THESIS

PREDATION AND NEST SUCCESS OF FOREST BIRDS IN NATIVE AND NON-NATIVE HABITAT ON SAIPAN, MARIANA ISLANDS

Submitted by

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WE HEREBY RECOMMEND THAT THE THESIS PREPARED UNDER OUR SUPERVISION BY THALIA SACHTLEBEN ENTITLED PREDATION AND NEST SUCCESS OF FOREST BIRDS IN NATIVE AND NON-NATIVE HABITAT ON SAIPAN, MARIANA ISLANDS BE ACCEPTED AS FULFILLING IN PART REQUIREMENTS FOR THE DEGREE OF MASTER OF SCIENCE.

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ABSTRACT OF THESIS

PREDATION AND NEST SUCCESS OF FOREST BIRDS IN NATIVE AND NON-NATIVE HABITAT ON SAIPAN, MARIANA ISLANDS

The primary causes of extinctions and declines of birds on islands are predation by non-native mammals (e.g., rats and feral cats) and the removal or alteration of indigenous forests. The impact of various exotic predators and extreme modification and removal of native forest on Saipan on the nesting success of the native avifauna had not been examined. Thus, I undertook this research on the island of Saipan in 2003 and 2004 to: (i) quantify nest densities in native and non-native forest, (ii) assess nest survival and predation rates on nests in native and non-native forest, (iii) determine whether certain nest site characteristics made nests more vulnerable to predators, and (iv) identify predators of nests of target bird species. My target species were the Nightingale Reed-warbler (*Acrocephalus luscinia*; federally listed as endangered) and three of the more common forest species, the Golden White-eye (*Cleptornis marchei*), Bridled White-eye (*Zosterops conspicillatus saypani*) and Rufous Fantail (*Rhipidura rufifrons saipanensis*). The Golden White-eye is an endemic species, while the latter two are endemic sub-species.

Nest densities were influenced by forest type in 2003, and by avian species, forest type, and sampling period in 2004. Daily nest survival rates varied by nest stage and species. For Bridled White-eyes, daily survival rates differed by year, whereas daily survival rates were constant across years for Golden White-eyes and Rufous Fantails. In 2003, daily nest survival in all stages for Bridled White-eyes was similar to that of Golden White-eyes and Rufous Fantails. Survival estimates were higher for Golden

White-eyes and Rufous Fantails than for Bridled White-eyes in the incubation and nestling stages in 2004. All predators identified by camera were avian, predominantly Micronesian Starlings (*Aplonis opaca*) and Collared Kingfishers (*Halcyon chloris*). Results from modeling the effect of nest site characteristics on daily nest survival rates were ambiguous, with much model selection uncertainty. However, there was weak evidence that higher numbers of nest support branches increased survival of Golden White-eye nests, while lower nest concealment (side cover) improved Bridled White-eye nest survival. These results were contrary to my predictions. No nest site variables that I measured appeared to affect Rufous Fantail daily nest survival rates.

Although not a target species of this study, I incidentally found the first nests of the Micronesian Honeyeater (*Myzomela saffordi rubratra*) on Saipan. I report my descriptions of the nests, nestlings, and parental behavior in Chapter 2. Measured nests (*n* = 3) averaged 46.7 mm in cup diameter, 65.7 mm in outer diameter, 41.3 mm in cup height, and 55.3 mm in external nest height. Nesting materials were primarily vine tendrils and what appeared to be *Casuarina equisetifolia* needles. Nests, nest placement and adult bird behavior appeared similar to those reported for this species prior to its extirpation on Guam.

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CHAPTER 1

PREDATION AND NEST SUCCESS OF FOREST BIRDS IN NATIVE AND NON-NATIVE HABITAT ON SAIPAN, MARIANA ISLANDS

INTRODUCTION

Two key threats to native wildlife are habitat destruction and alteration, and invasions of non-native species. In the United States, 85% of all imperiled species are threatened by habitat loss and/or degradation, while 49% are threatened by alien species (Wilcove et al. 1998). Evaluating imperiled species by taxonomic group, habitat loss/degradation threatens 81% of plants, 92% of vertebrates, and 87% of invertebrates, while alien species endanger 57% of plants, 47% of vertebrates, and 27% of invertebrates (Wilcove et al. 1998).

An extreme example of the effect of introduced species on native fauna is the extinctions and declines of birds of oceanic islands caused by mammalian predators (e.g., Atkinson 1985, Robinet et al. 1998, and references therein, Thibault et al. 2002). Predation is the most important cause of nesting mortality for passerines (Ricklefs 1969, Martin 1993). On islands, the avifauna often has evolved in the absence of mammalian predators, and if predators are introduced to islands nesting mortality can rise dramatically (e.g., Atkinson 1985). Since 1800, 103 bird species have become extinct, and > 90% of these have been on islands, many of which are in the Pacific (Birdlife International 2000). Invasive species have been implicated in the majority of these extinctions. This phenomenon has been documented repeatedly, e.g., Hawaii, New Zealand, and Lord Howe Island. One particularly well-known example is the extinction of most of the native birds of Guam by the brown treesnake (*Boiga irregularis*; Savidge 1987). Guam is the southern-most of the Mariana Islands, located in the western Pacific Ocean. The remainder of the islands in the Mariana archipelago form the Commonwealth of the Northern Mariana Islands (CNMI). The Mariana Islands have been listed as an Endemic Bird Area by Birdlife International (Stattersfield et al. 1998). Although depauperate in terms of overall species richness, 29% of the breeding species in the Northern Mariana Islands are considered threatened (Birdlife International 2000).

Many potential predators of bird nests have been introduced to the islands of the CNMI, including three species of rat (*Rattus rattus, R. norvegicus,* and *R. exulans*), the house mouse (*Mus musculus*), a monitor lizard (*Varanus indicus*), and feral cats (*Felis catus*) (Engbring and Pratt 1985). In addition, due to the high volume of traffic between Guam and the other inhabited islands in the Marianas (Rota, Tinian, Saipan), there is the potential for brown treesnake introduction. Rota's small native forest passerines have experienced unexplained declines, leading to speculation that the brown treesnake may already have reached this island (Rodda et al. 1998). To date, there have been > 75 plausible sightings and 11 hand-captures of brown treesnakes on Saipan (Gragg 2004).

In addition to animal introductions, the arrival of humans to oceanic islands often signals the beginning of wholesale habitat alteration and destruction. On Saipan, the native forests have been subjected to cutting, burning, browsing by introduced feral ungulates, and war-related damage (Craig 1992). Following World War II, large areas of the island were seeded with the fast-growing, non-native leguminous tree, tangantangan

(*Leucaena leucocephala*) to prevent erosion (Engbring and Pratt 1985). Because of introductions, tangantangan is now a common species of secondary growth in seasonally dry lowlands of Pacific islands (Mueller-Dombois and Fosberg 1998, Meyer 2004), and Smith (1985) lists tangantangan as one of Hawai'i's 13 worst weeds. Tangantangan exists in almost monotypic stands over large parts of Saipan and 77% of the remaining forest is non-native (Falanruw et al. 1989). Habitat alteration can impact avian populations and breeding success. Of all threatened species worldwide, 85% are endangered by habitat loss and degradation (Birdlife International 2000). However, damaging typhoons occur frequently in the Mariana Islands, which may have led to the evolution of dietary and habitat opportunism in the local avifauna (Engbring and Pratt 1985, Rodda et al. 1998). This may reduce the influence of non-native forest on nesting success on Saipan.

Little research has been done on the avifauna of the Mariana Islands, and basic life history information is unknown for most of the native and endemic species. Only 5 of 16 of the native land bird species have been the focus of an ecological study (Rodda et al. 1998, Mosher and Fancy 2002), and much of this work has been on foraging ecology and potential competition between the species studied (e.g., Craig 1989, Craig and Beal 2001). Information on the impacts of introduced predators on the native avifauna of the Northern Mariana Islands is lacking. Some of the introduced mammal species are known to have caused species declines and/or extinctions on other islands. Roof rats (*Rattus rattus*), in particular, are devastating nest predators (e.g., in New Zealand and Hawaii). However, the impact of rats appears to decrease with time since introduction (Best 1969, Veitch 1994); the historical introduction of roof rats to the Marianas archipelago presumably occurred with the arrival of the Spanish approximately 400 years ago. In addition, the impact of introduced predators has been greater on temperate than on tropical islands, possibly because birds on tropical islands have evolved with the presence of land crabs, which led to the development of some defensive behaviors that were also effective against exotic predators (Côté and Sutherland 1997). It is presently unknown how the birds utilize the non-native forest in comparison to the native forest and how rates of predation may differ between the two forest types.

Therefore, I undertook this research on the island of Saipan in the Northern Mariana Islands to assess what, if any, impact predation may be having on populations of four forest passerines, all open-cup nesters. My target species were the Nightingale Reed-warbler (*Acrocephalus luscinia*; federally listed as endangered) and three of the more common forest species, the Golden White-eye (*Cleptornis marchei*; an endemic species), Bridled White-eye (*Zosterops conspicillatus saypani*) and Rufous Fantail (*Rhipidura rufifrons saipanensis*). The latter two are endemic sub-species. Specific questions I attempted to address were (i) Does native forest have higher nest densities than non-native forest? (ii) Do nests in native forest experience higher nest survival than nests in non-native forest? (iii) Is predation at a level that would result in population declines of the target species? (iv) Does nest placement influence nest survival? and (v) Which species are responsible for nest predation?

I predicted that nest densities would be higher in native forest, reflecting a preference of the target species to nest in this forest type, and that *Rattus rattus* would be the primary predator of nests. Nest survival could be lower in native forest if it supported higher numbers of predators due to food availability or microhabitat sites. Alternatively, nest survival could be higher in native forest due to greater structural complexity and therefore increased difficulty for predators in finding nests. Influence of nest site characteristics on nest survival may be positive, negative, or neutral (e.g., Moors 1983, Murphy 1983, Filliater et al. 1994, Howlett and Stutchbury 1996, Burhans and Thompson 1998, Siepielski et al. 2001, VanderWerf and Smith 2002). I expected more nest support branches to increase the risk of predation, because of more stable and thus easier access to the nest for rats. I did not expect other characteristics such as nest concealment to influence nest survival since *Rattus rattus* are primarily opportunistic or olfactory predators. Because I only found two Nightingale Reed-warbler nests, and both fledged young, results reported are from the three common target species.

STUDY AREA

Saipan lies at 15°10' N and 145° 45' E, approximately midway between Japan and Papua New Guinea in the western Pacific (Figure 1.1). The island is 22 km long and 6 km wide and is the second largest island in the Marianas. Saipan has a tropical climate with an annual mean temperature of 28.3°C, monthly mean humidity of 79–86%, and mean annual rainfall of 200–250 cm. The dry season extends from December to June, and the wet season occurs from July to November. Typhoons may occur at any time but are most frequent between August and December (Young 1989, Mueller-Dombois and Fosberg 1998).

Two forest types were the focus of my study, native limestone forest and introduced tangantangan forest. Native limestone forest is estimated to cover only 5% of Saipan (Engbring et al. 1986) and is now restricted to cliffs and steep slopes that are not easily cultivated (Craig 1989, Stinson and Stinson 1994). This forest is typically dense, with *Pisonia grandis* and *Cynometra ramiflora* dominating the canopy, and *C. ramiflora* and *Guamia mariannae* the most common species in the understory (Craig 1996). Tangantangan forest is estimated to cover 28% of the island and grows in dense near-monocultures on flat lowlands and plateaus (Craig 1990). I selected 4 native and 4 non-native forest study sites (Figure 1.1). Because of the paucity and relative inaccessibility of native forest, one of my 'native' forest study sites on closer examination was actually mixed native/agriforest, including trees such as coconut (*Cocos nucifera*) and mango (*Mangifera indica*) (Craig 1996). Marpi, As Teo, and Kagman were native forest; Laolao Bay was mixed forest; and Bird Island, Cow Town, Obyan, and Naftan were non-native tangantangan forest (Figure 1.1). Study areas were delineated by flagged transects.

METHODS AND ANALYSES

Vegetation

I used the point-centered quarter method (Mueller-Dombois and Ellenberg 1974) to characterize vegetation and assess the percentage of native versus non-native forest among the study sites. I sampled different numbers of points along the transects depending on the size of the study site. I sampled 20 points at As Teo and Marpi, 30 points at Kagman, Cow Town, Bird Island and Naftan, and 40 points at Laolao Bay and Obyan. I calculated density, absolute frequency, dominance, and dominance rank for each tree species, and overall tree density for each site. Because I was primarily interested in the gradation of native to non-native forest, I also calculated the proportions of native and non-native trees within each study site.

Nest densities

I established line transects following distance sampling methodology (Buckland et al. 2001) to estimate nest densities in native and non-native forest. Random probability sampling should be used in transect placement to avoid unrepresentative and therefore biased samples. Thus, a random start point for the transects was chosen, after which transects were systematically placed throughout the study sites. Empirical estimates of the sampling variance of object (in this case, nest) encounter rates are derived from the number of detections obtained from individual lines, and a systematic sample allows a more precise estimate (Buckland et al. 2001). Due to the range in size and shape of study areas, transects had different spacing and individual lengths among the study sites, because transects extended from boundary to boundary across the sites (Buckland et al. 2001:234-235). To minimize disturbance to vegetation, transects were marked with flagging to denote the centerline. A 25-m buffer zone from the road was not searched for nests when surveying transects. Total line lengths in 2003 and 2004 in native/mixed forest were 4,211 m and 4,204 m, and in non-native forest were 4,195 m and 4,200 m, respectively. The empirical estimation of sampling variation in object encounter rates requires a sample size of ≥ 20 individual lines (Buckland et al. 2001); 29 transects were established in native/mixed forest, and 34 in non-native forest.

I surveyed transects once in 2003, from 16 - 27 June, and three times (approximately monthly) in 2004 to account for temporal spread in nesting that was observed in 2003. Two observers simultaneously walked each transect, with one observer remaining on the centerline, and the second observer walking in a sigmoidal pattern on either side of the centerline to detect additional nests (Anderson et al. 2001).

Only nests that were active when the survey was conducted were recorded. A nest was considered active if it contained eggs or nestlings and parental activity was observed. I never recorded the same nest twice, as nesting attempts were completed between surveys. Upon detection of an active nest, perpendicular distance to the line was measured using a steel measuring tape. Species and nest stage (egg-laying, incubation, or nestling) were also recorded.

Using program DISTANCE version 4.1 release 2 (Thomas et al. 2004), I fit a series of models (half-normal, half-normal cosine, hazard-rate, hazard-rate cosine, and Fourier series 1 and 2) to the pooled detection data (all species) and then for each species separately to evaluate whether detection probability differed by species, forest type, or year. I used Akaike's Information Criterion corrected for small sample sizes (AIC_c) to select between models (Burnham and Anderson 2002). Preliminary analyses suggested that the probability of detection was similar across forest types, years and species (Appendix 1.1), and the half-normal key model function with a cosine series expansion best approximated the data. Therefore, I used a global detection function for final analyses but stratified the estimates by species, forest type and time period (monthly survey). I then assessed potential differences in density using a maximum likelihood-based linear regression. This analysis incorporated sampling correlation (from use of the pooled detection function) and uncertainty in density estimates into the regression models by partitioning variance into two components. I assumed that residual variances (ε_i) were normally distributed, independent, and homogeneous. I did not make these assumptions for sampling variances (γ_i; variation in the estimate given the true value), which were given separately for each estimate by program DISTANCE. The variancecovariance matrix for the density estimates was derived using the formula $\hat{D}_1\hat{D}_2(CV(\hat{p}))^2$, where $CV(\hat{p})$ is the coefficient of variation of the probability of detection. I decided on several *a priori* regression models (Appendix 1.2), but also fit more refined exploratory models based on the results from the *a priori* models. I used AIC_c as a model selection criterion (Burnham and Anderson 2002) to examine which of the *a priori* models best fit the nest density data. Using summed Akaike weights (w_i), I then evaluated the relative importance of each variable in the best approximating model for 2004 by studying a balanced model set (each variable and interaction is present in the same number of models) composed only of the variables present in this top model and their interactions. Akaike weights (w_i) represent the weight of evidence in support of each candidate model, and the relative importance of each variable (or interaction) is obtained by summing the Akaike weights over all models in which that variable appears (Burnham and Anderson 2002). Summed Akaike weights of ≥ 0.40 indicate the variable is having an influence on the process of interest (G. C. White, unpubl. data).

Nest survival and the influence of nest site characteristics

Nest survival

I found Bridled White-eye (BRWE), Golden White-eye (GOWE), and Rufous Fantail (RUFA) nests while surveying line transects using distance sampling methodology (Buckland et al. 2001), or while moving through the forest to monitor existing nests. When found, each nest was flagged and assigned a unique nest identification number. Nest contents were visually inspected at approximately 3-day intervals (range = 1 – 5), using a mirror on a telescoping pole if necessary.

I analyzed nest survival after Stanley (2004), which allows the determination of stage-specific (egg-laying, incubation, nestling) daily survival probabilities when stage transition or failure dates are unknown. Initially, I fit models to ascertain whether nest survival differed across species, forest types, or years. Based on the results from these models, I next compared candidate models that treated nest stages separately with those that pooled across stages. These latter models were fit separately by species, and also separately by year for Bridled White-eyes. I used the relative differences between each candidate model and the model with the minimum AIC_c value ($\Delta\mathrm{AIC}_c$ values) and Akaike model weights to identify the most parsimonious models (Burnham and Anderson 2002). Models with $\Delta\mathrm{AIC}_c$ values ≤ 2 receive considerable support (Burnham and Anderson 2002). Finally, I used the best approximating model in each case to calculate daily survival probabilities. I used SAS version 9.1 for all analyses (SAS Institute Inc. 2004). I exponentiated daily survival probabilities to give overall nest survival. For all study species, there was much variation in lengths of nest stages. Therefore, I calculated overall nest survival using the shortest possible, longest possible, and an intermediate length nesting cycle. Confidence intervals for daily survival probabilities from SAS proc NLIN (utilized by Stanley (2004)) are incorrect when data are entered as sufficient statistics (Armstrong et al. 2002; T. R. Stanley, pers. comm.). Thus, I used the methods described by Burnham et al. (1987) and Armstrong et al. (2002) to compute confidence intervals for daily survival probabilities and overall nest survival. The approximate 95% confidence intervals for the daily survival rates ($\hat{r}$) are given by

$$\hat{r}_L = \frac{\hat{r}}{\hat{r} + (1-\hat{r})C} \quad \text{and} \quad \hat{r}_U = \frac{\hat{r}}{\hat{r} + (1-\hat{r})/C}$$

where $C = \exp\left[\frac{2se(\hat{r})}{\hat{r}(1-\hat{r})}\right]$ (Armstrong et al. 2002).

To calculate approximate 95% confidence intervals for overall nest survival, the standard error of the overall survival estimate was first calculated using the delta method, as illustrated by Armstrong et al. (2002). Because I did not pool daily survival rates across the egg-laying and incubation stages for Bridled White-eyes and Golden White-eyes, I extended the equation given by Armstrong et al. (2002) to

$$\sqrt{\left(se(\hat{r}_1)\left(\frac{\delta}{\hat{r}_1}\right)\right)^2 + \left(2\mathrm{cov}[\hat{r}_1\hat{r}_2]\left(\frac{\delta}{\hat{r}_1}\right)\left(\frac{\delta}{\hat{r}_2}\right)\right) + \left(2\mathrm{cov}[\hat{r}_1\hat{r}_3]\left(\frac{\delta}{\hat{r}_1}\right)\left(\frac{\delta}{\hat{r}_3}\right)\right) + \left(se(\hat{r}_2)\left(\frac{\delta}{\hat{r}_2}\right)\right)^2 + \left(2\mathrm{cov}[\hat{r}_2\hat{r}_3]\left(\frac{\delta}{\hat{r}_2}\right)\left(\frac{\delta}{\hat{r}_3}\right)\right) + \left(se(\hat{r}_3)\left(\frac{\delta}{\hat{r}_3}\right)\right)^2}$$

where $\frac{\delta}{\hat{r}_1} = t_1\hat{r}_1^{\,t_1-1}\hat{r}_2^{\,t_2}\hat{r}_3^{\,t_3}$, $\frac{\delta}{\hat{r}_2} = t_2\hat{r}_2^{\,t_2-1}\hat{r}_1^{\,t_1}\hat{r}_3^{\,t_3}$, and $\frac{\delta}{\hat{r}_3} = t_3\hat{r}_3^{\,t_3-1}\hat{r}_1^{\,t_1}\hat{r}_2^{\,t_2}$, and t_1 is the duration of the egg-laying stage, t_2 is the duration of the incubation stage, and t_3 is the duration of the nestling stage. The approximate 95% confidence interval for overall nest survival is then calculated using the same formulas as given for $\hat{r}_L$ and $\hat{r}_U$ above.

Based on results obtained from the nest density analyses, I fit exploratory models to determine whether nest survival differed by season in 2004. I classified season as early or late based on desiccation of the tangantangan forest. The months of February and March were classed as early season, and April and May as late season.

Nest site characteristics

I was interested in whether different placements affected the vulnerability of nests to predation. I predicted that *R. rattus* would be the primary nest predator, therefore I measured nest site characteristics that I thought would influence rat access to the nest. However, I also measured some characteristics that could influence visually cued (e.g., avian) predators. Characteristics measured included: tree and nest heights (using a clinometer), distance of the nest from the trunk (using a steel measuring tape), number of connecting branches from neighboring trees leading to the nest tree, number of nest support branches, diameters of the support branches (using a millimeter ruler), canopy cover (using a densiometer), side cover (by averaging visual estimates from the four cardinal directions at 1 m from the nest), ground cover (using Daubenmire plots; Daubenmire 1959), distance from the nearest road (grouped into 25-m intervals), and road type (e.g., heavily traveled dirt road, lightly traveled 2-track). I also recorded the tree type in which the nest was built (native or non-native species).

I utilized a non-linear mixed modeling approach using proc NLMixed in SAS version 9.1 (SAS Institute Inc. 2004) to model the influence of nest site characteristics on nest survival (Rotella et al. 2004, Shaffer 2004). This modeling approach is based on nest status (successful or failed) over the exposure interval of the nest, and estimates daily survival rates in relation to measured covariates. However, I did not use this method of analysis to provide daily and overall survival rates, as this modeling approach assumed homogeneity of daily survival rates, and results using the Stanley (2004) method indicated that daily survival rates differed by nest stage. In addition, I was unable to obtain information for all nest site characteristics for all nests. Comparison of models using AIC_c requires that data sets remain the same, and I censored nests for which I did not have complete nest site information (n = 80). I did not examine the effects of nest site characteristics on nest survival using the Stanley (2004) method, because it requires data to be partitioned into subsets, which leads to loss of information when transforming continuous variables into categorical variables. It is also difficult using the approach of

Stanley (2004) to examine more than one or two factors at a time. I had many continuous variables and wanted to assess possible additive effects of various factors. Thus, whereas it would have been better to examine nest survival and the influence of nest site characteristics on survival together, I instead used the Stanley (2004) method to obtain estimates of nest survival and proc NLMixed to model the influence of nest placement characteristics on nest survival.

I noticed a division of forest type use between BRWE and GOWE, with BRWE nesting predominantly in non-native forest and GOWE nesting primarily in native forest. Therefore, I fit the models separately by species. I decided on various *a priori* models for analysis (Appendix 1.3). Because all documented nest predation was avian, I concentrated my analysis on factors more likely to influence visually cued predators. I was concerned with potential differences in nest survival between years, due to the variation in field season timing between 2003 and 2004. Thus, I initially included year effects in all models. However, year was correlated with all other variables; final candidate models were therefore fit without a year effect.

Nest predator identification

I used both video and still cameras to identify nest predators. I placed cameras 1–2 m from the nest, and VCR base stations (where applicable) 15–30 m from the nest. I placed all still cameras at nests in the incubation stage. I set up 13 video cameras at nests in the incubation stage, and 3 in the nestling stage. I limited set-up time to blocks of ≤ 10 mins. In 4 cases (2 video cameras and 2 still cameras) I required more than 10 mins to complete camera set-up. In these cases, after 10 mins at the nest I took a 10-min break (or longer if the incubating adult had not yet left the nest) to allow the adults to resume incubation.

Still cameras

I used 10 Canon Sure Shot Owl PF Date 35mm automatic cameras, wired to be triggered by a bait egg. The use of an egg trigger involves securing a line to an egg added to the natural clutch in the nest. The way the line exits the nest is important, as it may impede incubation of the eggs by the female and thus influence hatching success (Farnsworth and Simons 2000). Design of the egg trigger followed Major and Gowing (1994), with the line exiting the bottom of the nest, as this did not appear to affect incubation.

Still cameras were housed in modified Rubbermaid® weatherproof containers, painted with a green and brown camouflage design, and disguised with natural vegetation where possible. I placed each camera at an active nest until the nest failed or fledged, at which time I moved the camera to the next available, suitable nest.

Because I initially encountered many problems with the still cameras, I used them to monitor 33 artificial nests in the first field season. Actual nests of the target species were used to construct the artificial nests as follows: recently active nests maintained in their original positions ($n = 3$); recently active nests collected and set up in the location of a previously monitored nest that was no longer present (e.g., due to predator destruction; $n = 13$); and collected nests set up in locations that appeared to be representative of active nest locations ($n = 17$; Appendix 1.4). Number of nests of each species used approximated their relative abundances by study site. Zebra finch (*Taeniopygia guttata*) eggs shipped from the mainland were all destroyed in transit. Therefore, most nests ($n = 24$) were baited with budgerigar (*Melopsittacus undulatus*) eggs obtained locally. Two nests were baited with GOWE eggs and 7 nests were baited with BRWE eggs collected from abandoned nesting attempts. Each nest was baited with one egg that also acted as the camera trigger egg. Cameras were left on nests for 3 – 12 days ($\bar{x} = 7.8$, mode = 9). Time of exposure depended primarily on bait egg deterioration as judged by odor.

I modified the still cameras between field seasons. Due to problems encountered with using fresh eggs as bait in 2003, I used wax eggs for triggering the cameras in 2004. These were made by blowing zebra finch eggs, and then filling them with white or blue candle-wax. Zebra finch eggs are thin and semi-transparent, and therefore the different wax colors simulated egg colors of the target species. The line securing the egg to the switch was set inside the wax, preventing adhesive failure. The wax eggs also provided an additional means of predator identification through imprints in the wax. I made additional minor modifications to allow the cameras to be mounted at the nest more quickly.

In 2004, I placed the cameras on 21 active nests. I placed three of these on BRWE nests, of which all were abandoned, despite modifications to placement and placement methodology between each set-up. Therefore, I deemed BRWE unsuitable to receive still cameras. Of the remainder, I placed 14 still cameras on RUFA nests and 4 on GOWE nests. The predominance of RUFA in this sample reflects both the accessibility of their nests relative to GOWE nests and that RUFA nesting attempts were more evenly spread throughout the field season.

Video cameras

To limit costs, I used quadriplex units (SuperCircuits QS20 B/W Quad Processor) that allowed up to four cameras to feed input into a time-lapse videocassette recorder (VCR; Nuvico NVVA-96N). Video systems were powered with rechargeable 12-volt, deep-cycle batteries that were replaced every 24 hours. I had 2 "base stations" (VCR, quadriplex unit and batteries) and 6 video cameras available. I used Sony E*View Super HAD black and white bullet cameras with a one-third-inch charge-coupled device and a homemade array of six 940nm infrared light-emitting diodes (LEDs) arranged in a circular fashion around the 6 mm lens. LEDs and lenses were shielded against the weather. Base stations were housed in weatherproof coolers and covered with a green and brown camouflage design tarpaulin. I disguised video cameras with natural vegetation found on site. As with the still cameras, I placed each set-up at an active nest until the nesting attempt was completed (failed or fledged), at which time I moved the set-up to the next available, suitable nest.

In 2003, the video cameras were not available until June. By this time, nesting attempts were fewer and nest densities lower, making it difficult to use multiple cameras with a base station. Over the two field seasons, a total of 16 nests (5 in 2003 and 11 in 2004) were monitored with time-lapse video: 4 BRWE, 4 RUFA, 1 GOWE and 2 NIRW in the four non-native sites, and 4 GOWE and 1 RUFA in native and mixed forest. The uneven distribution of camera placements across forest types and species reflects the relative accessibility of nests and the temporal distribution of nesting attempts. No nests were monitored by video at the As Teo and Kagman study sites due to difficult access.

RESULTS

Vegetation

Because of the high percentage of non-native trees at Laolao Bay (Table 1.1), this study site was treated separately as a "mixed" forest type in all further analyses. There was a much higher variety of tree species in the native and mixed forest sites, with the non-native sites overwhelmingly dominated by tangantangan (Appendix 1.5).

Nest densities

I found a total of 191 nests using line transect sampling (Figure 1.2). Nest density estimates were stratified by species, forest type, and survey period. Estimated nest densities ranged from 0 – 115/km^2 (Table 1.2; Appendix 1.6). BRWE nest densities ranged from 0 – 115/km^2 in non-native forest and 0 – 11/km^2 in native/mixed forest. GOWE nest densities ranged from 0 – 24/km^2 in non-native forest and 3 – 41/km^2 in native/mixed forest, and RUFA nest densities varied from 0 – 16/km^2 in native/mixed forest and 0 – 25/km^2 in non-native forest. Because of small sample sizes obtained per stratum, coefficients of variation of all density estimates varied from 24 – 94%, leading to overlap in 95% confidence intervals among many density estimates. However, some trends were evident from the estimates (Table 1.2, Appendix 1.6). In all surveys, GOWE had higher estimated nest densities in native forest than BRWE. In 3 out of 4 surveys, BRWE had higher estimated nest densities in non-native forest than GOWE (in the fourth survey, Apr/May 2004, both species had no nests in non-native forest). In general, estimated nest densities of GOWE in native and non-native forest were similar, while BRWE nest densities were much higher in non-native than in native forest. In 2004, estimated nest densities generally declined in the native and non-native forest from the Feb/Mar to the Apr/May survey periods. This decline was more pronounced in the non-native forest, where no nests of any species were found in Apr/May. Estimated nest densities in the mixed forest remained relatively similar over the 3 survey periods in 2004 and were also similar among the species.

Fewer regression models (Appendix 1.2) could be examined for the 2003 data. I only surveyed the transects once so there were no data on seasonal effects, and sample size was smaller. The best approximating model suggested the most influential factor on nest densities in 2003 was the proportion of native forest in each study site (Table 1.3; Appendix 1.7). In this model, the regression coefficient ($\hat{\beta}$) for proportion native forest was -21.35 (95% CI = -30.77, -11.93); thus, the highest nest densities were predicted to occur in non-native forest (Figure 1.3).

In 2004, the best approximating model included an additive species effect and a forest type by month interaction (Table 1.3; Appendix 1.7). This was an exploratory model not included in the original *a priori* model set. The species effect (BRWE) treated Bridled White-eyes separately while pooling Golden White-eyes and Rufous Fantails together. The forest type effect (native/mixed) pooled the native and mixed forest types and treated non-native forest separately, while the month effect (Feb/Mar) treated the first survey period separately from the latter two survey periods (Mar/Apr and Apr/May, pooled). The $\hat{\beta}$ (and 95% CI) for these effects were: BRWE = -2.31 (-4.51, -0.11), native/mixed = 2.02 (-0.37, 4.41), Feb/Mar = 25.81 (12.53, 39.10), and native/mixed*Feb/Mar = -24.87 (-38.48, -11.25; Table 1.4). Thus, this model predicted that nest densities were much higher in non-native forest than native forest during Feb/Mar, but that nest densities were slightly higher in native forest than non-native forest during Mar/Apr and Apr/May, with small differences in nest densities among the species in the forest types (Figure 1.4).

There were two other competing models, both with $\Delta \mathrm{AIC}_c$ values < 2. Based on the three top models, there is considerable model uncertainty concerning the species effect. The models include an additive species effect, no species effect, and an interaction effect between species and forest type.

Examination of the balanced model set for the species (BRWE), forest type (native/mixed), and survey period (Feb/Mar) effects indicated that the Feb/Mar and native/mixed effects were more important than the BRWE effect ($\sum w_i$ = 0.98, 0.96, and 0.73, respectively), however that the BRWE effect still had considerable influence on nest densities. The most important interaction was native/mixed*Feb/Mar ($\sum w_i$ = 0.94; Table 1.4). The BRWE*native/mixed and BRWE*Feb/Mar interactions did not appear to influence nest densities, having relative importance values < 0.40 ($\sum w_i$ = 0.20 and 0.14, respectively). Thus, it would appear that forest type and time of year have a greater influence on nest densities than avian species. However, the estimated nest densities suggested that BRWE nest densities were higher in non-native than native forest, and GOWE nest densities were fairly similar in these two forest types.

Nest survival and the influence of nest site characteristics

Nest survival

A total of 437 nests were monitored over both years (Table 1.5). BRWE nests were found almost exclusively in non-native tangantangan forest (160/198 nests). In contrast, GOWE nests were found predominantly in native and mixed forest (92/139 nests), while RUFA were relatively evenly distributed among forest types (40 in native and mixed forest, 60 in non-native forest). Preliminary analyses showed that nest survival differed among species. GOWE and RUFA nest survival did not differ by forest type or year (Table 1.6). BRWE nest survival differed by year (Table 1.6). The model treating BRWE nest survival separately by forest type was a competing model ($\Delta \mathrm{AIC}_c$ =

1.84), thus BRWE nest survival (both daily survival rates and overall nest survival) was calculated separately by forest type. Because of the small sample sizes of BRWE nests in native ($n = 7$) and mixed ($n = 31$) forest, these forest types were pooled in this analysis. There was a slight trend towards lower BRWE nest survival in non-native forest, however 95% CI for survival overlapped between non-native and native/mixed forest (Table 1.7), and BRWE nest survival was treated as constant across forest types in final analyses. Daily survival rates differed across all nest stages (egg-laying, incubation, nestling) for BRWE and GOWE, while the daily survival rate for RUFA was constant across the incubation and nestling stages (Table 1.8). Final models showed that BRWE had lower daily nest survival in the incubation (0.904) and nestling (0.928) stages in 2004 than either GOWE (0.958 and 0.969, respectively) or RUFA (0.967 in both stages). The confidence interval for the egg-laying stage for BRWE in 2004 was so broad that a comparison was meaningless. In 2003, confidence intervals for BRWE daily nest survival in all stages overlapped those for GOWE and RUFA. When the shortest stage lengths were used to calculate overall nest survival, BRWE in 2004 had lower overall nest survival (17.1%; 95% CI 11.1, 25.3%) than GOWE (40.0%; 95% CI 29.6, 51.5%) but not RUFA (28.0%; 95% CI 16.6, 43.1%; Table 1.9). When intermediate or longest stage lengths were used, confidence intervals of BRWE and GOWE nest survival began to overlap. However, there was still a tendency for BRWE nest survival to be somewhat lower than that of GOWE (Table 1.9). This is probably due to the greater change in nesting period from shortest to longest in GOWE (12 days) compared to BRWE (7 days). RUFA had the longest nesting period of all three species (up to 36 days); thus, the lack of difference in overall nest survival between RUFA and BRWE was presumably due to the longer exposure of RUFA nests to potential predation, despite a difference in daily nest survival. This underscores the value of examining and reporting daily survival probabilities in nest survival studies.

The exploratory analysis to assess whether nest survival differed seasonally indicated that GOWE and RUFA nest survival did not differ from early (February/March) to late (April/May) season in 2004 (Table 1.10). The model treating nest survival separately by season was a competing model in the case of BRWE (Table 1.10). BRWE daily and overall nest survival rates were calculated separately for early and late season, however neither differed between February/March and April/May (Table 1.11).

Nest site characteristics

Although I did not choose my study sites randomly, I modeled the study site effect as random to allow general inference to be made to the forest types, rather than to just my specific study sites. The random study site effects model was not included in the final model set for RUFA because the model did not converge. No nest site characteristics that I measured appeared to influence RUFA nest survival; the intercept only model was the best approximating model (Table 1.12). The 95% confidence intervals broadly overlapped zero for all regression coefficients ($\hat{\beta}$; Table 1.13). The number of nest support branches appeared to increase GOWE nest survival ($\hat{\beta} = 0.50$, 95% CI 0.08, 0.92; Table 1.14; Figure 1.5). The model including number and diameter of nest support branches seemed to be a competing model ($\Delta \text{AIC}_c \leq 2$), but the 95% confidence interval for the regression coefficient for support branch diameter evenly overlapped zero ($\hat{\beta} = -0.05$, 95% CI -0.29, 0.20; Table 1.15). Nest concealment (both side and canopy cover) appeared to decrease BRWE nest survival (Table 1.16). The 95%

confidence interval for the regression coefficient for side cover did not overlap zero ($\hat{\beta}$ = -1.64, 95% CI -3.23, -0.05), but the confidence interval for the canopy cover regression coefficient overlapped zero ($\hat{\beta}$ = -3.93, 95% CI -9.4, 1.6; Table 1.17). When these effects were plotted against nest survival for BRWE, only side cover appeared to have a real effect on BRWE nest survival (Figure 1.6). This would explain the model selection uncertainty in the results, as evidenced by small ΔAIC_c differences and similar model weights (Burnham and Anderson 2002).

Nest predator identification

Still cameras

Only 2 (possibly 3) of the 33 artificial nests monitored with still cameras in 2003 were depredated, and in no instance was the camera triggered to take a photograph of the predator. In all cases, adhesive failure caused parts of the trigger system to come apart. In 2004, 6 of 21 nests monitored with still cameras were depredated. No photographs of predators were taken, however imprints were left in the wax bait/trigger eggs. The imprints were examined under a dissecting microscope and all were identified as avian. Based on the size and characteristics of the bill imprints, 4 were made by Micronesian Starlings (*Aplonis opaca*), 1 by a Collared Kingfisher (*Halcyon chloris*), and 1 by an unidentified bird smaller than a Micronesian Starling. It is possible, though not certain, that this last case may have been the parent bird (RUFA) trying to remove the bait egg from the nest.

Video cameras

In 2003, one RUFA nest was depredated in the nestling stage by a Micronesian Starling. The other four videotaped nests fledged young. In 2004, 1 GOWE and 2 RUFA nests were depredated, all in the nestling stage; 2 of these (1 GOWE and 1 RUFA) were depredated by Micronesian Starlings, and the second RUFA nest was depredated by a Collared Kingfisher. Of the remaining nests monitored, 5 fledged, 1 was abandoned after ≥ 15 days of incubation when the egg (infertile) failed to hatch, 1 was still active at the time of camera removal, and the fate of the final nest was unclear. The nestling may have died from starvation.

Over both field seasons, in 228 days of recording, I experienced 3 blown fuses and 1 instance where the tape had stopped recording for no apparent reason. I also had 31 cases (29 in 2004) of premature battery failure and/or tape ejection. Battery life decreased by 1 – 66% over the span of the project. I suspect that after six months of storage, the batteries had begun to deteriorate and occasionally did not re-charge or retain the charge completely. No predation events were missed due to battery failure.

DISCUSSION

Nest densities

Model results suggested that the most influential factor on nest densities in 2003 was the proportion of native forest in each study site, with the highest nest densities predicted to occur in non-native forest. The best approximating model in 2004 included a species effect and a habitat by month interaction, and predicted the highest nest densities for all species were in non-native forest in late February/early March, with a slight difference depending on whether they were BRWE or GOWE and RUFA (Figure 1.4). In general, this model corresponded with estimated nest densities, with one exception. The model predicted higher nest densities of GOWE and RUFA than BRWE in non-native forest in Feb/Mar, whereas nest density estimates obtained from program DISTANCE suggested the opposite. The best approximating model did not explain a lot of the variation in the nest density estimates ($R^2 = 0.162$), and based on the three top competing models, there was uncertainty regarding the species effect. This could be due to the variability in the nest density estimates, or the lack of inclusion of other more important explanatory variables, such as prey availability and/or landscape context of individual study sites. Given the results from both years, and examination of the balanced model set in 2004, it would appear that forest type and time of year have the strongest influence on nest densities. However, forest type appears to affect the species in different ways. BRWE nest densities were much higher in non-native than in native forest, whereas GOWE nest densities tended to be similar in native and non-native forest (Table 1.2; Appendix 1.6).

The relative lack of BRWE in native forest and their predominance in non-native forest is interesting, as tangantangan forest is a relatively recent addition to the landscape. Foraging studies have found BRWE feed from relatively thin perches (< 5 mm), among small leaves or leaflets, in the canopy, and in sunlit areas (Craig 1989, Craig and Beal 2001). Tangantangan has a spindly growth form, and leaves are twice pinnately compound with 11–17 pairs of leaflets. It has short branches and thus does not provide much shade (Raulerson and Rinehart 1991). This description fits well with observed BRWE foraging habitat.

Nest survival and the influence of nest site characteristics

Nest survival

BRWE nest survival was lower than that of GOWE or RUFA in 2004. Otherwise nest survival was comparable between species and years. The timing of my fieldwork in 2004 coincided with the dry season. In 2003, fieldwork was performed during the transition from the dry to the wet season. The difference in the moisture content of tangantangan was visually obvious, with leaves becoming brown, dry and shriveled soon after my arrival on Saipan in 2004. This was not observed at any time during 2003, nor was the native or mixed forest ever observed to desiccate to the same extent as the tangantangan. BRWE nests were found almost exclusively in non-native tangantangan forest (160/198 nests). I suspected the drying out of the tangantangan had a strong influence on BRWE nest survival, perhaps through an effect on insect abundance. However, when I examined BRWE nest survival in early vs. late season in 2004 (time periods were defined by the desiccation of the tangantangan), there was no difference. This may be due to a density-dependent effect, e.g., earlier in the season there were more nesting pairs competing for resources, thus nest survival was similar to later in the season where there were fewer resources but also fewer nesting pairs competing for those resources. Thus, proportionately, the amount of resources available to nesting adults was the same. Alternatively, this may be related to predators. If, for example, the peak of breeding activity of the primary nest predators coincided with that of the focal (prey) species, there may be higher numbers of nests depredated earlier in the season, and fewer later in the season, but proportionately nest survival remains the same.

These results are of particular interest when viewed in combination with the nest density results. Estimated densities of BRWE nests were highest in non-native forest, however nest survival was lower in this habitat in 2004. It would appear that this species is not using optimal habitat for nesting. Van Horne (1983) first warned that density can be a misleading indicator of habitat quality. In a recent review, Bock and Jones (2004) found that a disconnect between avian density and reproductive success was more likely to occur in disturbed than in relatively natural areas. Bock and Jones (2004) suggested that this could be because species relationships are restructured through disturbance, or because birds may not correctly recognize sub-optimum habitats that they have not evolved with. Both of these conditions may apply to BRWE use of tangantangan forest.

Most published studies reporting daily and/or overall nest survival rates are from North America. However, in comparison with three studies I found reporting daily or overall survival rates for passerines in tropical forests, those of GOWE and RUFA on Saipan were higher (Roper and Goldstein 1997, Woodworth 1997, Robinson et al. 2000). Roper and Goldstein (1997) calculated daily survival rates of 0.928 and 0.923 in the egg-laying/incubation and nestling stages, respectively, for the Western Slaty Antshrike (*Thamnophilus atrinucha*) in Panama. Overall Mayfield nest survival in Panama ranged from 8.3 – 38.5% for various species (Robinson et al. 2000). Woodworth (1997) found mean daily survival rates of 0.923 ± 0.011 and 0.940 ± 0.014 in the egg-laying/incubation and nestling stages for the Puerto Rican Vireo (*Vireo latimeri*). BRWE daily nest survival was also higher in the nestling stage in 2003 when compared with these tropical forest studies, but lower in the incubation stage in both years and in the nestling stage in 2004 (Roper and Goldstein 1997, Woodworth 1997). Overall BRWE nest survival was generally higher than that reported for other tropical species (Robinson et al. 2000). Two of these 3 studies were conducted in mainland tropical forests, therefore I also looked for studies examining nest survival on islands. Most published studies reporting nest survival on islands were conducted in the Hawaiian Islands (Pletschet and Kelly 1990; Simon et al. 2000; Simon et al. 2001; Woodworth et al. 2001), with one study found from an offshore island of New Zealand (Armstrong et al. 2002). When compared with the passerines studied in the Hawaiian Islands, daily and overall survival rates of BRWE, GOWE, and RUFA were lower than 2 species, the Maui Parrotbill (*Pseudonestor xanthophrys*), average Mayfield nest success 42% (Simon et al. 2000) and the ‘Akohekohe (*Palmeria dolei*), average daily survival rate 0.989, Mayfield nest success 68% (Simon et al. 2001); similar to 1 species, the Hawai’i Creeper (*Oreomystis mana*), daily survival rate 0.960 ± 0.009 (Woodworth et al. 2001); and higher or lower than a fourth species, the Palila (*Loxioides bailleui*), depending upon nest stage and length of nesting cycle (Pletschet and Kelly 1990). Perhaps of most interest is that the daily and overall survival rates of BRWE, GOWE, and RUFA were approximately the same as those for New Zealand Robins (*Petroica australis*) on an island that is free of all introduced predators (Armstrong et al. 2002). Ninety-five percent CI for New Zealand Robin daily survival rates were 0.909 – 0.986 and 0.909 – 0.996 in the egg-laying/incubation and nestling stages, respectively, giving overall nest survival (95% CI) as 7 – 50% (Armstrong et al. 2002).

The three species that I studied on Saipan may nest multiple times in a year. Although I found distinct peaks in nesting activity (Figure 1.2), some nests were found in all months of my study. When combined with previous studies, nesting has been reported in most months of the year (Hartert 1898, Baker 1951, Jenkins 1983, Stinson and Stinson 1994, Craig 1996). In addition there were several instances where, following a nest predation event, a new nest was found within a week within 10 m of the failed nest. Since I did not mark individuals I cannot be certain, but it is likely that these were re-nesting attempts. Thus, not only is it possible for these species to nest throughout the year, they may also re-nest following a failed attempt. High daily nest survival rates combined with multiple nesting attempts explain the healthy populations of my target species, which may be at habitat saturation (Craig 1996).

Nest site characteristics

I found weak evidence for influence of nest placement on survival. However, the directions of the relationships were contrary to what I predicted. I expected nest survival to decrease with more support branches (e.g., Mezquida et al. 2004), but instead the number of nest support branches was positively related to the daily survival rate for GOWE nests (Figure 1.5). I had no initial prediction for the direction of an effect in the case of nest concealment (side cover and canopy cover), however based on prior studies, I would have expected increased cover to result in higher nest survival, as the nest would be more difficult for a predator to detect (Martin 1992). Again, I found the opposite, with BRWE daily nest survival inversely related to increasing side cover (Figure 1.6). Götmark et al. (1995) found that nest sites with intermediate concealment were chosen by Song Thrushes (*Turdus philomelos*), possibly so that incubating adults could see approaching predators. Bridled White-eyes do not seem to aggressively defend their nests (pers. obs.), and may prefer limited vegetation around the nest to be able to view predators and flee, perhaps additionally distracting the predator away from the nest.

Alternatively, it is possible that these nest site characteristics are not important, but instead that they are correlated with unmeasured factors that do influence nest survival. The birds may be selecting for specific locations based on other factors of importance (e.g., protection from weather). Amarasekare (1994) suggested that selection for nest sites by endemic Hawaiian birds was not driven by predation pressure, and the same may be true of Saipan birds. The nest placement characteristics that I measured were mostly aimed at examining vulnerability of the nest to rat predation. Instead, my results suggest that avian predation may be more predominant. The high degree of model selection uncertainty in my results (small ΔAIC_c values and model weights) suggests I am using models with poor explanatory variables.

Nest predator identification

Contrary to my original hypothesis that *Rattus rattus* would be the primary nest predator, all documented predation events ($n = 10$) were by native birds, primarily Micronesian Starlings and Collared Kingfishers. Because of the limited sample size, I cannot rule out other potential nest predators. It is possible that other predators avoided nests monitored with cameras. For example, it is well documented that *Rattus* species are neophobic (e.g., Innes 1978, Moors et al. 1992). However, many of the cameras were in place for ≥ 10 days and up to a month in some cases, and other authors have photographed *Rattus rattus* at nests (e.g., VanderWerf 2001). Other studies have found no effects of cameras on predation of nests (e.g., Thompson and Burhans 2003, Stake et al. 2004). If a potential nest predator depredated nests with a frequency likely to influence any of my target species at a population level, I would have expected to record at least one predation event by that species.

Thus, although I cannot rule out introduced species as nest predators, they did not appear to be detrimentally affecting the native forest passerines that were the focus of my study. This is also shown by the high nest survival rates estimated in this study. The introductions of many of the exotic species on Saipan occurred with discovery and settlement by the Spanish, about 400 years ago (Engbring and Pratt 1985, Mueller-Dombois and Fosberg 1998). It is possible that the birds now present are those that can tolerate these introduced predators (King 1984, Veitch 1994). Additionally, on Saipan the native birds evolved with the presence of land crabs, and may have developed defensive behaviors that are also effective against introduced predators (Atkinson 1985).

MANAGEMENT RECOMMENDATIONS AND SUGGESTIONS FOR FUTURE RESEARCH

Protection of the native forest that remains on Saipan is a high priority. My study indicates native forest appears to provide important habitat in the dry season, and to provide more consistent nest survival for BRWE when they are able to utilize it. Protection of tangantangan stands is also desirable. Tangantangan is extensively cut by locals for firewood, and I noticed an obvious recession in the forest boundary between my 2003 and 2004 field seasons at two of my study sites. However, tangantangan currently provides most of the forest habitat remaining to the native birds on Saipan, and was used by all three study species. Healthy seedlings and saplings of native tree species were observed growing in the understorey of the tangantangan study sites, and were used for nesting by the focal species. However, given the current paucity and small size of native seedlings in tangantangan after approximately 50 years, native forest may not be able to regenerate in established tangantangan stands. I recommend regular surveys of tangantangan forest to monitor the progress of native tree regeneration.

The three species I studied on Saipan do not appear to be negatively affected by predation on nests. However, this would certainly change with the introduction of brown treesnakes. Current trapping methods employed to detect brown treesnakes result in low capture rates, particularly at low snake and high prey densities (Gragg 2004). Therefore, I recommend intensive nighttime visual surveys by trained observers, focused around ports of entry. Visual surveys detected 95% of individuals across all size classes in a known brown treesnake population on Guam (G. H. Rodda, pers. comm.). Early detection of snakes is critical for the protection of native species, as control or eradication is currently not feasible once a large population of brown treesnakes is established.

This study found a relative lack of BRWE in native and mixed forest, and previously unreported temporal peaks in nesting. A year-round, multi-year study could determine whether peaks in nesting are influenced by changes in weather or resource abundance, or whether they occur cyclically, suggesting the birds have a fixed "recovery" period between breeding attempts. Regular line transect sampling to track nest densities would be desirable. It would also be helpful to pinpoint the reason for the predominance of BRWE in non-native forest, as this appears to be sub-optimal nesting habitat for them in some years.

It could also be interesting to further examine the question of the apparent lack of rat predation on nests. One way to do this would be to assess the densities of rats on Saipan. Active rats were observed diurnally in my study sites (pers. obs.), and anecdotally it appears that diurnal activity patterns are only observed when rat densities are very high. However, there could be an alternative explanation for this activity pattern on Saipan, and rat densities may be very low, leading to the lack of documented rat predation. I would have expected high rat predation on nests if rat densities were very high, and thus to have recorded at least one rat predation event with the cameras. The limited availability of video cameras in this study may have contributed to a lack of documented rat predation. An alternative approach to address this question in the future would be through monitoring a larger sample size of nests with video cameras.

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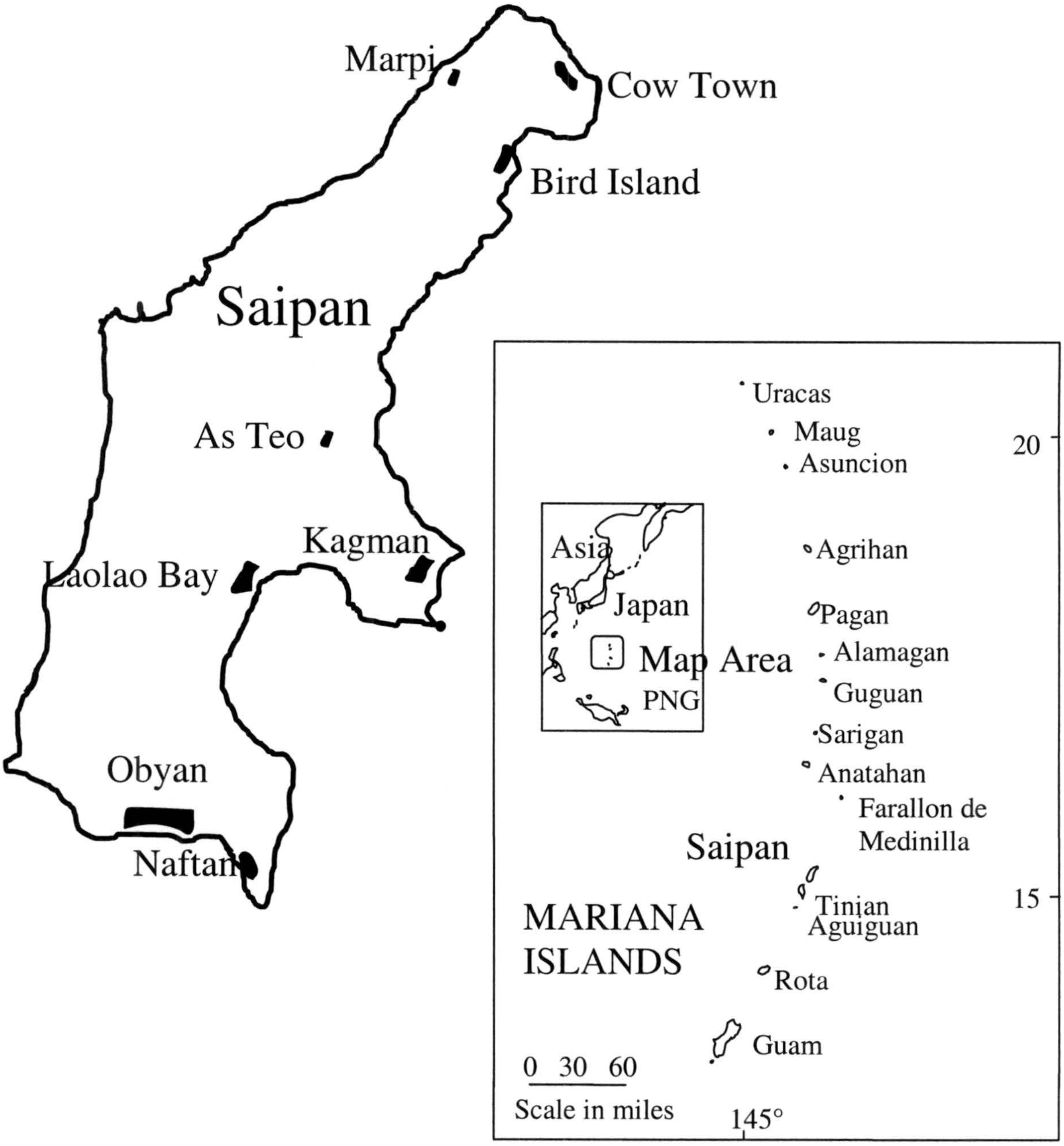

Figure 1.1. Study sites (shaded areas) and location of Saipan, Northern Mariana Islands.

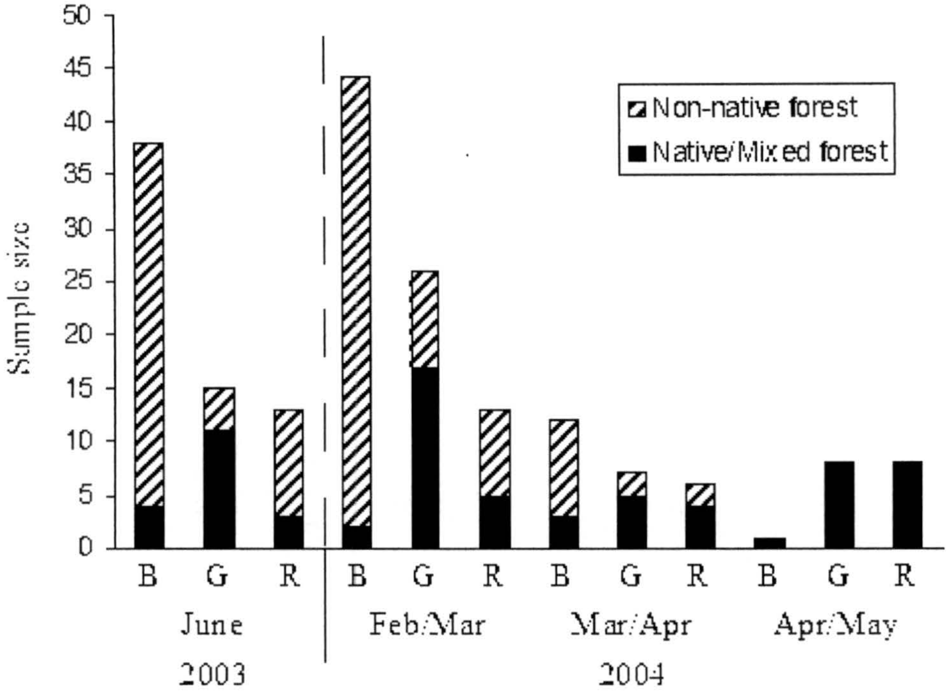

Figure 1.2. Numbers of Bridled White-eye (B), Golden White-eye (G), and Rufous Fantail (R) nests found using line transect sampling on Saipan, 2003 and 2004, illustrating temporal peaks in nesting.

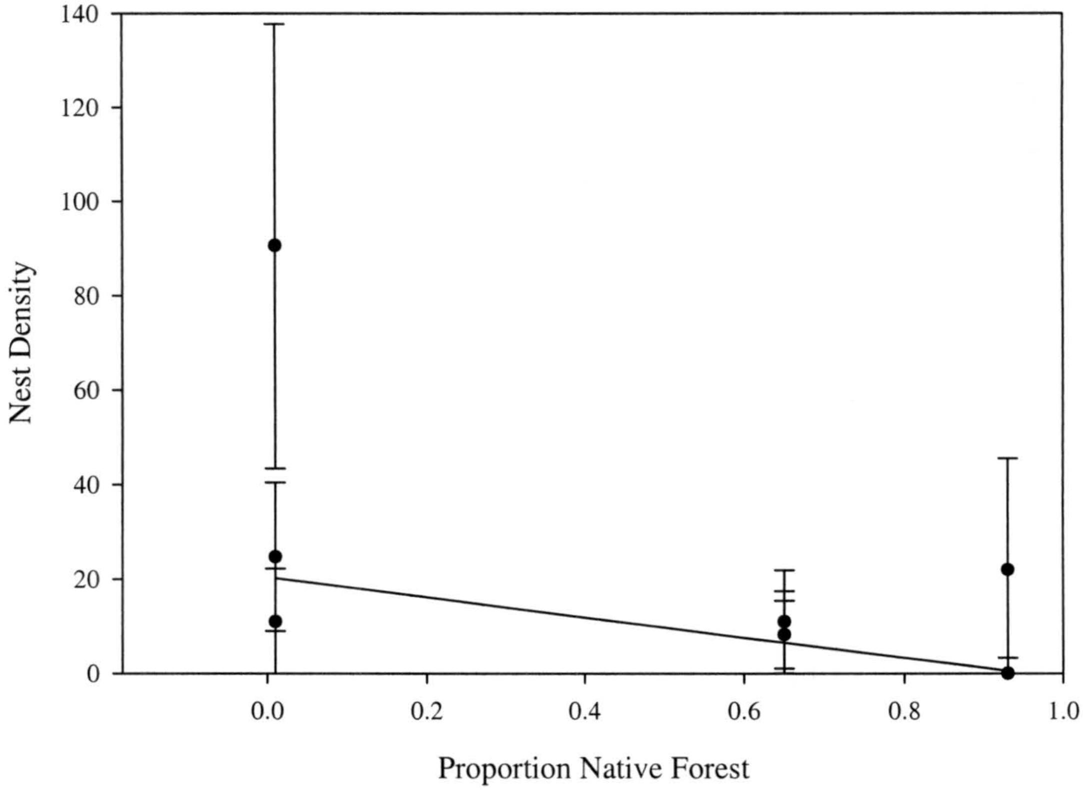

Figure 1.3. Regression line is from the best generalized linear regression model for 2003 [$\hat{D} = \beta_0 + \beta_1$(% Native forest)]. Dots with 95% CI are density estimates used to develop the model.

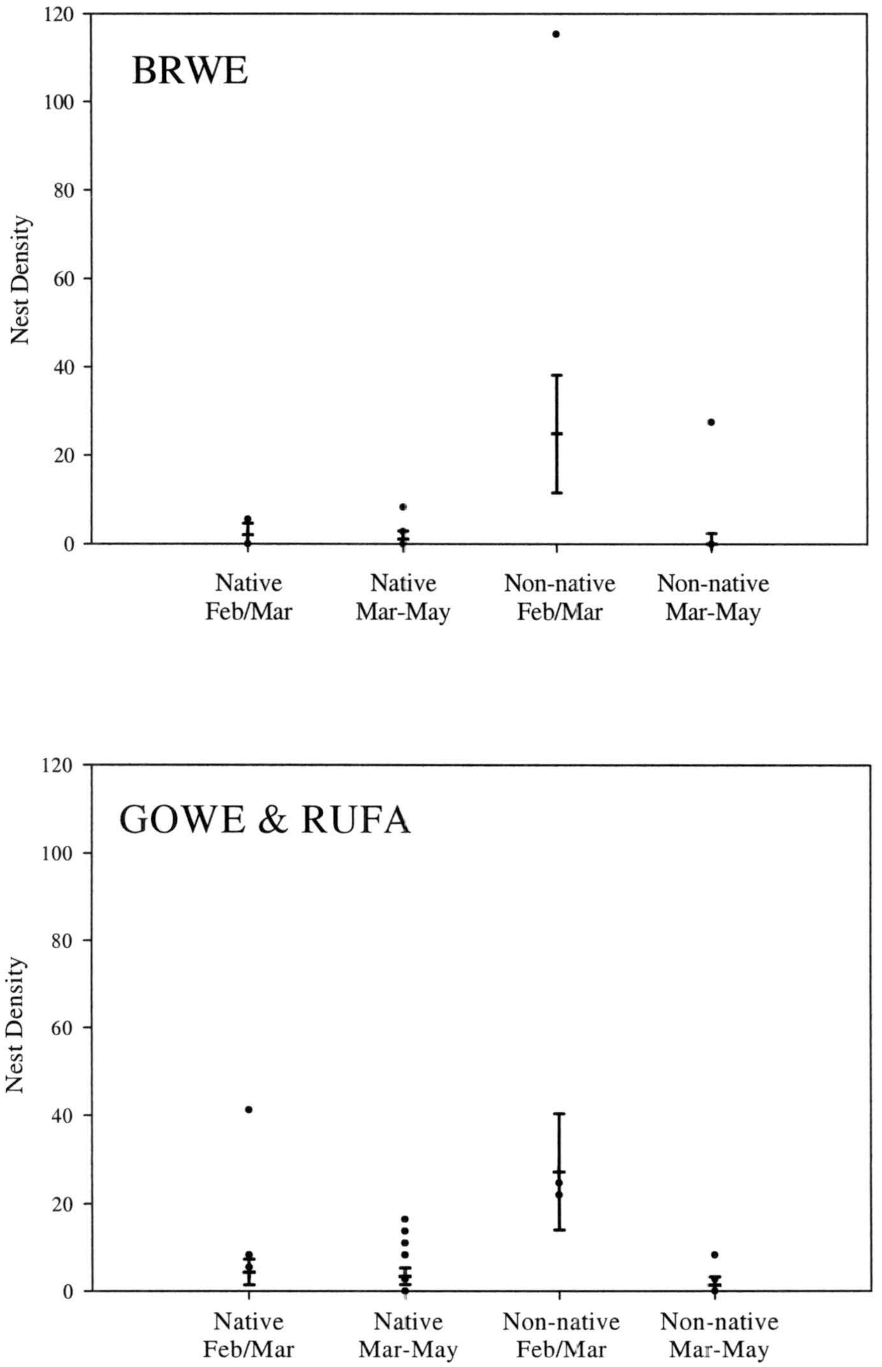

Figure 1.4. Bars are nest densities with 95% CI predicted by the best generalized linear regression model for 2004 [$\hat{D} = \beta_0 + \beta_1$(BRWE vs. others) + β_2(Native/mixed vs. non-native) + β_3(Feb/Mar vs. others) + β_4(Native/mixed*Feb/Mar)]. Dots are density estimates used to develop the model.

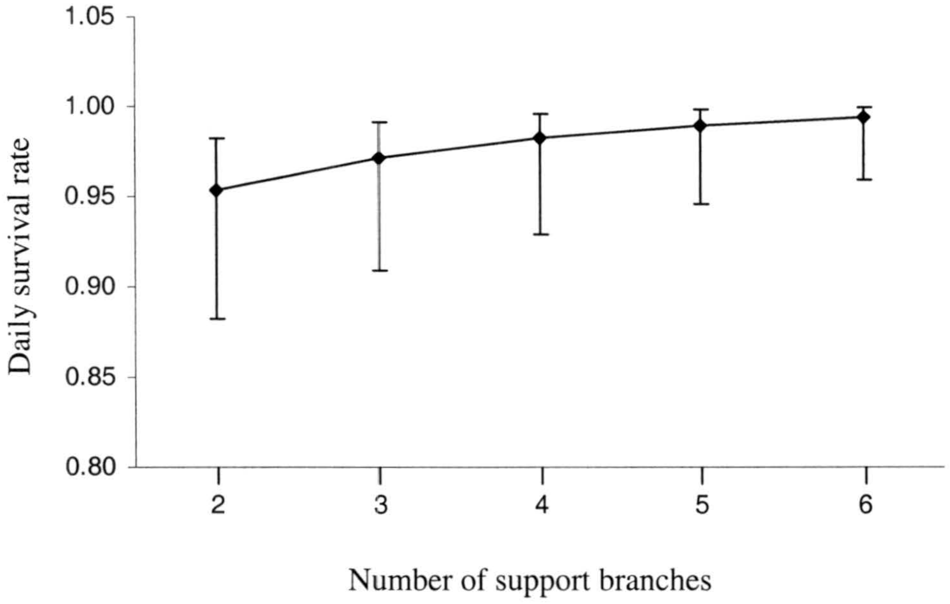

Figure 1.5. Golden White-eye daily nest survival rate in relation to number of nest support branches. Error bars give 95% confidence intervals around the means (♦).

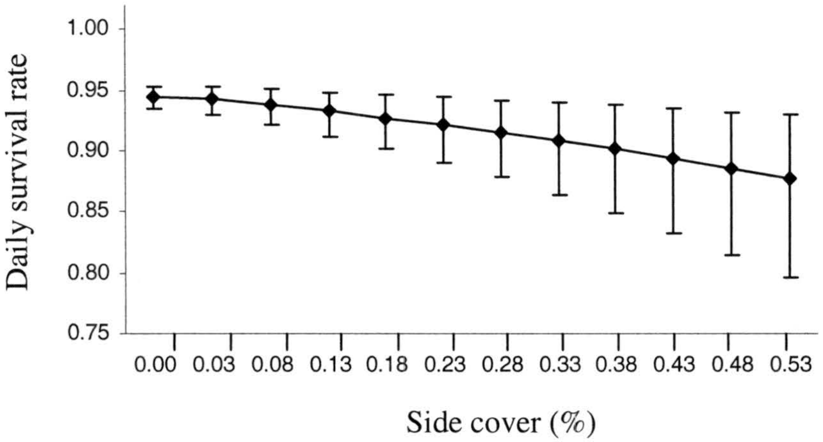

Figure 1.6. Bridled White-eye daily nest survival rate in relation to nest concealment (% side cover). Error bars give 95% confidence intervals around the means (♦).

Table 1.1. Percent native and non-native trees by study site on Saipan, 2003 – 2004.

Site	Forest Type	Tree density per $100m^2$	% Native	% Non-native
As Teo	Native	41	94	6
Kagman	Native	26	88	12
Marpi	Native	42	98	2
Laolao Bay	Mixed	34	65	35
Bird Island	Non-native	79	1	99
Cow Town	Non-native	65	0	100
Naftan	Non-native	84	2	98
Obyan	Non-native	39	0	100

Table 1.2. Density estimates (nests/km^2; ± SE) for Bridled White-eyes (BRWE), Golden White-eyes (GOWE), and Rufous Fantails (RUFA) in native, mixed, and non-native forest in 2003 and 2004 on Saipan. Full results with coefficients of variation and 95% confidence intervals are in Appendix 1.6.

			2004		
Species	**Forest type**	**2003**	**Feb/Mar**	**Mar/Apr**	**Apr/May**
BRWE	Native	0.0 (± 0.0)	5.4 (± 3.4)	0.0 (± 0.0)	0.0 (± 0.0)
	Mixed	10.9 (± 5.6)	0.0 (± 0.0)	8.2 (± 4.4)	2.7 (± 2.6)
	Non-native	90.5 (± 24.1)	115.2 (± 27.7)	27.4 (± 13.3)	0.0 (± 0.0)
GOWE	Native	21.9 (± 12.0)	41.1 (± 14.8)	10.9 (± 5.2)	10.9 (± 5.6)
	Mixed	8.2 (± 4.7)	5.4 (± 3.5)	2.7 (± 2.1)	10.9 (± 5.0)
	Non-native	10.9 (± 5.7)	24.7 (± 1.4)	2.7 (± 2.4)	0.0 (± 0.0)
RUFA	Native	0.0 (± 0.0)	8.2 (± 5.3)	0.0 (± 0.0)	16.4 (± 7.1)
	Mixed	8.2 (± 3.6)	5.4 (± 3.9)	8.2 (± 4.6)	13.7 (± 5.9)
	Non-native	24.7 (± 8.0)	21.9 (± 8.7)	8.2 (± 5.1)	0.0 (± 0.0)

Table 1.3. Model selection results from maximum-likelihood linear regression on nest densities of Golden White-eyes, Bridled White-eyes, and Rufous Fantails on Saipan, 2003 and 2004. Models with ΔAICc ≤ 8 are shown. Full results are in Appendix 1.7.

Model	**log-likelihood**	n[1]	K[2]	**AICc**[3]	**ΔAICc**[4]	w_i[5]
Weighted regression results 2003						
% native	-22.97	9	3	56.75	0.00	0.718
Native vs non-native/mixed	-24.09	9	3	58.99	2.23	0.235
Native vs non-native vs mixed	-22.47	9	4	62.95	6.20	0.032
Non-native vs native/mixed	-26.94	9	3	64.68	7.92	0.014
Weighted regression results 2004						
BRWE vs. others + non-native vs native/mixed * Feb/Mar vs. others	-62.51	27	6	141.22	0.00	0.344
Non-native vs native/mixed * Feb/Mar vs. others	-64.62	27	5	142.10	0.88	0.221
BRWE vs. others * non-native vs native/mixed, non-native vs native/mixed * Feb/Mar vs. others	-61.51	27	7	142.91	1.69	0.147
Species + non-native vs native/mixed * Feb/Mar vs. others	-62.19	27	7	144.28	3.06	0.074
Non-native vs native/mixed * month as continuous variable	-65.86	27	5	144.59	3.37	0.064
Non-native vs native/mixed * month	-62.73	27	7	145.37	4.15	0.043
Native vs non-native vs mixed * month as continuous variable	-63.30	27	7	146.50	5.28	0.024
BRWE vs. others * non-native vs native/mixed * Feb/Mar vs. others	-61.37	27	8	146.74	5.51	0.021
% Native * month as continuous variable	-67.91	27	5	148.69	7.46	0.008
Species + non-native vs native/mixed * month as continuous variable	-64.49	27	7	148.88	7.65	0.007
Species + non-native vs native/mixed * month	-60.28	27	9	149.15	7.93	0.006

[1] Sample size.
[2] Number of parameters in model.
[3] AIC corrected for small sample size (Burnham and Anderson 2002).
[4] Difference in AIC$_c$ value between best model and competing candidate model.
[5] Akaike weights, indicating the relative support for each of the candidate models.

Table 1.4. Estimated relative importance values (summed AIC$_c$ weights, $\sum w_i$), parameter estimates ($\hat{\beta}$), and 95% confidence intervals of variables and interactions in top regression model of nest densities in 2004.

Variable	Σw_i	$\hat{\beta} \pm SE$	LCL	UCL
Intercept	1.000	1.38 ± 0.95	-0.48	3.23
Feb/Mar	0.977	25.81 ± 6.78	12.53	39.10
Native/Mixed	0.965	2.02 ± 1.22	-0.37	4.41
BRWE	0.731	-2.31 ± 1.12	-4.51	-0.11
Native/Mixed * Feb/Mar	0.945	-24.87 ± 6.95	-38.48	-11.25
BRWE * Native/Mixed	0.203	-	-	-
BRWE * Feb/Mar	0.136	-	-	-

Table 1.5. Total numbers of nests of Bridled White-eyes (BRWE), Golden White-eyes (GOWE), and Rufous Fantails (RUFA) monitored on Saipan in 2003 and 2004. Includes all nests found, both incidentally and using line transect sampling.

Species	**2003**	**2004**	**Total**
BRWE	78	120	198
GOWE	66	73	139
RUFA	34	66	100
Total	178	259	**437**

Table 1.6. Results of preliminary model comparisons to determine the best approximating model with which to estimate nest survival for Golden White-eyes (GOWE), Rufous Fantails (RUFA) and Bridled White-eyes (BRWE) on Saipan, 2003 and 2004.

Model	**log-likelihood**	n[1]	K[2]	AIC_c[3]	ΔAIC_c[4]	w_i[5]
Species	1034.94	437	21	2114.11	0.00	0.995
Pooled	1055.20	437	7	2124.67	10.56	0.005
BRWE Pooled	454.47	198	4	917.16	0.00	0.715
BRWE Forest type	451.11	198	8	919.00	1.84	0.285
BRWE Year	448.24	198	8	913.25	0.00	0.876
BRWE Pooled	454.47	198	4	917.16	3.91	0.124
BRWE Pooled	454.02	198	4	916.25	0.00	1.000
BRWE Forest type&Year	439.59	198	24	934.12	17.87	0.000
GOWE Pooled	350.77	139	5	711.99	0.00	0.980
GOWE Forest type	349.02	139	10	719.76	7.77	0.020
GOWE Pooled	350.77	139	5	711.99	0.00	0.950
GOWE Year	348.08	139	10	717.88	5.89	0.050
GOWE Pooled	350.00	139	5	710.46	0.00	1.000
GOWE Forest type&Year	339.61	139	30	756.45	45.99	0.000
RUFA Pooled	292.61	100	4	593.64	0.00	0.955
RUFA Forest type	291.09	100	8	599.76	6.12	0.045
RUFA Pooled	292.61	100	4	593.64	0.00	0.953
RUFA Year	291.04	100	8	599.67	6.03	0.047
RUFA Pooled	294.30	100	4	597.02	0.00	1.000
RUFA Forest type&Year	287.60	100	24	639.20	42.18	0.000

[1] Sample size.

[2] Number of parameters in model.

[3] AIC corrected for small sample size (Burnham and Anderson 2002).

[4] Difference in AIC_c value between best model and competing candidate model.

[5] Akaike weights, indicating the relative support for each of the candidate models.

Table 1.7. Daily and overall nest survival of Bridled White-eyes in native/mixed and non-native forest on Saipan in 2003 and 2004.

Model	Daily Survival Rate Egg-laying and Incubation (95% CI)	Daily Survival Rate Nestling (95% CI)	Overall Survival Shortest (95% CI)[1]	Overall Survival Longest (95% CI)[2]
Native/Mixed forest	0.875 (0.793 - 0.928)	0.980 (0.947 - 0.993)	19% (10 – 31%)	10% (4 – 23%)
Non-native forest	0.872 (0.842 - 0.896)	0.941 (0.918 - 0.957)	11% (8 – 15%)	5% (3 – 9%)

[1] Shortest possible nesting cycle (22 days) of Bridled White-eyes used to calculate overall nest survival.
[2] Longest possible nesting cycle (29 days) of Bridled White-eyes used to calculate overall nest survival.

Table 1.8. Daily nest survival probabilities of Golden White-eyes (GOWE), Rufous Fantails (RUFA) and Bridled White-eyes (BRWE) on Saipan, 2003 and 2004. Golden White-eye and Rufous Fantail daily survival probabilities were constant across years. Daily survival rates differed between all nest stages for Golden White-eyes and Bridled White-eyes. Rufous Fantail daily survival rates were constant across the incubation and nestling stages.

Model	Egg-laying (95% CI)	Incubation (95% CI)	Nestling (95% CI)
GOWE	0.844 (0.591 - 0.953)	0.958 (0.938 - 0.972)	0.969 (0.955 - 0.979)
RUFA	0.835 (0.594 - 0.946)	0.967 (0.956 - 0.975)	0.967 (0.956 - 0.975)
BRWE 2003	0.790 (0.150 - 0.988)	0.920 (0.870 - 0.952)	0.970 (0.947 - 0.983)
BRWE 2004	0.985 (0.086 - 1.000)	0.904 (0.873 - 0.928)	0.928 (0.897 - 0.950)

Table 1.9. Overall nest success of Golden White-eyes (GOWE), Rufous Fantails (RUFA) and Bridled White-eyes (BRWE) on Saipan, 2003 and 2004. Nest success was constant across years for Golden White-eyes and Rufous Fantails.

Model	**Shortest stage lengths**[1]	**% Success (95% CI)**	**Intermed. stage lengths**[1]	**% Success (95% CI)**	**Longest stage lengths**[1]	**% Success (95% CI)**
GOWE	1,10,10	40.0 (29.6 - 51.5)	2,12,13	28.2 (17.0 - 42.9)	2,15,16	22.6 (13.3 - 35.6)
RUFA	2,27[2]	28.0 (16.6 - 43.1)	2,31[2]	24.4 (14.3 - 38.6)	2,34[2]	22.1 (12.7 - 35.6)
BRWE 2003	2,9,11	20.9 (4.7 - 58.8)	2,11,12	17.2 (3.9 - 51.5)	3,12,14	11.7 (1.4 - 55.5)
BRWE 2004	2,9,11	17.1 (11.1 - 25.3)	2,11,12	12.9 (8.0 - 20.1)	3,12,14	9.9 (5.6 - 16.9)

[1] Egg-laying, incubation, nestling.

[2] Egg-laying, incubation and nestling combined.

Table 1.10. Results of model comparisons examining whether nest survival for Golden White-eyes (GOWE), Rufous Fantails (RUFA) and Bridled White-eyes (BRWE) differed seasonally (early [February and March] vs. late [April and May]) on Saipan in 2004.

Model	**log-likelihood**	n[1]	K[2]	**AIC$_c$**[3]	**ΔAIC$_c$**[4]	w_i[5]
GOWE Pooled	178.45	102	7	372.09	0.00	0.999
GOWE by season	176.21	102	14	385.25	13.16	0.001
RUFA Pooled	186.86	92	7	389.06	0.00	0.999
RUFA by season	185.00	92	14	403.45	14.39	0.001
BRWE Pooled	293.04	158	5	596.48	0.00	0.596
BRWE by season	287.88	158	10	597.25	0.77	0.404

[1] Sample size.

[2] Number of parameters in model.

[3] AIC corrected for small sample size (Burnham and Anderson 2002).

[4] Difference in AIC$_c$ value between best model and competing candidate model.

[5] Akaike weights, indicating the relative support for each of the candidate models.

Table 1.11. Daily and overall nest survival of Bridled White-eyes on Saipan by season in 2004. February and March were considered early season, and April and May late season; these two periods were distinguished by desiccation of the tangantangan forest.

Model	Daily Survival Rate Egg-laying and Incubation (95% CI)	Daily Survival Rate Nestling (95% CI)	Overall Survival Shortest (95% CI)[1]	Overall Survival Longest (95% CI)[2]
February & March	0.865 (0.825 - 0.897)	0.944 (0.890 - 0.972)	11% (6 – 18%)	5% (2 – 12%)
April & May	0.895 (0.830 - 0.936)	0.924 (0.882 - 0.951)	12% (7 – 21%)	6% (2 – 15%)

[1] Shortest possible nesting cycle (22 days) of Bridled White-eyes used to calculate overall nest survival.

[2] Longest possible nesting cycle (29 days) of Bridled White-eyes used to calculate overall nest survival.

Table 1.12. Model comparisons of the influence of nest site characteristics on nest survival for Rufous Fantails on Saipan, 2003 and 2004.

Model	***K***[1]	**n-eff**[2]	**AICc**[3]	**ΔAICc**[4]	w_i[5]
Intercept (β_0)	1	1091	265.48	0.00	.251
Side cover	2	1091	266.56	1.08	.145
Nest height	2	1091	266.95	1.47	.120
% native trees	2	1091	267.27	1.79	.102
Tree (native vs. non-native)	2	1091	267.43	1.95	.094
Number of support branches + avg. diameter of support branches	3	1091	267.93	2.45	.073
Nest visit + minutes spent at nest	3	1091	268.41	2.93	.057
Side cover + canopy cover	3	1091	268.47	2.99	.056
Tree (native vs. non-native) + side cover	3	1091	268.58	3.10	.053
Nest height + side cover + tree (native vs. non-native)	4	1091	270.39	4.91	.021
Tree (native vs. non-native) + canopy cover + side cover	4	1091	270.48	5.00	.020
global	7	1091	274.41	8.93	.002

[1] Number of parameters in model.
[2] Effective sample size. Differs from sample size because of exposure intervals of nests.
[3] AIC corrected for small sample size (Burnham and Anderson 2002).
[4] Difference in AIC_c value between best model and competing candidate model.
[5] Akaike weights, indicating the relative support for each of the candidate models.

Table 1.13. Parameter estimates ($\hat{\beta}$) and their 95% confidence intervals from the non-linear mixed models examining the influence of nest site characteristics on nest survival for Rufous Fantails on Saipan, 2003 and 2004.

Model	$\hat{\beta}$	**LCL**	**UCL**
Intercept (β_0)	3.28	2.97	3.59
Side cover	-1.16	-3.41	1.10
Nest height	0.11	-0.20	0.42
% native trees	0.18	-0.60	0.96
Tree (native vs. non-native)	-0.07	-0.70	0.56
Number of support branches	0.19	-0.36	0.74
Avg. diameter of support branches	-0.09	-0.31	0.13
Nest visit	-1.64	-6.05	2.76
Minutes spent at nest	0.02	-0.12	0.17
Canopy cover	-1.12	-8.16	5.92

Table 1.14. Model comparisons of the influence of nest site characteristics on nest survival for Golden White-eyes on Saipan, 2003 and 2004.

Model	***K***[1]	**n-eff**[2]	**AICc**[3]	**ΔAICc**[4]	w_i[5]
Number of support branches	2	1338	298.60	0.00	.522
Number of support branches + avg. diameter of support branches	3	1338	300.48	1.87	.204
Intercept (β_0)	1	1338	302.86	4.25	.062
Tree (native vs. non-native)	2	1338	304.39	5.78	.028
Random location	2	1338	304.43	5.83	.028
Side cover	2	1338	304.47	5.87	.027
% native trees	2	1338	304.76	6.16	.024
Nest height	2	1338	304.80	6.19	.023
Tree (native vs. non-native) + side cover	3	1338	305.98	7.37	.013
Random location + tree (native vs. non-native)	3	1338	306.14	7.54	.012
Side cover + canopy cover	3	1338	306.18	7.57	.011
Nest visit + minutes spent at nest	3	1338	306.31	7.70	.011
Random location + % native trees	3	1338	306.44	7.83	.010
Random location + tree (native vs. non-native) + % native trees	4	1338	307.50	8.90	.006
Nest height + side cover + tree (native vs. non-native)	4	1338	307.79	9.18	.005
Tree (native vs. non-native) + canopy cover + side cover	4	1338	307.96	9.36	.004
global	8	1338	308.12	9.52	.004

[1] Number of parameters in model.
[2] Effective sample size. Differs from sample size because of exposure intervals of nests.
[3] AIC corrected for small sample size (Burnham and Anderson 2002).
[4] Difference in AIC_c value between best model and competing candidate model.
[5] Akaike weights, indicating the relative support for each of the candidate models.

Table 1.15. Parameter estimates ($\hat{\beta}$) and their 95% confidence intervals from the non-linear mixed models examining the influence of nest site characteristics on nest survival for Golden White-eyes on Saipan, 2003 and 2004.

Model	$\hat{\beta}$	**LCL**	**UCL**
Number of support branches	0.50	0.08	0.92
Avg. diameter of support branches	-0.05	-0.29	0.20
Intercept (β_0)	3.37	3.08	3.67
Tree (native vs. non-native)	-0.22	-0.85	0.41
Random location	0.06	-0.22	0.34
Side cover	-0.69	-2.84	1.46
% native trees	-0.11	-0.81	0.59
Nest height	-0.04	-0.33	0.25
Canopy cover	-1.54	-7.18	4.09
Nest visit	-0.60	-4.06	2.86
Minutes spent at nest	0.05	-0.10	0.21

Table 1.16. Model comparisons of the influence of nest site characteristics on nest survival for Bridled White-eyes on Saipan, 2003 and 2004.

Model	***K***[1]	**n-eff**[2]	**AICc**[3]	**ΔAICc**[4]	w_i[5]
Side cover + canopy cover	3	1256	378.27	0.00	.244
Side cover	2	1256	378.44	0.16	.225
Tree (native vs. non-native) + canopy cover + side cover	4	1256	380.22	1.94	.092
Intercept (β_0)	1	1256	380.34	2.06	.087
Tree (native vs. non-native) + side cover	3	1256	380.43	2.15	.083
Tree (native vs. non-native)	2	1256	381.89	3.61	.040
Random location	2	1256	382.03	3.76	.037
Nest visit + minutes spent at nest	3	1256	382.18	3.90	.035
% native trees	2	1256	382.19	3.91	.034
Nest height	2	1256	382.20	3.92	.034
Nest height + side cover + tree (native vs. non-native)	4	1256	382.40	4.12	.031
Random location + tree (native vs. non-native)	3	1256	383.62	5.34	.017
Random location + % native trees	3	1256	383.91	5.64	.014
Number of support branches + avg. diameter of support branches	3	1256	383.95	5.67	.014
Random location + tree (native vs. non-native) + % native trees	4	1256	385.50	7.23	.007
global	8	1256	387.29	9.02	.003

[1] Number of parameters in model.
[2] Effective sample size. Differs from sample size because of exposure intervals of nests.
[3] AIC corrected for small sample size (Burnham and Anderson 2002).
[4] Difference in AIC$_c$ value between best model and competing candidate model.
[5] Akaike weights, indicating the relative support for each of the candidate models.

Table 1.17. Parameter estimates ($\hat{\beta}$) and their 95% confidence intervals from the non-linear mixed models examining the influence of nest site characteristics on nest survival for Bridled White-eyes on Saipan, 2003 and 2004.

Model	$\hat{\beta}$	**LCL**	**UCL**
Side cover	-1.64	-3.23	-0.05
Canopy cover	-3.93	-9.42	1.57
Tree (native vs. non-native)	-0.19	-0.72	0.35
Intercept (β_0)	2.57	2.34	2.80
Random location	0.02	-0.14	0.19
Nest visit	14.09	-1956.56	1984.73
Minutes spent at nest	-0.02	-0.14	0.09
% native trees	-5.77	-34.44	22.91
Nest height	0.05	-0.20	0.29
Number of support branches	0.03	-0.17	0.23
Avg. diameter of support branches	0.06	-0.14	0.26

Appendix 1.1. Results of preliminary distance analyses on which I based use of a pooled detection function.

Data set	**Model**	***n***	**P_a**[1]	**P_a CV**[2]	**ΔAIC$_c$**[3]
Pooled	Pooled	192	0.54	0.06	0.00
	Forest type – native	73	0.50	0.08	
	Forest type – nonnative	119	0.56	0.08	1.16
	Year – 2003	64	0.51	0.10	
	Year – 2004	128	0.46	0.12	1.77
	Species – BRWE	95	0.56	0.09	
	Species – GOWE	55	0.52	0.10	
	Species – RUFA	42	0.54	0.07	3.64
BRWE	Pooled	95	0.56	0.09	0.00
	Forest type – native	10	0.50	0.29	
	Forest type – nonnative	85	0.56	0.09	2.02
	Year – 2003	37	0.56	0.13	
	Year – 2004	58	0.55	0.07	1.84
GOWE	Pooled	55	0.52	0.10	0.26
	Forest type – native	41	0.46	0.11	
	Forest type – nonnative	14	0.69	0.23	0.00
	Year – 2003	15	0.49	0.21	
	Year – 2004	40	0.33	0.38	1.49
RUFA	Pooled	42	0.52	0.12	0.00
	Forest type – native	22	0.57	0.12	
	Forest type – nonnative	20	0.50	0.21	0.90
	Year – 2003	12	0.50	0.33	
	Year – 2004	30	0.56	0.10	1.20

[1] Probability of detection.

[2] Coefficient of Variation of the probability of detection.

[3] Difference in AIC$_c$ (AIC corrected for small sample size) value between best model and competing candidate model (Burnham and Anderson 2002).

Appendix 1.2. *A priori* models proposed for weighted regression examining potential differences in nest densities between species, forest types and months. Basic *a priori* models were all based on forest type, avian species, and year effects, because I was interested in potential differences in nest densities across these factors. I use '+' to denote an additive effect, and '*' to denote an interactive effect in the models presented. Percent native, based on the point-centered quarter results, and percent available nest tree, based on nest trees used and nest trees available as determined by the point-centered quarter method, were proposed after forest type proved to be a poor predictor of differences in density in 2003.

2003

Intercept (β_0)
Species
Native vs. non-native vs. mixed
Native vs. non-native/mixed
Non-native vs. native/mixed
Species + Native vs. non-native/mixed
Species + Non-native vs. native/mixed

2004

Intercept (β_0)
Species
Month
Native vs. non-native vs. mixed
% native
Non-native vs native/mixed
% nest tree
Species + Month
Species * Month
Species + % native
Species + Non-native vs native/mixed
Species + % nest tree
Species * % native
Species * Non-native vs native/mixed
Species * % nest tree
Native vs. non-native vs. mixed + Month
Native vs. non-native vs. mixed * Month
Non-native vs native/mixed + Month
Non-native vs native/mixed * Month
Species + Non-native vs native/mixed + Month
Species + Non-native vs native/mixed * Month

Appendix 1.3. *A priori* models proposed for analyzing the influence of nest site placement on nest survival.

Model	**Explanation**
Random location	Study sites have intrinsic differences that could affect nest survival.
% native trees (in study site)	Golden White-eye nests seemed to be present in higher numbers in native forest, whereas Bridled White-eye nests seemed to predominate in non-native forest. Thus, it seemed possible that the % native trees present in a study site could influence nest survival of the avian species.
Random location + % native trees	These two variables together could explain more variation in nest survival than either factor alone.
Tree (native or non-native)	Whether a nest was located in a native or a non-native tree could influence nest survival due to structural differences between the types of tree. Native trees appeared more robust and to provide more nest cover than non-native trees.
Random location + tree (native or non-native)	A nest placed in a native tree in a non-native study site (or vice versa) could be more or less vulnerable to predation based on predator search strategies.
Random location + tree (native or non-native) + % native trees	Including % native trees with random location could explain more variation than random location alone, thus I included this model in addition to the one above.
Side cover (nest concealment)	Since all documented predation was avian, visual detectability of a nest could influence nest survival. This variable reflected cover around the nest.
Side cover + canopy cover	In addition to side cover around the nest, canopy cover over the nest could influence detection of the nest by avian predators.

Continued

Appendix 1.3 continued

Model	**Explanation**
Tree (native or non-native) + side cover	Since type of tree appeared to influence nest concealment, both variables together could explain more variation in nest survival.
Tree (native or non-native) + canopy cover + side cover	All of these variables in conjunction could influence detection of the nest by a predator more than one or two variables combined.
Nest height	Nests placed higher in trees could be more easily detected by avian predators flying over, thus height of the nest could impact survival.
Nest height + side cover + tree (native or non-native)	All of these variables in conjunction could influence detection of the nest by a predator more than one or two variables combined.
Number of (nest) support branches	More support branches could provide a more stable perch and thus easier access to the nest for a predator.
Number of support branches + avg. diameter of support branches	If nest support branches are thicker in diameter, this could provide additional stability for a predator trying to access the nest.
Nest visit (denotes the day a nest was visited) + Minutes at nest (number of minutes spent at the nest during a nest visit)	There have been conflicting findings between studies as to whether observers influence the probability of a nest getting depredated, thus I wanted to see if my visits affected nest survival. I hypothesized that I was more likely to cause a nest to fail by attracting a predator to the nest if I spent more time at the nest.
Intercept (β_0)	
Global (all above parameters)	

Appendix 1.4. Types and numbers of artificial nests of Golden White-eyes (GOWE), Bridled White-eyes (BRWE) and Rufous Fantails (RUFA) monitored with still cameras on Saipan in 2003. Nests of the target species were used and are referred to as "placed" if the nest was subjectively placed, or "actual" if in the original position of a previously active nest.

	GOWE		BRWE		RUFA		
Study Site	**Placed**	**Actual**	**Placed**	**Actual**	**Placed**	**Actual**	**Total**
Marpi	4	0	1	0	0	0	**5**
As Teo	0	0	1	0	0	1	**2**
Kagman	1	0	0	0	0	0	**1**
Laolao Bay	2	1	0	2	0	3	**8**
Bird Island	1	1	1	1	1	0	**5**
Cow Town	2	0	2	0	0	0	**4**
Obyan	1	0	1	2	0	2	**6**
Naftan	0	0	1	1	0	0	**2**
Total	**11**	**2**	**7**	**6**	**1**	**6**	**33**

Appendix 1.5. Tree species composition of study sites on Saipan in 2003 and 2004 as determined by the point-centered quarter method.

Mean distance of trees from point	Absolute density of trees per 100m^2	Tree Species	No. in quarters	No. in 100m^2	Dominance	Dominance Rank	Abs. Freq.
Native sites							
As Teo							
1.56	40.99	*Cynometra*	0.51	21.00	6505.78	1	0.80
		Pithecellobium	0.03	1.02	995.05	2	0.10
		Ficus	0.04	1.53	813.43	3	0.15
		Pisonia	0.01	0.51	798.16	4	0.05
		Guamia	0.24	9.73	542.71	5	0.55
		Ochrosia	0.04	1.53	295.57	6	0.15
		Unident.	0.04	1.53	290.12	7	0.15
		Pandanus	0.01	0.51	60.00	8	0.05
		Psychotria	0.04	1.53	23.38	9	0.15
		Eugenia	0.04	1.53	18.29	10	0.10
		Pipturus	0.01	0.51	0.46	11	0.05
Marpi							
1.54	42.15	*Cynometra*	0.28	11.59	2863.18	1	0.60
		Pisonia	0.08	3.16	2341.49	2	0.20
		Guamia	0.30	12.64	1561.82	3	0.75
		Unident.	0.03	1.05	514.92	4	0.10
		Psychotria	0.14	5.79	436.12	5	0.40
		Aidia	0.08	3.68	194.28	6	0.35
		Drypetes	0.03	1.05	103.03	7	0.50
		Neisosperma	0.03	1.58	19.28	8	0.10
		Eugenia	0.03	1.58	9.19	9	0.10

Continued

Appendix 1.5 continued

Mean distance of trees from point	Absolute density of trees per 100m^2	Tree Species	No. in quarters	No. in 100m^2	Dominance	Dominance Rank	Abs. Freq.
Native sites (cont.)							
Kagman							
1.97	25.51	*Pisonia*	0.05	1.27	8394.13	1	0.20
		Ficus	0.08	2.12	1457.33	2	0.20
		Psychotria	0.08	1.91	858.37	3	0.26
		Ochrosia	0.09	2.33	826.93	4	0.30
		Guamia	0.33	8.50	791.57	5	0.70
		Barringtonia	0.06	1.48	665.10	6	0.13
		Maytenus	0.06	1.70	392.78	7	0.23
		Unident.	0.06	1.70	271.40	8	0.23
		Aidia	0.03	0.85	235.67	9	0.10
		Cynometra	0.06	1.48	214.18	10	0.13
		Leucaena	0.03	0.63	175.54	11	0.10
		Albizia	0.03	0.63	66.44	12	0.03
		Drypetes	0.01	0.21	45.58	13	0.03
		Pandanus	0.01	0.21	43.62	14	0.03
		Eugenia	0.02	0.42	2.93	15	0.06
Mixed Site							
Laolao Bay							
1.71	34.09	*Pisonia*	0.03	0.85	2373.75	1	0.10
		Cynometra	0.10	3.64	1775.62	2	0.17
		Leucaena	0.26	8.79	1395.31	3	0.52
		Hibiscus	0.08	3.00	988.97	4	0.27
		Artocarpus	0.01	0.21	869.02	5	0.02

Continued

Appendix 1.5 continued

Mean distance of trees from point	Absolute density of trees per 100m^2	Tree Species	No. in quarters	No. in 100m^2	Dominance	Dominance Rank	Abs. Freq.
Laolao Bay (continued)							
		Guamia	0.16	5.36	537.78	6	0.37
		Cocos	0.02	0.64	457.38	7	0.07
		Unident.	0.01	0.21	456.77	8	0.02
		Morinda	0.05	1.71	208.53	9	0.20
		Albizia	0.05	1.92	174.23	10	0.12
		Pandanus	0.01	0.21	73.08	11	0.02
		Aidia	0.13	4.50	39.27	12	0.22
		Ficus	0.01	0.42	27.57	13	0.05
		Melanolepis	0.01	0.42	21.22	14	0.05
		Psychotria	0.03	1.07	20.44	15	0.12
		Bambusa	0.01	0.21	17.03	16	0.02
		Ochrosia	0.02	0.64	2.89	17	0.05
		Eugenia	0.01	0.21	2.17	18	0.02
Non-native sites							
Bird Island							
1.12	78.56	*Leucaena*	0.98	76.60	4444.51	1	1.00
		Morinda	0.01	0.65	8.02	2	0.03
		Carica	0.01	1.30	2.68	3	0.06
Cow Town							
1.24	64.98	*Leucaena*	0.96	62.27	3566.03	1	1.00
		Albizia	0.03	1.62	38.95	2	0.10
		Carica	0.01	1.08	21.74	3	0.06

Continued

Appendix 1.5 continued

Mean distance of trees from point	Absolute density of trees per 100m^2	Tree Species	No. in quarters	No. in 100m^2	Dominance	Dominance Rank	Abs. Freq.
Non-native sites (continued)							
Obyan							
1.59	39.49	*Leucaena*	0.89	35.05	1917.52	1	0.97
		Albizia	0.08	3.45	252.19	2	0.20
		Carica	0.02	0.74	31.24	3	0.02
		Pithecellobium	0.01	0.24	24.03	4	0.02
Naftan							
1.09	84.03	*Leucaena*	0.93	77.73	5226.84	1	1.00
		Bauhinia	0.02	1.40	34.78	2	0.06
		Albizia	0.02	2.80	13.03	3	0.13
		Carica	0.01	0.70	6.40	4	0.03
		Aidia	0.01	0.70	2.99	5	0.03
		Psychotria	0.01	0.70	1.43	6	0.03

Appendix 1.6. Nest density estimates/km^2 by species, habitat and month on Saipan, computed using program DISTANCE version 4.1 release 2 (Thomas et al. 2004). The best approximating model was the half-normal key function with a cosine series expansion.

Stratum codes:
B = Bridled White-eyes
G = Golden White-eyes
R = Rufous Fantails
3 = 2003
X = February/March, 2004
Y = March/April, 2004
Z = April/May, 2004
D = Mixed forest type
E = Native forest type
T = Non-native forest type
e.g., B3D = Bridled White-eyes in 2003 in mixed forest
p = Global detection function

```
                           Estimate/km²    %CV      df      95% Confidence Interval
                           ------------------------------------------------------
 Stratum: B3D
                  D         10.980        50.74   103.71    4.2498          28.369
 Stratum: B3E
                  D            0.00000
 Stratum: B3T
                  D         90.585        26.56   111.32    53.995          151.97
 Stratum: BXD
                  D            0.00000
 Stratum: BXE
                  D         5.4900        61.94   102.81    1.7731          16.998
 Stratum: BXT
                  D         115.29        24.02   113.80    72.118          184.31
 Stratum: BYD
                  D         8.2350        53.73   103.41    3.0334          22.356
 Stratum: BYE
                  D            0.00000
 Stratum: BYT
                  D         27.450        48.47   103.97    11.039          68.256
 Stratum: BZD
                  D         2.7450        94.21   101.78    0.56483          13.341
 Stratum: BZE
                  D            0.00000
 Stratum: BZT
                  D            0.00000
 Stratum: G3D
                  D         8.2350        57.19   103.13    2.8676          23.649
 Stratum: G3E
                  D         21.960        54.87   103.31    7.9395          60.740
 Stratum: G3T
                  D         10.980        52.22   103.56    4.1466          29.075
 Stratum: GXD
                  D         5.4900        64.09   102.69    1.7151          17.574
 Stratum: GXE
                  D         41.175        35.92   106.49    20.640          82.141

 Stratum: GXT
                  D         24.705        46.12   104.29    10.342          59.016
 Stratum: GYD
                  D         2.7450        76.78   102.17    0.71140          10.592
 Stratum: GYE
                  D         10.980        47.60   104.09    4.4815          26.902
 Stratum: GYT
                  D         2.7450        87.17   101.91    0.61786          12.195
```

```
                         Estimate/km2   %CV      df      95% Confidence Interval
                        ------------------------------------------------------

Stratum: GZD
                  D      10.980         45.78   104.34   4.6233         26.077
Stratum: GZE
                  D      10.980         50.86   103.70   4.2413         28.425
Stratum: GZT
                  D         0.00000
Stratum: R3D
                  D      8.2350         44.19   104.59   3.5642         19.027
Stratum: R3E
                  D         0.00000
Stratum: R3T
                  D      24.705         32.48   107.77   13.188         46.282
Stratum: RXD
                  D      5.4900         70.46   102.39   1.5584         19.340
Stratum: RXE
                  D      8.2350         64.09   102.69   2.5726         26.360
Stratum: RXT
                  D      21.960         39.39   105.54   10.342         46.629
Stratum: RYD
                  D      8.2350         55.36   103.27   2.9537         22.960
Stratum: RYE
                  D         0.00000
Stratum: RYT
                  D      8.2350         61.38   102.84   2.6832         25.274
Stratum: RZD
                  D      13.725         42.67   104.86   6.0993         30.885
Stratum: RZE
                  D      16.470         43.38   104.73   7.2301         37.519
Stratum: RZT
                  D         0.00000

              p          0.53787        5.83    191.00   0.47950      0.60333
```

Appendix 1.7. Full model selection results from linear weighted regression on nest densities of Golden White-eyes, Bridled White-eyes, and Rufous Fantails on Saipan, 2003 and 2004.

Model	log-likelihood	K^1	n^2	AICc^3	ΔAICc^4	w_i^5
Weighted regression results 2003						
% native	-22.97	3	9	56.75	0.00	0.718
Native vs. non-native/mixed	-24.09	3	9	58.99	2.23	0.235
Native vs. non-native vs. mixed	-22.47	4	9	62.95	6.20	0.032
Non-native vs. native/mixed	-26.94	3	9	64.68	7.92	0.014
% nest tree	-29.93	3	9	70.66	13.90	0.001
2003 Mean (β_0)	-32.81	2	9	71.62	14.86	0.000
Species + % native	-22.92	5	9	75.85	19.09	0.000
Species + native vs. non-native/mixed	-24.08	5	9	78.16	21.40	0.000
Species	-30.19	4	9	78.39	21.63	0.000
Species + non-native vs. native/mixed	-26.58	5	9	83.16	26.40	0.000
Species + % nest tree	-28.65	5	9	87.31	30.55	0.000
Weighted regression results 2004						
BRWE vs. others + non-native vs. native/mixed * Feb/Mar vs. others	-62.51	6	27	141.22	0.00	0.344
Non-native vs. native/mixed * Feb/Mar vs. others	-64.62	5	27	142.10	0.88	0.221
BRWE vs. others * non-native vs. native/mixed + non-native vs. native/mixed * Feb/Mar vs. others	-61.51	7	27	142.91	1.69	0.147
Species + non-native vs. native/mixed * Feb/Mar vs. others	-62.19	7	27	144.28	3.06	0.074
Non-native vs. native/mixed * month as continuous variable	-65.86	5	27	144.59	3.37	0.064
Non-native vs. native/mixed * month	-62.73	7	27	145.37	4.15	0.043
Native vs. non-native vs. mixed * month as continuous variable	-63.30	7	27	146.50	5.28	0.024

Continued

Appendix 1.7 continued

Model	**log-likelihood**	***K***	***n***	**AICc**	**ΔAICc**	w_i
BRWE vs. others * non-native vs. native/mixed * Feb/Mar vs. others	-61.37	8	27	146.74	5.51	0.021
% native * month as continuous variable	-67.91	5	27	148.69	7.46	0.008
Species + non-native vs. native/mixed * month as continuous variable	-64.49	7	27	148.88	7.65	0.007
Species + non-native vs. native/mixed * month	-60.28	9	27	149.15	7.93	0.006
Month as continuous variable	-71.29	3	27	149.63	8.40	0.005
2004 Mean (β_0)	-72.65	2	27	149.81	8.58	0.004
Native vs. non-native vs. mixed * month	-58.56	10	27	150.87	9.65	0.002
Non-native vs. native/mixed	-72.16	3	27	151.37	10.15	0.002
Species + month as continuous variable	-69.52	5	27	151.90	10.68	0.001
Species	-71.06	4	27	151.94	10.72	0.001
% native	-72.47	3	27	151.98	10.76	0.001
Native vs. non-native/mixed + month as continuous variable	-71.12	4	27	152.06	10.84	0.001
Native vs. non-native/mixed	-72.54	3	27	152.13	10.90	0.001
% nest tree	-72.57	3	27	152.19	10.97	0.001
Month	-71.27	4	27	152.35	11.13	0.001
Species + non-native vs. native/mixed	-69.75	5	27	152.36	11.13	0.001
Non-native vs. native/mixed + month as continuous variable	-71.27	4	27	152.37	11.15	0.001
Native vs. non-native vs. mixed	-71.42	4	27	152.67	11.45	0.001
Non-native vs. native/mixed + Feb/Mar vs. others	-71.44	4	27	152.71	11.48	0.001
Species + % native	-70.24	5	27	153.34	12.11	0.000
% native * Feb/Mar vs. others	-70.25	5	27	153.36	12.13	0.000
BRWE vs. others * non-native vs. native/mixed, BRWE vs. others * Feb/Mar vs. others	-67.05	7	27	154.00	12.78	0.000

Continued

Appendix 1.7 continued

Model	log-likelihood	K	n	AICc	ΔAICc	w_i
Species + non-native vs. native/mixed + month as continuous variable	-69.17	6	27	154.55	13.32	0.000
Native vs. non-native vs. mixed + month as continuous variable	-70.90	5	27	154.66	13.44	0.000
% nest tree * time	-70.94	5	27	154.73	13.51	0.000
Species + month	-69.27	6	27	154.75	13.53	0.000
Species + % nest tree	-71.04	5	27	154.94	13.71	0.000
Native vs. non-native/mixed * month as continuous variable	-71.10	5	27	155.06	13.83	0.000
Native vs. non-native/mixed + month	-71.12	5	27	155.10	13.87	0.000
Species + % native * Feb/Mar vs. others	-67.62	7	27	155.13	13.91	0.000
Non-native vs. native/mixed + month	-71.24	5	27	155.34	14.12	0.000
% nest tree * Feb/Mar vs. others	-71.42	5	27	155.70	14.47	0.000
Species * % native	-68.08	7	27	156.05	14.83	0.000
Native vs. non-native/mixed * month	-68.18	7	27	156.27	15.04	0.000
Species * % nest tree	-68.34	7	27	156.58	15.36	0.000
Species * non-native vs. native/mixed	-68.64	7	27	157.17	15.95	0.000
Species * time	-68.69	7	27	157.28	16.06	0.000
Species + non-native vs. native/mixed + month	-68.70	7	27	157.29	16.07	0.000
Native vs. non-native vs. mixed + month	-70.90	6	27	158.00	16.78	0.000
Species + % nest tree * Feb/Mar vs. others	-69.32	7	27	158.53	17.31	0.000
Species * month	-67.38	10	27	168.51	27.28	0.000

[1] Number of parameters in model.
[2] Sample size.
[3] AIC corrected for small sample size (Burnham and Anderson 2002).
[4] Difference in AIC_c value between best model and competing candidate model.
[5] Akaike weights, indicating the relative support for each of the candidate models.

CHAPTER 2

A DESCRIPTION OF THE FIRST MICRONESIAN HONEYEATER (*MYZOMELA RUBRATRA SAFFORDI*) NESTS FOUND ON SAIPAN, NORTHERN MARIANA ISLANDS

INTRODUCTION

Micronesian Honeyeaters (*Myzomela rubratra*) occur throughout the high islands of Micronesia, with subspecies endemic to Palau (*M. r. kobayashii*), Yap (*M. r. kurodai*), Chuuk (*M. r. major*), Pohnpei (*M. r. dichromata*), Kosrae (*M. r. rubratra*), and the Mariana Islands (*M. r. saffordi*; Pratt et al. 1987). Micronesian Honeyeaters, along with other native forest birds, were extirpated from Guam in the mid-1980s with the arrival and spread of the brown treesnake (*Boiga irregularis*; Savidge 1987, Wiles et al. 1995). On the inhabited Northern Mariana Islands (CNMI), most surveys have found Micronesian Honeyeaters to be less numerous on Saipan than on Rota or Tinian, and on Saipan, they are among the least abundant native passerines (Pratt et al. 1979, Ralph and Sakai 1979, Jenkins and Aguon 1981, Jenkins 1983, Craig 1996).

Little research has been done on the avifauna of the Mariana Islands, and basic life history information is unknown for most of the native and endemic species (Rodda et al. 1998, Mosher and Fancy 2002). This lack of information hampers the development and implementation of conservation measures. Saipan has experienced increased numbers of brown treesnake sightings in recent years (Rodda et al. 1998, N. B. Hawley pers. comm.). With the ever-present threat of brown treesnake introduction and establishment, information on the ecology and breeding biology of all species is urgently needed in the event that captive breeding programs need to be implemented.

I undertook a study to assess reproductive success of common forest passerines in native and non-native forest on Saipan (Chapter 1). Micronesian Honeyeaters were not a target species for this study, as they are reputed to be more common in coconut plantings, bushes and gardens of villages, scrub, second growth, coastal strand and mixed woodland habitats (Seale 1901, Safford 1902, Pratt et al. 1979, Jenkins 1983). However, I incidentally found seven Micronesian Honeyeater nests over the course of my study. To my knowledge, these are the first nests of this species found on Saipan. Here, I describe nests and nestlings.

STUDY AREA

Saipan is located in the western Pacific at 15°10' N and 145° 45' E (Figure 1.1). With a land area of 123 km^2, Saipan is the second largest island in the Marianas. Saipan has a tropical climate with an annual mean temperature of 28.3°C and mean annual rainfall of 200–250 cm. The wet and dry seasons may vary somewhat inter-annually, but the wet season extends from approximately July to November, and the dry season from December to June. Typhoons may occur at any time, but are most frequent between August and December (Young 1989, Mueller-Dombois and Fosberg 1998).

My study focused on two forest types, introduced tangantangan (*Leucaena leucocephala*) forest and native limestone forest. Most (77%) of the forest remaining on Saipan is non-native (Falanruw et al. 1989), and tangantangan forest is estimated to cover

28% of the island. This tree grows in dense near-monocultures on flat lowlands and plateaus (Craig 1990). Native limestone forest is restricted to cliffs and less accessible areas that are not easily cultivated (Craig 1989, Stinson and Stinson 1994), and is estimated to cover only 5% of Saipan (Engbring et al. 1986). *Pisonia grandis* and *Cynometra ramiflora* dominate the canopy of this forest type, and *C. ramiflora* and *Guamia mariannae* are the most common species in the understorey (Craig 1996). Four native and four non-native forest areas were selected as study sites (Figure 1.1). After assessing vegetation composition, one of the forest study sites initially considered native was determined to be a mixture of native and agriforest and included trees such as coconut (*Cocos nucifera*) and mango (*Mangifera indica*). Marpi, As Teo, and Kagman were native forest; Laolao Bay was mixed forest; and Bird Island, Cow Town, Obyan, and Naftan were non-native *Leucaena leucocephala* forest. Study areas were delineated by flagged transects.

METHODS

I conducted my study from April to July 2003 and February to May 2004. Micronesian Honeyeater nests were found while searching line transects following distance sampling methodology (Buckland et al. 2001), or incidentally while moving through the forest to monitor existing nests. When found, each nest was flagged and assigned a unique nest identification number. Nest contents were visually checked and described at 3-day intervals, using a mirror on a telescoping pole if necessary. I did not handle nest contents while the nest was still active. Thus, no egg measurements were possible, and I visually estimated nestling measurements using a millimeter ruler for comparison.

At the completion of the nesting attempt, I measured nest height, distance from trunk, and number and diameter of supporting branch(es). Tree species and tree height were also recorded. A clinometer was used to measure nest and tree heights (unless these could be directly measured with a steel measuring tape). Distance from the trunk was measured with a steel measuring tape, and diameters of supporting branches were measured with a millimeter ruler. The distance of the nest from the nearest road was estimated. Distances of nests from roads were grouped into 25-m categories (< 25 m, 26–50 m, 51–75 m, 76–100 m, and > 100 m). The nest was collected if possible and measured using a millimeter ruler.

RESULTS

I found a total of seven nests, one in 2003 and six in 2004. Nests were discovered on 31 May 2003, and 17 February, 9 March, 12 March, 7 April, 9 April, and 26 April 2004. Two nests had eggs, two had nestlings, and two did not yet have contents when located. The female was sitting on one nest and not disturbed, so contents were not determined when the nest was discovered. Four nests failed (three during incubation and one undetermined), and three fledged young. Four nests were located in the mixed forest, and one nest was located in each of the three native sites. All six nests where contents were seen contained two eggs or two young. I initially mistook two nests for Bridled White-eye (*Zosterops conspicillatus saypani*) nests, due to their similar size, structure and placement. However, I noticed that Micronesian Honeyeater nests tended to be thinner, and to deteriorate more rapidly than the nests of Bridled White-eyes and Golden White-eyes (*Cleptornis marchei*), whose nests they otherwise closely resembled.

Nest composition and structure

Only three nests were accessible and in adequate condition for measurement. Cup heights were 39, 40, and 45 mm ($\bar{x} = 41.3$ mm), and total nest heights were 41, 50, and 75 mm ($\bar{x} = 55.3$ mm). Internal diameters were 43, 47, and 50 mm ($\bar{x} = 46.7$ mm), while external diameters were 55, 69, and 73 mm ($\bar{x} = 65.7$ mm). Nests were comprised of vine tendrils and what appeared to be *Casuarina equisetifolia* needles (Figure 2.1). One of the nests also had part of a leaf skeleton from *Pandanus* sp. entwined around the outer base of the nest.

Nest Placement

Micronesian Honeyeater nests were located at various distances from roads (i.e., from < 25 m to > 100 m). Four nests were placed in *Guamia mariannae*, and three were placed in *Psychotria* sp. Nest (and tree) heights in *G. mariannae* were 1.5 m (5.6 m), 3 m (5 m), 3.5 m (6 m), and 5.1 m (not obtained), and in *Psychotria* sp. were 1.5 m (2 m), 1.7 m (2.3 m), and 3.8 m (8 m). Nests were placed 83–184 cm from the trunk in *G. mariannae* and 0–103 cm from the trunk in *Psychotria* sp., generally near the outer edge of the tree in which the nest was built (Figure 2.1). The number of nest support branches varied from two to five in both tree species, and average support branch diameter ranged from 1.5–9.7 mm in *G. mariannae* and from 1.5–2.5 mm in *Psychotria* sp.

Egg Description

Although four monitored nests each contained two eggs, I only clearly saw the eggs in one nest, found on 26 April 2004 at Marpi. The eggs were creamy white with two distinct rings of brown speckles, one ring near the broad end and the other near the narrow end of the egg.

Nestling Description

Two of the three nests that fledged young were found in the nestling stage, and one was found in the incubation stage. Micronesian Honeyeater nestlings are altricial and closely resemble Bridled White-eye nestlings in their development, until they develop red pin feathers. Because nestling development was variable, each nest is treated separately.

The 2003 nest initially contained eggs, and the two nestlings were first seen at day 0–1. At this age they were estimated at approximately 2 cm in length, had dark pink skin, and were downy on their wings and backs. On day 3–4, the nestlings had grown to 3–3.5 cm in length, were still covered with down, and skin color was dark pink/purple. They appeared to be well fed, with large, rounded stomachs noted. At day 6–7, the nestlings' eyes were cracking open. They were 4–4.5 cm long, with wing pins approximately 5 mm in length, and back pins beginning to erupt. The head was covered in long down. On day 7–8, the chicks were still 4–4.5 cm long, with wing pins 8 mm, back pins 2 mm, and bills beginning to curve. Head pins were still not present. Underlying skin color progressively lightened throughout nestling development and was pale pink by this stage. At day 9–10, the wing pins were 10 mm in length and tail and head pins had erupted 1 mm. Tan brown feathers had erupted from the wing pins, red feathers were beginning to erupt from the back pins, and 1–2 mm head pins were visible on day 10–11. Both nestlings prematurely fledged on day 13–14, when the observer was 1 m from the nest. One nestling was captured and returned to the nest, but the second could not be relocated and was left to the adults who were nearby and agitated. At this time, nestlings were estimated at 5.5 cm in length but were not yet fully feathered. Red feathers, 1 mm in length, were erupted on the back, gray feathers had erupted on the head, and 8 mm tail pins did not yet have feathers erupted. The breast was bare. On day 14–15, the remaining nestlings' tan brown wing feathers had turned dark gray, and it fledged at day 15–16.

The second nest that fledged young was found on 12 March 2004. At the time it was found, the two nestlings were already approximately 4 cm in length, with eyes open, and had 2 mm-long downy feathers erupting from the pins on the wings, back, and head. On 15 March only one nestling remained. This nestling prematurely fledged on 18 March with an observer distance of ≥ 3 m. The nestling could not fly, merely flutter, and was captured and returned to the nest. It was estimated at 4–4.5 cm in length and did not appear fully feathered. The erupted feathers were mostly black, with small red patches appearing on the head and back. This nestling fledged by 22 March when the final nest check was performed.

The last nest containing nestlings was found on 9 April 2004. The female was bringing food to the two nestlings at the time the nest was discovered. The nestlings were estimated at 3– 3.5 cm in length and were developing pin feathers. On 13 April the nestlings were approximately 4 cm long and were covered with long black pins from which feathers had erupted. Their eyes were open. Three days later the nestlings were 4–4.5 cm, and their bills were visible over the rim of the nest. They were black all over with no red visible. By 19 April the nestlings had fledged.

Parental Behavior

Only females were observed incubating (five nest checks) or brooding nestlings (one nest check). However, one or both members of the pair were often observed close to the nest. If observed, the adult(s) were always very agitated. Typically, one or both adults would feign injury, fluttering about low to the ground and drooping one wing. If only one adult was present, this behavior was sometimes accompanied by scolding. If both adults were present, often one adult would feign injury while the other scolded. I observed injury feigning behavior on 9/26 nest visits, and scolding at 5/26 nest checks. This behavior was only observed at nests containing nestlings. Micronesian Honeyeaters were very intolerant of disturbance at the nest during the incubation stage. Any time the incubating female was disturbed during a nest check ($n = 3$), the nest failed by the next visit.

DISCUSSION

Nests, eggs, and nestlings of the Micronesian Honeyeater have previously been found on Guam and described by various authors (Hartert 1898, Seale 1901, Yamashima 1932, Jenkins 1983, N. Drahos pers. comm.). Many similarities occur between Guam and Saipan nests. Measurements of individual nests seem quite variable, with the following ranges reported from Guam: cup height 25–50 mm, outer height 50–120 mm, internal diameter 25–60 mm, and external diameter 35–80 mm (Hartert 1898, Seale 1901, Jenkins 1983, N. Drahos pers. comm.). My measurements lie within these ranges. Nest heights are also similar, with nest heights on Guam varying from 1.2–4.6 m (Hartert 1898, Seale 1901, Yamashima 1932, Jenkins 1983, N. Drahos pers. comm.). As with

Saipan nests, Guam nests were described as "loosely constructed and fragile, often with daylight penetrating the walls" (Jenkins 1983:47), and were placed among the outer branches of the trees in which they were constructed (Seale 1901, Jenkins 1983).

Nesting materials appear similar, with *Casuarina equisetifolia* needles used on both islands. Other authors also report the use of leaves, although not *Pandanus* sp. specifically. The use of vine tendrils was not reported from Guam. The chief difference between my observations and those of other authors lies in the species of nest tree. Nests were placed in *Psychotria* sp. and *Guamia mariannae* (trees native to the Mariana islands) on Saipan, whereas on Guam, nests were reported in *Pithecellobium dulce*, *Casuarina equisetifolia*, *Delonix regia*, and *Bruguiera gymnorrhiza*, two of which are indigenous to the Marianas (*C. equisetifolia* and *B. gymnorrhiza*) and two of which are not (Raulerson and Rinehart 1991). This is likely a reflection of Guam authors working in different habitats than myself, rather than differences in habitat use between Guam and Saipan birds.

All reported clutch sizes are of two eggs. Micronesian Honeyeater eggs from Guam were described as white, off-white or cream-colored with rufous-brown speckling concentrated at the thicker end (Hartert 1898, Seale 1901, Jenkins 1983), or as white with gray and dark yellow-brown speckling concentrated near the sharper end (Yamashima 1932). I only clearly saw two eggs, but the markings differed slightly, showing two clearly defined rings of brown spots, one near the broad end and the other near the narrow end of the egg.

I could not find comparative descriptions of nestlings or data on fledging age. Several authors described fledgling Micronesian Honeyeaters from Guam. Seale

(1901:57) reported that “the young are olive brown above, yellowish on the under parts, washed with red on the sides of the fore breast and back; bill dark, yellowish on the base of lower mandible; feet and iris dark”. N. Drahos (pers. comm.) described a pair of fledgling Micronesian Honeyeaters recently out of the nest. The female was mouse gray with a faintly rusty red chin; bill black with a yellow stripe on its edge and top of the bill yellow at the base; eyes and feet black. He reported the male as similar, however with the middle of the back, chin and lower half of the head faintly cardinal red. Other author’s descriptions are similar although less comprehensive. Although my sample size is limited to two nests, it appears that Micronesian Honeyeater nestlings are apt to leap from the nest before they are fully ready to fledge, and a natural fledging age of 15–16 days seems reasonable. Parental behavior of Micronesian Honeyeaters on Guam appears the same as that of birds on Saipan, with females feigning injury (Stophlet 1946, Jenkins 1983, N. Drahos pers. comm.).

The Micronesian Honeyeater nests on Saipan were all located in native limestone forest or mixed native/agriforest. These habitats have not previously been reported as preferred by Micronesian Honeyeaters, and this underscores the importance of obtaining ecological information for all native species to further the development of conservation plans. Some of the habitats in which Micronesian Honeyeaters are reportedly common, for example backyard gardens, would appear unsuitable as nesting habitat, given this species’ intolerance of disturbance at the nest and the likelihood of disturbance in these areas. Overall, I found that Saipan Micronesian Honeyeaters have similar nesting requirements and behaviors as Guam birds prior to their extirpation. This information is useful, because it indicates that individuals could be sourced from Saipan for reintroduction to Guam, once brown treesnakes and other threats on Guam are controlled or eradicated.

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Figure 2.1. Micronesian Honeyeater (*Myzomela rubratra saffordi*) nest on Saipan, Northern Mariana Islands.