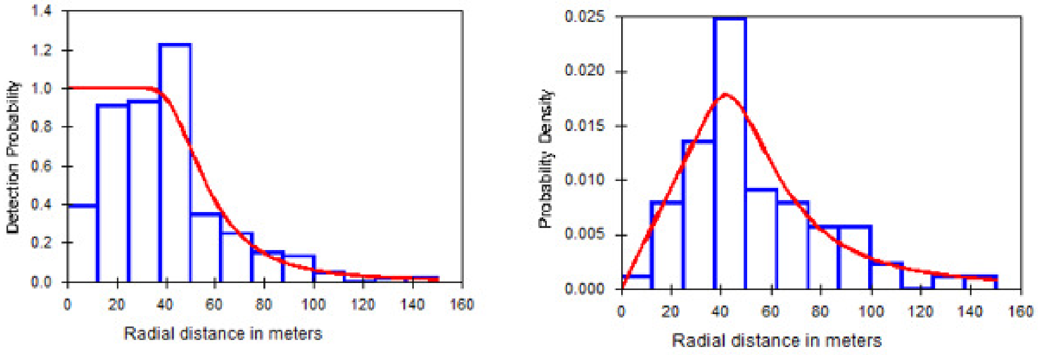
e) Lesser Coucal



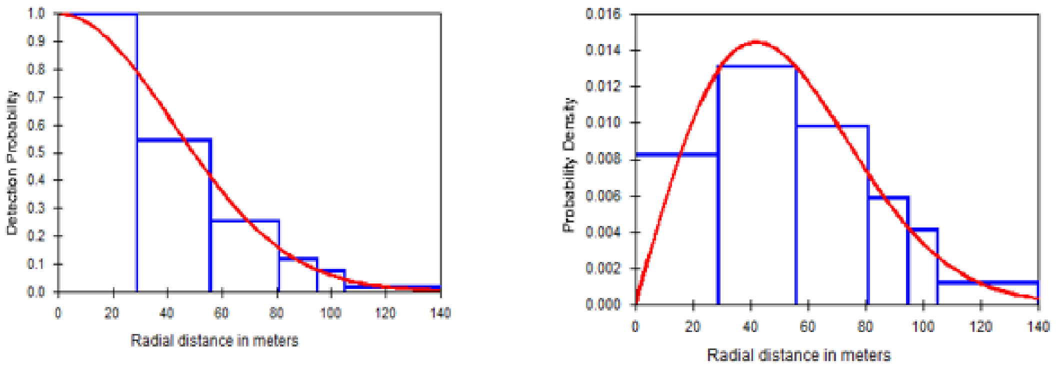
f) Western Koel



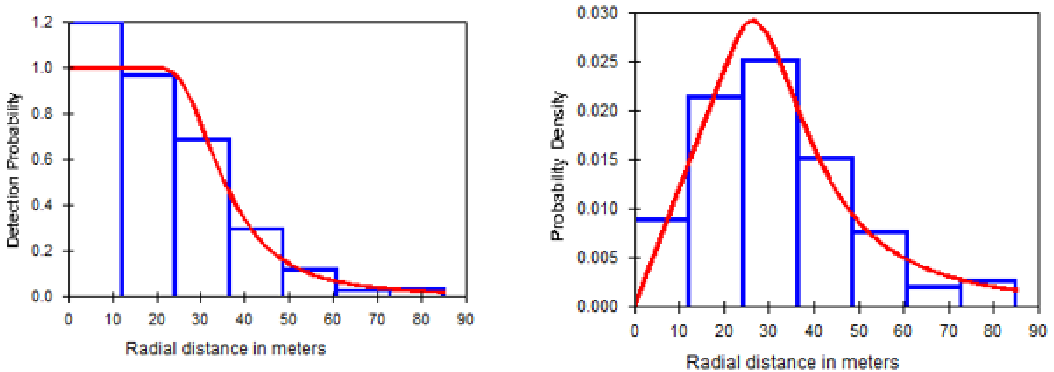
g) Red-and-blue Lory



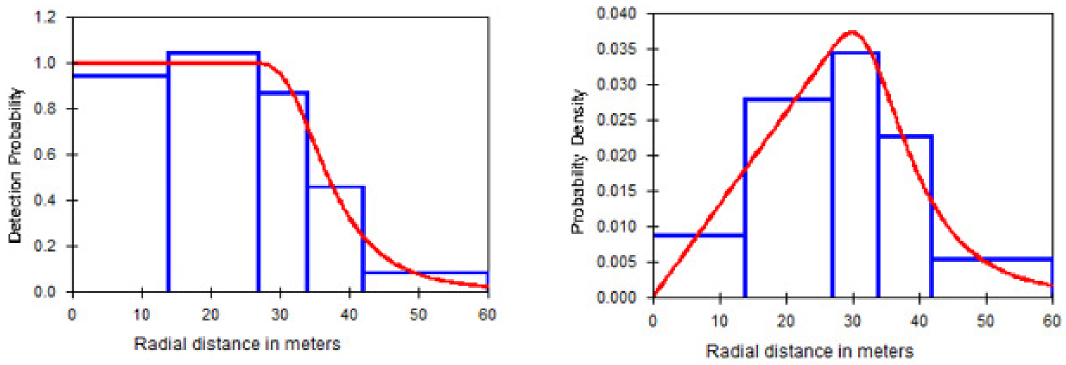
h) Golden-mantled Racquet-tail



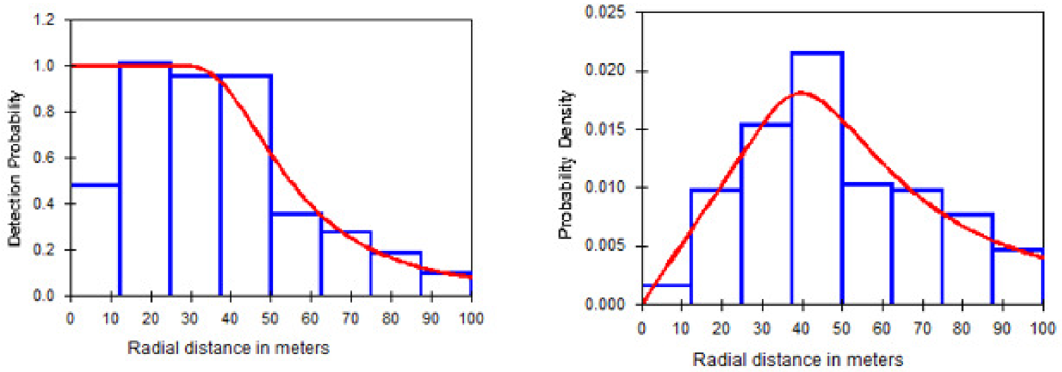
i) Talaud Pitta



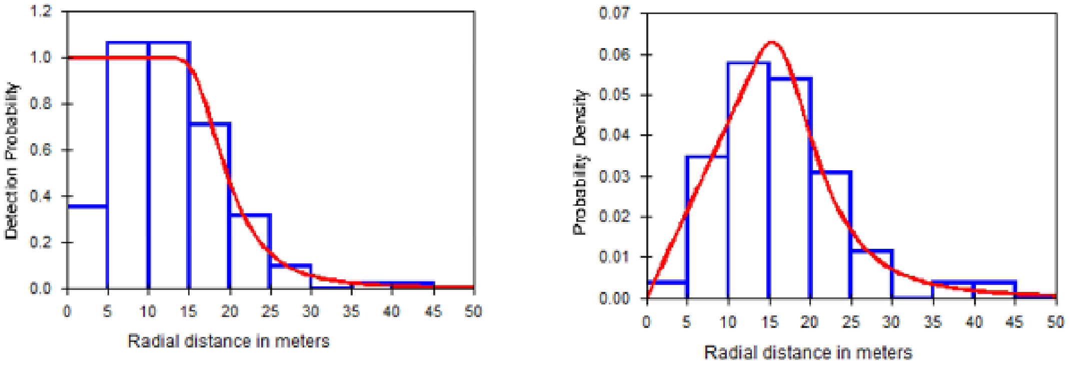
j) Black-naped Oriole



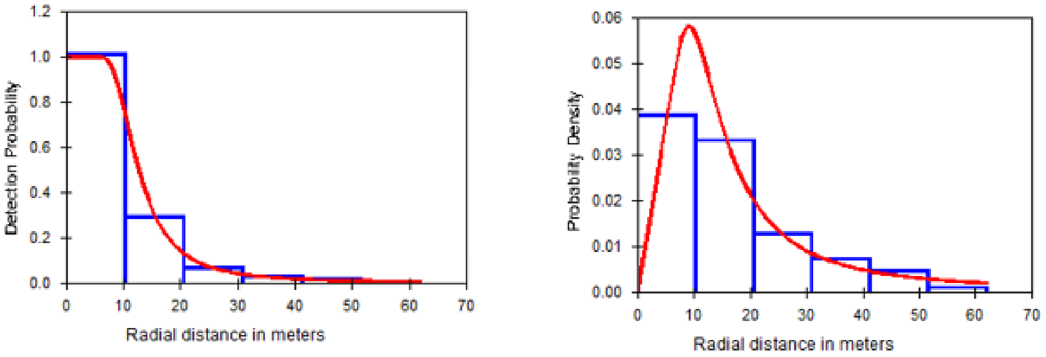
k) Island Monarch



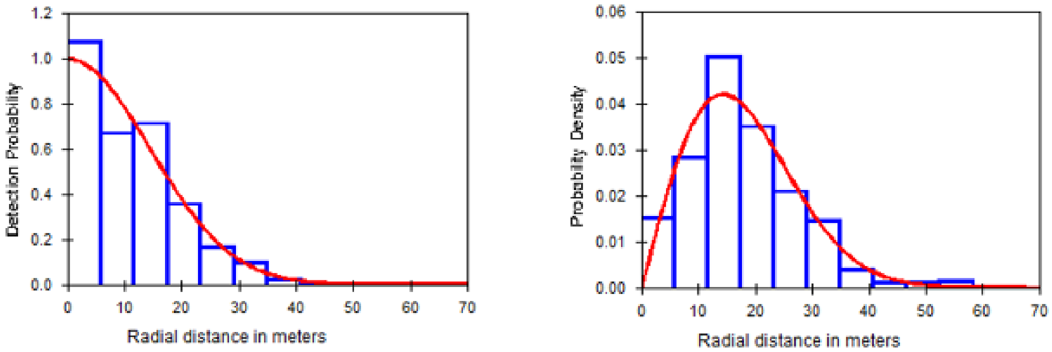
l) Southern Rufous Paradise-flycatcher



m) Gray's Grasshopper-warbler



n) Grey-sided Flowerpecker



o) Black Sunbird

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