# Supplementary Material

Status of endemic reed-warblers of the Mariana Islands, with emphasis on conservation strategies for the endangered Nightingale Reed-warbler *Acrocephalus hiwae*

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**Contents**

# Appendix S1. Description of reed-warbler trends in the Mariana Islands.

Appendix S2. Description of Mariana Islands occupied or previously occupied by reed-warblers.

# Appendix S3. Point-transect survey model evaluation.

# Appendix S4. Occupancy model evaluation.

# Appendix S5. Survey effort requirements

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# Appendix S1. Description of reed-warbler trends in the Mariana Islands

The Guam Reed-warbler has not been observed on Guam since 1969 despite numerous searches of all wetland sites on the island (Reichel *et al*. 1992). It was suggested that habitat loss (Jenkins 1983, Engbring and Ramsey 1984) or pesticide use caused their final decline (Drahos, unpublished data cited in Reichel *et al.* 1992). Although habitat loss likely played a role in its extinction, Reichel *et al.* (1992) believed that predation by the introduced brown tree snake *Boiga irregularis* played the key role in the final disappearance of the species on Guam. This suggestion is supported by the fact that the spread of the brown tree snake across Guam coincided with declines of almost all other bird species on the island (Savidge 1987, Reichel *et al.* 1992, Rodda and Savidge 2007).

It has been suggested that the Aguiguan Reed-warbler was common before WWII (Baker 1951), although observations in 1954 found it to be one of the least common forest birds on the island (GD Peterson, unpublished data, cited in Reichel *et al.* 1992). The 1982 surveys recorded four incidental observations and produced a population estimate of between 4 and 15 birds (Engbring *et al*. 1986). Small numbers of birds were observed at various times after the 1982 survey until 1995, when the Aguiguan Reed-warbler was last observed (Craig and Chandran 1993, Lusk 1993, Marshall *et al.* 1996). There are no wetlands on Aguiguan and past observations indicate the Aguiguan Reed-warbler used native limestone forest, particularly areas with dense canopy (Engbring *et al*. 1986; DFW unpublished data cited in Reichel *et al*. 1992), which is different from the habitat used by all other reed-warblers in the Mariana archipelago. The Guam and Pagan Reed-warblers appeared to have primarily occupied wetlands, while the Nightingale Reed-warbler uses wetlands and upland tangantangan (*Leucaena leucocephala*) thickets and forests on both Saipan and Alamagan (Reichel *et al*. 1992). Where the vegetation was drastically altered by feral ungulates, reed-warbler were found in cover consisting of partially open overstory and somewhat brushy understory (Reichel *et al.* 1992).

There is little information about the distribution of the Pagan Reed-warbler. Early collections and later observations suggested the species was locally common in the two wetlands on the island: Laguna Sanhalom (Inner Lake) and Laguna Sanhiyon (Lagoon Lake; Reichel *et al.* 1992). The species was present until at least the 1960s but has not been observed since, despite efforts to locate them in the late 1970s and 1980s, and in 2000 (Reichel *et al.* 1992, DFW 2000). Wetlands on Pagan were altered and reduced in the last century due to human development and feral ungulates, such that by 1979 there was little shoreline vegetation remaining around either wetland (Tenorio and Associates 1979). A volcanic eruption in 1981 resulted in elimination of all herbaceous and most woody vegetation at Laguna Sanhalom, which only began to recover in 1989 (Reichel *et al.* 1992). Pratt (2010) reported that both wetlands were primarily surrounded by ironwood trees (*Casuarina equisetifolia*) in 2010 and the only wetland vegetation recorded was a patch of the native saltgrass (*Paspalum vaginatum*) on Laguna Sanhiyon.

Following World War II, the Nightingale Reed-warbler populations on Saipan were thought to be localized due to extensive agricultural development on the island prior to the war (Stott 1947, Marshall 1949, Engbring *et al*. 1986). However, Pratt *et al.* (1979) characterized the species as common in the 1970s and Engbring *et al.* (1986) reported a population of around 4,800 birds in 1982. In 2007, an island-wide bird survey of Saipan showed that the Nightingale Reed-warbler has declined since 1982 to 2,742 birds (Camp *et al.* 2009). Saipan is the primary stronghold for the species, making this a result of great concern regarding the long-term conservation of the species in the Mariana Islands.

Limited information is available on the Nightingale Reed-warbler population on Alamagan prior to the 1980s. Reichel *et al.* (1992) estimated the population to be between 350 and 1,000 pairs based on limited surveys in 1988 and 1990. They reported the species occurred in forested areas with open or brushy understories and in forest edges adjacent to swordgrass *Miscanthus floridulus*. Stinson (1993) estimated the population to be around 2,000 individuals based on a survey of the northern half of the island in 1992. DFW (2000) estimated the population to be 173 pairs based on extensive surveys of the entire island in 2000.

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# Appendix S2. Description of Mariana Islands occupied or previously occupied by reed-warblers.

The Mariana Islands are located in the western Pacific Ocean east of the Philippines and south of Japan (Figure 1). The 290 km long archipelago is comprised of 15 islands that include the Territory of Guam and the 14 islands of the CNMI. The older southern islands are volcanic in origin, but composed of uplifted limestone terraces, while the islands north of Farallon de Medinilla are stratovolcanoes (Trusdell 2009). While Guam and the CNMI are politically separate, all islands in the archipelago are biologically similar although the plant and animal composition varies by island. The avifauna consists of 21species of native terrestrial and freshwater birds, 12 of which are endemic and 12 are federally listed as endangered, including the four reed-warbler species. At least three additional species have become extinct or extirpated in the last 50 years, including the Guam Flycatcher (*Myiagra freycineti*; extinct due to brown tree snake predation), the Mariana Mallard *Anas oustaleti*, and the White-browed Crake *Amaurornis cinerea*, as have some subspecies (for example, the Guam Bridled White-eye *Zosterops conspicillatus conspicillatus* (Pratt *et al.* 1987, Wiles *et al.* 1995, USFWS 2004).

Humans reached the Mariana Islands around 2,000 BC and may have impacted many, if not all the islands in the archipelago (Farrell 2001, Russell 1998). These impacts include agriculturally driven habitat conversion and fragmentation, the introduction of non-native species, and hunting (Steadman 1995). The Spanish reached the Mariana Islands in the early 15th century (Russell 1998) bringing with them European-based agriculture and husbandry of cattle *Bos taurus*, pigs *Sus scrofa*, and goats *Capra hircus* (Farrell 1992, Russell 1998). During the 19th and 20th centuries the islands were controlled by Germans, Japanese and Americans, all of whom further expanded agricultural production, human settlements and military development (Farrell 2001, Russell 1998, Spennemann 1999, Bowers 2001, Rottman 2004a,b).

Currently humans live year-round on Guam, Rota, Tinian, and Saipan. Anatahan supported a small human population until 2003 when a volcanic eruption lead to its evacuation. The other islands in the archipelago are unoccupied, although both Alamagan and Pagan are inhabited sporadically during some years. Guguan, Asuncion, Maug and Uracas see very little human use or visitation and have been designated as wildlife conservation areas by the CNMI (CNMI Constitution, Article XIV, Section 2 and CNMI Public Law 14-49). Asuncion, Maug, and Uracas are further protected as part of the Marianas Trench Marine National Monument (Proclamation No. 8335, 74 F.R. 1557). Farallon de Medinilla is currently used for training by the U.S. Navy and is off-limits to civilian access.

Aguiguan is approximately 7 km2 in area and is located 9 km off the southwest shore of Tinian and 150 km northeast of Guam (Figure 1). The island is composed of a series of step-like limestone terraces with sheer cliffs rising from the sea on all sides (Klawunder 1993). Because of the large number of feral goats on the island, Aguiguan is commonly referred to locally as “Goat Island.” The goats have had a significant impact to the vegetation (Rice 1991, 1993, Esselstyn *et al.* 2003). Although the island has been uninhabited since the end of World War II, human activities resulted in extensive alteration of the vegetation (Engbring *et al.* 1986). A recent land cover assessment (Amidon *et al.* 2017) indicates that approximately 55% of cover is native limestone forest, while 26% is scrub/shrub (primarily *Lantana camara*), 4% is tangantangan, 2% ironwood forest, 2% is mixed introduced forest, less than 1% is mixed grass/herbaceous, and the remainder of the island is unvegetated.

Alamagan is located 255 km north of Saipan and has an area of roughly 13 km2 (Figure 1). Approximately 48% of the island is covered in mixed grass/herbaceous vegetation, 35% is covered in forest and scrub vegetation (16% coconut forest, 8% hibiscus thicket, 6% mixed introduced forest, and 5% scrub/shrub), while the remainder is unvegetated (Amidon *et al.* 2017). The island was inhabited year-round until 2009 (de la Torre 2009) and the vegetation around the old village and homestead sites was extensively modified by humans (DFW 2000). The island also currently exhibits signs of substantial impacts from the feral ungulates (cattle, pigs, and goats), including erosion, likely due to population increases after human residents evacuated due to a typhoon.

Pagan, the fifth largest island of the Mariana archipelago at 48 km2 in area, is located approximately 305 km north of Saipan (Figure 1). The island is comprised of a northern and southern stratovolcano connected by a narrow isthmus. The northern volcano, Mt. Pagan, is active and was emitting steam and gas in 2016 (Global Volcanism Program 2016). Approximately 47% of the island is forested (34% ironwood forest, 10% coconut forest, and 3% mixed introduced forest) with the remainder dominated by mixed grass/herbaceous vegetation (25%), and bare rock and old lava flow (21%; Amidon *et al.* 2017). The island was inhabited until a volcanic eruption in 1981 caused the evacuation of 300 residents (Cave 2015). It is still occasionally inhabited, and many former residents wish to return (Cave 2015, Todino 2016).

Tinian, the third largest island in the Mariana archipelago at 101 km2 in area, is located 8 km southwest of Saipan and 166 km northeast of Guam (Figure 1). The island is formed of low plateaus with a gentle limestone ridge to the south (Camp *et al.* 2012). The vegetation is heavily dominated by non-native species and only around 4 percent of native limestone forest remains and that has been heavily altered by humans and nonnative species (Engbring *et al.* 1986, Amidon *et al.* 2017). After World War II, the island was re-vegetated with tangantangan that has since expanded into large thickets across the island (Engbring *et al.* 1986, Amidon *et al.* 2017). The northern 2/3 of the island is leased to the U.S. Navy (U.S. Navy 2010) for training exercises, although portions of the north are still used for public recreation, grazing, and agriculture. Most of the 3,136 people on island (Saipan Tribune 2011) live in or near the town of San Jose in the south. Goats were recently introduced to the southeast part of the island by a former Mayor and the area is already showing signs of overgrazing (APM, pers. obs.).

Saipan, the second largest of the islands in the Mariana archipelago at 119 km2, is the center of the CNMI government and has the largest human population. It is located 185 km north of Guam and is made up of a series of low-lying plateaus with a limestone ridge that is dominated by Mt. Takpochao (Figure 1). The vegetation is also highly altered by humans and nonnative species. The island was seeded with tangantangan after the war and tangantangan thickets and mixed introduced forest now dominate the island (Engbring *et al.* 1986, Amidon *et al.* 2017). Only around 16% of the island is covered in native dominated forest (Amidon *et al.* 2017). The latest census figures place the Saipan population at 48,220 (Saipan Tribune August 26, 2011, US Census Bureau 2010).

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# Appendix S3. Point-transect survey model evaluation.

Appendix S3. Figure 1. Histogram of reed-warbler detections used to derive the detection function and population estimates via distance sampling. A best fit line of these data were modeled using the hazard-rate detection function with the covariate survey.



Appendix S3. Table 1. Point-transect sampling model selection results for the reed-warbler data. Models are half-normal (H-norm) and hazard-rate (H-rate) key detection functions. Except for a covariate representing Survey (FWS 2010 Alamagan, DFW 2000 Alamagan, and DFW Saipan) no series expansion adjustment terms or sampling and site covariates were modeled because of over-fitting concerns due to the small numbers of detections. *K* is the number of estimated parameters, *L* is the estimate of the log-likelihood, AIC*c* is the 2nd-order Akaike's Information Criterion corrected for small sample sizes, ΔAIC*c* is the difference among the AIC*c* values, and *wi* is the Akaike model weight that the likelihood that model is the best model (of the 3 models evaluated).

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Model | *k* | *L* | AICc | ΔAICc | *wi* |
| H-rate Key Survey | 3 | -487.10 | 980.40 | 0.00 | 0.836 |
| H-rate Key | 2 | -490.42 | 984.94 | 4.54 | 0.086 |
| H-norm Key | 1 | -491.56 | 985.16 | 4.76 | 0.077 |

# Appendix S4. Occupancy model evaluation.

Selection of models describing the relation of reed-warbler occurrence to site and sampling covariates. Covariates are indicated in parentheses following the occupancy symbol “*ψ*” and detection probability symbol “*p*”. Site covariates “veg”, “understory”, “canhgt”, “cancov”, and “time”, refer to vegetation type, understory closure/openness, canopy height, and canopy cover, respectively. The sampling covariate “time” indicates the time of day (morning or afternoon) at which the observation was made. The term “global” refers to a model that includes all covariates. Column *k* is the number of model parameters; *L* is the -2\*log likelihood; ΔQAICc is the relative difference in QAICc values from the model with the smallest value; *w* is the QAICc model weight; ĉ is the estimated average over-dispersion parameter; and *Χ*2 *p*-value is the probability of observing a *Χ*2 greater than or equal to the test statistic based upon 999 parametric bootstraps. Shaded rows are top-ranked models for which occupancy estimates are provided in Table 2.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Model | *k* | *L* | ΔQAICc | *w* | ĉ | *Χ*2 p-value |
| *ψ*(canhgt-veg)*,p*(time) | 6 | 111.12 | 0.0 | 0.74 | 1.406 | 0.139 |
| *ψ*(veg)*,p*(time) | 5 | 111.86 | 2.7 | 0.19 | 1.332 | 0.139 |
| *ψ*(global)*,p*(time) | 8 | 100.74 | 6.0 | 0.04 | 1.252 | 0.198 |
| *ψ*(veg)*,p*(.) | 4 | 112.66 | 7.7 | 0.02 | 1.236 | 0.228 |
| *ψ*(canhgt-veg)*,p*(.) | 5 | 111.90 | 8.6 | 0.01 | 1.245 | 0.149 |
| *ψ*(understory-veg)*,p*(.) | 5 | 112.63 | 10.3 | 0.00 | 1.229 | 0.238 |
| *ψ*(cancov-veg)*,p*(.) | 5 | 107.32 | 12.1 | 0.00 | 1.149 | 0.238 |
| *ψ*(global)*,p*(.) | 7 | 101.54 | 12.7 | 0.00 | 1.135 | 0.287 |
| *ψ*(understory-canhgt)*,p*(.) | 4 | 115.48 | 12.8 | 0.00 | 1.200 | 0.188 |
| *ψ*(understory-veg)*,p*(time) | 6 | 111.82 | 13.0 | 0.00 | 1.214 | 0.208 |
| *ψ*(understory-canhgt)*,p*(time) | 5 | 114.74 | 15.5 | 0.00 | 1.186 | 0.238 |
| *ψ*(understory)*,p*(time) | 4 | 115.17 | 17.0 | 0.00 | 1.147 | 0.327 |
| *ψ*(canhgt)*,p*(.) | 3 | 115.90 | 20.3 | 0.00 | 1.095 | 0.347 |
| *ψ*(.)*,p*(time) | 3 | 115.83 | 20.3 | 0.00 | 1.094 | 0.317 |
| *ψ*(canhgt)*,p*(time) | 4 | 115.18 | 22.2 | 0.00 | 1.091 | 0.356 |
| *ψ*(cancov-canhgt-veg)*,p*(time) | 7 | 103.32 | 22.2 | 0.00 | 1.043 | 0.366 |
| *ψ*(understory)*,p*(.) | 3 | 115.93 | 22.5 | 0.00 | 1.072 | 0.297 |
| *ψ*(understory-cancov-canhgt)*,p*(time) | 6 | 106.64 | 24.0 | 0.00 | 1.035 | 0.396 |
| *ψ*(understory-cancov-veg)*,p*(.) | 6 | 103.95 | 24.2 | 0.00 | 1.007 | 0.455 |
| *ψ*(cancov)*,p*(time) | 4 | 112.75 | 24.7 | 0.00 | 1.043 | 0.356 |
| *ψ*(understory-cancov-canhgt)*,p*(.) | 5 | 107.45 | 25.8 | 0.00 | 1.004 | 0.475 |
| *ψ*(cancov-canhgt)*,p*(.) | 4 | 111.44 | 26.8 | 0.00 | 1.011 | 0.436 |
| *ψ*(.)*,p*(.) | 2 | 116.58 | 27.1 | 0.00 | 1.016 | 0.446 |
| *ψ*(cancov-canhgt-veg)*,p*(.) | 6 | 104.01 | 27.9 | 0.00 | 0.972 | 0.347 |
| *ψ*(cancov-veg)*,p*(time) | 6 | 106.55 | 29.1 | 0.00 | 0.985 | 0.436 |
| *ψ*(understory-cancov-veg)*,p*(time) | 7 | 103.12 | 30.2 | 0.00 | 0.964 | 0.406 |
| *ψ*(understory-cancov)*,p*(time) | 5 | 108.19 | 37.3 | 0.00 | 0.913 | 0.564 |
| *ψ*(cancov-canhgt)*,p*(time) | 5 | 110.75 | 38.1 | 0.00 | 0.928 | 0.545 |
| *ψ*(understory-cancov)*,p*(.) | 4 | 109.02 | 39.0 | 0.00 | 0.891 | 0.634 |
| *ψ*(cancov)*,p*(.) | 3 | 113.47 | 42.8 | 0.00 | 0.884 | 0.624 |

# Appendix S5. Survey effort requirements.

Detection probability (DP), in percent, of detecting one reed-warbler from a population of *n* birds randomly distributed across the known range of the species on Aguiguan and Pagan, CNMI, and the minimum number of survey visits (*Nmin*) to obtain 95% and 99% DP.

|  |  |  |
| --- | --- | --- |
| Pop (*n*) | Aguiguan | Pagan |
| Wetland and Forest | Wetland |
| DP (%) | *Nmin* 95% DP  | *Nmin* 99%DP | DP (%) | *Nmin* 95% DP  | *Nmin* 99%DP | DP (%) | *Nmin* 95% DP  | *Nmin* 99%DP |
| 20 | 99.9 | 0.4 | 0.7 | 29.5 | 8.6 | 13.1 | 97.7 | 0.8 | 1.2 |
| 15 | 99.4 | 0.6 | 0.9 | 23.1 | 11.4 | 17.5 | 94.0 | 1.1 | 1.6 |
| 10 | 96.7 | 0.9 | 1.4 | 16.0 | 17.1 | 26.3 | 84.7 | 1.6 | 2.4 |
| 5 | 81.9 | 1.8 | 2.7 | 8.4 | 34.2 | 52.6 | 60.9 | 3.2 | 4.9 |
| 2 | 49.5 | 4.4 | 6.7 | 3.4 | 85.5 | 131.5 | 31.3 | 8.0 | 12.2 |