Terrestrial Resource Surveys of Tinian and Aguiguan, Mariana Islands, 2008
- Final Report -

Top to bottom (left to right): Aguiguan Island by Curt Kessler, Tinian Monarch by Eric VanderWerf, Goat by Scott Vogt, Coconut Crab by Cheryl Phillipson, Red-taled Tropicbird by Scott Vogt, Mourning Gecko by Gordon Rodda, Shrew by Curt Kessler, Lepidoptera larvae by Shellly Kremer, Rufous Fantail by Eric VanderWerf, and Tinian Island by Curt Kessler.

FINAL REPORT

Terrestrial Resource Surveys of Tinian and Aguiguan, Mariana Islands, 2008

COMPILED AND EDITED BY

U.S. Fish and Wildlife Service
Pacific Islands Fish and Wildlife Office
Honolulu, HI

PREPARED FOR

Marine Force Pacific (MARFORPAC)
and
Naval Facilities (NAVFAC), Pearl Harbor
Honolulu, HI

DECEMBER 2009

ACKNOWLEDGEMENTS

These surveys would not have been possible without the support and participation of many people and organizations. Permits, participation in the surveys, and support were provided by the Commonwealth of the Northern Mariana Islands Department of Land and Natural Resources (DLNR), the DLNR Division of Agriculture, and the Tinian Mayor's Office. We would especially like to thank the Honorable Jose P. San Nicolas, Dr. Ignacio de la Cruz, Sylvan Igisomar, Joe Lizama, Henry Cabrera, Rosemary Camacho, Antonio Castro, David Evangelista, John King, Elvin Masga, Fabrious Muna, Jess Omar, Gayle Martin, Paul Radley, Laura Williams, Ignacio "Cutz" Aldan, Greg Camacho, Rodney Camacho, Vincente Camacho and all of the personnel from the Tinian Mayor's Office, Brown Treesnake Program, the Division of Fish and Wildlife's Fisheries Section, Division of Parks and Recreation, and the Division of Fish and Wildlife Admin Office that assisted with transect cutting, logistics, and administrative support for the surveys. Funding and assistance for the project was provided by the Naval Facilities (NAVFAC), Pearl Harbor and Marine Force Pacific (MARFORPAC) with administrative and field support from Navy biologists Vanessa Pepi and Scott Vogt. Research, analysis, and review for some of the surveys were provided by the U.S. Geological Survey, Biological Research Discipline and the University of Guam. Helicopter support for the surveys was provided by Americopters and pilots Rufus Crowe and Naoto Kogure. Additional field support for establishing the territory mapping plots on Tinian was generously provided by Yogesh Singh. We would also like to thank the following staff from the Pacific Islands Fish and Wildlife Office and Portland Regional Office for participating in the bird surveys and providing logistical support: Patrice Ashfield, Peter Dunlevy, Joshua Fisher, Adonia Henry, Shelly Kremer, James Kwon, Megan Laut, Patrick Leonard, Aaron Nadig, Jay Nelson, Jeff Newman, Cheryl Phillipson, Jeff Zimpfer, Elaine Bok, Mike Roy, John Nuss, and Kit Hershey. This report was compiled by Curt Kessler, Ann Marshall, and Fred Amidon, Pacific Islands Fish and Wildlife Office. Finally, we would like to give special thanks to the private landowners and people of Tinian for allowing access to their lands during the surveys and for their hospitality.

TABLE OF CONTENTS

1.0 INTRODUCTION 5

1.1 GOALS AND OBJECTIVES 5

1.2 BACKGROUND 5

2.0 TERRESTRIAL SURVEYS 8

2.1 VEGETATION SURVEYS ON TINIAN AND AGUIGUAN 8

2.2 INVERTEBRATE SURVEYS 19
- 2.2.1 GENERAL INSECT SURVEYS ON TINIAN AND AGUIGUAN 19
- 2.2.2 CANDIDATE BUTTERFLY SURVEYS ON TINIAN 50
- 2.2.3 COCONUT CRAB SURVEYS ON MILITARY LEASE LANDS ON TINIAN 61

2.3 REPTILE AND AMPHIBIAN SURVEYS 79
- 2.3.1 REPTILES AND AMPHIBIANS OF TINIAN 79
- 2.3.2 REPTILES AND AMPHIBIANS OF AGUIGUAN 131
- 2.3.3 POPULATION DENSITIES AND DIET OF MONITOR LIZARDS (*VARANUS INDICUS*) ON AGUIGUAN 139

2.4 AVIAN SURVEYS 150
- 2.4.1 GENERAL LAND BIRD SURVEYS ON TINIAN AND AGUIGUAN 150
- 2.4.2 SEABIRD SURVEYS 182
- 2.4.3 MICRONESIAN MEGAPODE ON TINIAN AND AGUIGUAN 201
- 2.4.4 NIGHTINGALE REED-WARBLER ON AGUIGUAN 219
- 2.4.5 TINIAN MONARCH SURVEYS 227

2.5 MAMMAL SURVEYS 240
- 2.5.1 RODENTS ON TINIAN 240
- 2.5.2 RODENTS ON AGUIGUAN 258
- 2.5.3 MARIANA FRUIT BAT ON TINIAN AND AGUIGUAN 269
- 2.5.4 SHEATH-TAILED BAT ON AGUIGUAN 273
- 2.5.5 FERAL GOATS ON TINIAN AND AGUIGUAN 279

3.0 APPENDICES 282
- 3.1 STATUS AND TRENDS OF THE LAND BIRD AVIFAUNA ON TINIAN AND AGUIGUAN, MARIANA ISLAND
- 3.2 SYSTEMATIC RODENT MONITORING: A STUDY OF THE SMALL MAMMALS IN THE MARIANA ISLANDS
- 3.3 ASSESSMENT FOR PACIFIC SHEATH - TAILED BATS (*Emballonura semicaudata rotensis*) ON AGUIGUAN, COMMONWEALTH OF THE NORTHERN MARIANA ISLANDS

Terrestrial snail surveys were conducted on Tinian. The individual hired to conduct the surveys has not provided a final report.

1.0 INTRODUCTION

1.1 GOALS AND OBJECTIVES

Pursuant to an Interagency Service Agreement (Agreement) between the U.S. Fish and Wildlife Service (USFWS), Marine Force Pacific (MARFORPAC), and the Department of Defense (DoD), the USFWS was contracted to conduct surveys of marine and terrestrial resources of some of the islands in the Mariana archipelago. The DoD is authorized to contract with appropriate Federal agencies under the provisions of the Sikes Act of 1960, as amended (31 U.S.C. 868) to promote planning, development, maintenance, and coordination of wildlife, fish and game conservation and rehabilitation on military reserves. This Agreement was developed in order to determine what resources may be impacted during the relocation of the U.S. Marine Corps Forces from Okinawa to the Mariana Islands and during training activities planned for various locations in the Mariana Islands.

Under the terms of the Agreement, some of the terrestrial surveys were conducted in 2008. The following report includes the results of the surveys conducted on the islands of Tinian and Aguiguan, Commonwealth of the Northern Mariana Islands. Further survey work will occur archipelago-wide in the following several years.

1.2 BACKGROUND

Tinian (100 square kilometers (km)) is the second largest island in the Commonwealth of the Northern Mariana Islands (Figures 1 and 2). It is 20 km long and 7 km at its widest point and the highest points on the island are Carolinas Ridge (178 meters (m)) and Mount Lasu (160 m) (Mueller-Dombois and Fosberg 1998). Aguiguan is a small island (7 km^2) approximately 9 kilometers south of Tinian (Figure 3). It is approximately 5 km long and 1.5 km wide at its widest point and the highest point on the island is 163 m (Engbring *et al.* 1986).

Both Tinian and Aguiguan constitute the Municipality of Tinian in the Commonwealth of the Northern Mariana Islands. In 2000, the population on Tinian was approximately 3,540 people, with the majority in the village of San Jose (U.S. Census 2008), while

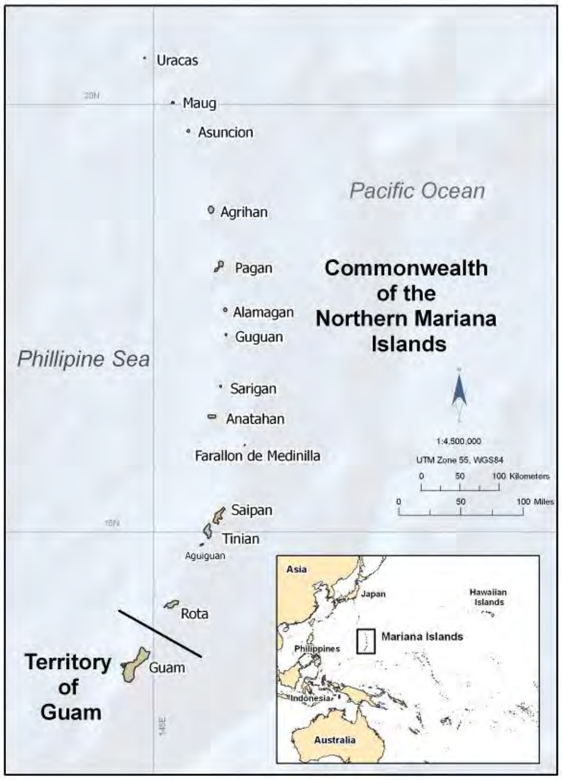


Figure 1. Location of the Commonwealth of the Northern Mariana Islands, Mariana archipelago.

Aguiguan is uninhabited. The climate on both islands is tropical and temperatures remain and relatively consistent during the year, ranging daily from 25 degrees to 30 degrees Celsius. Rainfall varies considerably between years but averages 218 centimeters annually, most of which falls from July to November. A dry season occurs between January and May when rains diminish to 8 to 15 centimeters per month.

Approximately 80 percent of arable land on Tinian was put into sugarcane production during the 1930s (Bowers 1950). Tinian was also the site of a major U.S. beach landing during WWII and much of the island developed into a major airbase to support bombing operations on Japan (Rottman 2004). The airbase was abandoned after the war but the Department of Defense currently leases approximately two-thirds of the northern part of the island for training purposes. Currently, training consists of once a year battalion size exercises that last for about two weeks and occasional helicopter touch and go practice along the airstrip for the rest of the year.

Aguiguan was also partially developed by the Japanese for sugar cane production and for timber harvest during the 1930s (Davis 1954). However, it

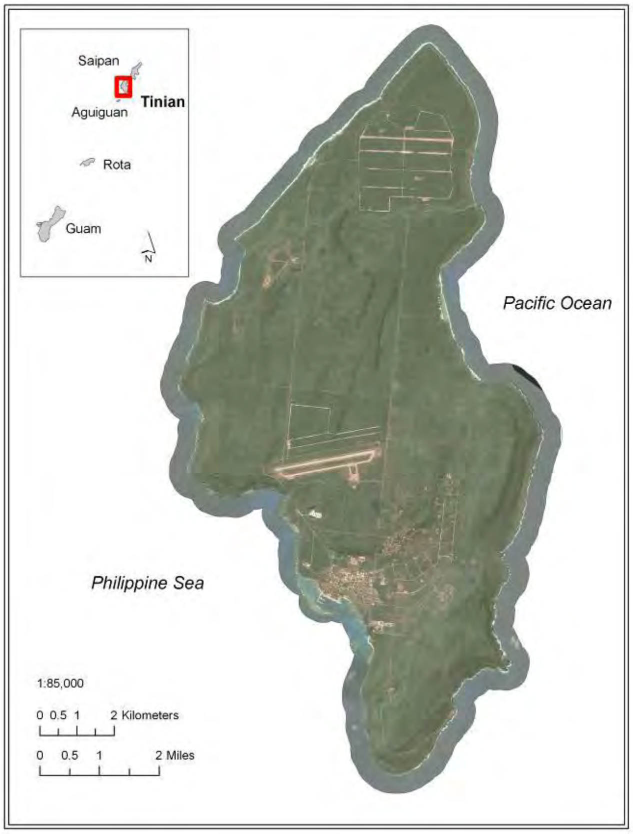


Figure 2. The island of Tinian, Commonwealth of the Northern Mariana Islands.

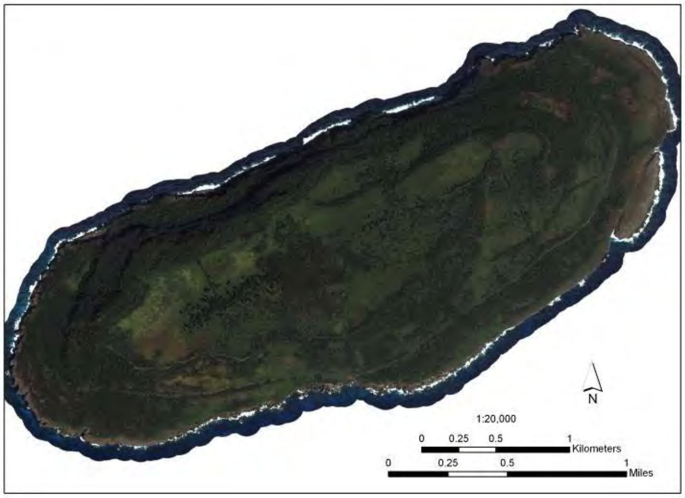


Figure 3. The island of Aguiguan, Commonwealth of the Northern Mariana Islands.

was spared invasion during WWII and has remained uninhabited since.

REFERENCES

Bowers, N. M. 1950. Problems of resettlement on Saipan, Tinian, and Rota Mariana Islands. Occasional Historical Papers Series No. 7. CNMI Division of Historic Preservation.

Davis, C. J. 1954. Report on the Davis expedition to Agiguan, July-August, 1954. Unpublished Report. 24 pp.

Engbring, J., F. Ramsey, and V. J. Wildman. 1986. Micronesian Forest Bird Survey, 1982: Saipan, Tinian, Agiguan, and Rota. U.S. Fish and Wildlife Service, Honolulu, Hawaii. 143 pp.

Mueller-Dombois, D., and F. R. Fosberg. 1998. Vegetation of the Tropical Pacific Islands. Springer, New York.

Rottman, G. L. 2004. Saipan and Tinian 1944: Piercing the Japanese Empire. Campaign 137. Ospery Publishing, Osceola, WI.

U.S. Census Bureau. 2008. Website of the U.S. Census Bureau, available online at <http://www.census.gov>.

2.0 TERRESTRIAL SURVEYS

2.1 VEGETATION SURVEYS ON TINIAN AND AGUIGUAN


Western Coast of Aguiguan. Photoby Curt Kessler


Pina Region of Tinian. Photoby Curt Kessler

Prepared by: **Fred Amidon, U.S. Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office, Honolulu, HI**

INTRODUCTION

The first large scale land cover mapping of islands in the Commonwealth of the Northern Mariana Islands was undertaken by the Forest Service in the 1980s (Falanruw *et al.* 1989). This mapping effort included the islands of Tinian, Rota, and Saipan but did not include the other islands in the archipelago. These maps were developed using 1976 aerial photographs and site visits during the 1980s (Falanruw *et al*. 1989) and an earlier version of these estimates were included in Hawaiian Argonomics (1985) and Engbring *et al*. (1986). Engbring *et al*. (1986) also mapped the land cover on Aguiguan as part of their bird surveys of the island sometime during the 1980s using 1968 aerial photographs.

In 2006, new land cover maps for Tinian, Rota, and Saipan were developed by the Forest Service. These updated maps were developed using 2.3-meter IKONOS (GeoEye®, Dulles, VA) multispectral data and 0.6-meter Quickbird (DigitalGlobe®, Longmount, CA) pan-sharpened natural-color imagery collected in 2000 or 2001 and fieldwork conducted in 2005 (Liu and Fischer 2006). Unfortunately, the remaining islands in the Commonwealth of the Northern Mariana Islands were not mapped during this effort.

In 2008, the Department of Defense contracted the U.S. Fish and Wildlife Service's Pacific Islands Fish and Wildlife Office to conduct terrestrial and marine surveys on Tinian and Aguiguan. The following report outlines the results of updating the Forest Service's 2006 land cover map of Tinian using recent satellite imagery and the development of a land cover map of the island of Aguiguan.

METHODS

Tinian

We utilized the Forest Service's 2006 land cover map of Tinian as our base map for evaluating habitat changes since this map was completed. A copy of this vegetation map is available online on the Forest Service's national website - http://www.fs.fed.us/r5/spf/fhp/fhm/landcover/islands/index.shtml. This map was created utilizing satellite imagery from 2000 or 2001 and fieldwork conducted in 2005 (see Liu and Fischer 2006 for more detailed information on how this map was created). Because the imagery utilized to develop this map was over five years old, we utilized 2006 DigiGlobe® satellite imagery of the island to note any new clearings or roads. Areas that were recently cleared were delineated directly on the digital image using ArcMap 9.2 (ESRI, Redlands, CA). These areas were then reclassified in the Forest Service vegetation cover layer using ArcTools (ESRI, Redlands, CA) and new acreages for each cover type were calculated using XTools Pro 5.2.0 (Data East, LLC, Novosibirsk, Russia).

Due to the different land cover categories used in the Forest Service's recent assessment of Tinian (13 categories) and their assessment in the 1980s (16 categories), it was not possible to assess overall land cover changes between the two time periods. However, Engbring *et al.* (1986) reported land cover estimates for Tinian in similar categories to those in the 2006 Forest Service assessment. These estimates were based on an early version of the land cover maps presented in Falanruw *et al.* (1989). Unfortunately, the methods utilized to develop both of these land cover estimates were not the same and so direct comparisons of acreages would not be appropriate. Therefore, we compared percent land cover in each of the land cover categories provided by Engbring *et al.* (1986) for the two time periods. As the 2006 Forest Service estimate included more land cover categories then those provided by Engbring *et al.* (10 categories) we combined several categories. Specifically, agroforest-coconut was combined with agroforest, casuarina (*Casuarina equisetifolia*) thicket was combined with mixed introduced forest for the secondary forest category, and urban vegetation was combined with urban.

Aguiguan

The land cover map of Aguiguan was produced using 0.6-meter resolution, 2001 QuickBird imagery as the primary source. Areas under cloud cover were assessed using 2006 imagery of the island in GoogleEarth (Google, Inc., Mountain View, CA). Six land cover types were delineated: native forest, secondary forest, tangantangan (*Leucaena leucocephala*) thicket, open field, coastal scrub, and non-vegetated. These categories were selected because they were identifiable and corresponded with important bird habitats. Native forest consisted of primarily native trees growing on limestone substrate. Important components of the forest included *Pisonia grandis*, *Cynometra ramiflora*, *Erythrina variegata* and *Guamia mariannae*. Secondary forest consisted primarily of forest dominated by *Delonix regia*, *Acacia confusa*, *Pithecellobium dulce*, and *Casuarina equisetifolia* trees in the canopy. Tangantangan thicket is a type of secondary forest almost exclusively dominated by *Leucaena leucocephala*. Open fields were dominated primarily by introduced *Lantana camara* but patches of *Chromolaena odorata* and *Miscanthus* spp. were also found in these areas. Coastal scrub included low scrubby species like *Pemphis acidula* and non-vegetated areas were primarily bare rock areas along the coast. Land cover types were differentiated based on differences in tone, texture, pattern, and color and were delineated directly on the digital image using ArcMap 9.2. Aerial photographs of Aguiguan from 1948 (black and white, 1:24,000), 1968 (black and white, 1:20,000), and 1994 (color, 1:20,000) were also utilized to assist with delineating secondary forest and limestone forest habitats. All images were registered using a second-order polynomial transformation with at least seven ground control points per photograph. A draft land cover map was ground-truthed in June and August 2008 and any changes were incorporated in the final land cover map.

RESULTS AND DISCUSSION

Tinian

Five sites on the island were identified as having been cleared or modified since the Forest Service's vegetation assessment of the island between 2001 and 2006 (Figures 1 and 2). All of these sites were previously disturbed and consisted of open fields, secondary forest, tangantangan thickets, and mowed or maintained areas (Table 1). Their clearing and modification resulted in their conversion to urban land cover (Table 1).

All modifications were incorporated into the Forest Service's 2006 land cover map and the total acreages for each land cover type were calculated (Table 2). Tangantangan thickets were the dominate cover type (34 percent of the island) followed by mixed secondary forest (27 percent) and other shrub and grass (19 percent). Native limestone forest only made up 5 percent of the land cover while urban areas (urban and urban vegetation) made up approximately 7 percent of the island.

When the recent land cover estimates were compared to those developed in the 1980s several changes were noted. First, overall coverage of open fields decreased while coverage of secondary forest increased (Table 2). This may be a result of succession over the last two decades as open areas are abandoned and claimed by secondary forest. A decrease in tangantangan was also found in addition to an increase in urban land cover (Table 2). The increase in urban cover is likely the result of increased development on the island (including homesteads, casinos, and the airport expansion) and the decline in tangantangan and open fields may also be associated with this development.

Aguiguan

The dominate land cover types on Aguiguan ranked from highest to lowest were native forest, open field, secondary forest, tangantangan thicket, no vegetation/barren, and coastal scrub (Table 3, Figure 3). Aguiguan is currently uninhabited, therefore urban and agricultural field cover types were not recorded. When compared to the land cover estimates by Engbring *et al.* (1986) for the 1982 surveys, the percentage of the island in open field decreased while the percentage in secondary forest and tangantangan increased (Table 3). This shows a transition from open field habitats to secondary forest type habitats over the two survey periods. Interestingly, tangantangan was not included in the landcover for 1982 despite it being a common landcover type on the neighboring islands of Tinian and Saipan (Engbring *et al.* 1986). Engbring *et al.* (1986) report a few small patches that were less than 5 hectares in size. Apparently the larger patches currently found on Aguiguan grew sometime during the last two decades. Based on the present state of extensive browsing of goats (*Capra hircus*) in this habitat type (Figure 4), the growth of tangantangan may have occurred when goat populations were suppressed between 1989 and 1990 (Rice 1991). An assessment of some areas currently containing tangantangan thickets with recently acquired aerial photographs of the island appears to support this theory (Figure 5). The emergence of tangantangan was observed on Sarigan Island after the eradication of goats (C. Kessler, USFWS, pers. comm.) and supports the theory that goats suppress this type of tree.

REFERENCES

Engbring, J., F. Ramsey, and V. J. Wildman. 1986. Micronesian Forest Bird Survey, 1982: Saipan, Tinian, Agiguan, and Rota. U.S. Fish and Wildlife Service, Honolulu, Hawaii. 143 pp.

Falanruw, M.C., T.G. Cole, and A.H. Ambacker. 1989. Vegetation survey of Rota, Tinian, and Saipan, Commonwealth of the Northern Mariana Islands. USDA Forest Service Resource Bulletin PSW-27. 11 pp.

Hawaiian Argonomics, Inc. 1985. Final report for flora and fauna survey of Tinian, Northern Mariana Islands, Volume I. Report to the U.S. Navy (Contract number N62742-84-C-0141), 13 December 1985. Honolulu, HI. 253 pp.

Liu, Z. and L. Fischer. 2006. Commonwealth of the Northern Mariana Islands Vegetation Mapping Using Very High Spatial Resolution Imagery: Methodology. U.S. Department of Agriculture, Forest Service, Pacific Southwest Region, Forest Health Protection, McClellan, California. 14 pp.

Rice, C.G. 1991. Goat removal from Aguijan Island: lessons for future efforts. Western Section of the Wildlife Society 27:42-46.

Table 1. Approximate land cover changes (in hectares) in six sites on Tinian that were cleared after the U.S. Forest Service's 2006 land cover assessment of Tinian. See Figures 1 and 2 for locations of sites.

Site	Land Cover Classification	Secondary Forest	Tangantangan	Open Field	Urban	Total
Airport	Forest Service	48	20	50	53	171
	Update	0	0	0	171	171
Voice of America	Forest Service	0	8	7	1	16
	Update	0	0	0	16	16
New Casino	Forest Service	4	0	1	0	5
	Update	0	0	0	5	5
Quarry	Forest Service	0	1	0	1	2
	Update	0	0	0	2	2
Sports Track	Forest Service	1	0	2	0	3
	Update	0	0	0	3	3
Old Roads	Forest Service	11	6	1	1	19
	Update	0	0	0	19	19

Table 2. Tinian - Acreage (hectares), percent cover, and change in percent cover of ten land cover types based on a 2006 Forest Service assessment, an update of the Forest Service assessment, and estimates by Engbring *et al.* (1986) for the 1980s. Change in land cover was the difference between the Engbring *et al.* estimates and the update of the Forest Service assessment.

Classification	Engbring *et al.*	Forest Service	Update	Percent Change
Native Forest	490 (4.9%)	549 (5.4%)	549 (5.4%)	+ 0.6%
Secondary Forest	1927 (19.2%)	2980 (29.5%)	2916 (28.8%)	+ 10.3%
Tangantangan	3852 (38.3%)	3453 (34.1%)	3417 (33.8%)	- 4.5%
Agroforest	0 (0.0%)	40 (0.4%)	40 (0.4%)	+ 0.4%
Open Field	3107 (30.9%)	2011 (19.9%)	1950 (19.3%)	- 11.6%
Cultivated	190 (1.9%)	134 (1.3%)	134 (1.3%)	- 0.6%
Strand	356 (3.5%)	223 (2.2%)	223 (2.2%)	- 4.5%
Urban	78 (0.8%)	616 (6.1%)	776 (7.7%)	+ 6.9%
Wetland	26 (0.15%)	26 (0.3%)	26 (0.3%)	+ 0.1%
Bare	33 (0.3%)	81 (0.8%)	81 (0.8%)	+0.5%
Total	10,048 (100%)	10,113 (100%)	10,113 (100%)	0.0%

Table 3. Aguiguan - Acreage (hectares), percent cover, and change in percent cover of six land cover types based on a 2008 assessment and estimates by Engbring *et al.* (1986) for the 1980s. Change in land cover was the difference between the Engbring *et al.* estimates and the recent estimates.

Classification	Engbring *et al.*	2008	Percent Change
Native Forest	281 (47%)	340 (49%)	+ 2%
Secondary Forest	21 (4%)	95 (14%)	+ 10%
Tangantangan	0 (0%)	44 (6%)	+ 6%
Open Field	256 (43%)	158 (23%)	- 20%
Coastal Scrub	15 (3%)	28 (4%)	+ 1%
Bare	23 (4%)	34 (5%)	+ 1%
Total	596 (100%)	699 (100%)	0%

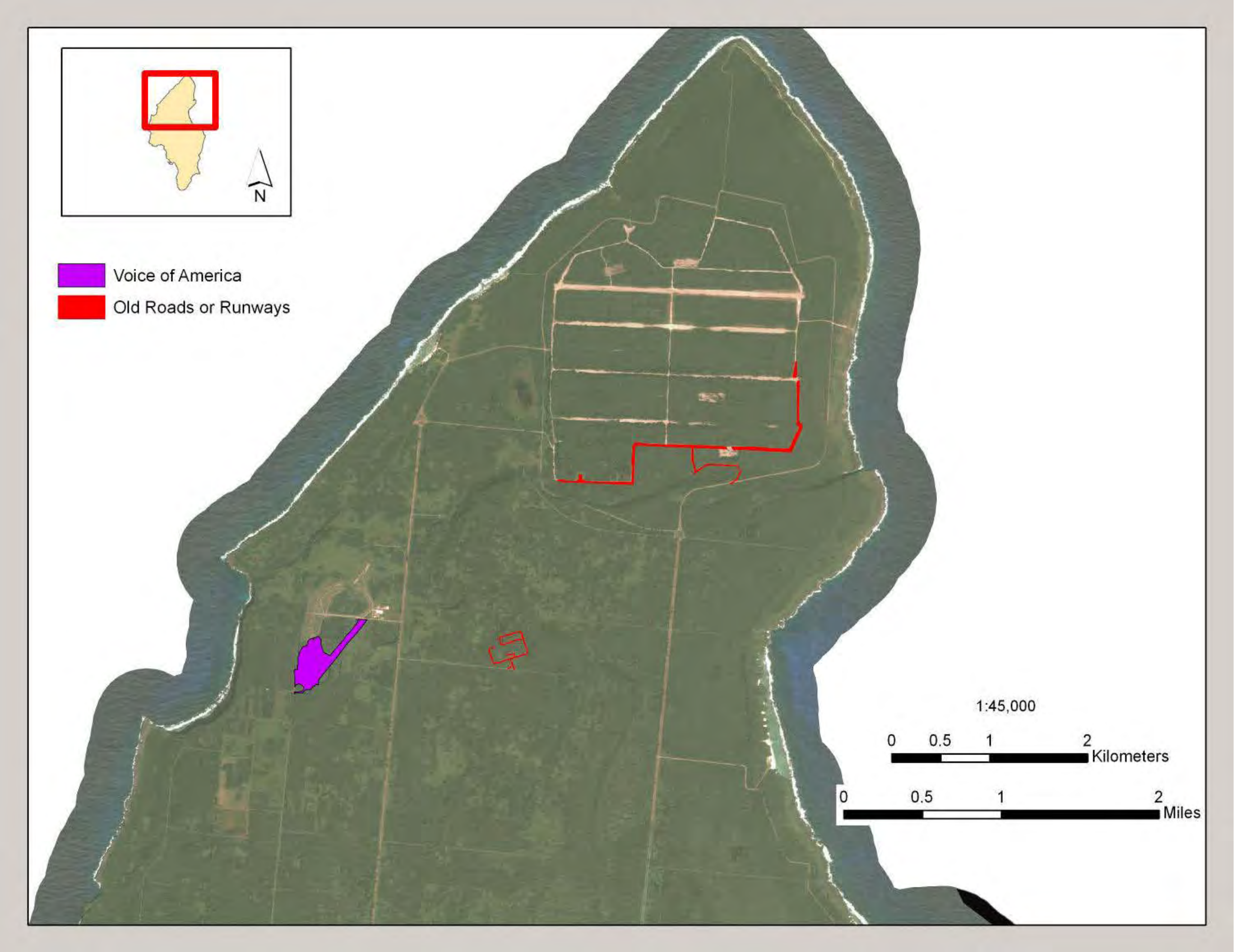


Figure 1. Locations of areas cleared in northern Tinian after the Forest Service's 2006 land cover assessment.

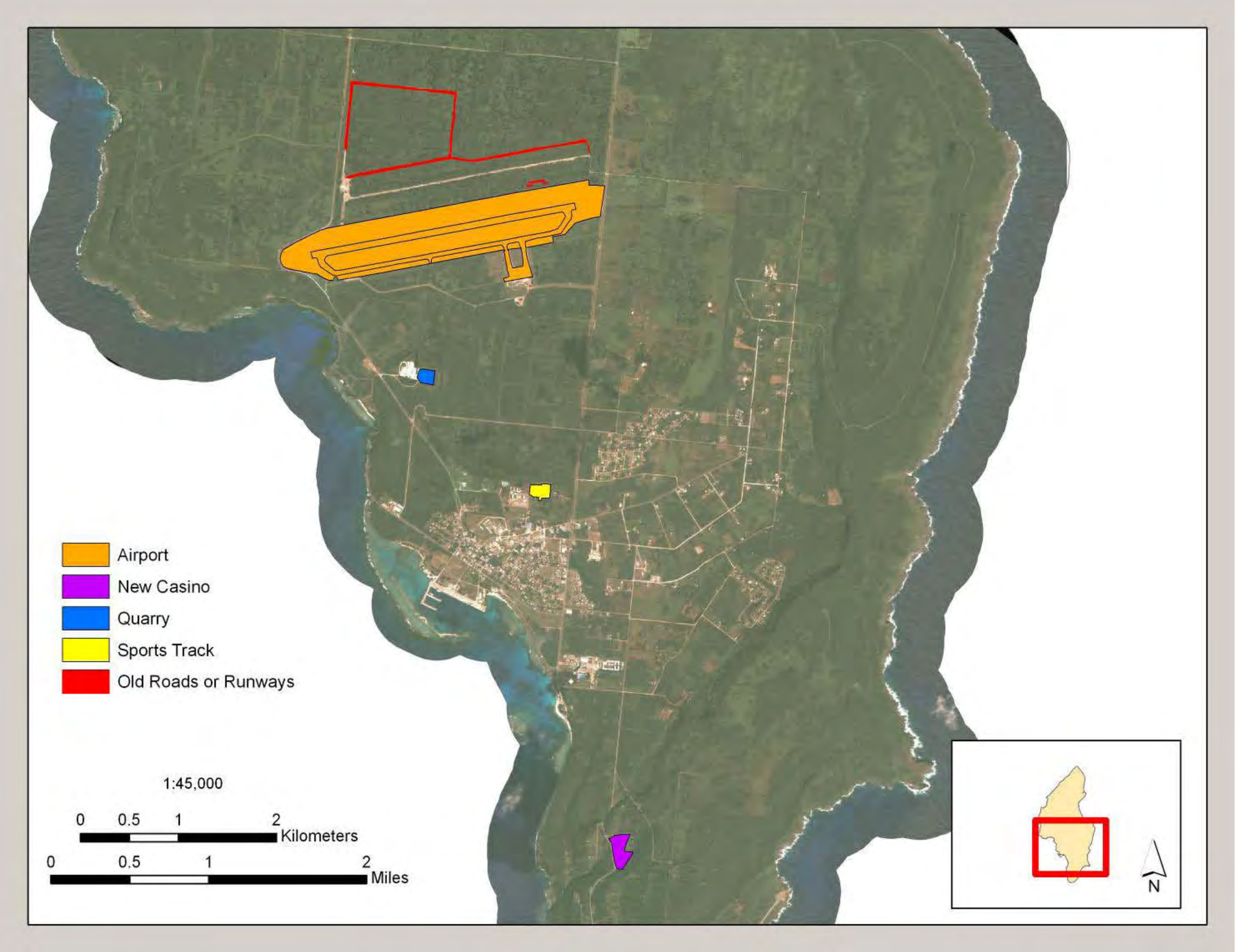


Figure 2. Locations of areas cleared in southern Tinian after the Forest Service's 2006 land cover assessment.

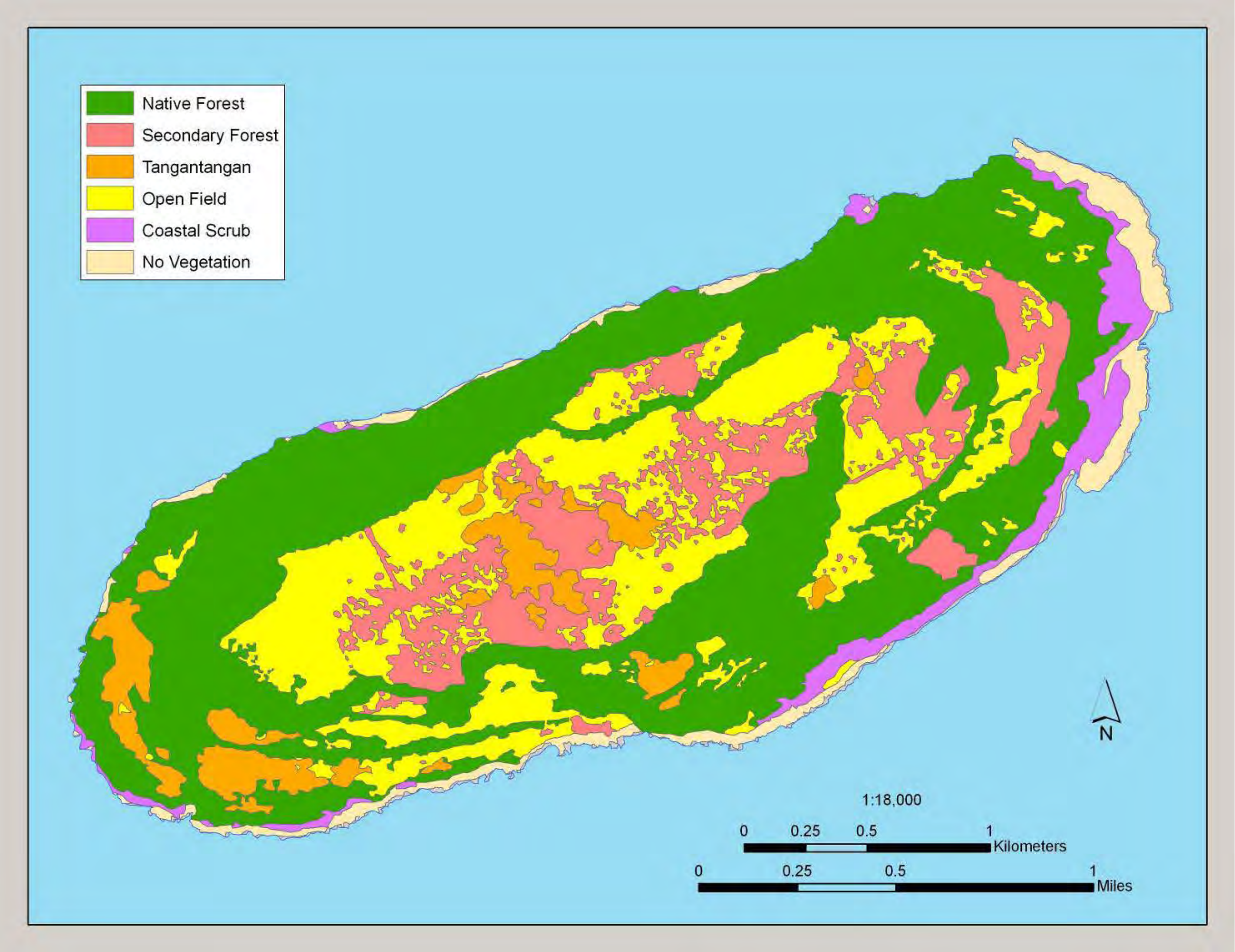


Figure 3. Land cover types on the island of Aguiguan, 2008.

Figure 4. A goat browsing in the understory of a tangantangan thicket. Note the lack of understory. Photo by Aaron Nadig, USFWS, August 2008.

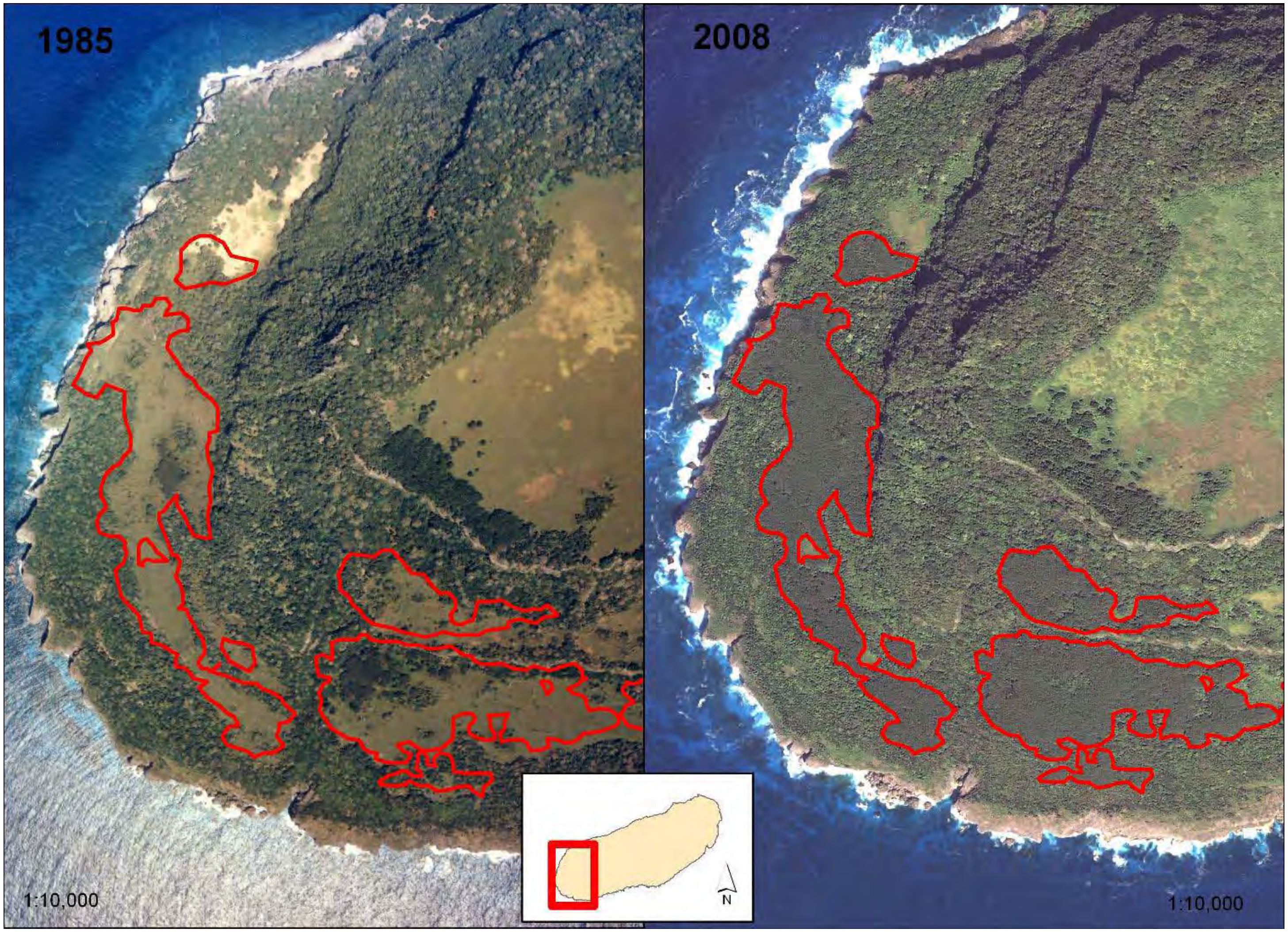


Figure 5. Approximate location of tangantangan thickets in 2008 (outlined in red) that were open fields in 1985 in the southwestern section of Aguiguan.

2.2 INVERTEBRATE SURVEYS

2.2.1 GENERAL INSECT SURVEYS ON TINIAN AND AGUIGUAN

Insect Sampling. Photo by Curt Kessler

Paper Wasps. Photo by Adonia Henry

Hypolimnas anomala.
Photo by Adonia Henry.

Prepared by: Michael Richardson, USFWS, Honolulu, Hawaii and Stephan Lee and Cory Campora, US Navy, NAVFAC Pacific, Pearl Harbor, Hawaii.

INTRODUCTION

A general entomological survey was conducted on the island of Tinian, Commonwealth of the Northern Mariana Islands (CNMI) from August 6-15, 2008; this survey included one day of survey work on the island of Aguiguan on August 12, 2008. In addition to generally surveying predominantly native limestone forest areas on Tinian and Aguiguan for native and nonnative arthropod species, surveys to determine the presence or absence of two butterfly species, *Vagrans egistina* and *Hypolimnas octocula mariannensis* (Lepidoptera: Nymphalidae) and their documented host plants were conducted. The primary efforts of the general entomological surveys involved a focus on the insect ecology within the native forest areas including those possibly associated with the two candidate butterflies. In particular, the status of insect species which may threaten the butterflies' host plant species or the butterflies directly through predation or parasitism was assessed. In addition, survey efforts were focused on ascertaining the presence/absence of invasive species of medical and socio-economic importance (i.e., mosquitoes, ants, termite, etc.), and additionally, aquatic species which may potentially be impacted within the project area (i.e., leased military training areas).

Prior Arthropod Surveys

Based on a review of available literature, very few entomological surveys have been conducted on the island of Tinian. The most comprehensive report found was delivered to the United States Navy (Contract N62742-84-C-0141) from Hawaiian Agronomics (International), Inc. in December, 1985. Hawaiian Agronomics spent a total of seventeen days from November 1984 to November 1985 over the course of four separate visits, collecting insects (including Lepidoptera) utilizing various methods of collection and accounting for temporal differences. The purpose of this study was not outlined in the report and one may only deduce that the intent was to produce a general status report for arthropods on the island at the request of the Navy. Hawaiian Agronomics spent a total of 355 field hours during their study, of which 102 hours were spent visiting coastal, mixed, and scrub limestone forest. Approximately 250 field hours were spent surveying a wide variety of sites across the island including San Jose, village, farms, tangantangan forest, wetlands and ponds, pastures, and even 10 hours at the primary dump (landfill site).

In their report, Hawaiian Agronomics provided a general discussion of their efforts and results by general habitat type, (i.e., wetlands, tangantangan forest, etc.), but did not describe survey results within specific localities. The overall conclusion of the report was a high abundance, yet low diversity of insects on Tinian. A single paragraph described results of efforts surveying within the upper elevation, mixed limestone forests. Notably, ants were listed as the most common insect collected, followed by termites. Other insects collected in these areas as described by the report included micro-lepidopterans, leafhoppers, wood-boring and girdling beetles. Within the leaf litter of these mixed limestone forest areas, Hawaiian Agronomics collected mites, centipedes, millipedes, flies, collembolans, and thysanurans.

The 1995 Hawaiian Agronomics report describes encounters with the nonnative wasp species, *Polistes stigma* (Family Vespidae) in some of the disturbed habitat areas surveyed. Specifically, the report describes the nests of this species occurring as close together as 10 feet in proximity within tangantangan and other nonnative secondary forest. However, no account of this species was given for surveys within the limestone forest areas surveyed. Interestingly, the Hawaiian Agronomics report briefly stated that no rare, threatened, or endangered species of insects or arthropods were observed within the areas surveyed; however, the date of the surveys precluded the establishment of Federal candidate arthropod species within the CNMI.

Candidate Butterfly Species

Currently, *Vagrans egistina* and *Hypolimnas octocula mariannensis* (Family Nymphalidae) are listed as candidates for listing under the Endangered Species Act. Both butterfly species are considered rare and have been recorded only on the islands of Saipan, Rota and Guam (Schreiner and Nafus 1996). *Vagrans egistina* was last collected on Guam in the late 1970s and on Rota in the1980s and 1995. *Hypolimnas octocula mariannensis* is historically known from Guam and Saipan and after an intensive survey for this species in 1995, investigators found only 10 populations on the island of Guam and none on the island of Saipan (Schreiner and Nafus 1996). During intensive surveys for these species in November of 2000 within the USFWS Ritidian Point Wildlife Refuge, only adults of *H. octocula mariannensis* were observed (M. Richardson, USFWS, pers. comm. 2008).

During their 1985 survey of Tinian, Hawaiian Agronomics collected 275 lepidopteran specimens comprised of 14 families, 36 species, and 10 unidentified species. Neither *Vagrans egistina* nor *Hypolimnas octocula mariannensis* were among the species collected, although neither species had been previously recorded on Tinian. The report does describe the difficulty encountered in differentiating the several species of adult nymphalid butterflies that were observed flying during their surveys. While neither species have ever been observed on Tinian, it is certainly possible, due to the close proximity of the islands in the Marianas archipelago and with episodic weather events, such as typhoons, that *V. egistina* or *H. octocula mariannensis* may have been distributed to islands outside of their known ranges in the past, including the island of Tinian. This is particularly true for those islands which do currently (including Tinian and Aguiguan) or have in the past, supported their host plants. Although not overtly cryptic, both *V. egistina* and *H. octocula mariannensis* are known to be fast fliers and only a trained biologist is likely to make a positive confirmation of their presence on islands where they have not been previously recorded.

The recorded host plant for *Vagrans egistina* is *Maytenus thompsonii* (Family *Celastraceae*), a small tree/shrub endemic to the Marianas and found primarily in the understory of native limestone forests (Vogt and Williams 2004, Schreiner and Nafus 1996). The recorded host plants for the *Hypolimnas octocula mariannensis* are *Procris pedunculata* and *Elatostema calcareum*, both forest herbs (Family *Urticaceae*) found growing on limestone outcrops in native limestone forest (Schreiner and Nafus 1996).

METHODS AND MATERIALS

Prior to arriving to Tinian for our August 2008 survey efforts, the following primary objectives to be modified as needed or as circumstance required were formulated:

Tinian 1st Priority Goals:

- Survey for the butterflies and host plants within remnant native forest areas on military lands.
- Survey for nonnative insect threats to the butterflies (predators/parasitoids) and to the host plants (true bugs, etc.) on military lands.
- Survey for other possible native insects, including aquatic species such as the odonates on military lands.

Tinian 2nd Priority Goals:

- Survey for the butterflies and host plants within remnant native forest areas on NON-military lands.
- Survey for nonnative insect threats to the butterflies (predators/parasitoids) and to the host plants (true bugs, etc.) on NON-military lands.
- Survey for other possible native insect species on NON-military lands.
- Survey for presence/absence of other important alien arthropod groups including mosquitoes and ants.

Aguiguan (Goat Island) Goals

- Survey for the butterflies and host plants within remnant native forest areas
- Survey for & collect nonnative insect threats to the butterflies (predators/parasitoids) and to the host plants (true bugs, etc.)
- Survey for & collect other possible native insects including aquatic species such as the odonates.
- Survey for presence/absence of & collect other important nonnative arthropod groups including mosquitoes and ants.

The habitat areas selected for surveys were identified using maps and reports prepared by U.S. Fish and Wildlife Service and Department of Defense biologists between June and August 2008. Due to the limited amount of time available for us to conduct the surveys, we decided to spend no more than a portion of each work day surveying along known or recently constructed transects within each of the several identified limestone forest sites. Most sites were visited during the daylight hours, although some sites were visited during the night to check traps or to run a blacklight. Equipment and methods used during surveys of terrestrial areas included sweep netting of vegetation, visual inspections of vegetation, caves, under rocks, and beneath rotting vegetation. Use of baiting and traps were also employed. For aerial insects, the following traps types were utilized: UV light with sheet and water pan, EVS light trap with LED light, BG Sentinel trap with BG-lure and Octenol lure, and a collapsible cone trap with protein bait. Ants were collected in every locality by hand collection, sweeping, aspiration, and with the use of 3”x5” index cards baited with both peanut butter and honey. Aquatic localities were surveyed directly by wading, use of a kayak, sweeping, use of a mosquito dipper, and benthic sampling with a D-frame aquatic net. All collection sites and transects are shown in Figures 1 through 4.

During visual searches for *Vagrans egistina* and *Hypolimnas octocula mariannensis* and their host plants, our own knowledge of the species as well as an information guide produced by USFWS personnel were utilized. The information guide provided photographs of both butterflies and their host plants as well as descriptions of the larval stages of the butterflies. Geographic positioning system (GPS) points were taken of all transects within each locality surveyed. Additionally, to facilitate future monitoring efforts, points were recorded for all host plants located during each survey. A total of 21 host plants sites were located on Tinian and 3 host plant sites were located on Aguiguan during our surveys. Nine host plant sites were located on Department of Defense leased land (see Figure 1 - Japanese Caves Transects North and South and Chiget Cliff) and 15 on CNMI public lands (see Figure 3 - Carolinas Nature Trail and Figure 4 - Aguiguan).

Identification of some specimens was completed by USFWS and Department of the Navy entomology staff; these specimens were submitted to the Bishop Museum, Department of Entomology for cataloguing and permanent storage. The majority of specimens collected were submitted to the Bishop Museum, Department of Entomology for identification, cataloguing, and permanent storage. See Appendix A for a list of all specimens collected during this survey.

RESULTS

A description of the survey efforts within each locality by date is outlined below. Note, within these descriptions, common names are primarily used; please refer to Appendix A for scientific names of all collected specimens.

August 7, 2008 – *Lake Hagoi*

(Inside Military Lease Area)
Coordinates: UTM 55 15.068083 145.625033
Habitat Type: Aquatic/Semi-aquatic and Mixed Introduced/Tangantangan Forest
One entomologist spent approximately 4 hours surveying this site and surrounding vegetation Access to Lake Hagoi was attempted from the road east of the lake. Aerial observation during the flight from Saipan to Tinian indicated the water level was low and that the lake had only two small pockets of open water. Navigation to the open water portions of the lake was extremely difficult due to the dense vegetation that surrounded the area. The water/mud-ooze level within the vegetation gradually deepened toward the center of the lake area, and once the reed portion of the lake area was reached, the water and mud was knee-deep, and passage by foot was no longer possible. A D-frame aquatic net was used to sample the water amidst the reeds and the substrate at the beginning of the reed area of the lake. An adult damselfly, damselfly nymphs, an adult aquatic beetle, aquatic beetle larvae, water boatmen, a spider, and aquatic snail shells were collected. There were many adult damselflies flying amongst the reeds, and they all appeared to be of the same of species. After leaving the lake area and returning to the entry point on the road east of the lake, insects were collected by net along the road. Specimens collected

included an adult dragonfly, two different species of leaf footed bugs, a carpenter bee, and one Nymphalid butterfly species.

Aerial view of Lake Hagoi showing the two small areas of open water.

In the evening a UV light trap was set up by one entomologist at the intersection southeast of the lake. The trap consisted of an ultraviolet bulb set against a white sheet with a pan of soapy water at the bottom. The trap was engaged just prior to sunset and then checked after approximately three hours. The trap was checked again the following morning prior to dismantling. Winged termites (alate form) were immediately attracted to the illuminated sheet and collected by hand. A water strider was attracted to the trap and was collected. The large marine toad, *Bufo marinus*, was also attracted to the trap due to large amount of insects swarming around the light. Upon checking the trap after three hours of operation, the toads had knocked over the pan of water. Specimens were salvaged from spilled water as much as possible. Aquatic species were targeted for collection, therefore water boatmen and other various adult beetles were collected while many of the flies and moths were not. Particularly abundant were what appeared to be koa haole, or tangantangan, moths, *Macaria abydata*.

On the same evening (August 7th), one entomologist set up a separate light trap west of the Lake. An established path that ran north from the road south of Lake Hagoi was used to locate a survey site west of Lake Hagoi. Before sunset (6:45 PM), an EVS light trap with a white LED as a light source and a BG-Sentinel trap with BG-Lure and Octenol lure were set up in the survey area in a shaded area. The BG-Lure mimics chemicals produced by human skin, and the Octenol lure has mosquito pheromone-like properties.

UV Light trap with sheet and water pan.

Koa haole, or tangantangan, moths, *Macaria abydata*, on light trap sheet.

<u>August 7, 2008 – *Mount Laso*</u>
(Inside Military Lease Area)
Coordinates: (Not Available)
Habitat Type: Mixed Native Limestone Forest

This site was visited by one entomologist for approximately 1 hour. An established trail from the shrine at Mount Laso that runs south was used to locate a survey site. The survey site was near the beginning of the trail in a shaded area. An EVS light trap with a white LED as a light source and a BG-Sentinel trap with BG-Lure and Octenol lure were set up in the survey area before sunset.

<u>August 7, 2008 - *Japanese Villiage Ruins*</u>
(Inside Military Lease Area)
Coordinates: (Not Available)
Habitat Type: Nonnative Shrub and Grassland

One entomologist set up an EVS light trap with a white LED as a light source was set up in an area off the road that leads to the Japanese Village Ruins away from direct sunlight. The EVS trap was set up before sunset, between Eighth Avenue and the arch that remains as part of the Japanese Village Ruins.

<u>August 8, 2008 – *Near Japanese Village Ruins*</u>
(Inside Military Lease Area)
Coordinates: (Not Available)
Habitat Type: Nonnative Shrub and Grassland and Mixed Forest and / Mixed Native Limestone Forest

One entomologist attempted to locate a semi-permanent pond that was in the vicinity of the Japanese village ruins. The intent was to collect aquatic insects. Unfortunately, there was no trail to this pond and passage through the grass and vegetation was very slow.

After approximately two hours, the search for the pond was abandoned and insects were collected along the road that leads past the ruins. A variety of butterflies were collected in this area, along with a large species of plant bug. There were also some adult dragonflies in the area, but capture of these fast flying insects was not possible. One of the dragonflies looked very different from the common red colored species – its coloration appeared to be grayish blue with maybe some yellow markings. At the end of this road, a small patch of native forest was found. Within the fringe of this native forest area, subterranean and drywood termites were collected from a living tangantangan tree and a dead coconut palm.

The EVS trap and BG-Sentinel traps at this location were checked. Both traps had failed to capture mosquito specimens, yet they were filled with a large number of fungus gnats. A decision was made to forgo setting subsequent mosquito traps at this location based on trap rate and time constraints.

August 8, 2008 - Lake Hagoi

(Inside Military Lease Area)
Coordinates: UTM 55 15.068083 145.625033
Habitat Type: Mixed Nonnative / Tangantangan Forest
This site was again visited by one entomologist for approximately 2 hours. Trapped specimens were collected from the EVS trap and BG-Sentinel trap that were placed west of the lake. Before sunset (6:44 PM), a blue LED light source was placed in the EVS trap, and the BG-Sentinel trap with BG-Lure and Octenol lure was reset. Before sunset at this location, an entomologist collected mosquitoes off of himself with an aspirator. Mosquito identification is pending authorization to utilize the taxonomic services of Bishop Museum.

Also in the evening the UV light trap was engaged, however on this occasion it was placed just off the road north of the lake area. Nothing new was collected in the trap relative to what was caught the night before. The koa haole, or tangantangan, moth was again present in high numbers.

August 8, 2008 – *Mount Laso*

(Inside Military Lease Area)
Coordinates: (Not Available)
Habitat Type: Mixed Native Limestone Forest
No mosquito specimens were collected from the EVS trap and BG-Sentinel trap, because no mosquitoes were caught in either trap. A decision was made to forgo setting subsequent mosquito traps at this location based on trap rate and time constraints. A malaise trap was set up in an open grassy area adjacent to tangantangan trees at this site.

August 8, 2008 – *Old Japanese Communications Center*

(Inside Military Lease Area)
Coordinates: (Not Available)

Habitat Type: Building
A collapsible Diptera/Hymenoptera cone trap was set up at the Old Japanese Communications Center to trap wasps. Canned chicken was used as protein bait. Subsequent visits to check the traps would reveal foraging ants to be a constant problem here.

August 8, 2008 – *Japanese Air Administration Building and Air Operations Building*
Coordinates: (Not Available)
Habitat Type: Building
Two collapsible Diptera/Hymenoptera cone traps were set up at the Japanese Air Administration Building and one cone trap was set up at the Japanese Air Operations Building to trap wasps. Canned chicken was used as protein bait. Wasps were very common around the Japanese Air Administration Building, but no wasps appeared to be interested in the protein bait. Subsequent visits to check the traps would reveal foraging ants to be a constant problem here.

August 9, 2008 - *Maga Transect*
(Inside Military Lease Area)
Coordinates: UTM 55 15.059444 145.621667
Habitat Type: Mixed Native Limestone Forest
Three entomologists spent the better portion of this day surveying this site along and off the established transect. Movement was slow as we developed an understanding for progressing carefully within the wet understory of the forest and avoiding the dreadful and unexpected sting of the *Polistes stigma* wasps, whose nests found along even the cut transect trail. Insects were sampled at several points by sweeping vegetation and hand collection from vegetation, under stones, and within substrate. Ants of several species were collected with baited cards. Neither adult candidate butterflies nor host plant species were located.

Native limestone forest collecting site on the Maga transect.

A nest of the paper wasp, *Polistes stigma*, in the forest.

August 9, 2008 - *Lake Hagoi*

(Inside Military Lease Area)
Coordinates: UTM 55 15.068083 145.625033
Habitat Type: Mixed Nonnative/Tangantangan Forest
One entomologist revisited established traps at this site for approximately 2 hours. Trapped specimens were collected from the EVS trap and BG-Sentinel trap before noon. Before sunset (6:44 PM), an EVS light trap with a green LED as a light source and a BG-Sentinel trap with BG-Lure and Octenol lure were set up in the survey area. Before sunset, mosquitoes were self-collected off himself with an aspirator. Mosquito identification is pending authorization to utilize the taxonomic services of Bishop Museum.

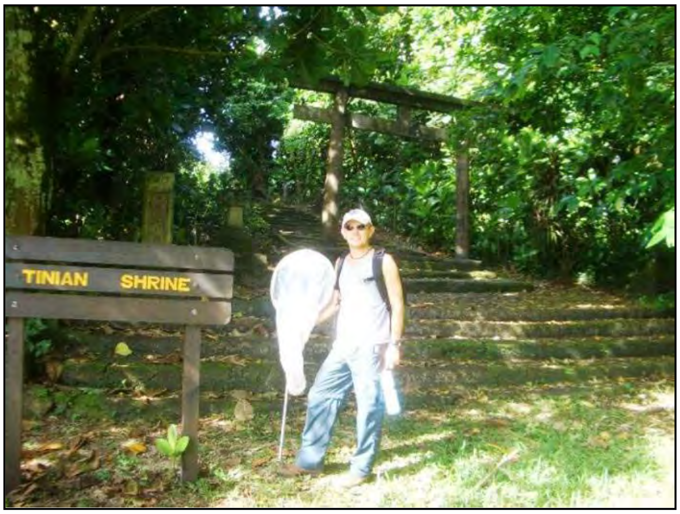
August 10, 2008 - *Carolinas Nature Trail (CNT) Area*

(Outside Military Lease Area)
Coordinates: UTM 55 14.939233 145.633783
Habitat Type: Mixed Native Limestone Forest
Based upon discussions with fellow biologists regarding known densities of both host plant species at this site, we (three entomologists) decided to visit this site to get a better feel for the distribution and appearances of the host plants' various life stages, as well as to sample insects from the host plants. A very large cluster of *Elatostema calcareum* was located in this area, growing along the numerous limestone outcroppings and within large corridor-like crevices in the stone. Very few insects were located on the plants themselves, most notably spittlebugs (Family Aphrophoridae) and ants of several species. Insects were sampled from the host plants and surrounding vegetation by sweep-netting, hand collection, and with the use of baited cards. The *E. calcareum* in various stages including with fruiting bodies, appeared quite healthy with no obvious signs of herbivory by either ungulates or insects. Some leaves were collected and bagged and two days letter yielded 2 small weevils (Family Curculionidae). Neither species of butterfly was located in either the adult or larval stage, despite the inspection of 30+ plants. As it turned out, the cluster of *E. calcareum* in this area was the largest in density and size that we would locate during the week long survey of all sites.

Several individual *Maytenus thompsonii* plants were also located within this site, but we did not have the opportunity to inspect them very closely prior to a very heavy downpour which hastened our retreat.

***Elatostema calcareum* in the Carolinas Nature Trail (CNT) area.**

August 10, 2008 - *Tinian Shinto Shrine (Carolinas Heights)*
(Outside Military Lease Area)
Coordinates: (Not Available)
Habitat Type: Mixed Native Limestone Forest
After a heavy rain shower mid-day, this mixed native limestone forest with some nonnative secondary forest components was surveyed by three entomologists for approximately two hours in the afternoon on this date. Neither adult candidate butterflies nor host plant species were located. The vegetation in this area was very dense and movement was slow due to lack of an established transect in this area. Ants of several species were very high in density within this locality as were termites of several species. One adult scorpion (*Liocheles australasiae*) and a colony of drywood termites were located within a rotten log. Several large mosquito larvae (identification pending), were collected in rain-filled portions of the old Shinto Shrine in this area.



Entomologist Mike Richardson at the Tinian Shinto Shrine

Liocheles australasiae

August 10, 2008 - *Lake Hagoi*
(Inside Military Lease Area)
Coordinates: UTM 55 15.068083 145.625033
Habitat Type: Mixed Nonnative/Tangantangan Forest
One entomologist revisited established traps in this area for approximately 1 hour. Trapped specimens were collected from the EVS trap and BG-Sentinel trap before noon. Mosquito identification is pending authorization to utilize the taxonomic services of Bishop Museum.

August 11, 2008 - *Makpo Wells (Tinian Pumphouse Area)*
(Outside Military Lease Area)
Coordinates: (Not Available)
Habitat Type: Nonnative marshland
Three entomologists visited the marshy terrain immediately surrounding the Tinian pumphouse on the morning of this date. No standing water could be located despite

recent heavy rains the day prior. No odonates were located. Several species of ants and one nonnative wasp species, (Family Vespidae, *Delta* sp.) were collected.

A wasp (*Delta* sp.) on its mud nest.

<u>August 11, 2008 - *Laderan Chiget Cliffs Transect (CC)*</u>

(Inside Military Lease Area)
Coordinates: UTM 55 15.060816 145.647866
Habitat Type: Mixed Native Limestone Forest

Three entomologists spent approximately 6 hours was spent walking this entire transect beginning from where the ridge intersects the paved road to the west until it ends at the Chiget beach area. Several individual *Maytenus thompsonii* plants were located and recorded with our GPS equipment. Only a single stand of *Elatostema calcareum* was located near the beach end of the transect. This stand was closely inspected for candidate larvae and other insects, but none were located. All individual plants of both host species appeared healthy and free of any appearance of herbivory. Very few insects were collected within surrounding native vegetation along this transect during this date, due to the amount of time needed to traverse the area and also because of our intention to return and conduct more sampling (which unfortunately did not occur due to time constraints). No adults of either candidate butterfly species were observed during this date. The densities of *Polistes stigma* nests along this transect were very high, in some areas two or more nests occurred within just five feet of each other. The nest height location varied from 1.5 feet from the ground to over 30 feet high from the ground. The single largest nest in terms of number of individuals was observed in this area, with a total of 40+ wasps. One single, large adult female coconut crab was observed within this area. Several mosquitoes were aspirated off the surveying entomologists.

Limestone cliffs that are skirted by the Laderan Chiget transect.

Entomologist Stephan Lee inspects *E. Calcareum* on the Laderan Chiget transect.

August 11, 2008 - *Korean Memorial & Saint Lourdes Shrine Cave (San Jose Village)*
(Outside Military Lease Area)
Coordinates: (Not Available)
Habitat Type: Mostly Nonnative Secondary Forest w/some Native Limestone Forest Components
Two entomologists visited the Saint Lourdes Shrine Cave and adjacent forest, located on the western edge of the San Jose village in the evening of this date. Unfortunately, a decent downpour prevented much of the work we had intended in the forest and we spent about one hour collecting insects and blacklighting within the cave itself and at the mouth of the cave. No insects were attracted to the blacklight and no scorpions were located.


Entrance to St Lourdes Shrine Cave


St Lourdes Shrine Cave interior.

August 11, 2008 – *Mount Laso*
(Inside Military Lease Area)
Coordinates: (Not Available)
Habitat Type: Mixed Native Limestone Forest
One entomologist revisited established traps within this area for approximately 1 hour. Insect specimens were collected from the malaise trap that was set up on August 8, 2008. Mosquitoes were separated from the insects collected.

August 11, 2008 – *Old Japanese Communications Center*
(Inside Military Lease Area)
Coordinates: (Not Available)
Habitat Type: Building
One entomologist revisited established traps within this area for approximately 1 hour. No wasps were trapped in the collapsible Diptera/Hymenoptera cone traps that were set up on August 8, 2008.

August 11, 2008 – *Japanese Air Administration Building and Air Operations Building*
(Inside the Military Lease Area)
Coordinates: (Not Available)
Habitat Type: Building
No wasps were trapped in the collapsible Diptera/Hymenoptera cone traps that were set up on August 8, 2008. No leftover bait was found at the Japanese Air Administration building.

August 12, 2008 - *Aguiguan Lower West Limestone Forest*

(Outside Military Lease Area)
Coordinates: (Not Available)
Habitat Type: Mixed Native Limestone Forest

One entomologist spent approximately two hours surveying the northeastern lower tier of mixed limestone forest. Neither adult candidate butterflies nor host plant species were located in this area. Arthropod samples were taken by net-sweeping surrounding native vegetation, use of baited cards, and hand sampling vegetation, under rocks, and in substrate. Individual and small herds of goats were observed during this time. Vegetation along this entire transect showed very obvious signs of ungulate herbivory, presumably by goats. *Polistes stigma* nest density was noticeably lower in this area than that observed on Tinian.

August 12, 2008 – *Aguiguan Cisterns Near Base Camp*

(Outside Military Lease Area)
Coordinates: UTM 55 14.851366 145.556800
Habitat Type: Aquatic/Semi-aquatic and Mixed Nonnative Forest

Two entomologists surveyed the aquatic environment inside of World War II era cisterns constructed by the Japanese during their occupation of island from approximately 1936 to 1945. There are numerous Japanese-era sites on the island with cisterns that contain water, but due to time constraints, the entomologists where only able to survey cisterns located in the center of the island, near the base camp. This area, which had once been a Japanese plantation village, had approximately 28 cisterns of various sizes and shapes (circular or rectangular with variable dimensions) and in various states of decay. Many of the cisterns were full of water, and some were empty or collapsed. Six of the cisterns containing water were sampled using a D-frame aquatic net and a mosquito dipper. Cistern biota sampled consisted of dragonfly nymphs, veliid bugs or broad shouldered water striders, chironomid or midgefly larvae, mosquito larvae, pleid bugs or pygmy backswimmers, annelid worms, a tetragnathid spider, and an adult beetle of unknown identification.

Some terrestrial insects were also collected in areas immediately adjacent to the cisterns. Both subterranean and drywood termites were collected from woody debris on the ground. A beetle larvae and an adult passalid beetle were also collected from wood debris (separate pieces of wood). Large millipedes were common in this area and were seen crawling within the substrate. Six of these millipedes were collected and were tentatively identified as *Trigoniulus lumbricinus*. Additionally, adult dragonflies (red species) were commonly seen flying through the cistern area. One of these dragonflies was caught in the base camp area, and was later identified by the Bishop Museum as *Trapezostigma transmarina*.

Approximately 1 hour was also spent surveying the mostly nonnative vegetation immediately surrounding the base camp area. Arthropod samples were taken by net-sweeping surrounding vegetation, use of baited cards, and hand sampling vegetation, under rocks, and in substrate. One immature scorpion (*Liocheles australasiae*) was located in the vegetation near the base camp.

Partially open cistern with vegetation.

Entomologist Stephan Lee sorting through a sample from open cisterns.

Collapsing cisterns with no water.

Intact cisterns.

<u>August 12, 2008 - *Aguiguan Upper West Limestone Forest*</u>
(Outside Military Lease Area)
Coordinates: UTM 55 14.854216 145.550683
Habitat Type: Mixed Native Limestone Forest
Three entomologists spent approximately two and a half hours surveying the northeastern upper tier of mixed native limestone forest. No adult candidate butterflies or *Elatostema calcareum* host plants were observed, but several individual *Maytenus thompsonii* plants were located and inspected carefully. These were the only flowering individuals of this species that we located during the week long survey. Most individuals showed signs of herbivory by goats along the base of the plant or on lower stems. Very few insects other than ant species and a couple of beetle species were collected from these host plants. Note, one *M. thompsonii* that was located within this area showed signs of insect feeding (chewing) on a large majority of leaves (see photos), but this feeding was not indicative of lepidopteran chewing, more likely chewing by beetles. Other arthropod samples were taken by net-sweeping surrounding native vegetation, use of baited cards, and hand sampling vegetation, under rocks, and in substrate.

A goat browsed *Maytenus thompsonii* on Aguiguan.

Signs of insect feeding on leaves of *Maytenus thompsonii*.

August 12, 2008 - *Aguiguan Lower Rock Shelf*

(Outside Military Lease Area)
Coordinates: (Not Available)
Habitat Type: Exposed Limestone Rock

Two entomologists spent approximately 15 minutes surveying small pools of water in the lower limestone shelf near the boat landing site. Drangonfly nymphs and mosquito larvae were found in the pools. A quick taste test suggested that the water was perhaps partly saline. Unfortunately more time could not be spent at this site due to an impending emergency evacuation of an injured biologist.

Entomologist Cory Campora examines the rock pools near the boat landing site on Aguiguan.

August 13, 2008 - *Japanese Defensive Caves Trail South (Laderan Lasu, Mt. Lasu)*

(Inside Military Lease Area)
Coordinates: UTM 55 15.039333 145.636833
Habitat Type: Mixed Native Limestone Forest

Three entomologist spent approximately six hours surveying the established southbound transect in this area. Several clusters of *Elatostema calcareum* host plants were located and recorded with our GPS unit. No plants contained phytophagous insects and none exhibited signs of any herbivory. A few scatter individual *Maytenus thompsonii* plants were also located and recorded with our GPS. None exhibited signs of any herbivory. Arthropod samples were taken by net-sweeping surrounding native vegetation, use of baited cards, and hand sampling vegetation, under rocks, and in substrate. *Polistes stigma* nests were very high in density along this transect. Several mosquitoes were aspirated off the surveying entomologists.

Elatostema calcareum **growing off the lower limestone cliff on the South Japanese defensive caves trail.**

August 14, 2008 - *Japanese Defensive Caves Trail North (Laderan Mangpang, Mt. Lasu)*

(Inside Military Lease Area)
Coordinates: UTM 55 15.042333 145.634000
Habitat Type: Mixed Native Limestone Forest

Three entomologist spent approximately six hours surveying the established southbound transect in this area. No *Elatostema calcareum* host plants were located; however, a few scatter individual *Maytenus thompsonii* plants were located and recorded with our GPS. None exhibited signs of any herbivory. Arthropod samples were taken by net-sweeping surrounding native vegetation, use of baited cards, and hand sampling vegetation, under rocks, and in substrate. Several of the deeper WWII Japanese defensive caves were examined closely with both a LED headlamp and separately with a blacklight with no scorpions or other arthropods observed. One cave did contain a mid-sized juvenile coconut crab. *Polistes stigma* nests were very high in density along this transect. Densities of both *Hypolimnas bolina* and *Hypolimnas anomala* (Family *Nymphalidae*) were quite high along the entire length of this transect.

August 15, 2008 - *Maga Transect*

(Inside Military Lease Area)
Coordinates: UTM 55 15.059444 145.621667
Habitat Type: Mixed Native Limestone Forest

On the final day of field work, one entomologist resurveyed this site for approximately three hours for the purpose of confirming absence of host plant species as noted on the August 9, 2008 visit to this site. No host plants were located despite careful searching along both the upper cliff edge, below the cliffs, and within the forest below the transect itself. Visual searches for the adult candidate butterflies were unsuccessful and no arthropods were collected on this date.

August 15, 2008 - *NKK Railroad Shrine Trailhead*

(Inside Military Lease Area)
Coordinates: (Not Available)
Habitat Type: Nonnative Secondary Forest

With a small amount of remaining time on the final day of field work, this nonnative, secondary forest site was surveyed for approximately 2 hours by one entomologist. Interestingly, but perhaps not surprisingly, insect diversity was much higher than that observed within the mixed limestone forest sites surveyed. Several specimens were collected including many insect and spider species not observed within the mixed limestone forest sites. Within the two acre area surveyed at this site, *Polistes stigma* nests were infrequently encountered and appeared to be less dense than that observed within the mixed limestone forest sites.

***NKK Railroad* Shrine Trailhead.**

August 15, 2008 – *Lake Hagoi*

(Inside Military Lease Area)
Coordinates: UTM 55 15.068083 145.625033
Habitat Type: Aquatic/Semi-aquatic Marshland

An entomologist returned to Lake Hagoi for approximately three hours with a kayak for better access to the open water portions of the lake. Passage through the vegetation was again difficult, but once the reeds were reached, movement over the water and mud was much easier. Unfortunately the height of the reeds prevented a clear view of where the open water portions of the lake were located. Due to this navigational difficulty and time constraints, the large open water portions of the lake were not visited. Small pockets of open water within the reeds where sampled however. These areas were teeming with water boatmen and water striders. Samples were taken with a D-frame aquatic, but no new benthic and aquatic organisms were collected compared to what was caught on August 7th. Many dragonflies were seen flying over the reeds, mostly the large red species, but a smaller species was also common. Damselflies were also common, but again only one species was apparent. This was the same species that was seen on August 7th. A sweep net was used to collect some more of the damselflies and two tetragnathid spiders were also collected.

Kayak being pushed through reeds in Lake Hagoi.

Path through reeds in Lake Hagoi created by Kavak.

DISCUSSION

The primary objectives of this week long survey were to determine the presence or absence of candidate species, *Vagrans egistina* and *Hypolimnas octocula. mariannensis*, to survey for and locate the host plant species, to inspect and survey the host plant habitat for potential insect threats to either the candidate butterflies or the host plants, and finally, to gather some information regarding the presence or absence of particular arthropod groups on the islands of Tinian and Augean, including insects of socie-economic importance (i.e., mosquito vector species, ants, and termites) and others including aquatic fauna such as odonates. The major limitation of this survey effort was the short duration of our visit which did not allow us to gather as much detailed information or to cover as many areas as we would have preferred. Heavy downpours affected our ability to survey on some days, but were not a significant factor. Surprisingly, the insidious *Polistes stigma* wasp was a substantial deterrent to moving quickly in most forested areas we surveyed. Coupled with the tricky terrain, humidity and high temperatures, and dense vegetation, the extent of the area surveyed at each site was less than we had hoped and intended to accomplish.

Candidate Butterflies (*Vagrans egistina* and *Hypolimnas octocula mariannensis*)

Neither adults nor larvae of either candidate butterfly species were observed during our survey efforts. All host plants (both *Elatostema calcareum* and *Maytenus thompsonii*) that we located appeared healthy and flush with new vegetation for the most part, and all plants were inspected to the best of our abilities. Neither host plant species at any of the sites we surveyed exhibited evidence of lepidopteran feeding. Conditions on both islands were fairly moist with several rain showers and a couple of downpours occurring during our week long survey effort, so it was likely not too dry for the butterflies to have been present. Many of the life stages including larvae, pupae, and adults of the several nymphalid butterflies known to be present on Tinian were frequently observed during our surveys of the limestone forest areas. The species most commonly and frequently observed were *Hypolimnas anomala* and *H. bolina*.

Both the number and the density of invertebrates collected on the two host plant species located on Tinian (only *Maytenus thompsonii* was located on the one day of surveying on Aguiguan), was surprisingly low. The most commonly collected insect on both host plant species were ants. Very few phytophagous insects were observed. The following groups of insects were collected on *Elatostema calcareum* in very low numbers: weevils (Family Curculionidae); tephritid flies (Family Tephritidae); spittle bugs (Family Aphrophoridae); mealybugs (Family Pseudococcidae); plant hoppers (Family Fulgoroidea); and one katydid (Family Tettigoniidae). No plants except one individual of *M. thompsonii* located on Aguiguan showed signs of insect herbivory (apparently by beetles). Only on the island on Aguiguan did we observe signs of ungulate herbivory, apparently goat browsing on *M. thompsonii*.

Polistes stigma and Other Wasps

Polistes stigma is a eusocial species of wasp belonging to the Vespidae family. They are generalist predators which forage for protein to nourish the larvae that are developing and housed within their paper nests. While this species may forage on nearly any sort of meat including that from human garbage, there is high potential for them to prey upon any invertebrates including butterfly larvae within their foraging area. Unfortunately, captured prey items are masticated (chewed up) when they are brought to the nest prior to being fed to the larvae, so it is difficult to ascertain what is being collected by the wasp. During our surveys of the mixed limestone forest areas, nests of this wasp species were generally observed to be as frequent as 4 nests within a 30 square foot area, occasionally occurring within 5 feet of each other.

Based upon our observations and prior survey work on the other islands within the CNMI, including Guam, we believe that densities of *Polistes stigma* nests on both Tinian and Aguiguan are extremely high and perhaps a significant factor which could preclude the two candidate butterfly species from utilizing the available host plant habitat on both islands. We believe it would be worthwhile to design a study to determine which groups of insects are being preyed upon by this species and to determine an actual measurement of nest density within the mixed limestone forest areas of Tinian and Aguiguan. Certainly, this information would be highly recommended prior to any efforts to augment populations of the two candidate butterflies that are discovered on these islands, or prior to any effort to translocate these species there as part of a larger recovery process. Additionally, it would be useful to understand and compare the density of *P. stigma* nests within habitat occupied by the candidate butterflies on Guam and elsewhere. Based upon our firsthand knowledge, nest density is certainly much lower within occupied areas on Guam.

Several other species of predatory wasps are known from Tinian. Of these, *Delta* spp. solitary wasps are the most likely possible predators of the candidate butterflies. Being solitary wasps, however, the impact of their predation is likely much lower than that of the eusocial *Polistes stigma*. We collected two specimens of this genus within areas outside the mixed native limestone forest sites.

Ants

Numerous species of ants collected during our surveys are pending identification, but it should be noted that certain species occurred in high densities within the mixed limestone forest areas that we surveyed. Most species are common tramp ants found throughout the Pacific region and some of these species are likely capable of inducing predation pressure on the larvae of the candidate butterflies.

Mosquitoes

The mosquitoes collected on Tinian for this survey are awaiting identification. Mosquitoes are vectors of human diseases and animal diseases and nuisance pests of humans. Mosquito transmission of bird disease on Tinian is not a concern of USFWS. However, the Navy and Marine Corps will be concerned about mosquito-borne human diseases on Tinian if Tinian will be utilized for Navy and Marine Corps training and berthing. Vector-borne disease transmission to humans relies on several factors: a competent disease vector, the presence of a pathogen, a host reservoir, and favorable environmental conditions. The 1985 Hawaiian Agronomics report listed two species of mosquitoes found on Tinian during their survey from November 1984 – November 1985: *Aedes albopictus* and *Culex quinquefasciatus*. *A. albopictus* can serve as a vector of dengue fever. Culex quinquefasciatus is a vector of West Nile virus, various viral encephalitides, dog heartworm, and avian malaria. Past surveys have found other *Aedes* and *Culex* species including *Culex tritaeniorhynchus*-vector of Japanese encephalitis, *Aedes aegypti*-a vector of dengue fever and yellow fever, and *Anopheles indefinitus* a possible vector of malaria. If human activity in Tinian increases, the chance of accidental introduction of a mosquito-borne disease infected host and a competent mosquito vector also could increase. Implementation of appropriate quarantine measures would be a logical course of action as well as implementation of mosquito surveillance programs.

Termites

In many parts of the world, termites are ranked among the most significant economic insect pests. The Mariana Islands are no exception. Termite species in the Mariana Islands are primarily recorded from collections on Guam and Saipan. Light (1946) listed the following three species from Guam collected in 1936: 1) *Cryptotermes hermsi* Kirby (a synonym of *Cryptotermes domesticus* (Haviland)) (Kalotermitidae), 2) *Neotermes* *connexus* Snyder (Kalotermitidae), and 3) *Prorhinotermes inopinatus* Silvestri (Rhinotermitidae); and one more species collected from an unknown location in the Mariana Islands, *Calotermes marianus* Holmgrem (species name no longer used - maybe a synonym of *Incisitermes marianus* Holmgren). Later in the 1970's *Coptotermes formosanus* Shiraki (Rhinotermitidae) was documented as established on Guam (Su and Scheffrahn 1998); however, it was subsequently found that this termite was actually *Coptotermes gestroi* (Wasmann) (Rhinotermitidae) (Su and Sheffrahn 1998), formerly known as *Coptotermes vastor* Light (Rhinotermitidae) (Yeap *et al.* 2007). In 1993 Su and Sheffrahn (1998) found three additional termite species on Guam: *Cryptotermes dudleyi* Bank (Kalotermitidae), a *Microceretermes* species (Termitidae), and a *Nasutitermes* species (Termitidae). A more recent paper describing the complex of termites in the Mariana Islands (Yudin 2002) mentions *Schedorhinotermes* (Rhinotermitidae) and *Macrotermes* (Termitidae) as present on Guam and Saipan.

Outside of Guam and Saipan the distribution of termite species in the Mariana Islands does not appear to be well documented. The Hawaiian Agronomics, Inc. (1995) report refers to termites as abundant on Tinian, and mentions both subterranean (Rhinotermitidae) and drywood (Kalotermitidae) termites; however, no further identifications are provided. Table III-2 of their report listed 3 different species as collected, but unidentified. The report from the Chiba expedition to the Northern Mariana Islands (Natural Museum and Institute, Chiba 1994) lists only two species of termites: 1) *Cryptotermes domesticus* Haviland (Agihan, Anatahan, Guguan, and Sarigan) and 2) *Prorhinotermes inopinatus* (Agrihan, Anatahan, and Sarigan).

Identifications are pending, but it appears that at least three different species of termites were collected during the current survey. A large forest drywood (Kalotermitidae – most likely *Neotermes connexus*) and a species of the subterranean genus *Prorhinotermes* (Rhinotermitidae - most likely *Prorhinotermes inopinatus*) were found on both Tinian and Aguiguan. A second subterranean species, *Coptotermes gestroi* was collected on Tinian only.*Coptotermes gestroi* is by far the most damaging economic pest of the three species collected. As a subterranean termite, it primarily lives in large colonies underground, but constructs mud tubes to forage above ground and connect to sources of wood or other cellulosic materials. Colonies of this termite can cause significant structural damage in relatively short periods of time; therefore any construction on Tinian should take into consideration building design and building materials that will minimize the risk of termite infestation.

Odonates

The odonate fauna of the Mariana Islands has been described as meager, with only two endemic species (*Anax piraticus* Kennedy and *Agrionoptera insignis guamensis* Leiftinck) among the14 recorded taxa (Polhemus 2000). However, 8 years ago a new species, *Ischnura luta,* was discovered on the island of Rota and determined to be the first endemic damselfly recorded for the Mariana Islands (Polhemus 2000). Due to the fact that the insect fauna on Tinian and Aguiguan has not been well studied, and considering that both islands have some freshwater bodies, it was determined that this order of insects should receive some additional focus.

Adult Anisoptera, or dragonflies, were commonly seen flying in most open areas on Tinian. Specimens of adult dragonflies were collected at Lake Hagoi, at the Tinian Dynasty Hotel Fountain, and within the town of San Jose. Species of adult Anisoptera collected included *Trapezostigma transmarina* Brauer, *Pantala flavescens* (Fabricius), and *Diplacodes bipunctata* (Brauer). These species are all indigenous to the region, and considered common. Lieftinck (1962) states that due to their strong migratory tendencies, *P. Flavescens* and *D. punctata* are the dominant dragonflies of Micronesia. *Trapezostigma transmarina* is not as widely distributed - the Micronesian range of its distribution only includes Bonin, Southern Mariana, Palau, and Yap (Lieftinck 1962). The only aquatic area sampled on Tinian for immatures, or nymphs, was Lake Hagoi, and although many adults were seen flying above the lake, dragonfly nymphs were not found within the lake. There appeared to be at least one species present on Tinian that we saw, but were unable to catch, and which was different from the three species that were collected. This species was medium sized with bluish gray and maybe some yellow coloration, and was seen near the Japanese Village Ruins and also at the trailhead of the Japanese Defensive Caves Trails.

One adult dragonfly was collected on Aguiguan near Japanese-era cisterns that are partially filled with water. This specimen was identified as *T. transmarina*. Dragonfly nymphs were collected from these cisterns, and from small rock pools near the ocean at the boat landing site. There were a number of these pools in the limestone rock, filled presumably by rainwater or spray from crashing waves. These nymphs were semi-translucent and greenish in color. Mosquito larvae were also present in these pools in high numbers. Identification is pending for all dragonfly nymphs collected on Aguiguan.

Adults and nymphs of Zygoptera, or damselflies, were collected at Lake Hagoi. The adults were identified as *Agriocnemis femina femina* (Brauer), and the identifications of the nymphs are pending. This species appeared common within close proximity to Lake Hagoi, however adult damseflies were not seen anywhere else on the island. *Agriocnemis femina femina* is considered a widely spread, indigenous species (Lieftinck 1962) and is not of any particular conservation concern.

It appears that neither of the two endemic dragonfly species were collected on Tinian or Aguiguan during this survey, but until the nymphs collected on Aguiguan are identified this cannot be stated with certainty. These two species are found only in the southern Mariana Islands, *Anax. piraticus* having been found on Guam and Saipan, and *Agrionoptera insignis guamensis* having been found only on Guam.

Other Aquatic Insects

Aquatic insects collected on Tinian in Lake Hagoi are awaiting further identification; however, the taxa match fairly well with the aquatic organisms recorded in the Hawaiian Agronomics survey (Hawaiian Agronomics, Inc. 1995). The aquatic organisms collected from cisterns on Aguiguan are interesting because there is apparently no record of any previous aquatic sampling within the island's Japanese-era structures. Identifications are pending, but the following is a preliminary list of organisms collected from the cisterns: 1) Heteroptera: Veliidae, 2) Heteroptera: Pleidae, 3) Diptera: Chironomidae, 4) Odonata, 5) Coleoptera, and 6) Aranea: Tetragnathidae.

Of particular interest are the pleids, or pygmy backswimmers, that were collected. There was no record of the family Pleidae in Micronesia until 2007 when a paper was published describing the discovery in 2006 of *Paraplea puella* (Barber) on Guam in a river outflow (Zack *et al.* 2007). *Paraplea puella* is a North American species of pleid and is speculated to have been accidentally transported to Guam recently via the aquaculture or aquarium trade. How pleids would have arrived on Aguiguan is somewhat of a mystery since it is an uninhabited island. The cistern in which the pleids were found is a remant structure of a Japanese-era plantation village that existed from 1936 to 1945 (Butler 1990). It is possible that pleids were introduced to Aguiguan by the Japanese at this time. If the pleid specimens can be identified to the species level, the history of their presence on Aguiguan may become better understood.

REFERENCES

Bourquin, O. 2001. Invertebrate Collections Northern Marianas Islands Status 2001. CREES Program Commonwealth of the Northern Marianas College, Saipan. Unpublished report.

Butler, B. M. 1990. An archeological survey of Aguiguan (Aguijan) northern Mariana Islands. Micronesian Archeological Survey Report Number 29. 260 pp.

Hawaiian Agronomics (International), Inc. 1985. Final Report for Flora and Fauna Survey of Tinian, Northern Mariana Islands (U.S. Navy Contract N62742-84-C-0141) Volume I of II. Honolulu, HI.

Lieftinck, M. A. 1962. Odonata. Insects of Micronesia 5:1-95.

Light, S. F. 1946. Insects of Guam. B. P. Bishop Museum Bulletin 189:9.

Miyano, S. 1994. Insects of the northen Mariana Islands, Micronesia, collected during the expedition. Natural Museum and Institute, Chiba Natural History Research Special Issue 1:199-215.

Polhemus, D. A. 2000. A new species of Ischnura from Rota (Odonata: Coenagrionidae), and a discussion of zygopteran zoogeography in the insular tropical pacific. Bishop Museum Occasional Papers 62:5-12.

Schreiner, I.H. and D.M. Nafus. 1996. Survey of rare butterflies in the Mariana Islands. Preliminary report. U.S. Fish and Wildlife Service unpublished report. 10 pp.

Schreiner I.H. and D.M. Nafus. 1997. Butterflies of Micronesia. Agricultural experiment station college of agriculture and life sciences. University of Guam. Mangilao, Guam. 30pp.

Su, N-y and R. H. Scheffrahn. 1998. *Coptotermes vastator* Light (Isoptera: Rhinotermitidae) in Guam. Proceedings of the Hawaiian Entomological Society 33:13-18.

U.S. Fish and Wildlife Service. March 2007. Species Assessment and Listing Priority Assignment Form *Hypolimnas octocula mariannensis*. Unpublished report, USFWS, Region 1.

U.S. Fish and Wildlife Service Region 1. March 2007. Species Assessment and Listing Priority Assignment Form *Vagrans egistina*. Unpublished report. March 2007.

Vogt, S.R. and L.L. Williams. 2004. Common Flora and Fauna of the Mariana Islands. WinGuide Publishers, Saipan, Northern Mariana Islands. 158pp.

Yeap, B-k, A. S. Othman, V. S. Lee, and C-y Yang. 2007. Genetic Relationship between *Coptotermes gestroi* and *Coptotermes vastator* (Isoptera: Rhinotermitidae). Journal of Economic Entomology 100: 467-474.

Yudin, L. 2002. Termites of Mariana Islands and Philippines, their damage and control. Sociobiology 40:71-74.

Zack, R. S., A. Moore, and R. H. Miller. 2007. First record of a pygmy backswimmer (Hemiptera: Pleidae) from Micronesia. Zootaxa 1617:67-68.

Figure 1. Northern arthropod collection sites and survey transects, Tinian, CNMI, 7-15 August 2008.

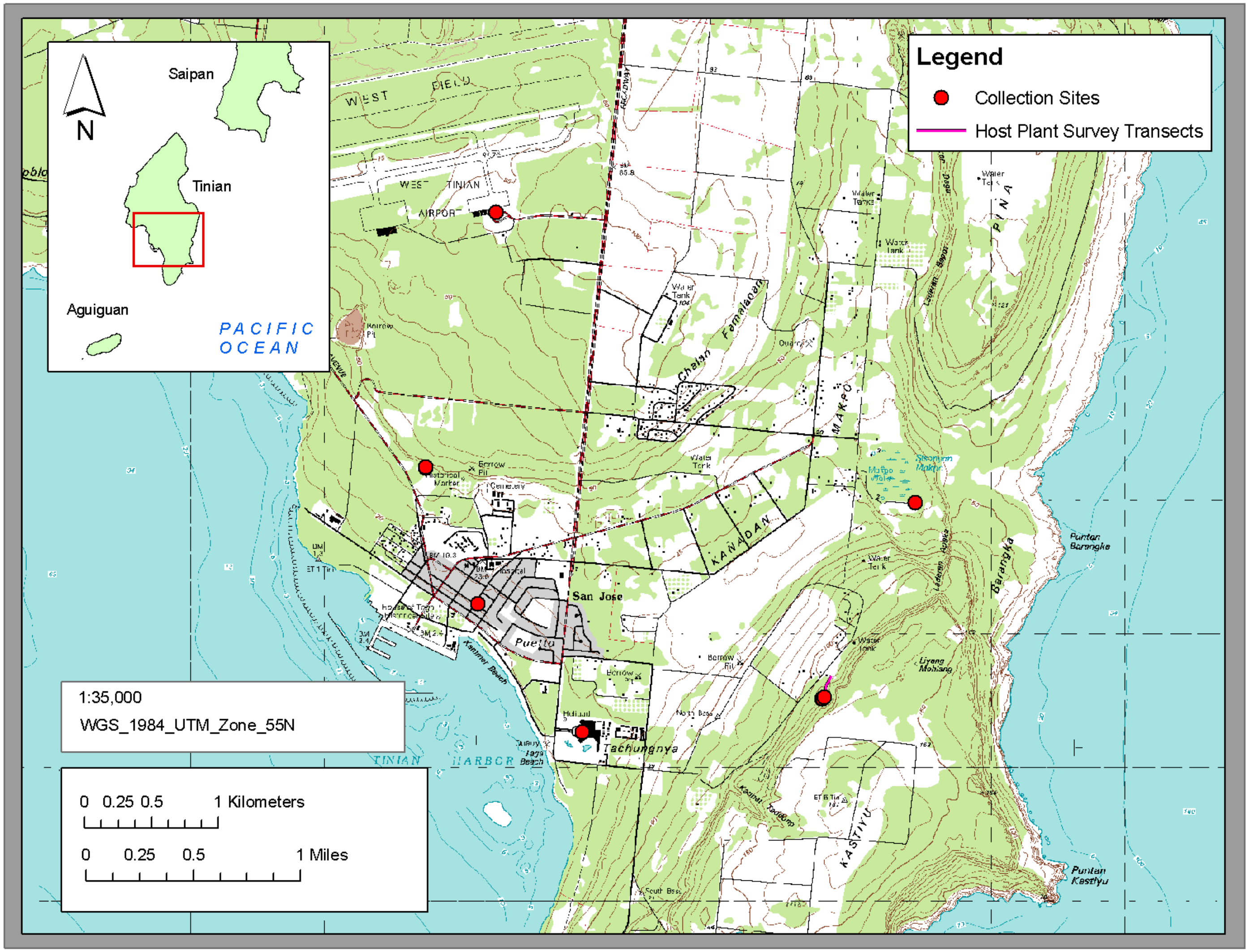


Figure 2. South central arthropod collection sites and survey transects, Tinian, CNMI, 8-11 August 2008.

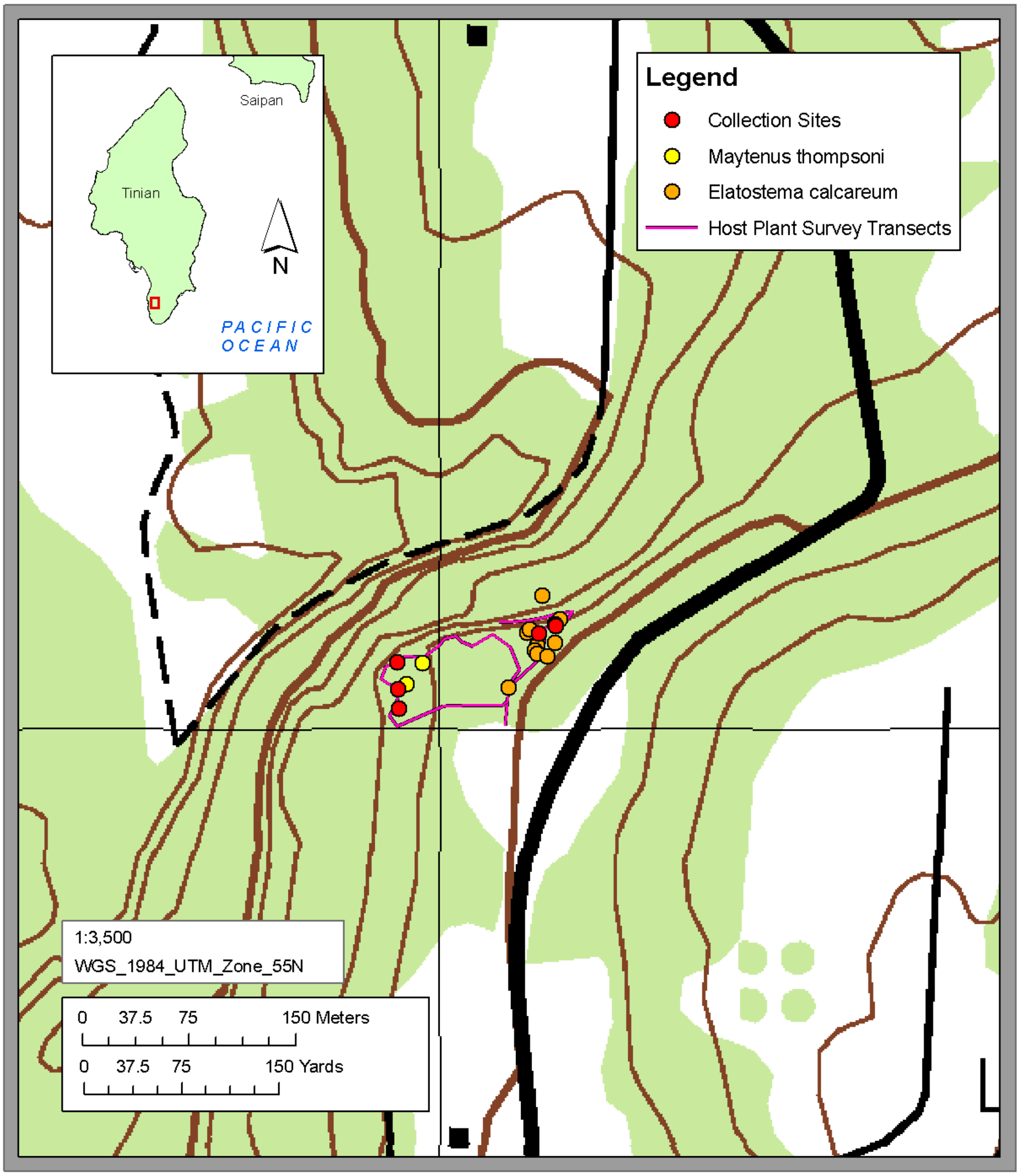


Figure 3. Southern arthropod collection sites and survey transects, Tinian, CNMI, 10 August 2008.

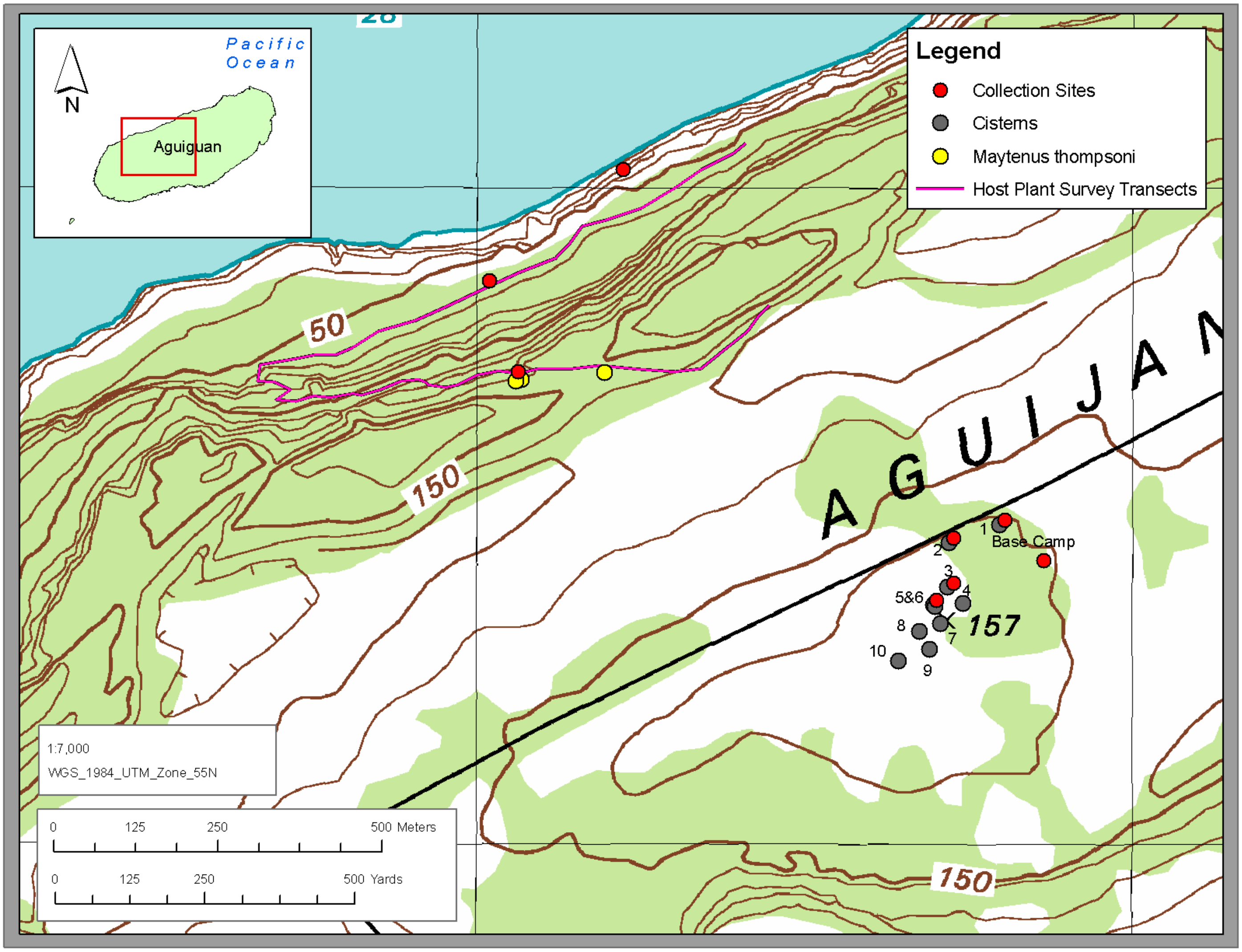


Figure 4. Arthropod collection sites and survey transects, Aguigan, CNMI, 12 August 2008.

2.2.2 CANDIDATE BUTTERFLY SURVEYS ON TINIAN

Prepared by: **Nathaniel B. Hawley, U.S. Fish and Wildlife Service, Saipan, MP and Antonio Castro, Division Fish and Wildlife, Tinian, MP**

INTRODUCTION

A survey was conducted on the island of Tinian, Commonwealth of the Northern Mariana Islands (CNMI) from June to October, 2008 to determine the presence or absence of two butterfly species, Marianas rusty butterfly *Vagrans egistina* and the forest flicker *Hypolimnas octocula mariannensis* (Family *Nymphalidae*). Currently, *V. egistina* and *H. octocula mariannensis* are listed as candidates for listing under the Endangered Species Act. Both butterfly species are considered rare and were only recorded on the islands of Saipan, Rota and Guam (Schreiner and Nafus 1996). *V. egistina* was last collected on Guam in the late 1970s and on Rota in the1980s and 1995. *H. octocula mariannensis* is historically known from Guam and Saipan and after an intensive survey for this species in 1995, investigators found only 10 populations on the island of Guam and none on the island of Saipan (Schreiner and Nafus 1996).

Based on a literature review, very few entomological studies have been conducted on the island of Tinian. The most comprehensive report available was prepared for the United States Navy (Contract N62742-84-C-0141) by Hawaiian Agronomics (International), Inc. in December, 1985. Hawaiian Agronomics staff spent a total of seventeen days between November 1984 to November 1985 collecting insects (including Lepidoptera) utilizing various methods of collection that accounted for temporal differences. They did not collect *Vagrans egistina* or *Hypolimnas octocula mariannensis*. This result was not surprising as both species have not been recorded on the island of Tinian, CNMI, previously. However, *Vagrans egistina* or *Hypolimnas octocula mariannensis* may have historically been found on Tinian or were missed during past survey efforts. A key factor in potentially finding either butterfly species is the presence of host plants on Tinian.

The recorded host plant for *V. egistina* is *Maytenus thompsonii* (Family *Celastraceae*), a small tree/shrub endemic to the Marianas and found primarily in the understory of native limestone forests (Vogt and Williams 2004, Schreiner and Nafus 1996). The recorded host plants for the *H. octocula mariannensis* are *Procris pedunculata* and *Elatostema calcareum*, both forest herbs (Family *Urticaceae*) found growing on limestone outcrops in native limestone forest (Schreiner and Nafus 1996).

METHODS

The primary objective of this survey was to determine the presence or absence of both Candidate species *Vagrans egistina* and *Hypolimnas octocula mariannensis* by locating host plant sites for each species and monitoring sites for life cycle stages. Since a large group of biologists were involved in surveying the flora and fauna on the island of Tinian, CNMI from June to August, 2008 an information guide (Figure 1) was produced and distributed to each biologist. The guide provided photographs of both butterflies and their host plants as well as descriptions of the caterpillars. The goal of the guide was to help identify host plants and *V. egistina* and *H. octocula mariannensis* in the field in order to collect GPS points of their locations for additional monitoring.

Four host plant sites were identified by U.S. Fish and Wildlife Service and Department of Defense biologists from June to August, 2008. Two host plant sites were located on Department of Defense leased land at Japanese Cave (Figure 2), and Chiget Cliff (Figure 3) and two on CNMI Public Lands at Carolinas Nature Trail (Figure 4) and Sisonyan Makpo (Figure 5).

A host plant area is an area with host plants for either *Vagrans egistina* or *Hypolimnas octocula mariannensis* within a 25 meter perimeter. For the purpose of this survey, only visual search hours were recorded while the observer was in a host plant area. Each identified host plant site was visually scanned for life cycle stages (eggs, caterpillar, chrysalis, and imagoes/adults) by one or two observers for up to two weeks at various time of the day. As time permitted a further scan of the area was conducted of up 1500 meters to determine if additional host plant sites had gone undetected.

Due to time constraints, this survey was unable to account for seasonal variation. The bulk of the survey occurred during the months of September and October which are considered part of the rainy season. Therefore, limited sampling was done during the dry season. However, Schreiner and Nafus (pers. comm. 2008) observed both species to be more numerous on islands they are recorded from during the rainy season although adults were observed year round.

Two butterfly bait traps (lip type obtained from BioQuip.com) were set at each host plant site for up to two weeks. The butterfly bait traps were re-baited every three days with locally obtained mashed, rotting bananas, a liberal dose of raw cane sugar, and a dash of water. The bait was prepared on the afternoon prior to the morning of use and typically became well fermented prior to being placed in the field. The traps were positioned within 5 meters of a host plant cluster and at approximately 3-4 meters above the ground.

RESULTS

No individuals of either candidate butterfly species were observed. A total of four host plant areas were identified and monitored from 9/10/08 till 10/21/08. A description of the search effort at each site is outlined below.

Carolinas Nature Trail (CNT) Area

Two sites, greater than 50 meters apart, were monitored at the Carolinas Nature Trail Area (UTM 55 P0353070 1652046). A total of 13 days were spent conducting visual searches (1,574minutes) with 624 hours spent trapping with 2 traps (Table 1). Common melon flies (*Bactrocera* sp.) were observed during thirteen of the fifteen times the traps were checked and were only recorded at one of the other monitoring areas, Chiget Cliff. However, no *Vagrans egistina* or *Hypolimnas octocula mariannensis* were recorded at this site. This area is not in the Military Lease Area and supported the single largest cluster of *Elatostema calcareum* found during this survey.

Japanese Cave (JC) Mount Lasu Area

The Japanese Cave Mount Lasu area (UTMs 55 P 0353408 1663125 and 55 P 0353357 1663105) is located in the Military Lease Area and consisted of several small clusters of host plants (*Maytenus thompsonii* and *Elatostema calcareum*). The two fruit baited traps were stationed at the two largest clusters (UTM site-1 = 55 P 0353408 1663125 and site-2 = 55 P 0353357 1663105). At Japanese Cave site-2, two caterpillars and four chrysalises of the blue moon butterfly *Hypolimnas bolina* were found feeding/pupating on *E. calcareum*, the chrysalises were reared in the lab for confirmation. Additional smaller clusters were visited several times over the 15 monitoring period with 1765 visual search minutes recorded and the two traps produced 696 total traps hours (Table 1). Over 8 kilometers of cliff line in the Mount Lasu area were surveyed for additional host plant sites during this period. Several small pockets consisting of 1-5 *E. calcareum* individuals were identified and scanned for life cycle stages of the candidate species, but none were found. Three *Melanitis leda* (Family *Satyridae*) and one *H. bolina* (Family *Nymphalidae*) were found feeding in the traps at Japanese Cave site-2. In addition, large congregations of mating *H. bolina* and *Hypolimnas anomala* (guardian butterfly) were observed, which supports Kemp's (2000) finding that the reproductive activity increases in *H. bolina* during months of higher rainfall and humidity levels.

Chiget Cliff (CC) -Laderan Chiget Area

Only one stand of *Elatostema calcareum* was identified and monitored at the Chiget Cliff site (UTM 55 P 0354957 1665402) after an extensive search (4 km) of the area. The site had approximately 175 individual *Elatostema calcareum* stems that were scanned for life cycle stages. Approximately 960 minutes were spent monitoring this site during the 12 day period, with 408 total trap hours (Table 1). No *Vagrans egistina* or *Hypolimnas octocula mariannensis* were recorded at this site. However, three *Gehyra oceanica* (Oceanic Gecko) were observed in the traps feeding themselves on melon flies (Bactrocera sp.). This site is entirely in the Military Lease Area.

Sisonyan Makpo (SM) Area

The Sisonyan Makpo site (UTM 55 P 0355868 1656618) was surveyed for 6 days with 505 visual search minutes and 120 total trap hours (Table 1). No *Vagrans egistina* or *Hypolimnas octocula mariannensis* were recorded at this site. This site is not in the Military Lease Area and was only surveyed as time permitted.

DISCUSSION

The primary objective of this survey was to determine the presence or absence of the two candidate species butterfly species, *Vagrans egistina* and *Hypolimnas octocula. mariannensis* After 4806 minutes (approx. 80 hours) of visual searching and 1848 documented trap hours (approx. 77 days) during the months of September and October, 2008 no life cycle stage of either species was collected. This finding was not entirely surprising as neither *V. egistina* or *H. octocula mariannensis* were previously collected or observed on Tinian despite the fairly comprehensive surveys conducted by Hawaiian Agronomics Inc. from 1984-85.

The bulk of the survey was represented by only two months of the rainy season, September and October, 2008. Several biologists however were actively investigating areas for host plant sites and individuals for at least 2 months prior to the survey, July and August, 2008. Even though the survey did not account for seasonal variation, it can be concluded that some life cycle stage of either species (*Vagrans egistina* and *Hypolimnas octocula mariannensis*) should have been collected after this level of effort. However it is recommended that host plants site should be observed monthly for at least one year to be certain.

Tinian is not known to be part of either species' historical range. However, the likelihood of introduced pests arriving to Tinian due to an increase in sea and air transports is a concern for a suite of native butterfly species. Additionally, any reduction of host plant sites for *Vagrans egistina* and *Hypolimnas octocula mariannensis* should be a conservation concern if translocation is considered as part of any future recovery or enhancement plans.

REFERENCES

Hawaiian Agronomics (International), Inc. 1985. Final Report for Flora and Fauna Survey of Tinian, Northern Mariana Islands (U.S. Navy Contract N62742-84-C-0141) Volume I of II. Honolulu, HI.

Schreiner, I.H. and D.M. Nafus. 1996. Survey of rare butterflies in the Mariana Islands. Preliminary report. U.S. Fish and Wildlife Service unpublished report. 10 pp.

U.S. Fish and Wildlife Service. March 2007. Species Assessment and Listing Priority Assignment Form *Hypolimnas octocula mariannensis*. Unpublished report, USFWS, Region 1.

U.S. Fish and Wildlife Service Region 1. March 2007. Species Assessment and Listing Priority Assignment Form *Vagrans egistina*. Unpublished report. March 2007.

Vogt, S.R. and L.L. Williams. 2004. Common Flora and Fauna of the Mariana Islands. WinGuide Publishers, Saipan, Northern Mariana Islands. 158pp.

Personal Communications

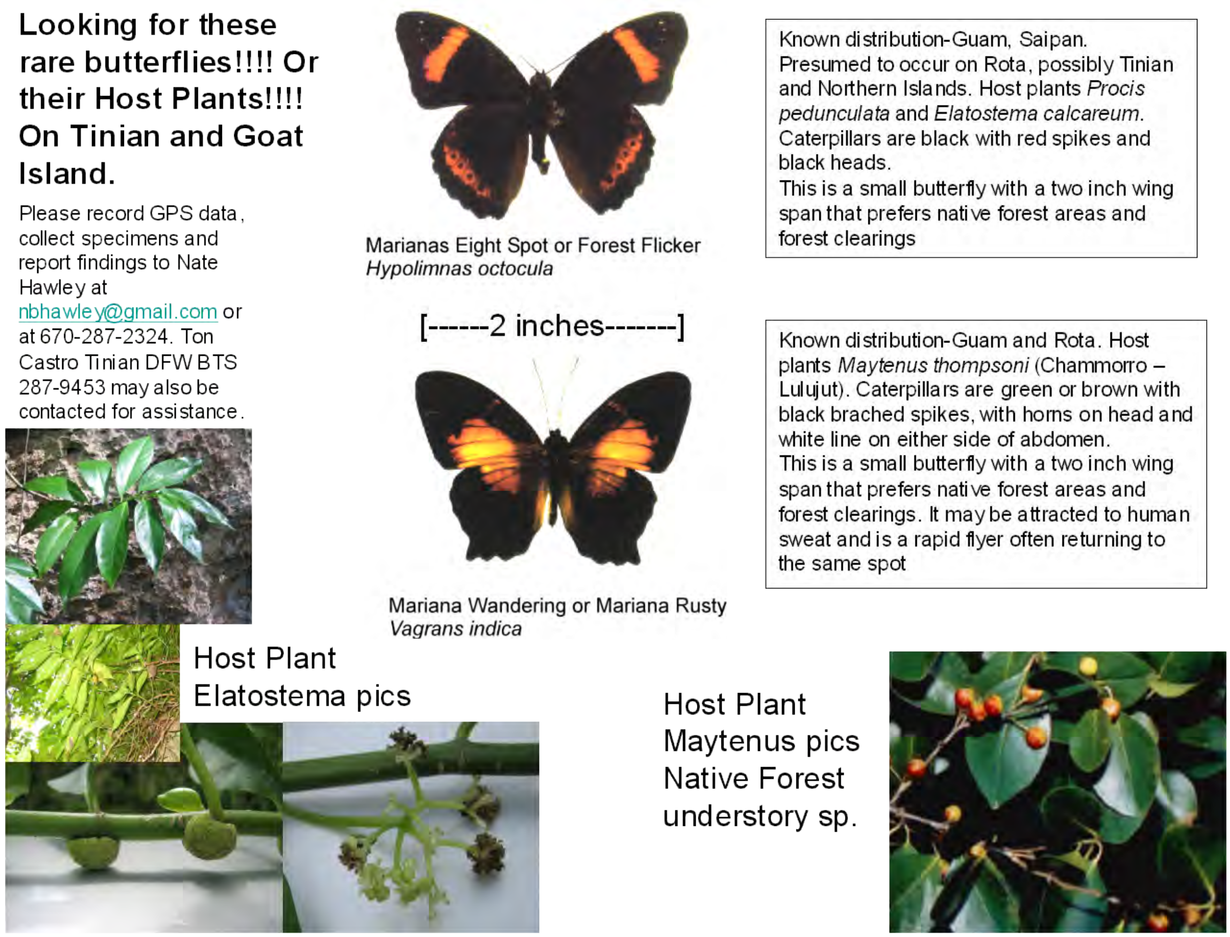
Schreiner, I.H., Oct. 3, 2008 note, via email communication.

Table 1. Level of effort: Visual and trap hours at four host plant sites, Tinian, CNMI. Site codes are CNT- Carolinas Nature Trail, JC – Japanese Cave, SM – Sisonyan Makpo, and CC – Chiget Cliff. Weather codes are CC – Cloud Cover, W – Wind, and R – Rain.

Site	Date	Observation Time (minutes)	Staff	# of Traps	Trap Hours	Weather Notes
CNT Site Set Up	9/10/08	45	1	0	0	na
CNT	9/11/08	75	2	2	48	CC:40% W:1-2 R:0
CNT	9/12/08	60	1	2	48	CC:90% W:1 R:0
CNT	9/13/08	150	1	2	48	CC:90% W:1-2 R:0
CNT	9/14/08	150	1	2	48	CC:95% W:2-3 R:Y
CNT	9/16/08	170	1	2	48	CC:90 W:1-2 R:Y
CNT	9/17/08	50	1	2	48	CC:80% W:1-2 R:Y
CNT	9/18/08	75	1	2	48	CC:70% W:2-3 R:Y
CNT	9/20/08	150	1	2	48	CC:70% W:2-3 R:Y
CNT	9/21/08	160	1	2	48	CC:65% W:1-2 R:0
CNT	9/22/08	120	1	2	48	CC:65% W:1-2 R:0
CNT	9/23/08	120	1	2	48	CC:65% W:1-2 R:0
CNT	9/24/08	120	1	2	48	CC:65% W:1 R:0
CNT	9/25/08	130	2	2	48	CC:65% W:1 R:0
CNT Total	**13 days**	**1575**			**624**	
JC Site Set Up	9/25/08	120	2	0	0	Set up traps host plant visual
JC	9/26/08	120	1	2	48	CC:65% W:1 R:Y
JC	9/27/08	135	1	2	48	CC:45% W:1 R:Y
JC	9/28/08	90	1	2	48	CC:70% W:1 R:0
JC	9/29/08	120	1	2	48	CC:75% W:1-2 R:Y
JC	9/30/08	120	1	2	48	CC:80 W:2-3 R:Y

Site	Date	Observation Time (minutes)	Staff	# of Traps	Trap Hours	Weather Notes
JC	9/31/08	120	1	2	48	CC:70% W:1 R:0
JC	10/1/08	120	1	2	48	CC:75% W:1-2 R:0
JC	10/2/08	120	1	2	48	CC:65% W:1 R:0
JC	10/3/08	120	1	2	48	NA
JC	10/4/08	60	1	2	48	NA
JC	10/5/08	60	1	2	48	CC:35% W:2 R:0
JC	10/6/08	100	1	2	48	CC: 50 W:1-2 R:0
JC	10/7/08	60	1	2	48	CC:40 W:1 R:0
JC	10/8/08	120	1	2	48	CC: 65% W:1-2 R:0
JC	10/9/08	180	2	1	24	CC:40% W:1 R:0
JC Total	**15 Days**	**1765**			**696**	
SM Site Set Up	10/9/08	85	1	0	0	NA
SM	10/10/08	120	1	1	24	CC:35% W:1 R:0
SM	10/11/08	90	1	1	24	CC:35% W:1 R:0
SM	10/12/08	90	1	1	24	CC:40% W:0 R:0
SM	10/13/08	60	1	1	24	CC:65% W:1 R:0
SM	10/15/08	60	1	1	24	CC:30% W:1 R:0
SM Total	**6 days**	**505**			**120**	
CC Site Set Up	10/10/08	120	2	0	0	CC:40% W:1 R:0
CC	10/11/08	60	2	1	24	CC:30% W:1 R:0
CC	10/12/08	90	2	1	24	CC:35% W:2 R:0
CC	10/13/08	80	2	1	24	CC:50% W:1 R:0
CC	10/14/08	80	2	1	24	CC:40% W:1 R:0
CC	10/15/08	90	2	1	24	CC:65% W:1-2 R:0
CC	10/16/08	60	1	2	48	CC:95% W:2-3 R:0

Site	Date	Observation Time (minutes)	Staff	# of Traps	Trap Hours	Weather Notes
CC	10/17/08	60	1	2	48	CC:70% W:1-2 R:0
CC	10/18/08	100	1	2	48	CC:40% W:1 R:0
CC	10/19/08	100	1	2	48	CC:40% W:1 R:0
CC	10/20/08	60	1	2	48	CC:30% W:1 R:0
CC	10/21/08	60	1	2	48	CC:40% W:1-2 R:0
CC Total	**12 days**	**960**			**408**	



Looking for these rare butterflies!!!! Or their Host Plants!!!! On Tinian and Goat Island.

Please record GPS data, collect specimens and report findings to Nate Hawley at nbhawley@gmail.com or at 670-287-2324. Ton Castro Tinian DFW BTS 287-9453 may also be contacted for assistance.

Marianas Eight Spot or Forest Flicker
Hypolimnas octocula

Known distribution-Guam, Saipan. Presumed to occur on Rota, possibly Tinian and Northern Islands. Host plants *Procis pedunculata* and *Elatostema calcareum*. Caterpillars are black with red spikes and black heads.
This is a small butterfly with a two inch wing span that prefers native forest areas and forest clearings

[------2 inches-------]

Mariana Wandering or Mariana Rusty
Vagrans indica

Known distribution-Guam and Rota. Host plants *Maytenus thompsoni* (Chammorro – Lulujut). Caterpillars are green or brown with black brached spikes, with horns on head and white line on either side of abdomen.
This is a small butterfly with a two inch wing span that prefers native forest areas and forest clearings. It may be attracted to human sweat and is a rapid flyer often returning to the same spot

Host Plant
Elatostema pics

Host Plant
Maytenus pics
Native Forest understory sp.

Figure 1. Butterfly Information Guide.

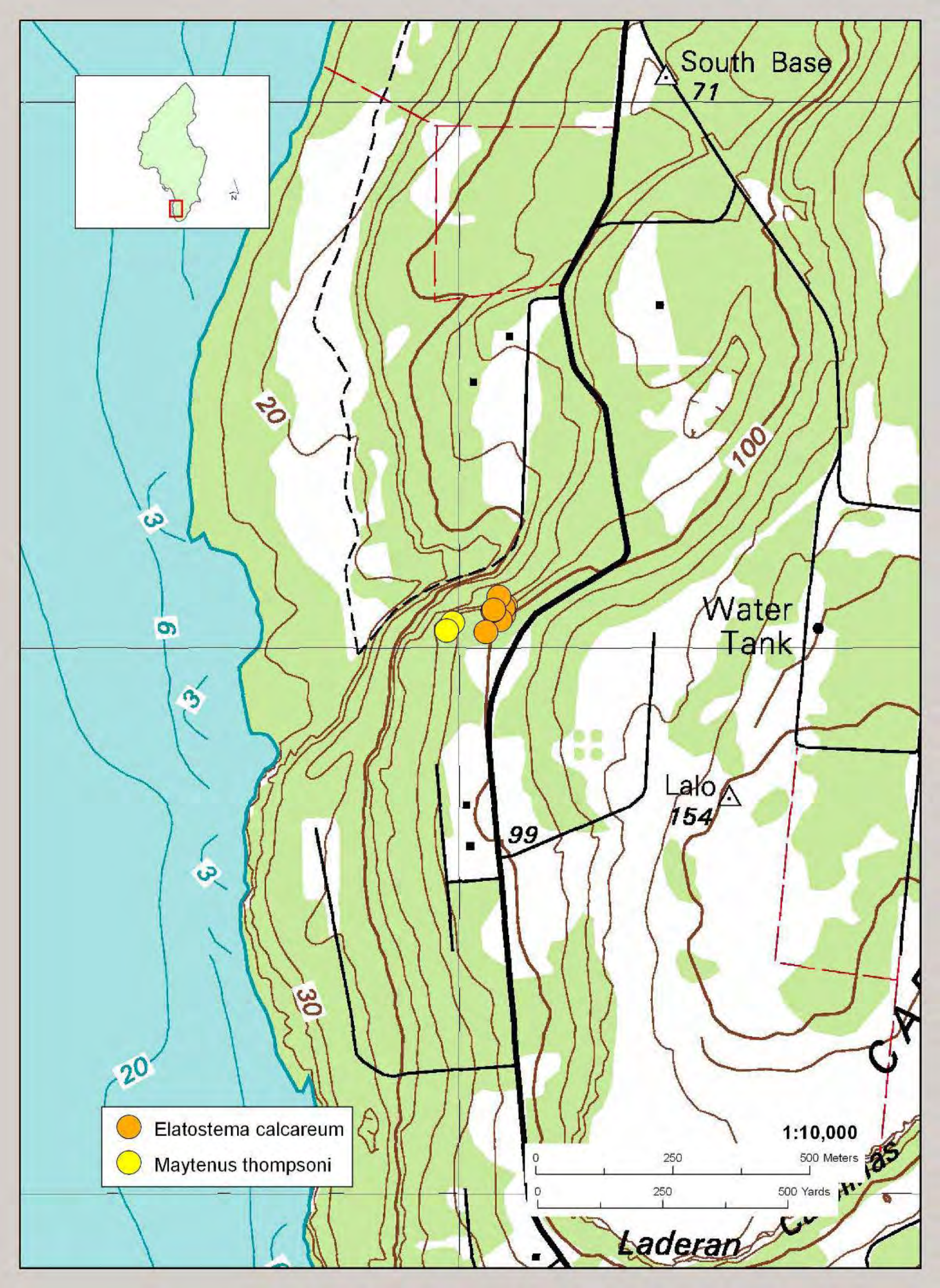


Figure 2. Host Plant Sites for *Vagrans egistina* and *Hypolimnas octocula* near the Carolinas Nature Trail, Tinian, CNMI.

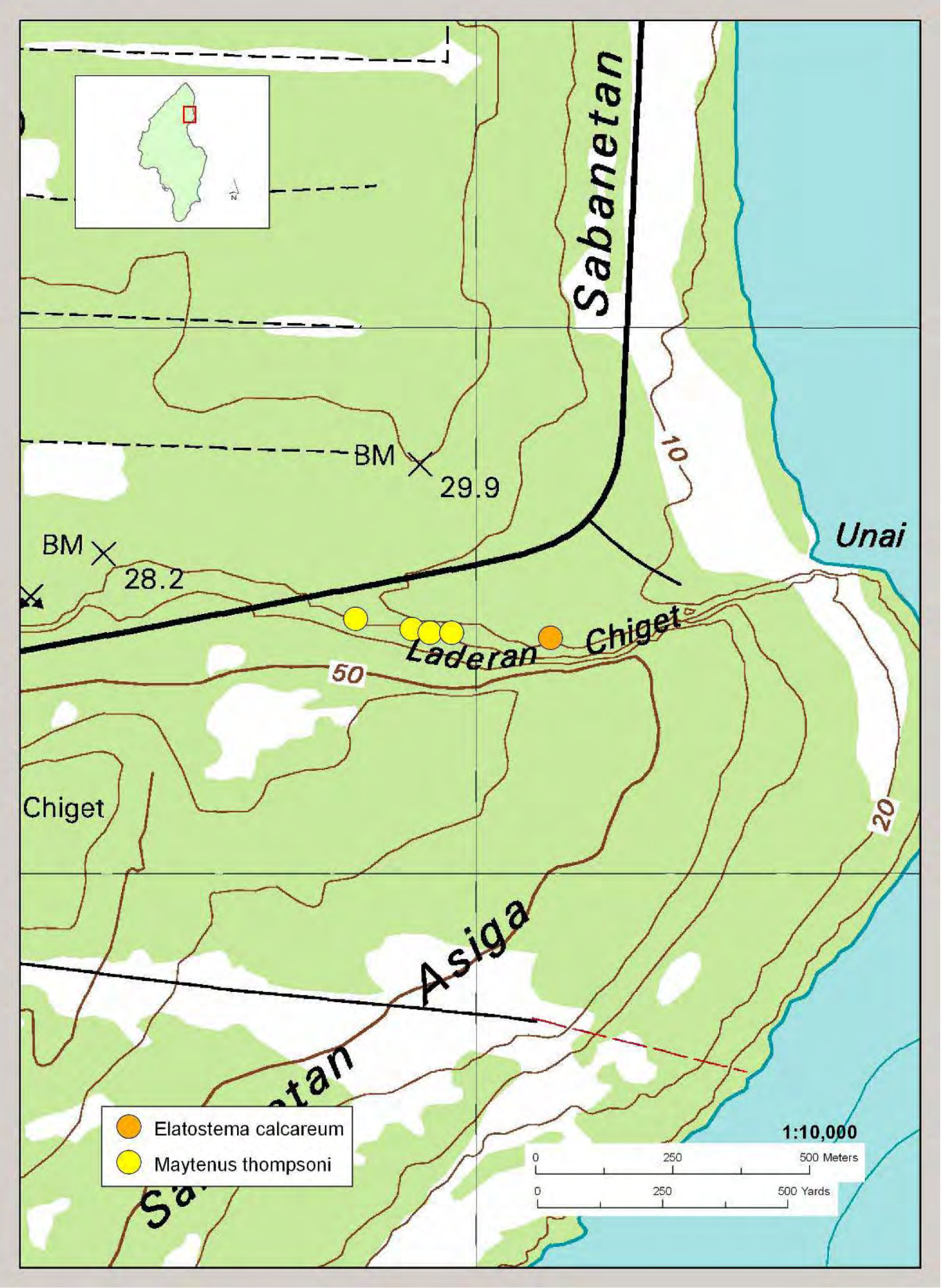


Figure 3. Host Plant Sites for *Vagrans egistina* and *Hypolimnas octocula* near Chiget Cliff, Tinian, CNMI.

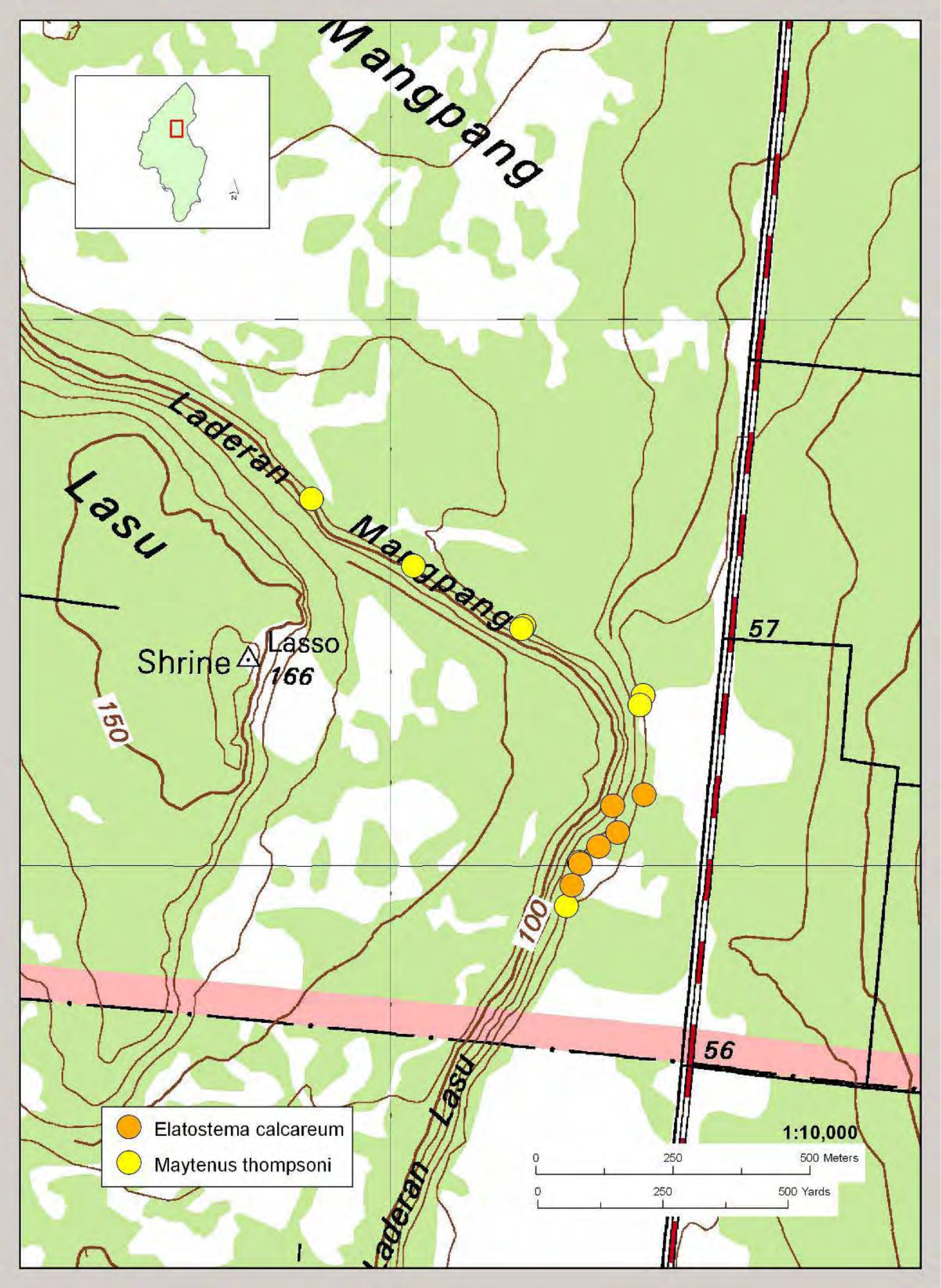


Figure 4. Host Plant Sites for *Vagrans egistina* and *Hypolimnas octocula* near the Japanese Caves, Tinian, CNMI.

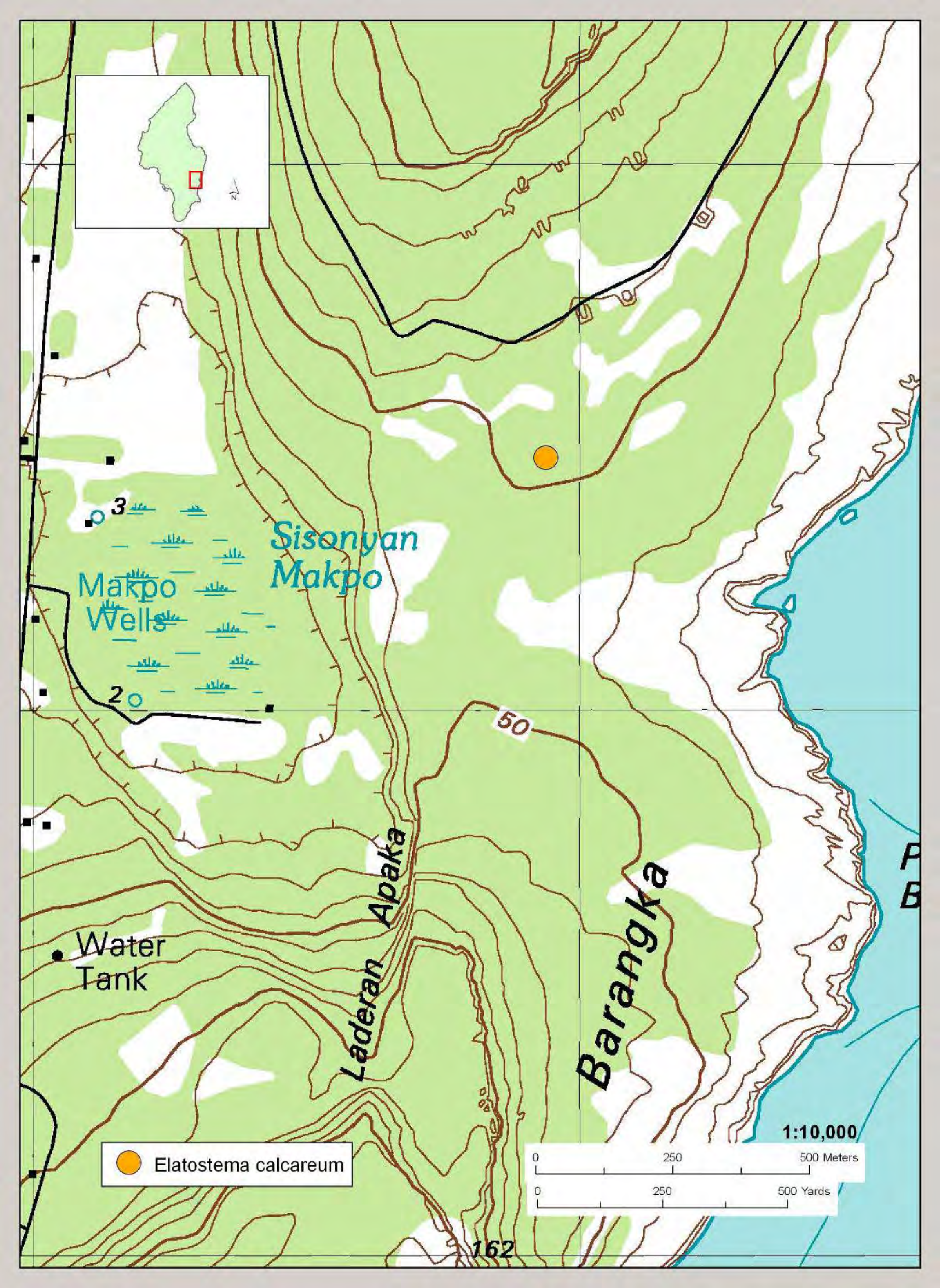


Figure 5. Host Plant Sites for *Vagrans egistina* and *Hypolimnas octocula* near Sisonyan Makpo, Tinian, CNMI – Sisonyan Makpo.

2.2.3 COCONUT CRAB SURVEYS ON MILITARY LEASE LANDS ON TINIAN


Coconut Crab. Photo by Scott Vogt.

Prepared by: **Scott Vogt, Wildlife Biologist, U.S. Navy, NAVFAC Pacific, Honolulu, HI**

EXECUTIVE SUMMARY

Coconut crabs (*Birgus latro*) were sampled on military lease lands on the island of Tinian and also on the uninhabited island of Aguijan just south of Tinian. Crabs were sampled by bait station transect lines and on 2 mark recapture bait station grids. Transect lines sampled 4 different habitat types (tangantangan forest, native forest, coastal forest , and grassland) and the trapping grids sampled 2 habitat types (tangantangan forest and native forest). Crab demographics and population densities were documented. The data show that on Tinian and Aguiguan, coconut crabs are being over-harvested and that present harvest rates cannot be sustained. Compared with unharvested populations, the Tinian and Aguiguan crab sizes are much smaller and densities are much lower.

INTRODUCTION

The Coconut or Robber Crab (*Birgus latro*) has a wide distribution ranging from Eastern Africa, through the Indian Ocean islands to the Pacific Ocean islands. Due to its large size, ease of collection and palatable flesh, the coconut crab is often over-harvested when it occurs in the vicinity of human habitation. The Mariana islands are no exception and surveys on Guam (USFWS, 2001) and Saipan (Kessler, 2006) have documented over-harvested populations.

Tinian is the second largest island in the Commonwealth of the Northern Mariana Islands (CNMI) with an area of 102 sq. km (39 sq. miles), and a human population of 3,500. Most habitat on Tinian is degraded due to a history of the grazing impacts of feral and domestic cattle, extensive sugar cane cultivation and disturbance during WWII. Vegetation is currently dominated by non-native species, principally the tangantangan tree (*Leucaena leucocephala*), Native limestone forest is generally confined to the sides of cliffs or other areas that were not suitable for cultivation. The northern 2/3rds of Tinian is leased by the U.S. Navy for training, although there are presently no military buildings there. A radio relay station for the Voice of America is on Tinian military lands and covers about 20 hectares.

The Commonwealth of the Northern Mariana Islands has a legal crab hunting season from September 15-November 15. Only crabs with a thoracic width larger than 3 inches (76 mm) are allowed to be taken and females carrying eggs (berried) of any size are prohibited. Coconut crabs are a type of hermit crab, however they drop the habit of residing in a shell at a small size and go through life with no shell. They are the largest land dwelling invertebrate in the world and can reach a weight in excess of 5kg. Coconut crabs breed on land but the female releases the eggs in the ocean where they immediately hatch. The oceanic larval stage lasts 2-3 weeks (Fletcher and Amos, 1994). Once on land the growth rate is slow and it probably takes 8-10 years for crabs to reach the CNMI legal size limit (Brown and Fielder, 1991).

The goals of this study were to establish coconut crab population densities and demographics on Tinian, and calibrate the catch per unit of effort index of bait station transect lines. With a calibrated index, crab densities can be calculated from the catch rates on transect lines of bait stations.

STUDY AREA

This study sampled coconut crabs on Navy leased lands on Tinian. The northern 2/3 of the island is leased by the U.S. Navy for training activities.

There are four main habitat types in this area: native limestone forest, introduced tangantangan (*Leuceana leucocephala*) forest, grassland, beach strand and mixed tangantangan grassland.

Crabs were sampled in February, April and May of 2007.

In October, 2006 a sampling trip was made to Aguiguan, an uninhabited island several miles south of Tinian. It was hoped that this island would provide a less harvested control population to compare with Tinian.

METHODS

Bait station transects

Coconut crab abundances were initially measured by transect lines of fermented coconut bait stations. The fermented coconut is a local technique (called "Poni"), coconut meat is grated, and allowed to ferment in a closed container for 5~10 days. One handful of poni bait was set on the ground (1 station) to attract crabs. Transect lines of poni bait stations, spaced every 20 meters, were used to sample native forest, introduced tangantangan, mixed tangantangan forest, grassland and beach strand habitat types. A total of 12 transect lines were monitored (Figure 1.). Four in native forest, 5 in tangantangan and mixed forest, 2 in grassland and 1 in beach strand. Each transect had 20-30 stations and was monitored for 1 or 2 nights. On Aguiguan 55 stations (20 meter spacing) on two lines were set in native forest for 3 nights and one line of 18 stations (20 meter spacing) was set in tangantangan mixed forest for 1 night.

Crab abundance was expressed as the catch per unit of effort (CPUE). CPUE was calculated by the number of crabs captured divided by the number of trapping nights. The number of trapping nights was the number of bait stations multiplied by the number of nights they were monitored. For example, 25 stations monitored for 2 nights was 50 trapping nights. Ten crabs captured on a 25 station transect monitored for 2 nights would be a CPUE of 0.20. Crabs captured between bait stations were also used in the CPUE calculation.

Figure 1. Location of bait station transect lines.

Mark recapture grids

Bait station grids were used to establish coconut crabs densities. Two grids of 100 coconut bait stations were set up in native forest and tangantangan forest (Figure 2.). Bait stations were spaced every 20 meters and the grids were 10 stations by 10 stations (100 stations total, 180 meters by 180 meters) and covered an area of 3.24 hectares. The locations of the grids were selected based on the CPUE of the bait station transects. The native forest grid was set on the cliff line facing North Field and approximately 1500 meters northwest of Mt Lassa. The tangantangan grid was placed in one of the forest blocks immediately off of the southwest end of runway able.

Figure 2. Location of mark-recapture grids.

Each bait station consisted of a whole coconut that had fallen off the tree but still had juice and had not yet sprouted. A small hole (~3 cm diameter) was cut into the coconut exposing the meat and juice. The coconut was wired to a tree or stuck onto a cut sapling tree so that the coconut was 10-30 cm off the ground. Vanilla extract was then poured into the coconut to magnify the scent. The crabs will stay on the coconut feeding and can be easily captured at night. Crabs on the ground at, and between, bait stations were also collected, marked and measured.

The grids were monitored for 5 nights within a one week period. Crabs captured on the grid were measured for thoracic length, weighed, sexed, marked and released. Crabs were marked with, fingernail polish, permanent magic marker or clear epoxy glue over a paper number.

Mark recapture data was analyzed using Program MARK to estimate the size of the population. This population estimate was then used with the effective area sampled (see below) to calculate the crab density for each habitat sampled. MARK also estimates the capture probability and recapture probability.

Telemetry and Thread Bobbins

A difficulty in trying to establish animal densities using trapping grids is determining how many animals come from outside the grid to be captured. So, the grid samples a larger area than its dimensions. It is often difficult to calculate the size of the sampled area and this is vital for accurate density estimates. A way to solve this problem is to fit animals with transmitters so that movements within and outside the grid can be ascertained. In this way a buffer strip can then be added to the grid boundary for density calculations. In other words, from telemetry data an extra 50 meters might be added to the grid boundary so that the true area sampled was 280m x 280m and not 180m x 180m. This problem is often overlooked or taken lightly in grid sampling but, small changes can have very large effects on the final density estimate.

Wildlife Track transmitters were attached to crabs with non-toxic marine putty. The transmitters weighed 10 grams and the putty added an additional 10 grams for 20 grams total. Crabs that were smaller than 150 grams were not fitted with transmitters. The transmitters have a 12 month battery life.

Crabs captured within the grids were weighed, measured for thoracic length and width, sexed and those larger than 150 grams were fitted with transmitters. Crabs were then released at the point of capture.

Barber thread bobbins were also attached (with non-toxic marine putty) to 8 crabs captured on the bait station transect lines. After attaching the bobbin crab was released. The thread feeds out and can be followed.

Comparison with a non-harvested population

Coconut crab fieldwork was conducted on the island of Diego Garcia at the Mini Mini conservation area in July 2003 and March 2004. Because of tight security and access restrictions this area contains a true non-harvested coconut crab population. Population density and demographics were documented for this area and are used here as a control population for comparison.

RESULTS

Transect lines

Transect lines in native forest had the highest CPUE with an average of 0.11 (range, 0.07-0.13). Tangantangan had a mean CPUE of 0.03 (range, 0-0.06). No crabs were captured in the grassland or beach strand habitats. The catch rates in tangantangan and native forest were very consistent.

Two bait station lines on Aguiguan produced a CPUE of 0.20. A bait station transect (20 stations, 20 meters spacing, monitored for 2 nights) on Diego Garcia in a mixed coconut/native forest produced a CPUE of 1.78.

Mark recapture grids

The native forest grid captured 34 crabs and the tangantangan grid captured 12 animals. This was consistent with the line transect data for these habitat types in that the native forest grid captured ~3 times as many crabs as the tangentangen grid.. There were no recaptures on either grid. The majority of the crabs were captured within the first 3 days: 82% on the native forest grid and 75% on the tangantangan grid. Since there were no recaptures, the data were analyzed as removal plots. Due to the short sampling period, the population was assumed to be closed (no deaths, births, immigration or emigration). Because of the behavioral response to being captured the model "Full Closed Captures with Heterogeneity" in program MARK was chosen for the analyses. This model takes into account the behavioral response to being captured and also individual heterogeneity in capture (different individuals have different capture probabilities). See Table 1 for the population estimates and capture probability estimates for the 2 mark recapture grids.

Table 1. Population Estimate and Capture Probability of the Trapping Grids.

	Population estimate	95% confidence interval	Capture probability	95% confidence interval
Native forest plot	38.78	34.93-58.50	0.33	0.18-0.52
Tangantangan plot	14.32	12.21-37.57	0.28	0.08-0.65

Telemetry and Thread Bobbins

Not as much data was collected as desired due to problems with the radio receivers. Both units initially worked for 1-2 days and then stopped picking up the signals. Another receiver was borrowed from COMNAVMAR but several days of data were missed because of this. All crabs left the grid area, or lost/destroyed the transmitter, 1-14 days after having the transmitter attached.

An apparent behavioral response was shown by all crabs with transmitters. All crabs took shelter in rock crevices and stayed there for 1-5 nights before resuming foraging.
Eight crabs on the native forest grid were fitted with transmitters and 6 on the tangantangan grid. Twenty eight out forty six (60%) of the crabs, captured on the grids were too small for transmitters (less than 150 grams).

Two transmitters were found in the field having apparently fallen off the crab. One, in the native forest grid, was within 2 meters the capture point. The other was off of the tangantangan grid and was found 210 meters from the point of capture just outside of the forest on the runway tarmac.

The thread bobbins did not provide as much data as hoped due to the thread breaking within 40 meters of the release point. All crabs released with bobbins climbed trees within 15 meters of capture. The thread broke on 6 of the crabs within 15 meters after climbing down from the tree. One crab after climbing down the tree took refuge in a karst crevice where the thread broke. One crab stayed in a crevice 2 days and nights before moving out and then the thread broke.

Population densities
While not as much telemetry data was collected as hoped, enough was collected to estimate the actual size of the area trapped. The mean of the maximum distance moved (MMDM) by each crab, within the one week grid sampling period, was added as a buffer to the dimensions of the trapping grids. The data from both grids were pooled for this due to the low sample size. This was 50 meters. So the effective trapping area was 280 meters x 280 meters or 7.84 hectares for both grids. See Table 2 for the density estimates for each habitat type.

Table 2. Comparison of Crab Densities in Native Forest and Tangentangen Habitats.

	Population estimate	95% confidence interval
Native forest plot	4.95 crabs/hectare	4.46-7.46 crabs/hectare
Tangantangan plot	1.83 crabs/hectare	1.56-4.79 crabs/hectare

A similar crab mark recapture project with a bait station grid on Guam had an MMDM buffer strip of 44 meters (USFWS 2001). This MMDM was calculated by recaptured crabs on the grid (the distance between recaptures) and not telemetry but the similarity is noteworthy.

The density estimate for the Diego Garcia population, using quadrat sampling, was 233 crab/ha, 107-358. This is roughly 42 times higher than Tinian.

Using the area of each habitat type (Liu and Fisher 2006) we can estimate the total crab population for Navy lands on Tinian. Native forest (150ha) = 742 crabs; secondary growth forest (2021 ha) and tangantangan forest (2302 ha) = 7,911 crabs. Total population estimate for these forest communities is 6,653 crabs. Of this estimate, 421 crabs (6.33%) are legal size. Note, that since the catch rates of the bait station transect lines were similar for tangantangan and secondary forest, the densities are presumed to be similar.

Our study, to a degree, has calibrated the catch rate index (Charts 3, 4, and 5). A higher catch rate on the transect lines did in fact predict a higher density of crabs. Using the Aguiguan catch rate of 0.20 we can reasonably conclude that the density is at least double that of Tinian native forest which had an average catch rate of 0.11. So, we can reasonably conclude that Aguiguan has crab densities of about 10 crabs/ha. The area of Aguiguan is 699 ha and of this, 479 ha (68%) is forested (Chapter 2. this report). For the forested area, we estimate the Aguiguan population to be 4,790 crabs. We did not survey the non-forested areas (mostly dominated by introduced *Lantana camara*). So crab numbers on Aguiguan are undoubtedly higher than this estimate.

Population Demographics
Counting captures from the transect lines, the grids and road captures, a total of 79 crabs on Tinian, were measured and weighed in this study. Out 79 crabs, 5 were legal size (6.33%). Of the 79 crabs, 41 were female and 38 were male.

Comparing the mean sizes and weights with the Diego Garcia (non-harvested) population is interesting (Table 3). The differences are stark (Charts 1. and 2.). The graphs show that the reason the average weights at Diego Garcia are roughly 4 times higher than Tinian is due to an almost total lack of crabs above the legal size limit (~37 mm TL) on Tinian.

Table 3. Coconut Crab lengths and weights: Diego Garcia vs Tinian

	Diego Garcia: Mean with 95% confidence interval	Tinian: Mean with 95% confidence interval
Male thoracic length, mm	48, 45-52	30, 28-32
Female thoracic length, mm	42, 40-45	27, 26-29
Total thoracic length, mm	46, 44-48	29, 28-30
Male weight, grams	988, 722-1254	238, 197-279
Female weight, grams	529, 391-667	183, 158-208
Total weight, grams	844, 649-1038	209, 185-234

Chart 1. Size Classes of Coconut Crabs on Tinian

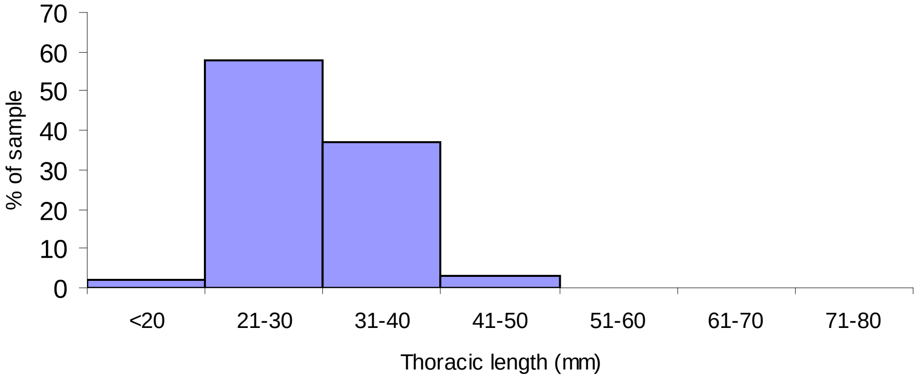
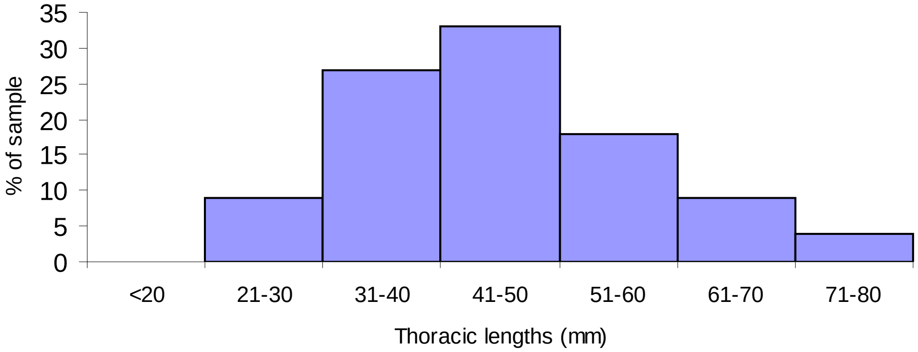


Chart 2. Size Classes of Coconut Crabs on Diego Garcia

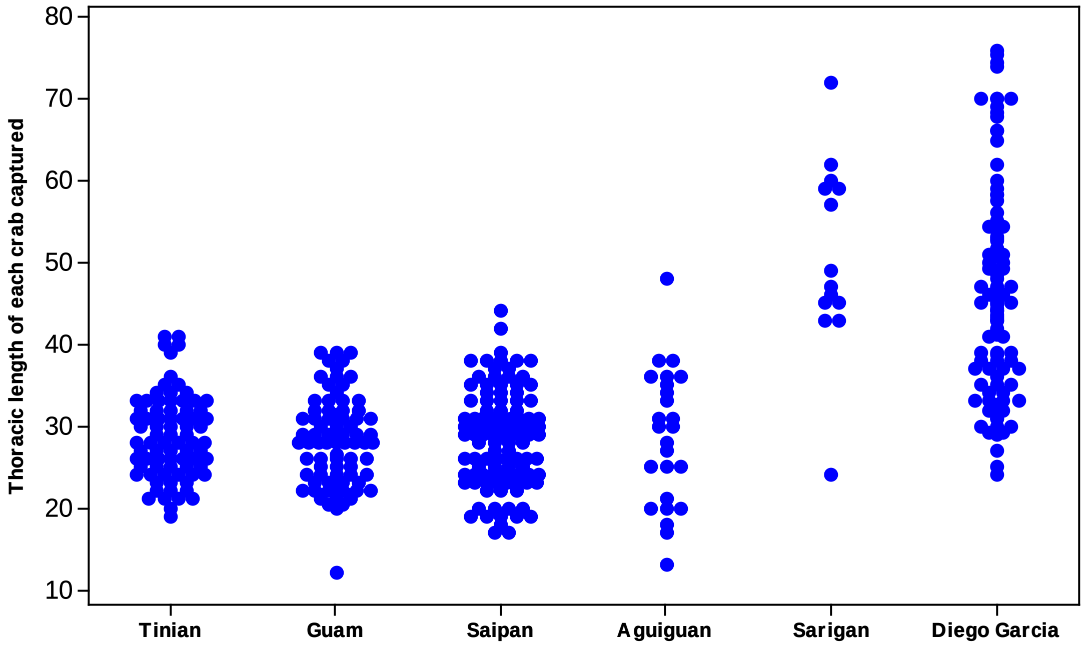


On Diego Garcia the most common size class, 41-50mm, represents about 35% of the population. On Tinian this size class is only 3% of the sample, and there were no crabs found that were larger than this size class. On Diego Garcia crabs larger than the 41-50mm size class represent an additional 31% of the population.

This is further graphically shown in chart 3, comparing Guam, Saipan, Tinian and Aguiguan data with the Diego Garcia non-harvested standard. On these Marianas islands there is nothing above 50 mm TL and very few that are above 40 mm TL, while on Diego Garcia the majority of the population is above 40mm TL.

The data from Aguiguan is hampered by small sample size due to the short time spent on the island. While the catch rate was almost double that of Tinian the demographics were similar to Tinian (Guam and Saipan also) and indicate that Aguiguan is also overharvested (Chart 3, Appendix 2.). The proportion of the Aguiguan sample over the legal size limit, 12% (4 out of 25) was also double what it was on Tinian. This produces an estimate of 574 legal size crabs on the forested areas of Aguiguan.

Chart 3. Comparison of Coconut Crab Sizes from the Marianas and Diego Garcia



We documented only 1 crab smaller than 20mm TL on Tinian. Crabs smaller than 20rnm TL are typically not well documented with surveys like these (Chauvet, C. and T. Kadiri-Jan.1999). The habits of these smaller crabs make them harder to find and they are typically nearer to the ocean (Kadiri-Jan and Chauvet 1998). So we cannot comment on juvenile recruitment with this study even though this is obviously an important factor.

Calibration of the CPUE index

The CPUE index was strongly correlated with the number of crabs captured on the grid and the estimated densities (Charts 4, 5, and 6). The previously mentioned Guam study produced a CPUE of 0.15, captured 45 crabs and produced a density estimate of 14 crab/ha.

The Guam, and Tinian data fit well together with a very strong correlation between the CPUE, the total number of crabs captured (Chart 4.) and the estimated crab densities (Chart 5.).

Chart 4. The correlation between CPUE and the total number of Coconut Crabs captured on bait station grids on Tinian and Guam

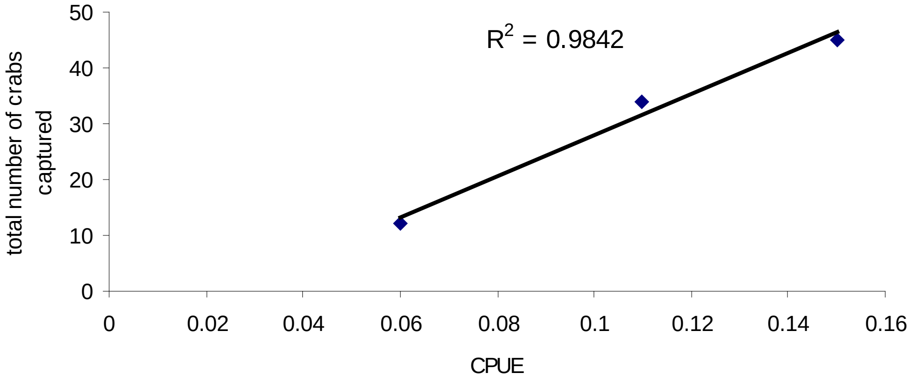
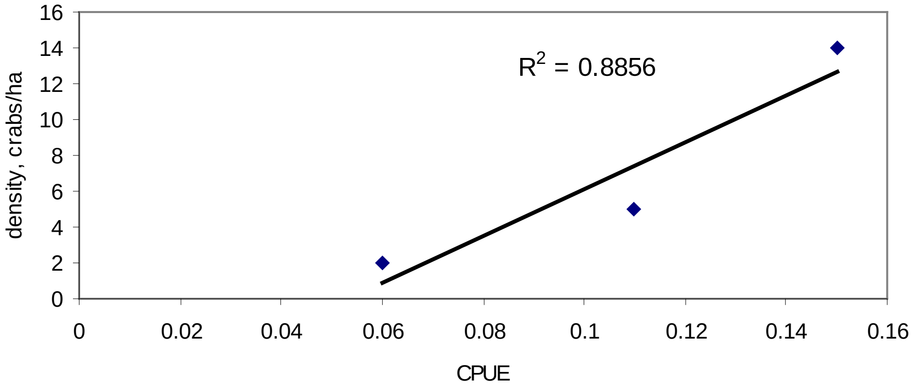
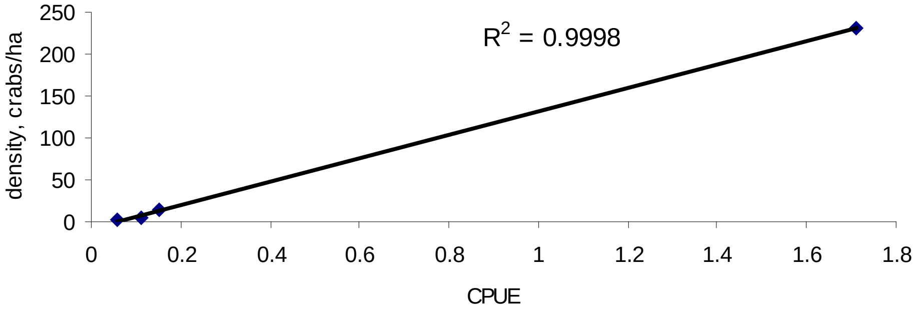


Chart 5. The correlation between CPUE and Coconut Crab densities on Tinian and Guam



When the Diego Garcia data is added, the correlation is very strong (Chart 6.). The relationship on the regression graph appears linear however there is a very large data gap between the Marianas and Diego Garcia. The relationship might not be linear as the CPUE for Diego Garcia was 15 times higher than the CPUE for native forest on Tinian but, the true density was in fact 46 times higher. We suspect that there is a leveling off or saturation point with the bait stations, in that one can only get so many crabs on one station. So, the relationship is probably sigmoid and not linear.

Chart 6. The correlation between CPUE and Coconut Crab densities on Tinian, Guam and Diego Garcia



DISCUSSION

This study has shown that the demographics of coconut crabs on Tinian favor much smaller crabs than a non-harvested population (Diego Garcia) There are few crabs over the legal size limit and population densities are much lower than the non-harvested population. Over-harvest is the suspected reason for this.

Crab surveys on Aguiguan in 2000 (Cruz *et al.* 2000) pointed to an over harvested population. The average width and weight (thoracic length was not measured) was 66 mm and 293 grams. In 2006 the average width and weight was slightly smaller at 61 mm and 245 grams. The catch rate in 2000, however, was much higher with 3.7 crabs/bait station for 1 night of trapping.

When compared with a non-harvested population, the differences are stark. At all sampling sites old bait stations from crab hunters were seen. Anecdotally, out of season poaching is reported to be rampant. There are also reports of crabs being sold (this is illegal) for $50-$100 per crab (depending on the size) (C. Sanchez pers. com.). The CNMI economy is very poor at present and it is possible that crabs are being sold to supplement household income.

The telemetry data shows 2 predominant movement patterns by the crabs. Crabs occupy a small area for a short period of time or constantly move in random directions and distances. The movement patterns increase the likelihood of over-harvest as one does not need to move trap lines after an area has been trapped out. Crabs will quickly move back into the area. Kessler (2006) documented similar movements on Saipan, where a 3.25 hectare grid was trapped out in one week and 2 months later a similar number of crabs were caught on the same grid. This pattern is likely to contribute to the belief that crabs are very abundant on Tinian. Most people when informed we were doing a crab study commented on the abundance of crabs on Tinian. Many people also talked about how 15-20 years ago one could drive the North Field roads and runways at night and collect many crabs on the road.

Both grid trapping sites had similar capture probabilities (0.28 and 0.33). On both habitat types ~ 30% of the population can be captured on a given night. This is a high capture probability that also makes the crabs susceptible to over-harvest.

Given the movement patterns of the crabs and that North Field on Tinian has a fairly extensive road system, slowly cruising the roads at night would be a low effort method to harvest coconut crabs. Three crabs were captured on roads during this study, on the way home after checking bait stations on the grids or the transect lines. Fresh bait stations were also observed placed just off the road, 2-3 meters in the forest, leading to the Mt Lasso over look. This would also provide a quick and easy method for crab harvest.

Size data and telemetry suggest that small crabs are being harvested. The transmitter found on the tarmac of North Field appeared to have been purposefully removed by a poacher, as it was 210 meters from the release site out on the tarmac with no cover or vegetation The other transmitter that fell off did so within 2 meters of the capture site, suggesting that the putty had not dried sufficiently before the crab was released. This crab was far below legal size at 28 mm TL, 58 mm width and 154 grams weight.

Sustainable crab harvest should not exceed the natural, annual, mortality of the legal sized population (Fletcher and Amos, 1994). This was estimated to be 5% of the population. With the Tinian population estimate for Navy lands this calculates to 21 crabs that can be harvested this year and 20 crabs for Aguiguan.

Given the evidence for large scale harvesting, poaching, and the skewed demographics it is obvious that the Tinian (and quite possibly Aguiguan) population is being harvested at an unsustainable rate.

MANAGEMENT RECOMMENDATIONS

The obvious recommendation is to stop out of season poaching with increased law Enforcement, strictly enforce size and bag limits during the crab season and keep records of this sizes of harvested crabs. In light of the current economic situation on Tinian and the dollar value for a single crab, this could prove very difficult.

The best way to help the crab population recover would be a 5 year harvest moratorium, with population monitoring. After the 5 year period is up, allow a well regulated and controlled harvest. The population would be estimated as was done in this study and then a harvest goal would be set. Once this goal is reached, then the season would be closed.

The 5 year time period is somewhat arbitrary but enforcing a longer period of time is probably not feasible.

Because the coconut crab is such a popular game species a total crab moratorium might not be feasible. A more realistic approach might be to close select zones to crabbing for 5 years while allowing crab harvest elsewhere. At the end of 5 years the closed zone is opened and other areas are closed.

More information is needed on small sized crabs (< 20mm TL). A specific study on this size class is recommended to quantify juvenile densities, juvenile recruitment and establish distribution patterns.

We also recommend doing more work on calibrating the CPUE index with the true population density. The data gap between the lower and higher densities needs to be filled. As a management tool, being able to quickly estimate population densities from 2 or 3 night bait station transects would be invaluable.

REFERENCES

Brown, I.W., and D.R. Fielder. 1991. The coconut crab: aspects of the biology and ecology of Birgus latro in the Republic of Vanuatu. ACIAR Monograph No. 8, 136pp. Australian Centre for International Agricultural Research, Canberra, Australia.

Chauvet, C. and T. Kadiri-Jan. 1999. Assessment of an unexploited population of coconut crabs, Birgus latro (Lenni', 1767) on Taiaro atoll (Tuamotu archipelago, French Polynesia). Coral Reefs (1 999) 18:297-299.

Cruz, J., L. Arriola, N. Johnson and G. Beauprez. 2000. Wildlife and vegetation surveys Aguiguan 2000. Technical Report #2 CNMI-DFW, unpublished.

Esselstyn, J. Cruz, J.B. Williams, LL. and N. Hawely. 2003. Wildlife and vegetation surveys, Aguiguan 2002. Technical report #9. CNMI Division of Fish and Wildlife, Wildlife Section. P.O. Box 10007, Lower Base, Saipan, MP 96950.

Fletcher, W.J. and M. Amos. 1994. Stock assessment of coconut crabs. ACIAR Monograph No. 29. 32pp. Australian Centre for International Agricultural Research. GPO Box 157 1, Canberra, Australia 2601.

Helber Hastert and Fee, Planners. 2004. Integrated natural resources management plan: Farallon De Medinilla and Tinian military lease areas, Commonwealth of the Northern Mariana Islands. Prepared for Commander, U.S. Naval Forces Marianas.

Kadiri-Jan T. 1995. Biology and population dynamics of the coconut crab (*Birgus 1atro*) in Loyalty islanad. New Caledonia. Theses of Universite' Francaise du Pacifique New Caledonia: 155pp.

Kadiri-Jan T. and C. Chauvet. 1998. Distribution of the juvenile coconut crab, *Birgus latro* (L.), on the island of Lifou, New Caledonia. Ecoscience, 5(2): 275-278.

Kessler, C. C. 2006. Management implications of a coconut crab (*Birgus latro*) removal study in Saipan, Commonwealth of the Northern Mariana Islands. Micronesica 39 (1):3 1-39.

Liu, Z. F. and L. Fischer. 2006. Commonwealth of the Northern Mariana Islands vegetation mapping using very high spatial resolution imagery: methodology. USDA Forest Service Pacific Southwest Region, Forest Health Protection. McClennan, California.

USFWS. 2001. Assessment of the coconut crab (*Birgus Latro*) population on the Naval computer and telecommunications master station. Prepared for: Department of the Navy, Environmental Section, 'Commander U.S. Naval Forces, Marianas. Prepared by: U.S. Fish and Wildlife Service, Guam National Wildlife Refuge, Dededo Guam.

Attachment 1.

Coconut Crab Telemetry Data

Crab #	**Sex**	**TL mm**	**Weight grams**	**Distances moved (meters)**	**Native forest or tangan tangan**
45	F	32	263	Crab moved off of grid immediately and could not be found till the following month. It had moved 435 meters from the capture point	native forest
50	F	26	163	20,5,5,5,32	native forest
54	M	28	194	20,20,5	native forest
56	F	32	275	75	native forest
57	M	33	366	59,61, 10	native forest
58	F	28	160	5, crab stayed in the same crevice for 5 days and nights. The following month the crab had left the grid.	native forest
59	F	33	375	Transmitter fell off within 2 meters of capture point	native forest
63	F	33	238	50	native forest
2	M	35	306	Unable to collect data due to the transmitter frequency being the exact same frequency as that of the Voice of America relay station	tangan tangan
3	F	39	371	28	tangan tangan
5	F	31	241	15	tangan tangan
8	M	28	154	Transmitter fell off within 210 meters from capture point	tangan tangan
9	M	27	172	89, 50	tangan tangan
10	F	30	204	120, 140	tangan tangan

Attachment 2.

Weights and Measures of Sampled Crabs
Aguiguan Crabs

Sex	Thoracic Length (millimeters)	Weight (grams)	Width (milimeters)
m	25	140	52
m	36	360	77
m	36	400	80
m	33	270	72
m	28	na	na
m	21	60	40
m	25	105	50
m	35	315	76
m	30	230	65
m	36	370	81
m	38	440	78
m	30	220	64
m	20	72	43
m	34	365	72
m	20	90	43
m	48	990	110
m	17	50	36
m	13	25	28
m	38	460	82
f	27	178	56
f	18	55	39
f	31	260	68
f	31	240	64
f	25	110	50
f	20	65	44

Attachment 2. Continued
Tinian Crabs

Sex	Thoracic Length (millimeters)	Weight (grams)	Width (milimeters)
f	28	205	63
f	22	64	42
f	32	315	69
m	33	345	69
m	34	360	72
m	32	230	65
f	24	102	43
f	29	158	59
m	23	98	47
m	41	535	83
f	33	325	70
f	26	158	59
m	32	255	59
m	40	525	83
m	33	265	67
f	29	206	58
f	33	325	69
m	40	490	80
m	24	115	48
m	31	234	64
m	21	76	43
m	29	250	58
m	31	270	66
f	27	178	58
m	24	114	49
f	26	140	52
m	31	200	63
f	35	274	73
f	23	93	48
m	34	334	69
m	30	181	59
m	26	115	52
f	28	205	62
m	23	90	47
m	41	490	82
m	28	175	56
f	22	93	45
m	36	390	72
f	30	220	62
f	27	173	58
m	31	245	65
f	25	135	47
m	31	245	61
f	26	155	55

Tinian Crabs Continued

Sex	Thoracic Length (millimeters)	Weight (grams)	Width (milimeters)
f	30	250	69
m	27	220	58
f	32	263	67
f	20	72	41
f	25	98	54
m	30	193	64
m	34	313	70
f	26	163	57
f	24	123	52
f	21	80	43
m	22	84	47
m	28	194	60
f	32	328	72
m	27	126	51
m	33	346	74
f	28	160	54
f	33	313	71
f	25	143	55
m	25	102	50
f	26	163	57
f	33	238	62
f	26	130	52
m	35	306	70
f	39	371	79
f	21	94	45
f	31	241	62
m	25	122	52
f	24	116	49
m	28	154	58
m	27	172	54
f	30	204	61
f	19	62	39
f	24	146	52
m	21	92	49
f	31	211	65

2.3 REPTILE AND AMPHIBIAN SURVEYS

2.3.1 REPTILES AND AMPHIBIANS OF TINIAN

Littoral Skink, Photo courtesy of USGS, BRD

Cane Toad, Photo courtesy of USGS, BRD

Prepared by: **Gordon H. Rodda, Robert N. Reed, Shane R. Siers, Thomas J. Hinkle, Thomas H. Fritts, and Robert P. Reynolds; U.S. Geological Survey, Biological Resources Disicpline, Fort Collins Science Center, Fort Collins, CO**

INTRODUCTION

This work covers the terrestrial (as opposed to marine) reptiles and amphibians of Tinian. For marine turtles see US Fish and Wildlife Service (1996), Pultz *et al.* (1999), and Kolinski *et al.* (2004). This report describes all herpetofauna within the land environment, including subterranean (= fossorial), terrestrial, and arboreal species.

The data reported herein were generated primarily during field work supported by the U.S. Fish and Wildlife Service using funding from the U.S. Navy. Why might the Navy care about the status of reptiles on Tinian? Presumably the Navy is interested in conserving the diversity of life on lands that they manage by preserving required habitats, or minimizing detrimental habitat change. Accordingly, the 2008 fieldwork was conducted exclusively on Navy-leased lands of Tinian and the geographic scope of this report is Tinian's Military Lease Area. In addition, military operations of all services may impact Tinian's biodiversity and quality of life by accidentally introducing new species to Tinian. Thus a focus of this study is the interaction between native and introduced species. Finally, the health and resilience of ecosystems is often reflected in the composition of the ecological communities found in a place, and the Navy may wish to conserve that health and resilience both for the direct benefits to residents of the Northern Mariana Islands and for the greater latitude it provides for addressing ecological problems created elsewhere. For example, Tinian might become a refuge for bird or lizard species extirpated on Guam, thereby minimizing operational restrictions that might otherwise impact military activities on Guam. Alternately, if the Brown Treesnake or another invasive species were to become established on Tinian, military use of Tinian would be negatively impacted because shipments leaving Tinian for pest-free areas would present a biosecurity risk to destination sites; this would necessitate inspection and interdiction efforts for military cargo, with associated costs in time and money. Thus the Navy can prevent future expenses by maintaining a healthy natural community on Tinian. Toward that end, we provide a snapshot of the terrestrial reptiles of Tinian in 2008, and compile historical records indicating change over time.

In this report we assume that the conservation of native species is warranted, but the protection of introduced species is not. Why do native reptiles deserve precedence over introduced species? The easiest example is the ecological disaster associated with the introduction of the Brown Treesnake on Guam (Savidge 1987; Fritts and Rodda 1998). In that case, the introduction of a single new reptile eliminated many of the native birds, bats, and lizards of Guam (Rodda *et al.* 1999). The net result was a spectacular loss of global biodiversity, but despite this, Guam now has more reptile and amphibian species than it did prior to the arrival of the Brown Treesnake. The additions to Guam were common widespread species, whereas the lost species were unique local ones. From Guam's perspective, local biodiversity increased. The snake's arrival spread a common species over an additional island, but eventually removed unique local bird and bat species from the entire world – global biodiversity decreased. As the bottom line is global biodiversity, species introductions are normally detrimental and should be avoided.

At 10,180 hectares, Tinian is the sixth largest of the ~2500 islands of Micronesia. Despite its relatively large size, Tinian has experienced only two herpetological inventories. The first islandwide compendium was published in 1948 (Downs 1948), though the entomologist Townes (1946) gave earlier useful notes. Downs collected 35 specimens, but missed one common species (Mourning Gecko, *Lepidodactylus lugubris*) and at least five rarer ones. Owen (1974) provided much useful information, but did not attempt a comprehensive inventory. Forty years later Wiles *et al.* (1989) produced the first comprehensive review, including not only original field work but also a review of all earlier literature records. Rodda *et al.* (1991) put the Tinian inventory of Wiles *et al.* (1989) into the context of the entire Mariana archipelago. The methods of Wiles *et al.* (1989) however did not lend themselves to quantification of the population densities of any species; their surveys provided only qualitative descriptions of relative density. Thus Wiles *et al.* (1989) provided a baseline inventory, but did not attempt the task of monitoring populations. This study provides the first quantification of population densities, including field data from 1989 and 2008.

Pregill (1998) made a major contribution to our understanding of the nativeness of various species by sampling subfossil cave remains on several Mariana Islands including Tinian. He found that only the Brahminy Blindsnake, *Ramphotyphlops braminus* was unequivocally native (occurred in prehuman layers), though the prehuman sediment sample was very small. On biogeographic grounds, and species' occurrence in the earliest prehistoric strata, it is likely that several other species arrived in the Marianas unaided by humans, as detailed in the species accounts. An unexpected result of Pregill's study was the discovery that several species previously thought to have been in the Marianas since antiquity arrived on Tinian only following the time of Western contact

(Oceanic Gecko, *Gehyra oceanic;* Mutilating Gecko, *Gehyra mutilata;* Mangrove Monitor, *Varanus indicus*) and were undoubtedly human introductions on Tinian.

MATERIALS AND METHODS

Two methods were used islandwide in 1989: glueboard surveys and visual searches. Three methods were used in 2008, exclusively in the Military Lease Area: glueboard surveys, visual searches at night, and total removal plots. All study sites are mapped in Figure 1. Habitat names follow Mueller-Dombois and Fosberg (1998), with the exception of the "mixed" category, which is intermediate in composition and structure between *Leucaena* forest and limestone forest.

Glueboard sampling. - Glueboard sampling methods are detailed in Rodda *et al.* (1993, 2005b). The glueboards used in the 1989 sampling on Tinian were of several manufacturers, and were subsequently determined to be suboptimal for density studies. Therefore, the 1989 samples (Table 1) are not directly comparable to the 2008 samples (Table 2), which were conducted with paper mouse glueboards (Victor, Lititz PA). These traps were set individually in lines of 12 on the ground in shade. The traps were separated by at least 5 meters and aligned in either a straight line parallel to a road, shoreline, or trail edge, or in a ring surrounding a total removal plot. The traps were checked every 30 min for three morning hours (2008) or periodically throughout the day for 24 h (1989 and 2008). Capture rates are expressed as captures per trap-hour.

Table 1. Glueboard sampling of Tinian, 1989. Throughout this document, latitudes and longitudes are given to the precision implied by the number of significant digits and all are in the WGS84 projection.

Site	Latitude	Longitude	Micro-habitat	Time of day	Trap -Hrs	Date (1989)	Lizard Captures
***Pemphis* coastal zone**							
PTAH	N 15.1011	E 145.6449	Rocky shore	0730-0730	1152	15-17 Aug	24
Limestone forest							
CPNB	N 14.94	E 145.640	Forest floor	0930-1200	318	17-18 Aug	6

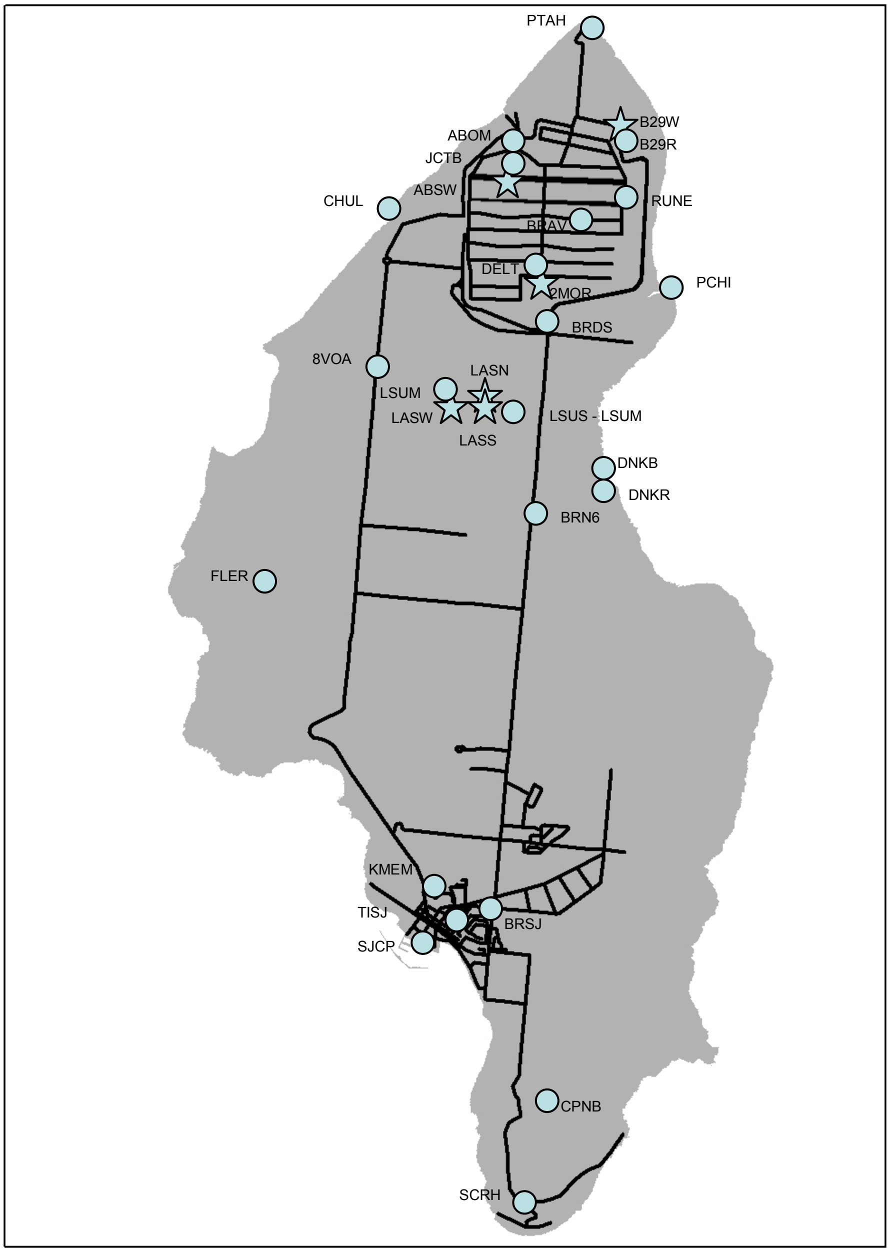


Figure 1. Outline map of Tinian showing sites sampled in 1989 or 2008. Starred sites incorporated total removal sampling. Coordinates for each site are given in the appropriate accompanying tables.

Table 2. Glueboard sampling of military lease lands on Tinian, 2008.

Site	Latitude	Longitude	Micro-habitat	Time of day	Trap -Hrs	Date (2008)	Lizard Captures
***Pemphis* coastal zone**							
PTAH	N 15.1011	E 145.6449	Rocky shore	0800-1100	36	17 July	4
CHUL	N 15.073	E 145.616	Rocky shore	0837-1137	42	17 July	10
Strand coastal zone							
CHUL	N 15.073	E 145.616	Forest floor	0830-1150	39.6	17 July	10
DNKB	N 15.030	E 145.648	Forest floor	0800-1100	36	17 July	11
***Leucaena* forest**							
ABSW	N 15.0776	E 145.6327	Forest floor	0733-1033	36	24 June	22
ABSW	N 15.0776	E 145.6327	Tree trunks	0730-0715	285	24-25 June	0
B29W	N 15.0855	E 145.6495	Forest floor	0758-1058	36	24 June	12
B29W	N 15.0855	E 145.6495	Tree trunks	0800-0730	282	24-25 June	1
FLER	N 15.0198	E 145.5884	Forest floor	0740-1055	72	17 July	44
DNKR	N 15.030	E 145.648	Forest floor	0740- 1040	36	17 July	23
Mixed (*Leucaena* – Limestone) forest							
2MOR	N 15.063	E 145.638	Forest floor	0810-1110	36	24 June	13
2MOR	N 15.063	E 145.638	Tree trunks	0740-0740	288	24-25 June	0
LASW	N 15.042	E 145.626	Forest floor	0819-1134	39	28 June	25
LASW	N 15.042	E 145.626	Tree trunks	0735-0735	288	28-29 June	0
Limestone forest							
LASN	N 15.0422	E 145.6302	Forest floor	0905-1205	36	28 June	16
LASN	N 15.0422	E 145.6302	Tree trunks	0833-0803	282	28-29 June	0
LASS	N 15.0410	E 145.6298	Forest floor	0900-1200	36	28 June	8
LASS	N 15.0410	E 145.6298	Tree trunks	0850-0835	286	28-29 June	0
LSUS	N 15.04	E 145.63	Forest floor	0800-1105	72	6 July	6

Visual search method. - Visual search methods are detailed in Rodda *et al.* (2005a). Briefly, the searchers worked individually, walking at about 0.5 km/h, scanning the vegetation on one side of a trail or road, usually at night with the aid of a headlamp. Each reptile seen was identified to species and characterized by its perch height and perch taxon, though the latter data will not be reported here. Relative densities are expressed as captures per unit effort (detections per searcher-hour); sample sizes are given separately for 1989 (Table 3) and 2008 (Table 4). The headlamps used in 1989 were relatively dim narrow-beam dry cell lights (Justrite, Des Plaines IL). Those used in 2008 were the brighter and broader beam Brunton (Riverton WY) and Mila (Sweden) lamps used by orienteering teams (similar to Mila lamps whose effectiveness is reported in Lardner *et al.* 2007, under review).

Table 3. Visual searches of Tinian, 1989. All but the one at Puntan Chiget (PCHI) were at night.

Site	Latitude	Longitude	Search-Hrs	Date (1989)	Lizard Detections
Urban					
JCTB	N 15.080	E 145.633	0.83	8 Aug	10
SJCP	N 14.966	E 145.621	0.53	17 Aug	35
TISJ	N 14.97	E 145.625	1.74	13-16 Aug	30
Coastal habitats					
CHUL	N 15.073	E 145.616	2.36	12 Aug	47
PCHI	N 15.06	E 145.655	0.18	15 Aug	2
***Leucaena* forest**					
ABOM	N 15.08	E 145.63	11.64	9-10 Aug	80
RUNE	N 15.075	E 145.65	3.83	8-13 Aug	9
SCRH	N 14.924	E 145.632	4.56	14 Aug	30
Mixed (*Leucaena* - Limestone) Forest					
BRDS	N 15.055	E 145.639	0.47	15 Aug	12
BRN6	N 15.03	E 145.637	1.00	15 Aug	12
BRSJ	N 14.97	E 145.632	0.60	12 Aug	16
KMEM	N 14.975	E 145.623	1.14	12 Aug	13
Limestone Forest					
LSUS	N 15.04	E 145.63	1.20	15 Aug	0
CPNB	N 15.94	E 145.640	1.99	16 Aug	21

Table 4. Nighttime visual searches of military lease lands on Tinian, 2008.

Site	Latitude	Longitude	Search-Hrs	Date (2008)	Lizard Detections
***Leucaena* forest**					
8VOA	N 15.0505	E 145.6134	1.30	24 July	20
B29R	N 15.09	E 145.64	4.62	15 July	28
BRAV	N 15.074	E 145.650	5.83	15 July	32
DELT	N 15.07	E 145.64	3.80	15 July	24
DNKR	N 15.030	E 145.648	3.96	16 July	39
FLER	N 15.020	E 145.588	4.14	16 July	10
Mixed (*Leucaena* - Limestone) Forest					
BRN6	N 15.026	E 145.637	4.10	18 July	215
LSUM	N 15.04	E 145.63	4.20	16 July	27
Limestone Forest					
LSUS	N 15.04	E 145.63	5.88	9 July	10

Total removal methods. - Total removal methods are described in detail in Rodda *et al.* (2001). In brief, our objective was to physically isolate a 10 × 10 m patch of forest such that no lizards (other than very large *Varanus indicus*) could leave or enter. Arboreal lizard movement was blocked by canopy separation, and terrestrial movement was prevented by erection of a 0.4 m-tall fence of aluminum flashing which was buried in the ground to block shallow subterranean escape, and sprayed with white lithium automotive grease to discourage climbing. The vegetation was then cut down, carefully inspected, and removed in small quantities to discover all non-fossorial non-volant vertebrates present.

To prevent arboreal lizards from fleeing during canopy separation, canopy separation was conducted during the day, when almost all of Tinian's arboreal species are in refugia. To prevent terrestrial lizards from fleeing during erection of the aluminum flashing, fence emplacement occurred at night when the terrestrial species (almost all are diurnal) were in refugia. Three species of lizard and one toad could potentially escape because their activity periods are anomalous in this regard: Cane Toads (*Rhinella marina*) and Pacific Slender-toed Gecko (*Nactus pelagicus*) lizards were potentially capable of escaping on the ground because they are terrestrially-active at night while the fence was being erected; Green Anoles (*Anolis carolinensis*) and Emerald Skinks (*Lamprolepis smaragdina*) are likewise theoretically capable of escaping because they are active in the trees during the day (for example, they might flee the area during canopy separation). We do not believe that these species avoided detection on a large scale by these measures, but we were not able to rigorously quantify any leakage that might have occurred.

The locations of the total removal plots on Tinian (Table 5) were selected by the Navy representative Scott Vogt, in order to best accommodate the technical challenges of erecting a lizard-proof fence while surrounding an area of characteristic vegetation. Thus the exact plot localities were chosen to maximize vegetation representativeness and soil depth (Table 6). They were not chosen with any knowledge of the constituent reptile densities and therefore should be unbiased reptile density samples.

Table 5. Characteristics of total removal sampling plots used on military lease land on Tinian, 2008. The date range is for vegetation modification stages only. The listed "Person-Hrs" is the effort needed to remove and inspect the vegetation. Roughly an equal amount of time was required for other tasks.

Site	Latitude	Longitude	Area (m^2)	Dates (2008)	Person-Hrs	Lizard Captures
***Leucaena* (Tangantangan) Forest**						
ABSW	N 15.0776	E 145.6327	100	25-26 June	49	17
B29W	N 15.0855	E 145.6495	100	5-7 July	72.5	33
Mixed (Limestone-*Leucaena*) Forest						
2MOR	N 15.0631	E 145.6377	100	2-4 July	64	23
LASW	N 15.0422	E 145.6264	100	8-10 July	79	50
Limestone Forest						

LASN	N 15.0422	E 145.6302	100	11-14 July	122	41
LASS	N 15.0410	E 145.6298	80	29 June – 1 July	100	32

We can more precisely understand the relationship between species density and habitat features using the vegetation measurements of each plot (summarized in Table 6). In general, the basal area of *Leucaena* and the count of all woody stems (> 1 cm dbh) decreased in the sequence *Leucaena*-mixed-limestone forest and the number of large stems (>10 cm dbh) and the total vegetative biomass increased in the same habitat order. The dominant ground cover was about 60% leaf litter in all cases, but the amount of coarse woody debris tended to increase in the *Leucaena*-mixed-limestone forest sequence.

Table 6. Vegetative characteristics of the total removal plots. Reported woody percentages are of basal area for the entire plot (all woody stems > 10 mm diameter breast high (dbh)). Reported groundcover percentages are mean ground coverage. The secondary groundcover "CWD" is coarse woody debris.

	Leucaena forest		Mixed forest		Limestone forest	
Site	ABSW	B29W	2MOR	LASW	LASN	LASS
Dominant tree (%)	*Leucaena* (100%)	*Leucaena* (100%)	*Morinda* (73%)	*Leucaena* (66%)	*Leucaena* (40%)	*Premna* (29%)
Secondary tree (%)	-	-	*Leucaena* (24%)	*Aglaia* (21%)	*Premna* (18%)	*Pisonia* (20%)
Canopy height (m)	6.3	6.5	6.3	6.3	7.5	8.0
Wet veg biomass (K kg)[1]	1.4	2.2	2.3	2.1	3.2	3.5
Stems > 10 mm dbh[1]	191	91	112	136	75	44
Stems > 100 mm dbh[1]	0	6	3	7	16	9
Dominant groundcover (%)	Litter (60%)	Litter (53%)	Litter (58%)	Litter (68%)	Litter (64%)	Litter (59%)
Secondary groundcover (%)	Grass (31%)	Herbs (31%)	Ferns (25%)	CWD (13%)	CWD (20%)	CWD (16%)
Litter depth (mm)	22	25	19	13	17	14

[1]. Values for 80 m^2 LASS adjusted to 100 m^2. All other counts based on 100 m^2 plots.

Detection probability estimation and missed ratios. - We paired glueboard sampling with total removal plots to assess the trap detection probabilities of various species. The glueboards were placed as close to the plot as is possible while assuring a 7 m separation between traps (to minimize intertrap interference). On Tinian we always conducted the paired glueboard trapping prior to disturbance of the plot vegetation.

Similarly, we paired visual surveys with total removal sampling whenever the geography of surrounding vegetation made practical visual surveys in the same habitat sampled by total removal. This too allowed an estimate of detection probability from the absolute densities documented in the total removal plots. Detection probability is of interest primarily because it can be used to estimate the proportion of animals that are overlooked in a survey. For example, if the detection probability is 0.25, one out of four animals were detected, on average. Another way to state this is that four animals were usually present for every one that was seen (in other words, we missed three of four). This is the way we state it in this paper, a value we will call the "missed ratio," the inverse of detection probability. In our example the missed ratio of 4.0 indicates that if we saw 8, 32 were most likely present.

When the "missed ratio" is multiplied by the mean detection rate for a habitat-by-species combination (e.g., the Indo-Pacific House Gecko, *Hemidactylus frenatus,* in mixed forest) we obtain an estimate of absolute density for the habitat sampled. This allows a key check on whether the absolute densities reported from total removal plots were representative of the habitat. For example, using missed ratios we estimated that the Mourning Gecko, *Lepidodactylus lugubris,* was generally more common in *Leucaena* habitat than they were in our *Leucaena* total removal plots, but that our total removal samples of the Mutilating Gecko, *Gehyra mutilata,* were very closely matched to their sampled densities in almost all habitats (species specific evaluations included under species accounts).

While missed ratios are especially useful in this way, the pattern of missed ratios is also helpful in understanding which habitats, islands, and species are particularly favorable for detection (few missed) or particularly difficult (many missed). Tables of missed ratios are given under the relevant species accounts.

Validity of the sampling methods. - Of the various sampling techniques, total removal has the highest face validity (Rodda *et al.* 2001), in that the local population is totally enumerated ("censused") rather than sampled ("surveyed"), but the total removal method is not strictly applicable to all species. For example, it is not intended for use on subterranean species such as the Brahminy Blindsnake, *Ramphotyphlops braminus*. Species that may aestivate underground – here the Cane Toad, *Rhinella marina* - could also be missed. Large climbing lizards such as the Mangrove Monitor, *Varanus indicus* can probably vault the barrier and are at such low density that quantification using total removal plots is unlikely to be informative. For the appropriate species, however, total removal sampling is unequivocal and precise, with no ambiguity about the size of the area sampled (unlike index methods and mark-recapture, for which quantification of the area sampled can be elusive) or the number of individuals found therein. Total removal is also the only method under consideration that provides size distributions, biomasses, sex ratios and other unbiased demographic information.

Glueboard sampling seems to work best for strictly terrestrial species such as the Curious Skink, *Carlia ailanpalai* (Rodda *et al.* 2005b). The technique's primary weakness is that it is imprecise (wide confidence limits) and index values are often not strictly proportional to absolute abundance. Thus, for example, an index value of 6 cannot be interpreted as having twice the absolute population density found in a site where the same species has an index value of 3. Glueboard capture rates cannot be legitimately compared among species, and other restrictions may apply (Rodda *et al.* 2005b). For example, it may not give appropriate relative abundances when making comparisons among habitats or islands.

Visual surveys may be the best choice for estimating relative abundance of arboreal species (Rodda *et al.* 2005a), though the confidence intervals may be even wider than for glueboard samples, indicating low precision. Species-specific modulators of visual detection can be inferred from the visual missed ratio tables given in the species accounts.

RESULTS

Glueboard sampling. - Glueboard yields are given in Table 7 (1989) and Table 8 (2008).

Table 7. 1989 Glueboard capture rates (captures per trap-hr). An empty cell indicates that appropriate trapping did not occur for the indicated species at the indicated site. See species accounts for details of appropriate conditions. See Table 1 for placement and number of trap-hours at each locality. Single additional specimens of *Lepidodactylus lugubris* and *Varanus indicus* were caught at CPNB. Glueboards used in 1989 were of a different adhesive and configuration than in 2008, so capture rates are not directly comparable.

	Terrestrial native			Terrestrial introduced	Arboreal introduced	
	Cryptoblepharus. poecilopleurus	*Emoia atrocostata*	*Emoia caeruleocauda*	*Carlia ailanpalai*	*Gehyra mutilata*	*Hemidactylus frenatus*
Coastal habitats						
PTAH	0	0	0	0.005		
Limestone forest						
CPNB	0		0.006	0.062		

Table 8. Glueboard capture rates 2008 (captures per trap-hr) based on appropriate conditions for each species (arboreal species only in tree traps; nocturnal species only in night sets, etc.). An empty cell indicates that appropriate trapping did not occur for the indicated species at the indicated site. See species accounts for details of these conditions. See Table 2 for placement and number of trap-hours at each locality. Omitted lizards had capture rates of zero.

	Terrestrial native			Terrestrial introduced	Arboreal introduced	
	Cryptoblepharus. poecilopleurus	*Emoia atrocostata*	*Emoia caeruleocauda*	*Carlia ailanpalai*	*Gehyra mutilata*	*Hemidactylus frenatus*
Coastal habitats						
CHUL	0.012	0.095	0	0.748		
PTAH	0		0	0.083		
DNKB	0		0	0.306		
$\overline{X}$	0.004		0	0.379		
***Leucaena* forest**						
ABSW	0		0	0.611	0	0.004
B29W	0		0	0.333	0	0
FLER	0		0	0.611		
DNKR	0		0	0.639		
$\overline{X}$	0		0	0.549	0	0.002
Mixed forest						
2MOR	0		0	0.361	0	0
LASW	0		0	0.641	0	0
$\overline{X}$	0		0	0.501	0	0
Limestone forest						
LASN	0		0.028	0.417	0	0
LASS	0		0	0.139	0.014	0
LSUS	0		0	0.083		
$\overline{X}$	0		0.009	0.213	0.007	0

Visual searching. - Visual detection rates are in Table 9 (1989) and Table 10 (2008).

Table 9. Detection rates (sightings/person-hr) during visual surveys on Tinian, 1989. See Table 3 for person-hrs and sample sizes. Omitted lizards had sighting rates of zero.

	Arboreal Native	Terrestrial introduced	Arboreal introduced (or potentially introduced)				
	Lepidodactylus lugubris	*Carlia ailanpalai*	*C. poecilo-pleurus*	*Gehyra mutilata*	*Gehyra oceanica*	*Hemidactylus frenatus*	*Lamprolepis smaragdina*
Urban							
JCTB	3.077		0	0	0	12.048	0
SJCP	0		0	0	0	66.038	0
TISJ	1.724		0	4.598	0	10.345	0.575
Coastal habitats							
CHUL	0.758		1.695	2.119	0	11.441	0
PCHI		11.111	0				0
***Leucaena* forest**							
ABOM	6.186		0	0.172	0	0.515	0
RUNE	9.512		0	0.261	0	0.522	0
SCRH	1.667		0	2.632	0.658	1.096	0.219
Mixed (*Leucaena* - Limestone) Forest							
BRDS	2.128		0	0	0	23.404	0
BRN6	1.000		0	3.000	0	9.000	0
BRSJ	15.000		0	0	0	10.000	1.667
KMEM	4.386	0.877	0	4.386	0	0.877	0.877
Limestone forest							
LSUS	0		0	0	0	0	0
CPNB	2.010		0	3.015	3.518	2.010	0

Table 10. Detection rates (sightings/person-hr) during nighttime visual surveys on military lease lands, Tinian, 2008. Values for BRN6 are somewhat approximate, as the large number of simultaneous detections made it difficult to scrupulously avoid double counting of moving individuals. See Table 4 for person-hrs and capture sample sizes. Omitted lizards had sighting rates of zero. BRN6 was also unusual in being isolated trees in the median of a road; therefore it was not averaged with LSUM to characterize mixed forest visual detection rates. *Anolis carolinensis* and *Lamprolepis smaragdina* were actively spreading their ranges at the time of our samples; therefore it was not appropriate to characterize their relative abundances in *Leucaena* forest by combining sites where the species was present with those sites not yet reached.

	Arboreal Native	Arboreal introduced (or potentially introduced)				
	Lepidodactylus lugubris	*Anolis carolinensis*	*Gehyra mutilata*	*Gehyra oceanica*	*Hemidactylus frenatus*	*Lamprolepis smaragdina*
***Leucaena* forest**						
8VOA	3.077	4.615	0	0	3.077	4.615
B29R	0.433	0	1.082	0	4.329	0
BRAV	0.172	0	0	0	4.631	0
DELT	1.316	0	2.105	0	2.105	0
DNKR	0.758	0	0	0.253	7.828	0.505
FLER	1.691	0	0.242	0	0.242	0.242
$\overline{X}$	1.241		0.572	0.042	3.702	
Mixed forest						
BRN6	9.512	0	1.463	0	39.3	1.707
LSUM	1.667	1.190	0.952	0.238	0.238	0.476
Limestone forest						
LSUS	0.340	0	0.340	0.170	0.170	0.340

Total removal sampling. - For comparisons within a species it is appropriate to consider the absolute densities revealed by total removal sampling (Table 11), but for comparisons among species it is perhaps more appropriate to consider the biomass distribution, as one individual of a large species may consume many times the energy and space that is occupied by a smaller species (Table 12, Figure 2). The most striking attribute of the biomass distribution by fundamental niche (Table 12) is the paucity of terrestrial species biomass, especially in limestone forest, where it constituted only about 1% of lizard biomass. Native species did not constitute as much as half of the lizard biomass in any sampled habitat.

Table 11. Densities of each herpetofauna species on Tinian, as revealed by total removal plots on military lease lands, 2008. Densities are given in units of individuals per hectare, based on sampling of approximately 0.01 hectare.

	Leucaena forest		Mixed forest		Limestone forest	
	ABSW	B29W	2MOR	LASW	LASN	LASS
Subterranean native species						
Ramphotyphlops braminus	0	0	0	100	0	125
Terrestrial native species						
Cryptoblepharus poecilopleurus	0	100	0	0	0	0
Emoia caeruleocauda	0	0	0	600	0	0
Arboreal native species						
Lepidodactylus lugubris	100	800	1000	2600	2700	1875
Perochirus ateles	0	0	0	0	100	0
Terrestrial introduced species						
Carlia ailanpalai	1100	500	800	300	0	125
Rhinella marina	0	0	100	200	400	125
Arboreal introduced (or potentially so) species						
Anolis carolinensis	0	0	0	300	0	0
Gehyra mutilata	0	400	500	1100	500	1125
Gehyra oceanica	0	0	0	0	500	875
Hemidactylus frenatus	500	1500	0	0	0	0
Lamprolepis smaragdina	0	0	0	100	300	0

Table 12. Biomass densities of each lizard species on Tinian, as revealed by total removal plots on military lease lands, 2008. Biomasses are given in units of kilograms per hectare, based on sampling of approximately 0.02 hectare (two total removal plots in each habitat). Omitted lizards had capture rates of zero.

	Leucaena forest		Mixed forest		Limestone forest	
Terrestrial native						
Cryptoblepharus poecilopleurus	0.09		0		0	
Emoia caeruleocauda	0		0.48		0	
Terrestrial native subtotal (% of total)	0.09	(2)	0.48	(9)	0	(0)
Arboreal native						
Lepidodactylus lugubris	0.35		1.80		1.70	
Perochirus ateles	0		0		0.15	
Arboreal native subtotal (% of total)	0.35	(8)	1.80	(33)	1.85	(16)
Terrestrial introduced						
Carlia ailanpalai	1.90		0.78		0.09	
Terrestrial introduced subtotal (% of total)	1.90	(45)	0.78	(14)	0.09	(1)
Arboreal introduced (or possibly so)						
Anolis carolinensis	0		0.44		0	
Hemidactylus frenatus	1.52		0		0	
Gehyra mutilata	0.36		1.29		1.08	
Gehyra oceanica	0		0		5.78	
Lamprolepis smaragdina	0		0.63		2.42	
Arboreal introduced total (% of total)	1.88	(45)	2.36	(43)	9.28	(83)
Terrestrial combined (native and introduced): % of total	1.99	(47)	1.26	(23)	0.09	(1)
Arboreal combined (native and introduced): % of total	2.23	(53)	4.16	(77)	11.13	(99)
Native combined (arboreal and terrestrial): % of total	0.44	(10)	2.28	(42)	1.85	(16)
Grand total (kg/ha)		4.22		5.42		11.22

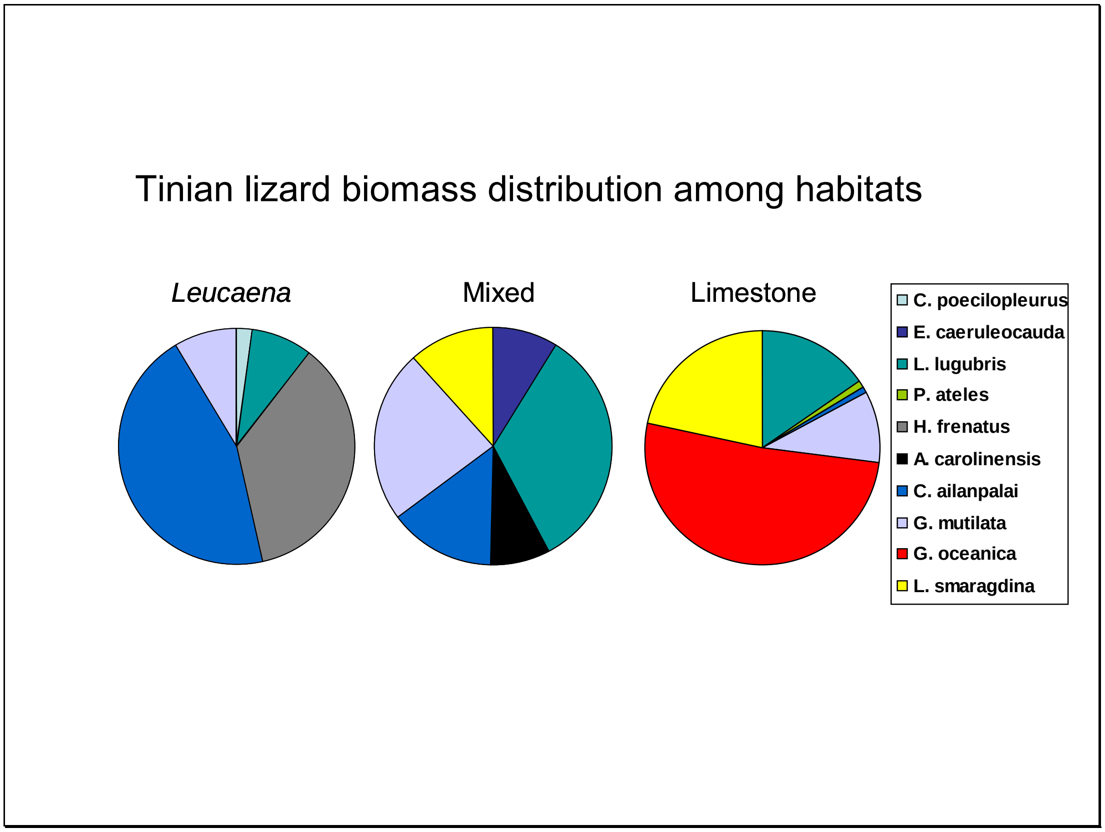


Figure 2. Distribution of lizard biomass among lizards and habitats on military lease lands on Tinian, 2008, as indicated by total removal plots. Amphibians and snakes have been omitted from this table to reflect their uncertain sampling. Species are listed in the same order as the tables (terrestrial native, then arboreal native, and so forth), beginning at twelve o'clock and continuing clockwise.

Species accounts

Native species

Oceanic Snake-eyed Skink, *Cryptoblepharus poecilopleurus*

Body length[1]: 22 - 47 mm Mass: 0.2 - 1.8 g

Figure 3. The Oceanic Snake-eyed Skink, *Cryptoblepharus poecilopleurus.*

Previous studies – Note that the nominal species is under revision and is likely to contain a number of island endemics in the South Pacific (Horner 2007, G. Zug, 2008 pers. comm.), but the form in the Mariana Islands is relatively widespread in the northwestern Pacific. Because this littoral clade has extensively speciated on islands (reflecting an evolutionarily long residence in the area of speciation), and because this particular species is endemic to the northern Pacific, it is assumed that this species reached many islands on its own (i.e., it was not introduced by man). Because it is very small and its skeleton is fragile, it is not a good candidate for preservation as a subfossil in prehuman remains. Thus we assume it is likely native despite the absence of reported subfossils in prehuman strata (Pregill 1998).

This species is found in a variety of microhabitats, including *Casuarina* (Australian pine) groves, rocky and sandy areas, grass, leaf litter around *Cocos* (coconut) palms, etc. (McCoid *et al.* 1995). The unifying factor in this range of habitat types, however, is that these microhabitats must be closely associated with the littoral zone. For example, *C. peocilopleurus* has been collected on and around *Casuarina*, but only when the trees are immediately adjacent to the shore. Vogt and Williams (2004) report occasional specimens from upland situations (limestone forest implied by not explicitly stated) on Saipan and the Northern Mariana Islands, though these may be associated with cliffs (also found around upland cliffs on Rota (Rodda, pers. obs.). The exception is on Guguan, where this species occurs throughout the island as a sand swimmer in ash fields (McCoid *et al.* 1995). Vogt (2008) comments that its former presence on Sarigan

---

[1] Sizes given above photographs are ranges for specimens from the Mariana Islands.

(Northern Mariana Islands) may have been attributable to soil disturbance associated with dense populations of goats. It is possible that goat-churned soil could provide a loose soil niche similar to that found in ash fields on Guguan.

The first record of this species on Tinian was three specimens collected by Norm Scott and Herman Muna from *Leucaena* trees at the shrine at Puntan Tahgong (our site PTAH) in 1985 (Wiles *et al.* 1989). These *Leucaena* trees were immediately adjacent to a *Pemphis* belt that occurs in the salt spray zone of the point. We trapped the same place four years later (Table 1), but found only *Carlia* (Table 73). We did find *Cryptoblepharus poecilopleurus* in strand forest at nearby Unai Chulu in 1989 (Tables 3, 9). These were the only published records for Tinian prior to this study.

This study (2008) – We found the Snake-eyed Skink (*Cryptoblepharus poecilopleurus*) at only two sites: adjacent to Unai Chulu (beach) and in the B29W total removal plot just northeast of North Field. The Unai Chulu population was expected, as the species is associated with strand habitat. However, the B29W total removal plot was nearly 1 km inland, in North Field's characteristic monotypic stands of *Leucaena*. To the best of our knowledge, this is the first record for this species anywhere in *Leucaena* habitat or at a great distance from cliff or strand habitat. On the windward side of North Field the *Leucaena* stands were stunted by the prevailing winds and salt spray, and therefore the B29W site was not as far inland in a habitat sense as simple distance from the coast would suggest. Nonetheless, this discovery undermines our confidence in predicting the full distribution of the species on Tinian. Prior to this discovery we would have confidently predicted that the species would occur only immediately adjacent to salt water, especially in the vicinity of *Casuarina* stands, strand vegetation or cliffs. However, the detection in a *Leucaena* stand indicates that it might occur in a variety of sites on the military lease lands of Tinian. It is not, however, present throughout, as we found it in only the two sites mentioned.

Management recommendations - The nominal species has an extensive distribution throughout the northwestern Pacific, though some of these localities may be of closely-related species. It is found along the coast of virtually all of the Mariana Islands, including the far northern islands. As presently understood the species is not considered to be at risk of endangerment or in need of special management. As with all of Tinian's native species, the most important protection is prevention of new introductions. It is notable that Hawley (2008) and Vogt (2008) observed a recent apparent decline of this species on Sarigan; Vogt suggested that the species may benefit from soil disturbance by ungulates. Monitoring soil conditions in conjunction with monitoring populations of this species may shed light on limiting factors.

Littoral Skink, *Emoia atrocostata*

Body length: 28 - 85 mm Mass: 0.6 – 11.0 g

Figure 4. The Littoral Skink, *Emoia atrocostata.*

Previous studies – This species was not previously known to occur on Tinian, though it was known to occur on small islands both north (e.g., Saipan outlier Maigo Luao) and south (Aguiguan) of Tinian (Rodda *et al.* 1991). Throughout its vast range it occurs in two habitat types: mangrove mud flats and rocky coasts having tide pools. In Palau, it occurs in both of these habitat types. In the Marianas it is known only from tide pool areas (hence the alternate common name: Tide-pool Skink), especially associated with the shrub *Pemphis*, which occurs in the salt-sprayed area immediately inland of the intertidal zone on high-energy rocky shorelines. This is the only suitable habitat for this species on military lease lands of Tinian.

Because this species is strictly limited to the littoral zone, it would not be expected to be found in the upland caves sampled for subfossil material by Pregill (1998), and it was not found there. However, as this taxon is a superlative colonizer of remote islands, and is endemic to the Pacific basin (including Indo-Pacific areas), we treat it as a native species.

This study (2008) – We found the Littoral Skink (*Emoia atrocostata*) to be reasonably common in the *Pemphis* zone north of Unai Chulu. We did not find it in similar habitat at Puntan Tahgong (Ushi Point), or in strand forest 10-20 m inland from the *Pemphis* zone of Unai Chulu. *Carlia ailanpalai* is the dominant terrestrial lizard in the *Pemphis* zone at Puntan Tahgong and several other places where we looked, but it is likely that additional populations of *E. atrocostata* will be discovered in *Pemphis* habitat.

Management recommendations – As this skink is widely distributed (Pacific and Indian Oceans) and common, this is not a species of special concern. However, the absence of the skink from typical habitat such as that found at Puntan Tahgong, and its possible replacement by the introduced Curious Skink (*Carlia ailanpalai*) suggests that monitoring of these two species be continued in the *Pemphis* zone to determine whether the introduced skink displaces the native one.

Pacific Blue-tailed Skink, *Emoia caeruleocauda*

Body length: 21 – 56 mm Mass: 0.1 – 3.7 g

Figure 5. The Pacific Blue-tailed Skink, *Emoia atrocostata.*

Previous studies – The colorful and conspicuous Pacific Blue-tailed Skink (*Emoia caeruleocauda*) is found from Borneo to Vanuatu and throughout the western Pacific on the ground and low in vegetation in forested areas (Brown 1991). Pregill (1998) found it in early prehistoric subfossil material, but did not record it in prehuman strata. However, it is endemic to western Oceania and therefore is presumably native to at least some of the islands therein. For that reason we treat it as native to the Mariana Islands.

In the Marianas it is the only common native skink still found throughout most islands (Rodda *et al.* 1991). However, on Tinian it has been largely replaced (Wiles *et al.* 1989) by the introduced Curious Skink. It is not known if this replacement has been due to direct interaction between the species or an indirect interaction, such as a reciprocal response to a habitat feature (e.g., one species prefers drier areas; the other prefers wetter areas). Previous studies have found the blue-tailed skink to be largely missing from the extensive *Leucaena* stands on military lease lands of Tinian (Wiles *et al.* 1989).

This study (2008) – Our observations corroborated earlier studies showing this species to be rare or possibly absent from most *Leucaena* habitat on military lease lands of Tinian. We found it only in or near native forest on Mt. Lasu (LASW total removal plot; seen near LASS; one trapped at LASN) in low numbers. It was present in the one of our mixed forest total removal sites that was on Mt. Lasu, but was absent from the one on North Field. From this limited information it is impossible to determine whether the difference is geographic or due to proximity to native forest.

The pattern of its abundance in the Marianas (Figure 6) indicates a dramatic difference in abundance between Saipan/Tinian and the islands further south (Guam/Rota). Although this skink appears to be significantly reduced in the presence of the Brown Treesnake (compare upper left bar with the one directly below it in Figure 6), the skink reaches very high abundances on Guam even in the presence of snakes, especially in *Pandanus* habitat, but also in all forested habitats. For example, there are individual localities in *Leucaena* habitat on Guam where the blue-tailed skink attained densities of 6300 per hectare, so we can infer that *Leucaena* forest is appropriate habitat for this species. However, we did not find it in *Leucaena* habitat on either Saipan or Tinian, and even in its preferred habitat of limestone forest it was rare on both islands. The commonality between Saipan and Tinian and the distinction with Rota and Guam may be the Musk Shrew, *Suncus murinus*. This large terrestrial shrew is a notorious consumer of skinks (Barbehenn 1974) and was extremely abundant on Saipan and Tinian, but rare on Guam and absent from Rota (Wiewel *et al.* in press).

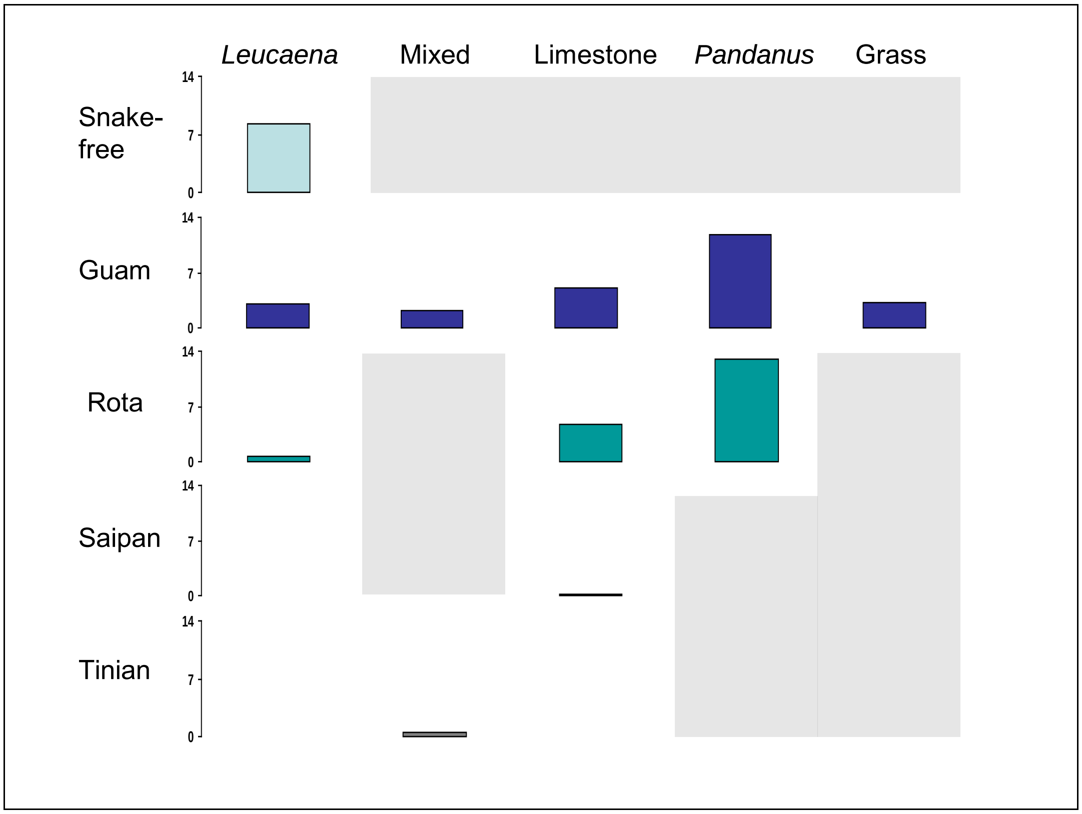


Figure 6. Patterns of abundance of *Emoia caeruleocauda*, as deduced from 40 total removal plots (each 10 × 10 m) on the four large Mariana Islands. Values given are biomass densities (kg/ha). Sample sizes are two plots per island by habitat condition, with the exceptions of Guam (9 Leucaena, 1 Mixed, 1 Pandanus, and 7 Grassland), and 3 *Leucaena* plots on Rota. The grayed-out combinations of habitat and island were not sampled. White-backed areas with no bars indicate zero abundance. The snake-free plots were samples collected in snake exclosures 12 months after snake-removal on Guam, and are provided as a contrast to the abundance indicated immediately below it

(i.e., Guam – *Leucaena*) to indicate the great short-term reduction in lizard abundance associated with snake presence on Guam. Over a longer term we expect and see evidence of contrary, indirect impacts on lizard abundance, probably via the abundance of other lizard-eating animals (shrews, kingfishers, rails, etc.) also reduced by snake predation.

Management recommendations – Although not globally rare (due to its extensive geographic range), this characteristic native species of Mariana forests appears to have been extirpated from most of the military lease lands on Tinian. Retention of the populations that remain probably hinges on retention of limestone forest habitat and prevention of new species introductions. The economic benefits of such retention are unknown, and the ecological benefits have been little studied vis-à-vis this lizard. Townes (1946) and McCoid (1997) found the lizards to be generalized insectivores of non-ant species. Bailey (1976) found a degree of specialization on lepidopteran larvae; thus their presence could benefit agriculture. However, the most important economic contribution of this species may be as a food item for species valued by tourists: Slifka *et al.* (2004) found this to be a highly nutritious prey for kingfishers. Presumably it provides the same benefits to other saurophagous (lizard-eating) birds such as bitterns.

Mariana Skink, *Emoia slevini*

Body length: 20 – 77 mm Mass: 0.4 – 10.4 g

Figure 7. The Mariana Skink, *Emoia slevini*.

Previous studies – The Mariana Skink, *Emoia slevini*, is found only in the Mariana Islands. Pregill (1998) did not detect this species in prehuman strata, but did find it to dominate skink remains in all prehistoric strata. We assume it is therefore native to all of the Mariana Islands. It was first found on Tinian immediately after World War II (Brown and Falanruw 1972, Rodda *et al.* 1991), but has not been detected on Tinian since then (Wiles *et al.* 1989).

This study (2008) – We did not detect this species.

Management recommendations – This species has disappeared from the large southern Mariana Islands in the last 50 years, for no obvious reason (McCoid *et al.* 1995a). Whatever the reason, it may apply to all four large Mariana Islands. Study of this species where it still occurs (far northern Mariana Islands: Alamagan, Asuncion, Guguan, Pagan, Sarigan: Rodda *et al.* 1991) is needed to develop a management strategy to preserve this species, which is endemic to the Mariana Islands and has been extirpated from the bulk of its historic range.

Mourning Gecko, *Lepidodactylus lugubris*

Body length: 19 – 49 mm Mass: 0.1 – 2.7 g

Figure 8. The Mourning Gecko, *Lepidodactylus lugubris.*

Previous studies – As currently understood, the triploid species *Lepidodactylus lugubris* is a parthenogenetic hybrid derived from diploid *Lepidodactylus moestus* and an undescribed species (Radtkey et al. 1995). Because the present distributions of the parental stocks overlap only in Micronesia, the presumption is that the species arose in Micronesia, or at least somewhere in Oceania. Thus even though no prehuman fossils of this very delicate species have yet been detected in the fossil record (Pregill 1998) we presume this species to be native. This species has been found to be widely distributed in Oceania and reasonably common throughout the Mariana Islands, including Tinian (Wiles *et al.* 1989, Table 9).

This study (2008) – We found evidence of this species in all habitats and all localities considered. It was numerically the most abundant lizard in both mixed and limestone forest habitats (Table 11), but due to its small size it was not responsible for the greatest portion of biomass in any site (Table 12, Figure 2). The Mourning Gecko was one of only two species (the other *Hemidactylus frenatus*) sighted at every locality subjected to visual searches (Table 10). It is known to occur from intertidal habitats to undisturbed upland forest (Sabath 1981), and the evidence from Tinian supports the general conclusion that this species may be found everywhere on the island.

Although widespread, the Mourning Gecko tends to be less conspicuous than its numerical abundance would suggest. In comparison to the Indo-Pacific House Gecko (*Hemidactylus frenatus*) in *Leucaena* habitat on Tinian for example, the Mourning Gecko was about half as abundant as the house gecko in the total removal plots (mean of 450/ha v. 1000/ha: Table 11). Yet in terms of visual sighting rates (Table 10) it averaged only

1.24 detections per hour compared to the house gecko's 3.70 sightings per hour, a *Hemidactylus/Lepidodactylus* ratio of about threefold (compared to an absolute abundance ratio of about twofold). This suggests that the Mourning Gecko is more difficult to sight than the house gecko.

We can quantify the species' visual detectability with reference to the missed ratios reported in Table 13. The anticipated association between higher missed ratios and visually-obstructed habitats was observed on both Guam and Tinian (compare *Leucaena* and mixed habitats to the other, denser vegetation types). On both islands, limestone forest and *Pandanus* forest habitats had elevated missed ratios (about threefold that of other habitats). However, Mourning Geckos were also about threefold more difficult to detect on Guam than on Tinian (compare matched habitat types). This might be due to the presence of Brown Treesnakes (*Boiga irregularis*), as suggested by the much lower missed ratios for the snake-free samples in *Leucaena* habitat on Guam (mean of 815 vs. 3034). Comparably lower missed ratios of Mourning Geckos are estimated for Tinian, suggesting that Mourning Geckos in snake-infested areas of Guam are less visible, presumably because they are also hiding from foraging snakes on Guam. The mean sighting rate in *Leucaena* forest on Tinian (1.24: Table 10) in combination with the missed ratio in *Leucaena* forest on Tinian (1223: Table 13) implies a mean absolute population density of about 1517/ha in *Leucaena* forest on Tinian. This is about threefold the 450/ha mean density observed in our two total removal plots, suggesting that our total removal plots may have inadvertently sampled areas of relatively low Mourning Gecko density for that habitat type.

Table 13. Patterns of visual missed ratios of *Lepidodactylus lugubris* in the Mariana Islands. The layout and sample sizes of this table follows those of Fig 6. The value expressed is the mean ratio of absolute density assessed in total removal plots to number of detections per hour in visual searches of adjacent vegetation. Higher values therefore indicate lower detectability and a higher proportion of individuals overlooked. "Undefined" indicates a mean value of zero in the numerator (total removals = 0) or denominator (no visual detections). We expect higher missed ratio values in habitats with low visual penetration (limestone forest, *Pandanus,* grass).

Island	*Leucaena*	Mixed	Limestone	*Pandanus*	Grass
Snake-free	815				
Guam	3034	3934	11085	10958	Undefined
Rota	Undefined		Undefined	Undefined	
Saipan	Undefined		Undefined		
Tinian	1223	1158	3155		

Despite this species' high density, we did not detect it with tree-based glueboards (Table 8). Thus care should be taken when judging the density of *Lepidodactylus* from detection or capture rates. Based on the perches occupied by the *Lepidodactylus* seen, the species appears to have a preference for twig-end or foliage perches. Such places are more difficult to search visually, or to trap, than are the trunk/limb locations favored by other geckos. This generalization may not be applicable to all twig-end species: day-active lizards sighted at night in trees (*Anolis, Lamprolepis*) are conspicuous because they are bright green and sleep at the very ends of branches, often hanging out into trails, roadways, or other easily-searched venues.

Throughout the Marianas (Figure 9), the Mourning Gecko is ubiquitous and reasonably abundant in all forested habitats. It does better when snakes have been removed (see increase on Guam when "snake-free": Figure 9), but does not show any dramatic differences in abundance between the Mariana islands, suggesting that none of the other introduced predators (e.g., shrews or rats) greatly affect its numbers. It appears to do slightly better in more mesic habitats, such as limestone forest or *Pandanus*.

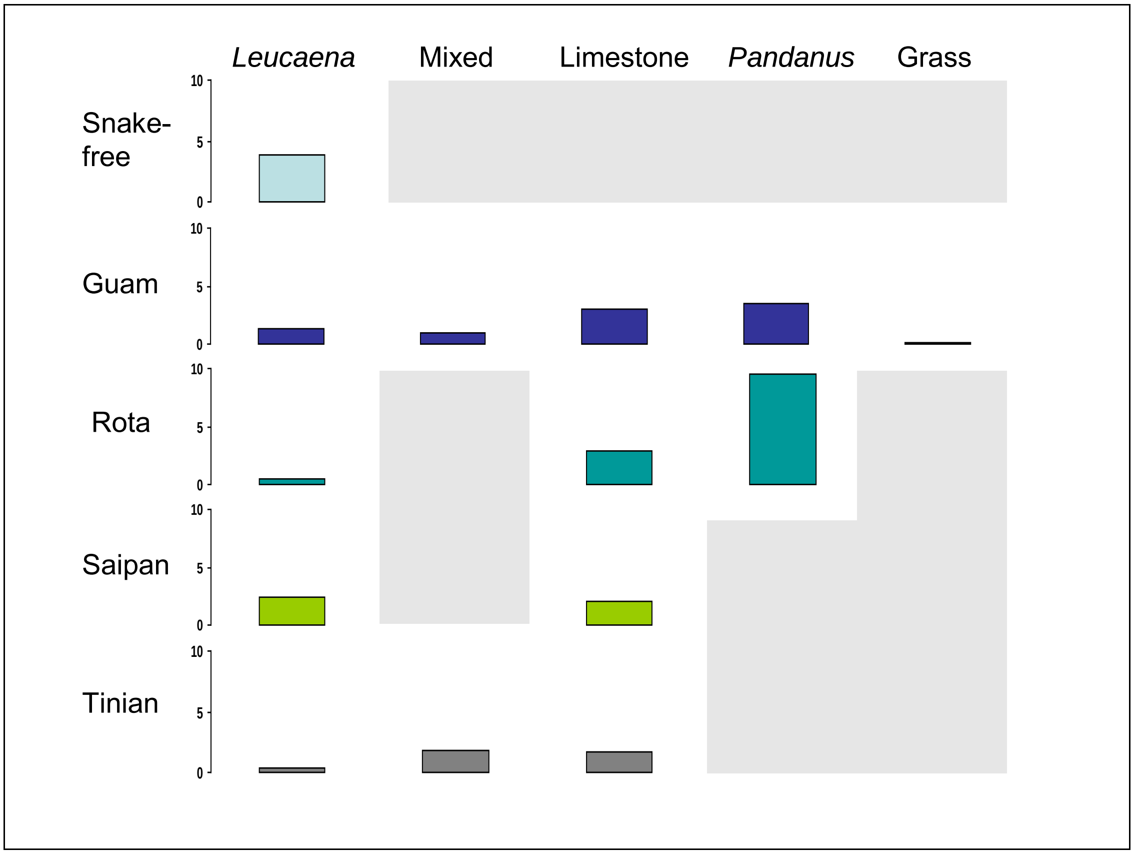


Figure 9. Patterns of abundance of *Lepidodactylus lugubris*, as deduced from 40 total removal plots (each 10 × 10 m) on the four large Mariana Islands. Symbols and sample sizes as in Figure 6.

Time trends. - There are two gecko species with sufficient data to begin exploring possible changes in abundance over time on Tinian: *Lepidodactylus lugubris* and *Hemidactylus frenatus*. The best samples are forested habitats, especially *Leucaena* and limestone forests. In 1989 the Mourning Gecko was relatively abundant in *Leucaena* forest, especially in the vicinity of North Field. The mean sighting rate in *Leucaena* forest in 1989 (Table 9) was 5.79/hr, but this includes relatively high sighting rates (mean 7.85/hr) at North Field (ABOM and RUNE) and a relatively low sighting rate (1.67/hr) at the southern tip of Tinian (SCRH). In 2008 our mean sighting rate (Table 10) at North Field (B29R, BRAV, DELT) for this species was only 0.64/hr, a 92% reduction in sighting rate.

Before treating sighting rates as proportional to absolute abundance it is important to consider the possibility that the 1989 searchers were more effective than the 2008 searchers. It is notable that the 1989 searches took place in early August (Table 3), almost exactly the same season as the 2008 searches (July). The gecko species most visually similar to *L. lugubris* is *Hemidactylus frenatus*, which exhibited a sharp increase in the number of sightings 1989 to 2008. Considering only North Field localities (ABOM and RUNE in 1989; B29R, BRAV, and DELT in 2008), the mean sighting rate for *Hemidactylus frenatus* increased from 0.52/hr to 3.69/hr, a 600% increase. Thus it appears unlikely that the decrease in *Lepidodactylus* sightings was due to reduced searcher skill.

Combining the contrary trends in these two gecko species we find that the total number of visual detections in 1989 at North Field for these two species were in the ratio 78:8 or about 93% *Lepidodactylus*, whereas the comparable data for 2008 were 8:55 or only 13% *Lepidodactylus*. From the absolute sighting rates we derive the impression that *Lepidodactylus* has declined in density at North Field and *Hemidactylus* has increased. As these two species are presumably each other's closest competitor, these trends may not be independent; the ascendance of the larger species (*Hemidactylus*) may be partially or wholly spurring the decline of *Lepidodactylus*.

We know of no clear hypothesis to account for this change, as the species have coexisted on Tinian for at least a thousand years (Pregill 1998). However, it is notable that a similar decline in *Lepidodactylus* and concurrent increase in *Hemidactylus* has been noted in total removal plots for similar habitat (*Leucaena* forest north of Northwest Field) on Guam in the period 1995-1999: at the beginning of the period *Lepidodactylus* was numerically slightly dominant 1750/ha v. 1550/ha and constituted about 53% of the two species' counts, but in four years time had declined to only 23% of the combined counts (800/ha v. 2650/ha). These absolute densities are consistent with the Tinian experience in indicating both a decline in *Lepidodactylus* and an increase in *Hemidactylus*. Though the scale of the change is smaller on Guam, the time interval is also shorter; the annual rate of change is roughly comparable. We have no explanation for the change.

<u>Management recommendations</u> – As the species concept is presently applied, the Mourning Gecko is broadly distributed throughout the world (having been introduced in both Africa and the New World), ubiquitous in all habitats on Tinian, and common in all habitats. If what we perceive to be a single Mourning Gecko species turns out to include several cryptic species, some of them rare or highly localized, we would need to reevaluate the assumption that the conservation of this species is assured. The latter scenario is possible, as the species concept is difficult to apply to this parthenogenetic (clonally reproducing) form, and there are many identified strains or clones of this nominal species (Ineich 1988). The clonal representation on Tinian has not been investigated or quantified as it has for nearby areas (Yamashiro *et al.* 2000). Even if the

current conception of the species concept is correct, an introduced insectivorous lizard occupying the same nocturnal twig-end niche could potentially displace it in the Mariana Islands. The most likely competitive displacement of the Mourning Gecko would be by other clones of the same superspecies, as has been suggested by Yamashiro *et al.* (2000). Clarification of clonal composition on Tinian would be of value in understanding the species' apparent population decline.

Micronesian Gecko, *Perochirus ateles*

Body length: 19 – 65 mm Mass: 0.2 – 6.7 g

Figure 10. The Micronesian Gecko, *Perochirus ateles*.

Previous studies – This species is endemic to Micronesia, and it was the only gecko found in prehistoric subfossil remains on Tinian (Pregill 1998). Thus although it has not yet been detected in prehuman strata, it is highly likely that it is native to the Mariana Islands. Only two specimens of this species were reported prior to this (2008) work; one was collected on Mt. Lasu in 1946 by H. K. Townes (Wiles *et al.* 1989; see also Townes 1946), and the second by Haldre Rogers on southern Tinian (Carolinas Plateau) 12 Aug 2003 (USNM 561148). Scott R. Vogt (US Navy) reported a recent (February 2007) sighting in the vicinity of Mt. Lasu. McCoid and Hensley (1993, 1994a, b) provided useful natural history data from elsewhere in the Marianas.

This study (2008) – A single specimen was taken from a limestone forest total removal plot on Mt. Lasu. No others were seen or trapped.

Management recommendations – Based on the few specimens recently detected in the Mariana Islands (Cocos Island, Rota Island, Saipan Island), and the suggestion by Pregill (1998) that the prehistorically common *Perochirus* tends to be displaced by introduced *Gehyra oceanica*, it seems prudent to consider the endemic Micronesian Gecko to be at risk from the introduced *Gehyra*. Due to the large number of islands on which *Perochirus ateles* occurs naturally in Micronesia, it would appear to be less threatened with global extinction than is the more narrowly endemic *Emoia slevini*, and the prospect for retaining this species on Tinian is much greater in that it still occurs there, albeit in extreme rarity. Surviving populations in the southern Marianas appear to be largely limited to limestone forest, or at least habitats with large diameter perches (McCoid and Hensley 1994a). Where it is common (Buden 2007), the Micronesian Gecko is found in a diversity of habitats, including edificarian habitats as well as native forest, suggesting that the negative pressures against it in the Marianas are currently more forceful in secondary habitats. In the absence of information on what those forces might be it is difficult to ascertain whether reduction in those forces is practical, or whether preservation of the Mariana populations will hinge on maintaining the native forest habitats where the causes of endangerment apply with less force. One conundrum associated with this species' endangerment in the Marianas is that if it is endangered due to predation or competition by *G. oceanica*, why is it most common in the habitats (esp. limestone forest) where the Oceanic Gecko is most common (see species account for Oceanic Gecko)? This suggests that other hypotheses for its rarity should be evaluated. Sabath (1981) reported that Micronesian Geckos were present only in limestone forest in Guam in 1969, but events since this observation underscore the vulnerability of this species, as it is now extirpated from Guam (Rodda and Fritts 1992). In addition to protecting this species by preventing new species introductions and retaining or restoring native limestone forest, research into the ecology of this species would be useful in understanding its habitat requirements. The Micronesian Gecko is yet common in a variety of habitats in the Caroline Islands (Buden 2007), providing practical opportunities for its study.

Brahminy Blindsnake, *Ramphotyphlops braminus*

Body length: 59 – 151 mm Mass: 0.1 – 1.2 g

Previous studies – Pregill (1998) found the blind snake to be present in the Mariana Islands since at least early prehuman times; thus is unquestionably native. A variety of reports document its presence on Tinian (Cagle 1946c, Downs 1948, Wiles *et al.* 1989), but none has endeavored to establish its distribution or abundance on Tinian.

This study (2008) – We found *Ramphotyphlops braminus* in both mixed and limestone forest total removal plots, but we did not actively search for it in any sites and the total removal method is poorly suited to detection of this species. Elsewhere in the Marianas we have found this species in *Leucaena* forest among many other habitats. We have no reason to believe that it is not common throughout military lease lands on Tinian.

Management recommendations – This parthenogenetic snake presently has a pan-tropical distribution, probably due to the ability of single individuals (they are all females) to found a new population, and the propensity of this species to stow-away in plants, soil, and other protective materials. No biodiversity concerns have been suggested regarding this species.

Potentially native species

Indo-Pacific House Gecko, *Hemidactylus frenatus*

Body length: 20 – 59 mm Mass: 0.1 – 3.9 g

Figure 12. The Indo-Pacific House Gecko, *Hemidactylus frenatus*.

Previous studies – The Indo-Pacific House Gecko (*Hemidactylus frenatus*) may be a complex of several species (N. Arnold, 2007 pers. comm.; A. Bauer 2007 pers. comm.), but as presently recognized it is one of the world's most widespread geckos, introduced throughout the New and Old World tropics and sub-tropics. For this reason, many authors assume that this species was carried to Oceania only through human agency, but Pregill (1998) found it in prehistoric strata that predate the arrival of all other introduced vertebrates, including rats. Thus it may be native to the western part of Micronesia, though evidence from eastern Micronesia suggests it was a human introduction there and in Polynesian sites further east (Pregill 1998). It was the first gecko studied on Tinian (during World War II: Cagle 1946a, b). Although more abundant and conspicuous in the *Leucaena* forests that make up much of the military lease lands on Tinian, it has been recorded in virtually all forest and edificarian environments (Wiles *et al.* 1989).

This study (2008) – We found this species to be the most abundant and conspicuous gecko in *Leucaena* forests (Tables 8, 10-12), but it was not found in our total removal plots in other habitats on Tinian (Tables 11, 12). Based on samples of the other habitats on nearby islands (Figure 13), however, it can survive in relatively low numbers in such habitats. It is possible that the other habitats have some as yet unrecognized limitation that prevents the house gecko from occurring in them on Tinian, but it is as least as plausible that we would have detected it in such sites on Tinian if our sampling had been more extensive. The pattern of densities in the Mariana Islands suggests that the Indo-Pacific House Gecko prefers drier, more disturbed habitats, such as *Leucaena* forest, but the gecko's absence from this habitat on Saipan (the island most similar in ecology to Tinian) has not been explained.

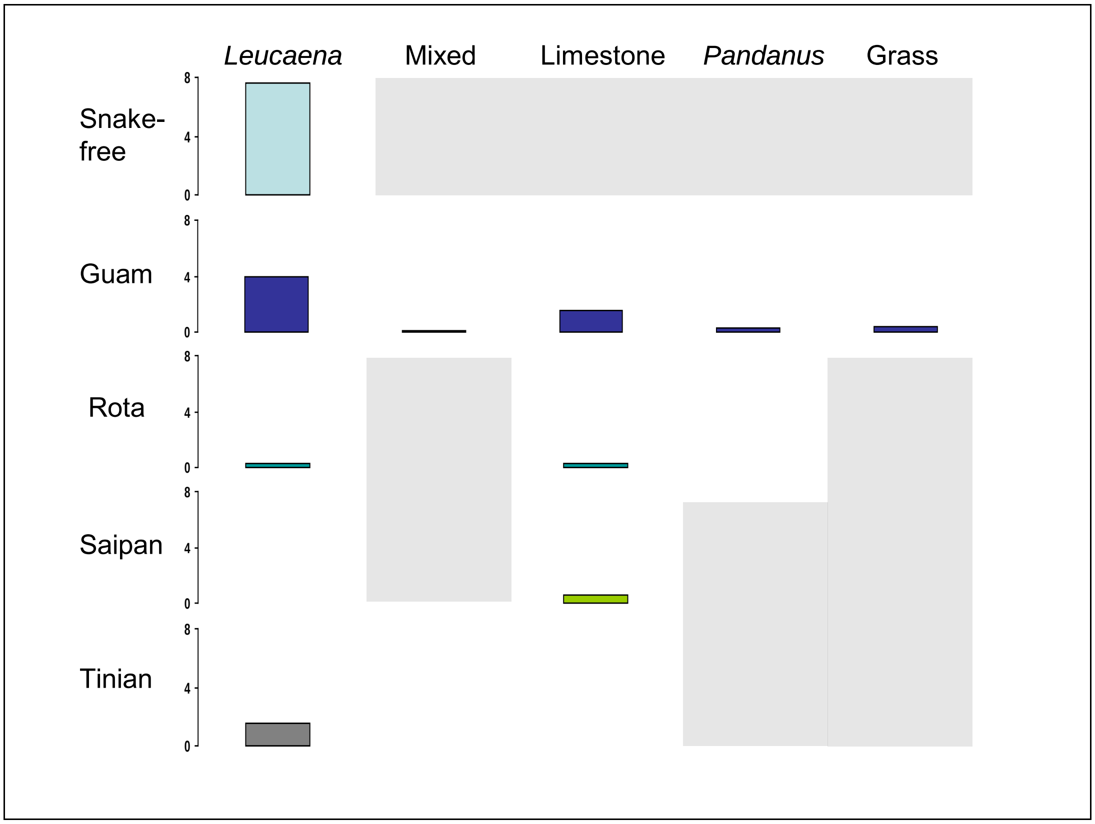


Figure 13. Patterns of abundance of *Hemidactylus frenatus*, as deduced from 40 total removal plots (each 10 x 10 m) on the four large Mariana Islands. Symbols and sample sizes as in Figure 6.

The record of missed ratios of the Indo-Pacific House Gecko (Table 14) is sparse, but suggests that the species, like the Mourning Gecko (Table 13), reacts strongly to the presence of Brown Treesnakes, becoming much harder to detect where the snakes are present. Compare the low missed ratio in Brown Treesnake-free *Leucaena* habitat of Guam (174) and Tinian (259) to the much higher values obtained in snake-occupied parts of Guam (2383 and 1151). The mean missed ratio for house geckos in *Leucaena* habitat on Tinian (259) combined with the mean detection rate of 3.70 indicates a probable mean absolute density of around 958/ha for this species in that habitat on Tinian. This comports very well with the mean of 1000/ha indicated by the total removal plots (Table 11), suggesting that the *Leucaena* total removal plots were well representative of that habitat on Tinian.

Table 14. Patterns of visual missed ratios of *Hemidactylus frenatus* in the Mariana Islands. The layout and sample sizes of this table follows those of Figs 6, 9. The value expressed is the mean ratio of absolute density assessed in total removal plots to number

of detections per hour in visual searches of adjacent vegetation. Higher values therefore indicate lower detectability and a higher number of animals overlooked. “Undefined” indicates a mean value of zero in the numerator (total removals = 0) or denominator (no visual detections). We expect higher values in habitats with low visual penetration (limestone forest, *Pandanus,* grass).

Island	*Leucaena*	Mixed	Limestone	*Pandanus*	Grass
Snake-free	174				
Guam	2383	Undefined	1151	Undefined	Undefined
Rota	Undefined		Undefined	Undefined	
Saipan	Undefined		Undefined		
Tinian	259	Undefined	Undefined		

See the discussion of density changes of *Lepidodactylus* and *Hemidactylus* from 1989 to 2008 on Tinian in the *Lepidodactylus* species account.

Management recommendations – This increasing pan-tropical species presents no obvious biodiversity concerns, unless the nominal species turns out to be composed of a variety of species, some of which are rare. The form in the Marianas appears to be of a widespread genotype however (Moritz *et al.* 1993).

Pacific Slender-toed Gecko, *Nactus pelagicus*

Body length: 23 – 68 mm Mass: 0.1 – 7.0 g

Figure 14. The Pacific Slender-toed Gecko, *Nactus pelagicus*.

Previous studies – *Nactus pelagicus* is widespread in the northwestern Pacific, apparently derived from a species complex in Melanesia (Zug and Moon 1995). As an all-female species (parthenogenetic), it would be an excellent candidate for natural dispersal. However, the uniformity of this species in Micronesia suggests an evolutionarily recent and human-aided dispersal. Pregill (1998) found some prehistoric but no prehuman remains; thus there remains some question as to whether this species was introduced by prehistoric settlers. There is only one recorded specimen of this species from Tinian: collected at a "forested" site in 1924 (Wiles *et al.* 1989).

This study (2008) – We did not record this species. Given the dearth of terrestrial lizard biomass on Tinian (see Table 12 and Figs 2, 6, 17), and the apparent vulnerability of this semi-terrestrial species to introduced shrews (Rodda 1992, Rodda and Fritts 1992, Fritts and Rodda 1998), it seems probable that it was eliminated from Tinian by the introduction of the Musk Shrew (*Suncus murinus*).

Management recommendations – Should the shrew be eradicated from Tinian, it would be prudent to take advantage of this event to recover this species on Tinian. However, the nominal species is globally widespread (Zug and Moon 1995) and the species is not at risk of endangerment.

Introduced species

Green Anole, *Anolis carolinensis*

Body length: 24 – 73 mm Mass: 0.2 – 9.2 g

Figure 15. The Green Anole, *Anolis carolinensis*.

Previous studies – The current *Anolis carolinensis* population on Tinian is believed to date from the late 1990s, when it was found only very near the port (G. Perry 1998 pers. comm.). It was not found during extensive surveys around San Jose 1984-1985 by Wiles *et al.* (1989) or in 1989 by us (Tables 1, 3, 7, 9). However Mayer and Lazell (1992) reported that it had colonized San Jose in 1978. If that colonization persisted into the present, it must have been exceedingly rare during the studies of the late 1980s and early and mid 1990s.

This study (2008) – We did not systematically study the abundance of this lizard around San Jose, but it was conspicuously common there during our stay. John Gourley (2008 pers. comm.) reported it to be very abundant in *Sanseveria* thickets south of the airport in 2008. We found it to be very numerous along 8th Avenue opposite the Voice of America facility and along the adjacent road to the summit of Mt. Lasu (Table 10). We detected it in the total removal plot part way up that road, but did not detect it in the total removal plots at the summit of Mt. Lasu. This leads us to suspect that the population is yet patchy and is still expanding. Further monitoring of this population expansion is warranted. This species is the only diurnal arboreal insectivorous lizard on Tinian other than the introduced *Lamprolepis smaragdina*. It is not generally known to be a threat to native lizards, though Suzuki and Nagoshi (1999) reported that *Cryptoblepharus poecilopleurus nigropunctatus* was apparently disappearing from Hahajima (Ogasawara (= Bonin) Islands) in association with expansion there of the Green Anole colonization.

Management recommendations – Unless new information emerges to suggest an adverse interaction with native lizards, management action need not extend beyond monitoring the spread of this new invader.

Curious Skink, *Carlia ailanpalai*

Body Length: 21 – 67 mm   Mass: 0.1 – 7.2 g

Figure 16. The Curious Skink, *Carlia ailanpalai*.

Previous studies – This species was introduced to Saipan prior to 1964 (Wiles *et al.* 1989), but the date of introduction to Tinian is unknown. Owen (1974) observed a similar species on Tinian, but did not collect any, so the identity of Owen's sighting cannot be determined. *Carlia ailanpalai* was widespread and abundant on Tinian by the 1984-1985 sampling of Wiles *et al.* (1989), especially in *Leucaena* forests. Note that many earlier documents refer to this species in the Mariana Islands as *Carlia fusca*, a nearly identical species found in New Guinea. Zug (2004) clarified that the form found in the Mariana Islands is that found on the island of Manus, Papua New Guinea, *Carlia ailanpalai,* and introduced in the Marianas.

This study (2008) – In our total removal plots (Table 11) we found this species to be the most abundant terrestrial species in *Leucaena* forest (mean = 800/ha) and mixed forest (mean = 550/ha), but relatively rare in limestone forest (mean = 63/ha), although all terrestrial species were rare in limestone forest, constituting only 0.09 kg/ha (about 1% of lizard biomass: Table 12). Despite its absolute rarity in limestone forest (Table 11), *Carlia ailanpalai* was the commonest terrestrial lizard in that habitat, as no other terrestrial lizard species was detected in limestone forests. Thus it was relatively the most successful species among terrestrial lizard species in all habitats, but it was not absolutely very successful compared to how well this species does on other islands (Figure 17). The value given in Figure 17 for Rota is potentially misleading for this comparison, as the colonization by *Carlia ailanpalai* of Rota at the time of sampling included only one of three total removal plots in *Leucaena* (and none in other habitats). For occupied sites, the mean absolute density in *Leucaena* forests for the Curious Skink is lower on Tinian than all other sampled sites, and an order of magnitude lower (800/ha) than on snake-free sites in *Leucaena* habitat on Guam (mean 7950/ha). As noted above for the blue-tailed skink, the conspicuously low density of Curious Skinks on Tinian is most likely attributable to the very high density of shrews on Tinian (Wiewel *et al.* in press). The density of shrews is unlikely to fully explain the scarcity of Curious Skinks in all habitats on Tinian in that the shrew itself is less common (by a factor of about 2) in limestone forest than in *Leucaena* forest (means of 24.2/ha and 52.8/ha in limestone and *Leucaena* forest respectively on Tinian: Wiewel *et al.* in press). On Guam, where shrew densities were uniformly and immeasurably low in *Leucaena*, mixed, and limestone forest (Wiewel *et al.* in press), *Carlia ailanpalai* was appreciably more dense in *Leucaena* (5456/ha: Figure 17) than in mixed (1100/ha: Fig 17) or limestone forest (350/ha: Figure 17). Thus habitat differences appear to play a modulating role in addition to the depressing influence of predatory shrews and snakes.

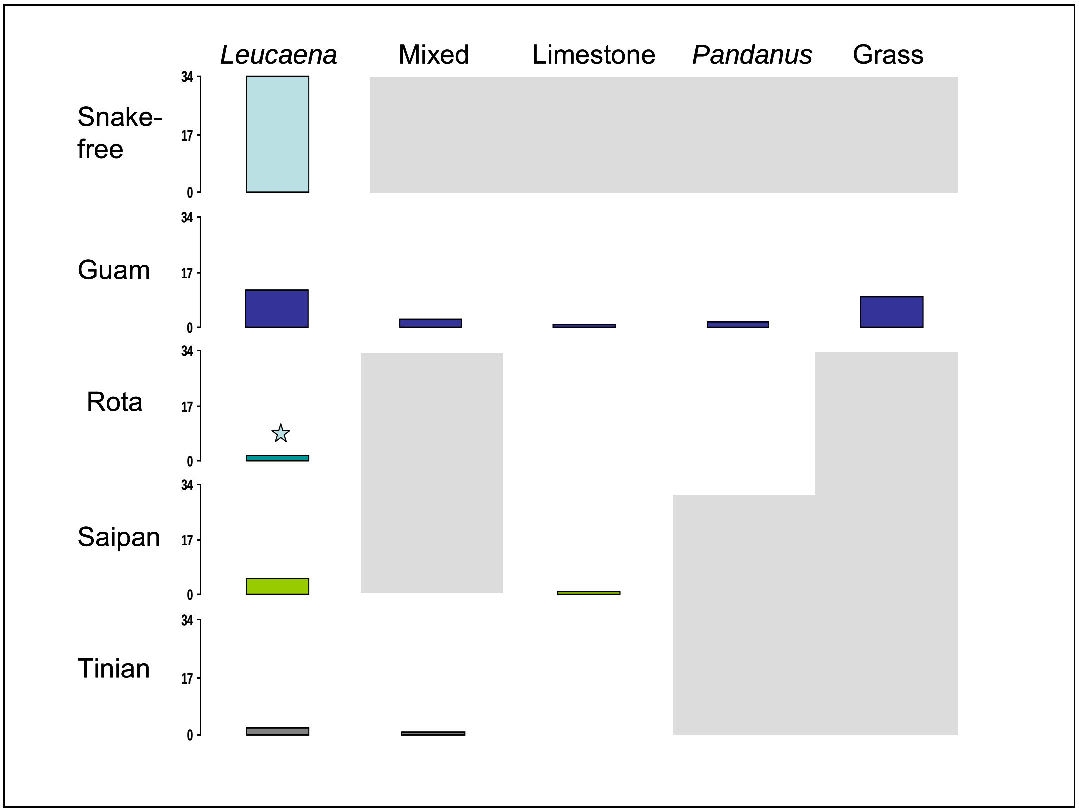


Figure 17. Patterns of abundance of *Carlia ailanpalai*, as deduced from 40 total removal plots (each 10 × 10 m) on the four large Mariana Islands. Symbols and sample sizes as in Figure 6. Although *Carlia ailanpalai* was known to be present on Rota at the time the Rota plots were sampled, the extent of colonization covered only one of the three *Leucaena* plots (star indicates bar of concern) and none of the others. In the one occupied *Leucaena* plot the biomass density was therefore three times (5.4 kg/ha) the mean shown, though this higher value may reflect densities prior to achievement of a population density equilibrium.

Missed ratios of *Carlia ailanpalai* (on traps) did not show an obvious pattern (Table 15), though the lower values on Tinian suggest trap capture may be relatively better there. To better understand the influences on detectability of Curious Skinks we built a general linear model on single plot values (not the means shown in Table 15) considering snake presence, island, density, and habitat. Density and detectability were natural log transformed to obtain normal distributions. Unfortunately, several of these variables were partially confounded, but none was found to be associated with ln(detectability) except ln(density), which had a highly significant relationship ($P < 0.0001$), with a slope of 0.454. The positive slope between the missed ratios given in Table 15 and the density shown in Figure 17 implies an inverse relationship between detectability and density. At higher densities, Curious Skinks are less trappable (see Rodda *et al.* 2005b for a comparable result). One plausible explanation is that at high densities subordinate animals are cowed into reduced activity. Another possibility is that at higher densities a skink that might otherwise run onto a glueboard is more likely to be warned of the glueboard's hazards by the struggling or presence of a previously-caught individual. Whatever the cause, the inverse relationship between density and detectability complicates the interpretation of glueboard capture rates, as there would not be a proportional correspondence between the density of the lizard and the trap capture rate. This lack of correspondence limits the utility of an index in novel situations, but does not greatly impact our use of Tinian missed ratio estimates, which are venue-specific.

Taking the observed Tinian mean missed ratios by habitat in Table 15 and the observed mean capture rates in Table 8 we compute estimated mean densities for the three habitat types of 906/ha (*Leucaena*), 672/ha (mixed), and 192/ha (limestone forest). These are reasonably congruent with the total removal plot means of 800/ha, 550/ha, and 63/ha, respectively. This suggests that our total removal plots were reasonably representative of the habitats sampled.

Table 15. Patterns of trap missed ratios of *Carlia ailanpalai* in the Mariana Islands. The layout and sample sizes of this table follows those of Figs 6, 9. The value expressed is the mean ratio of absolute density assessed in total removal plots to number of detections per trap-hour in 3 morning-h glueboard samples of adjacent vegetation. Higher values therefore indicate lower detectability and a higher number of untrapped individuals. “Undefined” indicates a mean value of zero in the numerator (total removals = 0) or denominator (no trap detections).

Island	*Leucaena*	Mixed	Limestone	*Pandanus*	Grass
Snake-free	4336				
Guam	6766	3960	1883	1008	5595
Rota	1231		Undefined	Undefined	
Saipan	2719		6300		
Tinian	1650	1342	900		

Management recommendations – Where Curious Skinks reach high densities they have been suspected of displacing native lizards through predation or competition (Vogt and Williams 2004). It is imaginable that they provide a dietary subsidy for predators such as shrews, leading to greater pressure on alternate prey, an example of “apparent competition.” However, Curious Skinks have such low densities on Tinian at present that such negative impacts are unlikely to be a major problem. Periodic monitoring should suffice to assess whether Curious Skinks remain at low density. At a landscape level it may be difficult to manipulate the density of this species, but retention or restoration of limestone forest would appear to be an effective measure (Figure 17).

Mutilating Gecko, *Gehyra mutilata*

Body length: 19 – 56 mm Mass: 0.1 – 4.3 g

Figure 18. The Mutilating Gecko, *Gehyra mutilata*.

Previous studies – Pregill (1998) established that this species was introduced to Tinian about 500 years ago. Wiles *et al.* (1989) found it to be reasonably common in the Military Lease Area and to have a patchy distribution ("widely dispersed locations") on Tinian.

This study (2008) – We found this species present in moderate numbers in all habitats studied, but the distribution was patchy in *Leucaena* habitat. For example, we found it in one of two *Leucaena*-habitat total removal plots (Table 11), and three of six *Leucaena*-habitat visual surveys (Table 10). It was found in all total removal plots and visual surveys in the other habitats. In all sites it was less numerous and represented less biomass than the Mourning Gecko, *Lepidodactylus lugubris*, but it was nowhere rare. As a proportion of the total lizard biomass (Table 12), it represented 8.5%, 24%, and 9.6% in *Leucaena,* mixed, and limestone forests respectively. This suggests that it does relatively best in mixed forest (Figure 19), which is also the habitat type in which it had the highest mean sighting rate (Table 10).

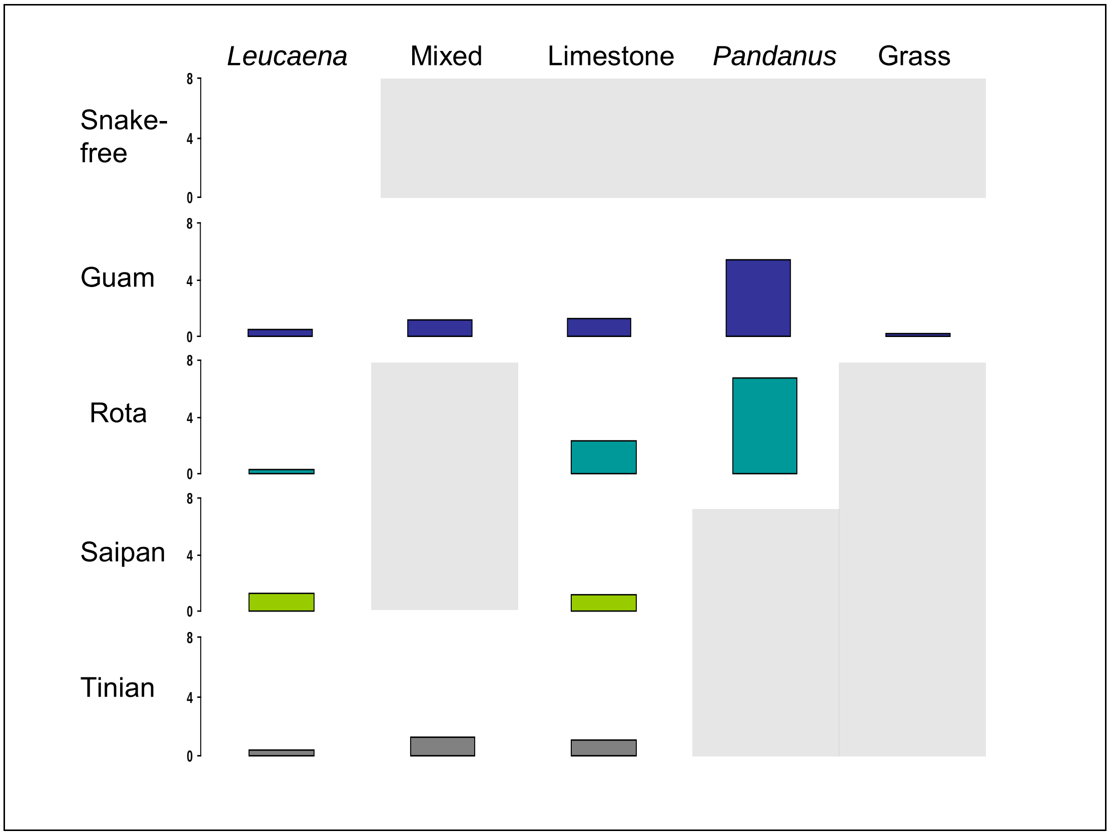


Figure 19. Patterns of abundance of *Gehyra mutilata*, as deduced from 40 total removal plots (each 10 × 10 m) on the four large Mariana Islands. Symbols and sample sizes as in Figure 6.

Gehyra mutilata is relatively difficult to see in limestone forest and *Pandanus* habitat (high values: Table 16). Although those habitats are visually obstructed, and therefore expected to have higher values than those for *Leucaena* or mixed habitats, the absolute magnitude of the missed ratios for this species (Table 16) are noticeably higher than those for *Hemidactylus frenatus*, a similar-size gecko species that relies to a similar degree on larger diameter perches (Table 14: visual missed ratio 1511 in limestone forest, compared to 14516 for *G. mutilata*).

Taking our Tinian habitat-specific missed ratios (Table 16) and mean visual detection rates for this species (Table 10), we estimate mean densities of 211/ha, 852/ha, and 813/ha for *Leucaena,* mixed, and limestone forest respectively. These estimates comport well with the corresponding total removal estimates of 200/ha, 800/ha, and 813/ha, and suggest that our total removal plots were representative of their habitats for this species.

Table 16. Patterns of visual missed ratios of *Gehyra mutilata* in the Mariana Islands. The layout and sample sizes of this table follows those of Figs 6, 9. The value expressed is the mean ratio of absolute density assessed in total removal plots to number of detections per hour in visual searches of adjacent vegetation. Higher values therefore indicate lower detectability and more overlooked individuals. “Undefined” indicates a

mean value of zero in the numerator (total removals = 0) or denominator (no visual detections). We expect higher missed ratio values in habitats with low visual penetration (limestone forest, *Pandanus,* grass).

Island	*Leucaena*	Mixed	Limestone	*Pandanus*	Grass
Snake-free	Undefined				
Guam	Undefined	1686	14516	10303	Undefined
Rota	Undefined		Undefined	Undefined	
Saipan	Undefined		Undefined		
Tinian	370	896	2390		

Management recommendations – Although this introduced gecko has the potential to eat smaller native geckos (i.e., *Lepidodactylus lugubris* or juveniles of other species), and to compete with similar-sized lizards, we see no evidence that it is having an adverse impact on Tinian. This conclusion should be re-evaluated in light of new findings when they become available.

Oceanic Gecko, *Gehyra oceanica*

Body length: 29 – 86 mm Mass: 0.7 – 14.2 g

Figure 20. The Oceanic Gecko, *Gehyra oceanica*.

Previous studies – Pregill (1998) determined that this species was introduced to Tinian about 500 years ago, and suggested that it may have negatively influenced the survival of the native gecko *Perochirus ateles*. Downs (1948) and Wiles *et al.* (1989) collected this species, with the latter declaring it to be common and widespread, with records in all forested habitats. This conclusion was consistent with our 1989 surveys of Tinian (Tables 7, 9).

This study (2008) – Although we obtained visual sightings of this species in all habitat types (Table 10), *Gehyra oceanica* was rarely found on small diameter perches, and we did not record it in either *Leucaena* or mixed forest total removal plots (Table 11). In our limestone forest plots it constituted about half (52%) of lizard biomass, a fraction over twice that of *Lamprolepis smaragdina,* the species with the next highest biomass. The preference for limestone or *Pandanus* forest is evident on all snake-free islands of the Marianas (Figure 21). This species' absence from Guam (it persists only locally in one suburban area of Guam) is presumably related to vulnerability to Brown Treesnakes (Rodda and Fritts 1992). Given this species' predilection for large diameter trees, we were surprised at the absence of individuals on large diameter trees in the middle of north Broadway (BRN6: Table 10). However, it was missing from the several searches of trees on Broadway in 1989 also (Table 9). Perhaps the isolated trees of Broadway are too dry an environment for this moisture-favoring species (Figure 21).

We have too few data comparing visual sighting rates to total removal yields for a meaningful analysis of missed ratios.

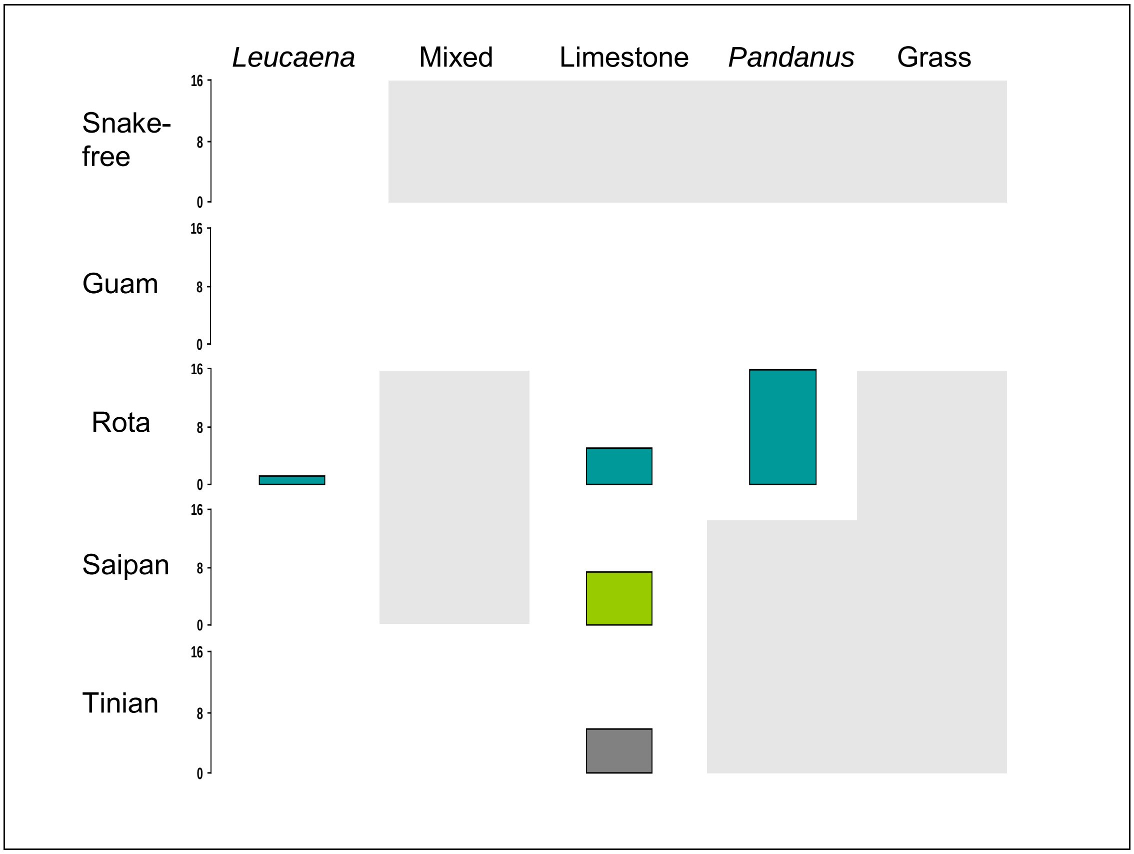


Figure 21. Patterns of abundance of *Gehyra oceanica*, as deduced from 40 total removal plots (each 10 × 10 m) on the four large Mariana Islands. Symbols and sample sizes as in Figure 6. Although *Gehyra oceanica* was known to be present on Guam throughout the

period when these Guam total removal plots were sampled, it was not known to be present in the vicinity of the sampled plots.

Management recommendations – Because this species is non-native and potentially hazardous to native geckos such as the Micronesian Gecko, *Perochirus ateles,* conservation of this species is neither necessary nor desirable. However, its high sensitivity to Brown Treesnake presence may allow it to be an early indicator of the presence of Brown Treesnake populations, and any dramatic declines in this species ought to be investigated as potential evidence of Brown Treesnake colonization.

Emerald Skink, *Lamprolepis smaragdina*

Body length: 60 – 110 mm Mass: 5.0 – 26.8 g

Figure 22. The Emerald Skink, *Lamprolepis smaragdina*.

Previous studies – The first definite record of this highly conspicuous species in the Mariana Islands is from the island of Saipan in 1978 (Wiles and Guerrero 1996). Owen (1974) did not see it on Tinian in 1974, and it is presumed to have reached Tinian from Saipan. Wiles *et al.* (1989) observed it only in south-central Tinian in 1984-1985. With one exception all of their records were near San Jose village, and they commented that it "does not appear to have spread islandwide." Wiles *et al.* (1989) found it primarily in *Leucaena* or secondary habitat, but within that habitat type they found it almost exclusively on flame trees (*Delonix regia*). We found it on Tinian in 1989 (Table 9) in areas that Wiles *et al.* (1989) had observed it, as well as at the southern end of the Carolinas Plateau (site SCRH). We did not find it in native forest in 1989, but we did observe it in a diversity of habitats, including the cliffside vegetation at the Korean Memorial (Table 9). Thus it may have spread somewhat in the 4-5 y between the Wiles *et al.* (1989) study and ours. However, the difference might also be attributable to differential sampling. Perry and Buden (1999) observed this species in a variety of sites in southern Tinian, but did not endeavor to map its distribution.

This study (2008) –We found this species at all sampled locations except North Field (Tables 10-12, Fig 23). Notably, observations included all habitat types including limestone forest (Tables 10-12). The absence from North Field (2MOR, B29W, B29R, ABSW, ABLE, BRAV, DELT) is striking because that region is almost exclusively *Leucaena* habitat (only 2MOR is mixed), which Wiles *et al.* (1989) identified as the primary habitat of this species. Therefore, it seems probable that its absence from North Field is a temporary condition attributable to lack of dispersal to that locality. Note that the Emerald Skink was found in the *Leucaena* total removal plots we conducted on Saipan (Figure 23), but not on Tinian, as our *Leucaena* plots on Tinian were all north of apparently-occupied habitat. We predict that future surveys will eventually find it around North Field as well as suitable habitat to the north of the runway complex.

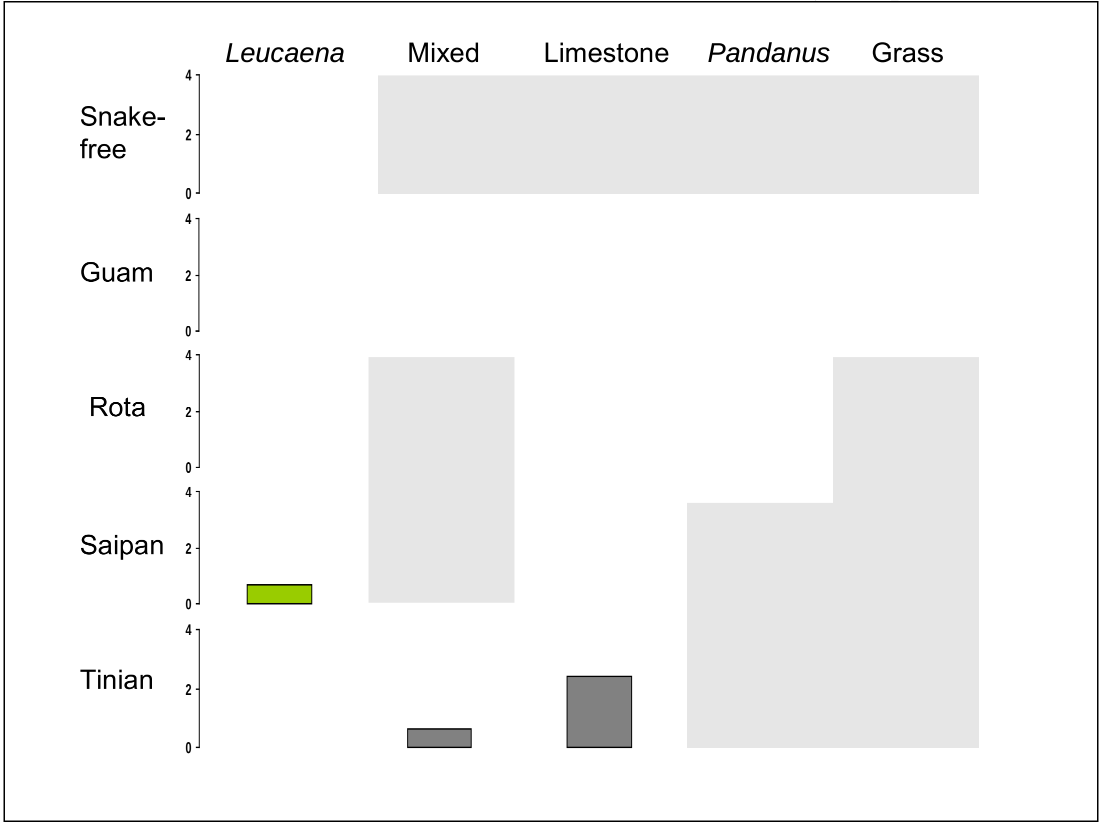


Figure 23. Patterns of abundance of *Lamprolepis smaragdina*, as deduced from 40 total removal plots (each 10 × 10 m) on the four large Mariana Islands. Symbols and sample sizes as in Figure 6. *Lamprolepis smaragdina* was not known to occupy either Guam or Rota at the time of sampling.

Management recommendations – This species is capable of consuming smaller lizards (Perry and Buden 1999). The smaller day-active native species of concern include *Emoia caeruleocauda*, *Emoia slevini*, and *Cryptoblepharus poecilopleurus*. Of those species, none is primarily arboreal (although *Cryptoblepharus* is locally arboreal), and the Emerald Skink is almost exclusively so (Brown and Alcala 1980, Buden 1995, Buden 1996a, 1996b, Perry and Buden 1999). Therefore it seems unlikely that the predatory impact of this large skink, if any, will be significantly detrimental to the continued survival of any native species. Nonetheless, the recorded presence of *Cryptoblepharus poecilopleurus* at B29W, a non-littoral *Leucaena* site north of the known distribution of the Emerald Skink is notable in that the Snake-eyed Skink's absence further south may reflect predation by the Emerald Skink. Perry and Buden (1999) found that Emerald Skinks on Tinian usually (67%) perched head down with an abnormally low mean perch height (0.89 m); thus they would be in a physical position to capture the much smaller semi-arboreal Snake-eyed Skink. Future monitoring should take note of the geographic, habitat, and microhabitat occupancies of these two species.

Mangrove Monitor, *Varanus indicus*

Body length: 99 - 540 mm   Mass: 10 – 3650 g

Figure 24. The Mangrove Monitor, *Varanus indicus*.

<u>Previous studies</u> – Pregill (1998) established that monitor lizards on Tinian were likely introduced during the western period (less than 500 years ago); the earliest written observation was by De la Corte (mid 1800s: Wiles *et al.* 1989). Although apparently rare prior to the 1950s (Wiles *et al.* 1989), monitor lizards were documented but not quantified by all observers since Owen (1974). Wiles *et al.* (1989) commented that they were "seen most often in tangantangan [=*Leucaena*] forest and weedy fields and openings."

<u>This study (2008)</u> – Removal plots and glueboards are of low utility for this species. Instead, Scott Vogt (Navy) monitored this species. We saw monitor lizards opportunistically, including in limestone forest.

<u>Management recommendations</u> – Given the non-native status of this species, we see no need to be concerned for its conservation. Although potentially detrimental to smaller native species, we are aware of little evidence suggesting a significant impact.

Cane Toad, *Rhinella marina*

Body length: 23 – 130 mm   Mass: 1.0 – 224.0 g

Figure 25. The Cane Toad, *Rhinella marina*.

Previous studies – Prior to Chaparro *et al.* (2007) the species now known as *Rhinella marina* was termed *Chaunus marinus* and before Frost *et al.* (2006) it was known as *Bufo marinus*. Frost now accepts *Rhinella* and it is under that name that it appears in Amphibian Species of the World 5.2 (http://research.amnh.org/herpetology/amphibia/references.php?id=4034). Under any scientific name it is most commonly called the Marine or Cane Toad. The Cane Toad was introduced to Guam in 1937 (Anon. 1940, Easteal 1981) and introduced from there to Tinian prior to 1944, presumably during World War II (Stohler and Cooling 1945, Townes 1946, Downs 1948). Subsequently it was observed islandwide (e.g., Owen 1974), especially in proximity to standing water. Wiles *et al.* (1989) found it in all habitats of the Military Lease Area.

This study (2008) – We casually observed it crossing roads throughout the island, but did not quantify its abundance except in total removal plots (Figure 26), where it was found in mixed and limestone forested plots. Its absence from the *Leucaena* forest plots is probably coincidental, as the toads congregate in moist areas, and moist areas may have been missing by chance from our two *Leucaena*-forest plots. Figure 26 shows this species' widespread but irregular distribution in the Mariana Islands. The lack of regularity in the total removal results no doubt at least partially reflects proximity to water, which was neither controlled for nor quantified in the selection of plots.

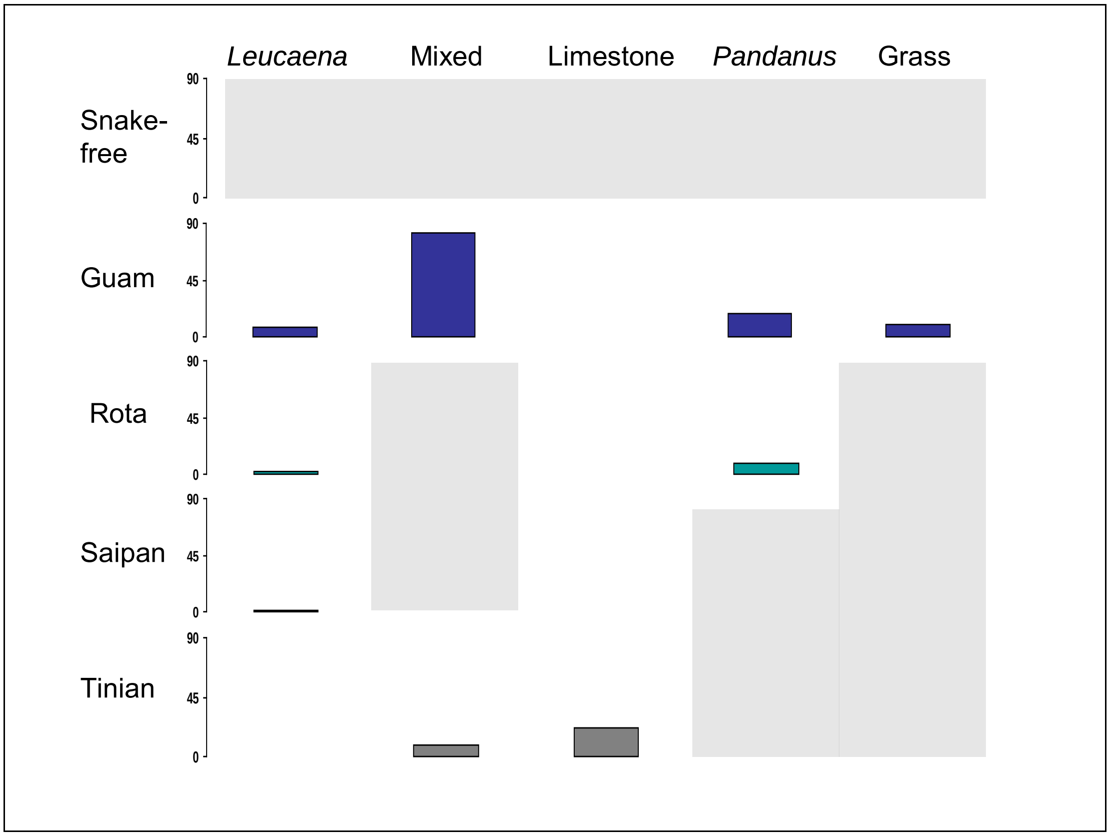


Figure 26. Patterns of abundance of *Rhinella marina*, as deduced from 40 total removal plots (each 10 × 10 m) on the four large Mariana Islands. Symbols and sample sizes as in Figure 6. *Rhinella marina* had been intentionally extirpated from the snake-free plots at the time of sampling, and is therefore grayed-out, though it was present in substantial numbers in the immediate vicinity.

Management recommendations – This recently introduced species is generally viewed as detrimental to native amphibians and predators that attempt to eat the poisonous toads, of which there are none on Tinian. No special management is necessary or desirable for this species' conservation, but reduction in its numbers would be desirable. At this time there are no methods for control of this toad that would be easily applicable to Tinian, but consideration should be given to invoking new control measures should they become available.

Overarching management recommendations

Avoid introducing new species of any type – The greatest threat to the extant herpetofauna of Tinian is likely to come from newly introduced mammalian, avian, or reptilian predators. Biosecurity measures to prevent new colonizations by either inadvertent (i.e., stowaway) or intentional pathways (e.g., release of pets by service personnel) should be the highest priority for conservation of reptiles and amphibians on Tinian.

Conserve and promote native forest habitats – Although the Marianas' rare native lizards (esp. *Perochirus ateles* and *Emoia slevini*) can thrive in all types of habitat under natural conditions, present conditions appear to favor their survival in native forest, especially limestone forest. Tinian's current depauperate native herpetofauna may reflect a long history of agricultural development, and restoration of these hard-pressed or extirpated natives may be compatible with the forest environments appropriate for military activities.

Monitor prey species of Brown Treesnake-control interest – Species that are key prey for Brown Treesnakes (shrews, rats, mice, anoles, and geckos) influence the efficacies of control tools for the snake (Rodda *et al.* 2001, Gragg *et al.* 2007). Periodic monitoring of those species would facilitate effective selection of control tools should the snake arrive on Tinian.

Acknowledgments

Funding: Scott Vogt (U.S. Navy), Earl Campbell (U.S. Fish and Wildlife Service), and Tiffany Taylor (U.S. Department of the Interior's Office of Insular Affairs). Logistics: Lea' Bonewell, Nate Hawley, Scott Vogt, Ton Castro, Joe Lizama, Sylvan Igisomar. Field assistance: U. Krystal Kim, Joseph F. Entilla, Jeremy R. Cruz, Ton Castro, Björn Lardner, Scott Vogt, James Stanford, Emilie Fidler, Amy Yackel Adams, Lea' Bonewell, and the clouds when they came. Improvements to the manuscript were suggested by Björn Lardner, Amy Yackel Adams, Lea' Bonewell. Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Literature Cited

Anon. 1940. The toad introduced into Guam in 1937. The Guam Recorder 17(2):68, 84.

Bailey, P. 1976. Food of the Marine Toad, *Bufo marinus* and 6 species of skink in a cacao plantation in New Britain, Papua New Guinea. Aust. Wildl. Res. 3:185-188.

Barbehenn, K. R. 1974. Recent invasions of Micronesia by small mammals. Micronesica 10:41-50.

Brown, W. C. 1991. Lizards of the genus *Emoia* (Scincidae) with observations on their evolution and biogeography. Mem. Calif. Acad. Sci. 15:1-94.

Brown, W. C., and A. C. Alcala. 1980. Philippine Lizards of the Family Scincidae. Silliman Univ. Natur. Sci. Monogr. Ser. Num. 2, Dumaguete City, Philippines.

Brown, W. C., and M. V. C. Falanruw. 1972. A new lizard of the genus *Emoia* (Scincidae) from the Marianas Islands. Proc. Calif. Acad. Sci. 39:105-110.

Buden, D. W. 1995. Reptiles, birds, and mammals of Mokil and Pingelap atolls, eastern Caroline Islands. Micronesica 28:9-23.

Buden, D. W. 1996a. Reptiles, birds, and mammals of Ant Atoll, eastern Caroline Islands. Micronesica 29:21-36.

Buden, D. W. 1996b. Reptiles, birds, and mammals of Pakin Atoll, eastern Caroline Islands. Micronesica 29:37-48.

Buden, D. W. 2007. The reptiles of Satawan Atoll and the Mortlock Islands, Chuuk State, Federated States of Micronesia. Pac. Sci. 61:415-428.

Cagle, F. R. 1946a. A lizard population on Tinian. Copeia 1946:4-9.

Cagle, F. R. 1946b. Tail loss and regeneration in a Pacific gecko. Copeia 1946:45.

Cagle, F. R. 1946c. *Typhlops braminus* in the Mariana Islands. Copeia 1946:101.

Chaparro, J. C., J. B. Pramuk, and A. G. Gluesenkamp. 2007. A new species of arboreal _Rhinella_ (Anura: Bufonidae) from cloud forest of southeastern Peru. Herpetologica 63:203-212.

Downs, T. 1948. Amphibians and reptiles of Tinian Island. Trans. Kans. Acad. Sci. 51:112-116.

Easteal, S. 1981. The history of introductions of *Bufo marinus* (Amphibia: Anura); a natural experiment in evolution. Biol. J. Linn. Soc. 16:93-113.

Fritts, T. H., and G. H. Rodda. 1998. The role of introduced species in the degradation of island ecosystems: a case history of Guam. Ann. Rev. Ecol. Syst. 29:113-140.

Frost, D. R., T. Grant, J. Faivovich, R. H. Bain, A. Haas, B. F. B. Haddad, R. O. De Sa, A. Channing, M. Wilkinson, S. C. Donnellan, C. J. Raxworthy, J. A. Campbell, Blotto Boris L., P. Moler, R. C. Drewes, R. A. Nussbaum , J. D. Lynch, D. M. Green, and W. C. Wheeler. 2006. The amphibian tree of life. Bulletin of the American Museum of Natural History (297):1-370.

Gragg, J. E., G. H. Rodda, J. A. Savidge, G. C. White, K. Dean-Bradley, and A. R. Ellingson. 2007. Response of Brown Treesnakes to reduction of their rodent prey. J. Wildl. Manage. 71:2311-2317.

Hawley, N. B. 2008. Herpetological surveys and small mammal surveys, Sarigan 2006. In G. Martin, L. L. Williams, J. B. de Cruz, N. B. Hawley, S. Vogt, B. D. Smith, O. Bourquin, S. Kremer, and C. Kessler (eds.), Wildlife and Vegetation Surveys of Sarigan Island April 13-25, 2006, pp. 4-1 to 4-4. Northern Mariana Islands, Div. of Fish and Wildlife, Saipan MP.

Horner, P. 2007. Systematics of the snake-eyed skinks, *Cryptoblepharus* Weigmann (Reptilia: Squamata: Scincidae) – an Australian-based review. The Beagle, Rec. Mus. Art Gall. Northern Terr., Suppl. 3: 21-198.

Ineich, I. 1988. Mise en évidence d'un complexe unisexué-bisexué chez le gecko *Lepidodactylus lugubris* (Sauria, Lacertilia) en Polynésie francaise. C. R. Acad. Sci. Paris 307:271-277.

Kolinski, S. P., L. I. Ilo, and J. M. Manglona. 2004. Green turtles and their marine habitats at Tinian and Aguijan, with projections of resident turtle demographics in the southern arc of the Commonwealth of the Northern Mariana Islands. Micronesica 37:97-118.

Lardner, B., J. A. Savidge, and G. H. Rodda. 2007. Spotting cryptic animals in the dark: what light properties should a good headlamp have? In G. W. Witmer, W. C. Pitt, and K. A. Fagerstone (eds.), Managing Vertebrate Invasive Species: Proceedings of an International Symposium, pp. 234-245. USDA-APHIS Wildlife Services National Wildlife Research Center, Fort Collins, CO.

Lardner, B. M., J. A. Savidge, G. H. Rodda, R. N. Reed, and A. A. Yackel Adams. In review. Nocturnal visual survey results are dependent on lamp properties. Herpetol. Rev.

Mayer, G. C., and J. D. Lazell, Jr. 1992. Identity and distribution of the introduced *Anolis* lizard of Hawaii and other Pacific islands (abstract). Suppl. Bull. Ecol. Soc. Amer. 73:265.

McCoid, M. J. 1997. Interactions of *Carlia* cf. *fusca* (Scincidae) with the herpetofauna of Guam. unpub. M.S. thesis, Texas A&M - Kingsville, Kingsville, TX.

McCoid, M. J., and R. A. Hensley. 1993. Shifts in activity patterns in lizards. Herpetol. Rev. 24:87-88.

McCoid, M. J., and R. A. Hensley. 1994a. Distribution and abundance of *Perochirus ateles* (Gekkonidae) in the Mariana Islands. Herpetol. Rev. 25:97-98.

McCoid, M. J., and R. A. Hensley. 1994b. Eggs, hatchling sizes, and oviposition sites of lizards on Guam, Mariana Islands. Herpetol. Rev. 25:98-100.

McCoid, M. J., G. H. Rodda, and T. H. Fritts. 1995a. Distribution and abundance of *Emoia slevini* (Scincidae) in the Mariana Islands. Herpetol. Rev. 26:70-72.

McCoid, M. J., G. H. Rodda, R. A. Hensley, and T. H. Fritts. 1995b. Habitat selection by *Cryptoblepharus poecilopleurus* (Scincidae) in the Mariana Islands. Micronesica 28:103-108.

Moritz, C., T. J. Case, D. T. Bolger, and S. C. Donnellan. 1993. Genetic diversity and the history of Pacific island house geckos (*Hemidactylus* and *Lepidodactylus*). Biol. J. Linn. Soc. 48:113-133.

Mueller-Dombois, D., and F. R. Fosberg. 1998. Vegetation of the Tropical Pacific Islands. Springer, New York.

Owen, R. P. 1974. Environmental Impact Study on the Terrestrial Fauna and Flora of Tinian With Respect to the Proposed Establishment of a U.S. Military Base on That Island. Trust Terr. of the Pac. Island, Biol. Lab., Koror, Palau

Perry, G., and D. W. Buden. 1999. Ecology, behavior and color variation of the Green Tree Skink, *Lamprolepis smaragdina* (Lacertilia: Scincidae), in Micronesia. Micronesica 31:263-273.

Pregill, G. K. 1998. Squamate reptiles from prehistoric sites in the Mariana Islands, Micronesia. Copeia 1998:64-75.

Pultz, S., D. O'Daniel, S. Krueger, H. McSharry, and G. Balazs. 1999. Marine turtle survey on Tinian, Mariana Islands. Micronesica 32:85-94.

Radtkey, R. R., S. C. Donnellan, R. N. Fisher, C. Moritz, K. A. Hanley, and T. J. Case. 1995. When species collide: the origin and spread of an asexual species of gecko. Proc. Roy. Soc. Lond. B Biol. Sci. 259:145-152.

Rodda, G. H. 1992. Loss of native reptiles associated with introductions of exotics in the Mariana Islands. Pac. Sci. 46:399-400.

Rodda, G. H., E. W. Campbell, III, and T. H. Fritts. 2001. A high validity census technique for herpetofaunal assemblages. Herpetol. Rev. 32:24-30.

Rodda, G. H., E. W. Campbell, III, T. H. Fritts, and C. S. Clark. 2005a. The predictive power of visual searching. Herpetol. Rev. 36:259-264.

Rodda, G. H., K. Dean-Bradley, and T. H. Fritts. 2005b. Glueboards for estimating lizard abundance. Herpetol. Rev. 36:252-259.

Rodda, G. H., and T. H. Fritts. 1992. The impact of the introduction of the Brown Tree Snake, *Boiga irregularis*, on Guam's lizards. J. Herpetol. 26:166-174.

Rodda, G. H., T. H. Fritts, M. J. McCoid, and E. W. Campbell, III. 1999. An overview of the biology of the Brown Treesnake, *Boiga irregularis*, a costly introduced pest on Pacific Islands. In G. H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka (eds.), Problem Snake Management: the Habu and the Brown Treesnake, pp. 44-80. Cornell Univ. Press, Ithaca, New York.

Rodda, G. H., T. H. Fritts, and J. D. Reichel. 1991. The distributional patterns of reptiles and amphibians in the Mariana Islands. Micronesica 24:195-210.

Rodda, G. H., M. J. McCoid, and T. H. Fritts. 1993. Adhesive trapping II. Herpetol. Rev. 24:99-100.

Sabath, M. D. 1981. Gekkonid lizards of Guam, Mariana Islands: reproduction and habitat preference. J. Herpetol. 15:71-75.

Savidge, J. A. 1987. Extinction of an island forest avifauna by an introduced snake. Ecology 68:660-668.

Slifka, K., P. McGill, A. Oiler, and D. Kesler. 2004. Nutrient composition of lizards consumed by Micronesian Kingfishers (*Todirhamphus cinnamomina reichenbachii*) on Pohnpei. Proceedings of the Fifth Comparative Nutrition Society Symposium.

Stohler, R., and A. G. Cooling. 1945. Toads in the Marianas (*Bufo marinus*). Science 101:678.

Suzuki, A., and M. Nagoshi. 1999. Habitat utilizations of the native lizard, *Cryptoblepharus boutonii nigropunctatus*, in areas with and without the introduced lizard, *Anolis carolinensis*, on Hahajama, the Ogasawara Islands, Japan. In H. Ota (ed.), Tropical Island Herpetofauna: Origin, Current Diversity, and Conservation, pp. 155-168. Elsevier, St. Louis MO.

Townes, H. K. 1946. Entomology. Part 1. Non-agricultural plants. In Anon., Vol. 12, Economic Survey of Micronesia, pp. 1-53. US Commercial Company Report, Honolulu, HI.

US Fish and Wildlife Service. 1996. Status and Distribution of Marine Turtles on the Island of Tinian, Commonwealth of the Northern Mariana Islands-1994&1995. Honolulu, HI.

Vogt, S. R. 2008. Ground skink surveys on Sarigan Island, Commonwealth of the Northern Mariana Islands. In G. Martin, L. L. Williams, J. B. de Cruz, N. B. Hawley, S. Vogt, B. D. Smith, O. Bourquin, S. Kremer, and C. Kessler (eds.), Wildlife and Vegetation Surveys of Sarigan Island April 13-25, 2006, pp. 5-1 to 5-6. Northern Mariana Islands, Div. of Fish and Wildlife, Saipan MP.

Vogt, S. R., and L. L. Williams. 2004. Common Flora and Fauna of the Mariana Islands. privately printed, Saipan, Northern Mariana Islands.

Wiewel, A. S., A. A. Yackel Adams, and G. H. Rodda. 2009. Distribution, density, and biomass of introduced small mammals in the southern Mariana Islands. Pac. Sci. 63(2):in press.

Wiles, G. J., A. B. Amerson, Jr., and R. E. Beck, Jr. 1989. Notes on the herpetofauna of Tinian, Mariana Islands. Micronesica 22:107-118.

Wiles, G. J., and J. P. Guerrero. 1996. Relative abundance of lizards and Marine Toads of Saipan, Mariana Islands. Pac. Sci. 50:274-284.

Yamashiro, S., M. Toda, and H. Ota. 2000. Clonal composition of the parthenogenetic gecko, *Lepidodactylus lugubris*, at the northernmost extremity of its range. Zool. Sci. 17:1013-1020.

Zug, G. R. 2004. Systematics of the *Carlia "Fusca* Lizards (Squamata: Scincidae) From New Guinea and Nearby Islands. Bishop Mus. Bull. Zool. 5, Bishop Mus. Press, Honolulu HI.

Zug, G. R., and B. R. Moon. 1995. Systematics of the Pacific slender-toed geckos, *Nactus pelagicus* complex: Oceania, Vanuatu, and Solomon Islands populations. Herpetologica 51:77-90.

2.3.2 REPTILES AND AMPHIBIANS OF AGUIGUAN

The Mutilating Gecko (*Gehyra mutilate*)
Photo courtesy of USGS, BRD

Pacific Blue-tailed Skink (*Emoia caeruleocauda*)
Photo courtesy of USGS, BRD

Prepared by: Gordon H. Rodda, Robert N. Reed, and James W. Stanford; U.S. Geological Survey, Biological Resources Disicpline, Fort Collins Science Center, Fort Collins, CO.

INTRODUCTION

Note: Refer to Section 2.3.1, the Reptiles and Amphibians of Tinian, for a general introduction to the rationale for this work, and a more detailed description of the methods used.

Aguiguan Island (or Aguijan Island as it is spelled on USGS maps; the island is also popularly referred to as Goat Island, in reflection of the high density of these introduced ungulates) is a relatively small (720 ha) island lying off the south coast of Tinian. It is a "high" island, with raised limestone terraces and cliffs on all sides. Because it has no beaches or protected anchorages, it can be impossible to land or leave in windy conditions and thus is uninhabited. Nonetheless, it has been the subject of several biological inventory expeditions in the last two decades, and for its size is better known herpetologically than many larger islands such as Tinian. Particularly noteworthy are the visits of Davis (1954), Campbell (1995), Cruz *et al.* (2000), and Esselstyn *et al.* (2003). An enduring element of herpetological interest has been the high densities of monitor lizards found on Aguiguan, which we did not survey in our work; Scott Vogt (U.S. Navy) conducted concurrent surveys of monitor lizards. The other two conspicuous elements of the fauna are the apparent high densities of introduced rats (*Rattus exulans*) and goats (*Capra hircus*). Population densities of rats are covered in the accompanying report (Yackel Adams *et al.* 2009), though the visual surveys for nocturnal reptiles reported in this paper also tracked rat sightings and relevant sighting data are included herein. We recorded no data on goat distribution or abundance, but note here that a high goat density on "Goat" Island is responsible for very open understory vegetation, which greatly enhances sight lines. Thus lizard sighting rates on "Goat" are elevated in comparison to sighting rates on Tinian or the other major islands of the Mariana archipelago.

Our objective in this study was to conduct a modest number of spot searches, to see if any major components of the non-varanid herpetofauna had been overlooked. Although Aguiguan is herpetologically well known for its size, there were a number of species expected to be present that had not been documented; we targeted these species rather than attempting to verify the distribution or status of known resident species. We also added a small body of data on search per unit effort, but we did not stratify sampling by habitat type nor attempt absolute population density estimation as was performed on Tinian.

MATERIALS AND METHODS

See Rodda *et al.* (2009) for detailed methods; of the methods in that report we utilized glueboard sampling (Table 1, a total of 1420.05 trap-hours) and visual searches (Table 2, total of 12.74 search-hours). Most of the surveys were conducted along the indicated bird transects (see Esselstyn *et al.* 2003 for GPS coordinates), with no attempt to delineate geographic or habitat distribution of species within the island. We assume that the island is small enough that any suitable habitat will be occupied throughout the island. Our sampling was concentrated on the west end of the island; if there are any species limited to the eastern end, we would not have encountered them.

Table 1. Glueboard sampling of Aguiguan Island, 2008.

End Date	Transect	Time of day	Elapsed (h)	Target	Traps	Trap-h	Lizard captures
20-Jul-08	A4	0800-1110	3.15	Skinks	12	37.8	2
20-Jul-08	A4	2000-0715	11.25	Geckos	11	123.75	3
22-Jul-08	A2	0845-1145	3	Skinks	13	39	0
22-Jul-08	A2	0850-1150	3	Skinks	12	36	6
23-Jul-08	A4	1930-0816	12.5	*Nactus*	12	150	0
24-Jul-08	A2	2000-0730	11.5	Geckos	12	138	0
24-Jul-08	A4	2000-0630	10.5	*Nactus*	12	126	0
24-Jul-08	A5	1740-0640	13	Geckos	12	156	5
26-Jul-08	Second camp	1111-1011	23	*Cryptoblepharus*	12	276	6
27-Jul-08	Second camp	1125-1425	3	*Cryptoblepharus*	12	36	6
30-Jul-08	W end of plateau	1300-1000	21	Geckos	12	252	3
31-Jul-08	Orig. helispot	1430-0700	16.5	Green lizards	3	49.5	2

Table 2. Nighttime visual surveys of Aguiguan Island, 2008.

Date	Transect	Time of day	Search-hours	Lizard detections
19-Jul-08	Camp Trail and A4	1955-2110	1.25	13
21-Jul-08	A4	1952-2107	1.25	3
21-Jul-08	A4	1854-2000	1.06	33
22-Jul-08	A2	2023-2218	1.92	11
22-Jul-08	A2	2024-2219	1.92	15
23-Jul-08	A4	2021-2123	1.03	5
23-Jul-08	A4	2021-2123	1.03	3
23-Jul-08	A5	1932-2032	1	2
25-Jul-08	A2	1900-2003	1.03	3
30-Jul-08	A4 and Camp Trail	1915-2030	1.25	5

RESULTS

Glueboard sampling. – Mean glueboard rates (captures per trap-h) were computed on the basis of nighttime trap-h only for nocturnal species and daytime trap-h only for diurnal species. Geckos: *Hemidactylus frenatus*: 0.0066; *Gehyra mutilata*: 0.0197; and *Gehyra oceanica*: 0.0033. Skinks: *Emoia caeruleocauda*: 0.0134; and *Cryptoblepharus poecilopleurus*: 0.0081.

Visual sightings. – Ten sightings were sufficiently fleeting that the species of gecko was not determined. Of the 88 sightings for which the species could be unequivocally determined, the gehyras were the most often seen: *Gehyra mutilata*: 2.98/search-h and *Gehyra oceanica*: 3.45/search-h. The two smaller geckos were infrequently seen: *Lepidodactylus lugubris*: 0.157/search-h and *Hemidactylus frenatus*: 0.314/search-h.

SPECIES ACCOUNTS – see Tinian report (Rodda *et al.* 2009) for photographs and body sizes of each species.

NATIVE SPECIES

Oceanic Snake-eyed Skink, *Cryptoblepharus poecilopleurus*

Previous studies – This species was collected by Campbell (1995), Cruz *et al.* (2000) and Esselstyn *et al.* (2003), primarily along the coast (all three studies) and in the mouth of a cave (Campbell 1995).

This study – We collected a substantial number (9) on glueboards in the same general area where they had been previously documented. It is noteworthy that no observers have found this species any appreciable distance from the coastline, although it is not evident what habitat feature is directly responsible for this limitation, as they occur along the coastline in vegetation that appears similar in structure and species composition to sites not occupied further inland.

Littoral Skink, *Emoia atrocostata*

Previous studies – This species is strictly limited to the intertidal zone, and has been found on Aguiguan Island in that habitat by Campbell (1995) and Esselstyn *et al.* (2003).

This study – We did not sample the intertidal zone to confirm that this species remains present in that habitat on Aguiguan.

Pacific Blue-tailed Skink, *Emoia caeruleocauda*

Previous studies – Campbell (1995), Cruz *et al.* (2000), and Esselstyn (2003) found this species to be common throughout the island in a variety of habitats.

This study – Our mean glueboard capture rate for this species (0.0134) was the highest of any diurnal lizard and was similar to that recorded on Tinian, where it was found only in limestone forest, with a mean capture rate of 0.009 (see Rodda *et al.* 2009).

Mourning Gecko, *Lepidodactylus lugubris*

Previous studies – An unspecified number of this species was detected in native and introduced forest by Campbell (1995). Cruz *et al.* (2000) recorded one individual on a glueboard in introduced forest, and Esselstyn *et al.* (2003) did not document this species.

This study – We detected this species by visual searches only (mean sighting rate was 0.156, the lowest of the species seen). It is possible that additional individuals were seen too poorly for species identification (i.e., scored as unknown gecko), but we doubt that many such events transpired, as this species is relatively unwary and easy to identify. For comparison, the sighting rate on Tinian averaged 1.241 sightings/h in *Leucaena* forest and 0.340 sightings/h in limestone forest. Our searches on Aguiguan focused on limestone forest, so while the mean sighting rate was low, our sample size was modest and the detection rate was not beyond the range of values expected for this limestone forest habitat.

POTENTIALLY NATIVE SPECIES

Indo-Pacific House Gecko, *Hemidactylus frenatus*

Previous studies – Campbell (1995) reported the first detection of this species on Aguiguan. He found it only in introduced forest, as did Cruz *et al.* (2000), whereas Esselsyn *et al.* (2003) found it only in limestone forest.

This study – We found it in moderate numbers, by both visual surveys (0.314 detections/h) and glueboard surveys (0.0066 captures/trap-h). These detection rates are in line with those recorded on Tinian.

Pacific Slender-toed Gecko, *Nactus pelagicus*

Previous studies – This species has not been detected in historic times on Aguiguan, despite an abundance of apparently suitable habitat (On Guam it is restricted to relatively undisturbed limestone or ravine forest (Rodda and Fritts 1996), a habitat now reasonably abundant on Aguiguan). Pregill (1998) reported it present in prehistoric strata.

This study – We targeted this species in our searches (see Table 1) but failed to detect it. It may have gone undetected, it may have been extirpated prehistorically and not have recolonized Aguiguan in historic times (though it remained on or recolonized the islands to the north (Tinian, Alamagan, Anatahan, Sarigan) and to the south (Rota, Guam)), or it may be vulnerable to predation from the numerous introduced *Rattus exulans*.

INTRODUCED SPECIES

Curious Skink, *Carlia ailanpalai*

Previous studies – This species is ubiquitous on Tinian and Saipan, but has not previously been detected on Aguiguan.

This study – Although we did not specifically target this species, we sampled extensively in the places were it would be expected to be detected if it were present. We think that it is unlikely to have yet colonized Aguiguan.

Mutilating Gecko, *Gehyra mutilata*

Previous studies – Campbell (1995) first recorded this species on Aguiguan. Cruz *et al.* (2000) and Esselstyn *et al.* (2003) found it in moderate numbers.

This study – This species was the second-most commonly seen gecko (2.98 sightings/h) and the most frequently trapped gecko (0.0197 captures/trap-h). The reported sighting rate is undoubtedly an underestimate, as many of the unknown geckos were probably of this species, which is wary and hard to distinguish from juvenile Oceanic Geckos. The sighting rate for Aguiguan was higher than that reported for this species in any habitat on Tinian, though Tinian has relatively little good habitat for this species and Aguiguan has ample prime habitat.

Oceanic Gecko, *Gehyra oceanica*

Previous studies – Campbell (1995) noted this species presence in both limestone and introduced forest. Cruz *et al.* (2000) trapped it commonly, but Esselstyn (2003) did not trap it at all.

This study – We captured only one of this species on glueboards, but it was the commonest species detected in visual surveys (3.45 sightings/h) and it was extremely dense (subjectively up to $1/m^2$) on the walls in two of the caves we visited. As with its congener, this species probably accounts for some of the unknown geckos, as juveniles of this species are hard to distinguish from *G. mutilata*. However, the species is not particularly wary, and its habitat preferences greatly overlap those of *Nactus pelagicus*, which we targeted. This species elsewhere in the Mariana Islands is found to favor limestone forest as a habitat (Rodda *et al.* 2009). Thus our sighting rate on Aguiguan may overestimate its abundance throughout Aguiguan. Sighting rates on Tinian were substantially lower, and it was not detected by glueboards on Tinian. The failure of Esselstyn *et al.* (2003) to detect it on Aguiguan is likely related to that study's reliance on glueboard captures.

ANCILLARY OBSERVATIONS

Mangrove Monitor, *Varanus indicus*

Previous studies – Peterson (1954, cited in Davis 1954), Davis (1954), Campbell (1995), Cruz *et al.* (2000), and Esselstyn *et al.* (2003) have all provided useful data or observations on the conspicuous abundance of the monitor on Aguiguan Island.

This study – We saw many monitors during our sampling, but did not attempt to quantify their abundance. Refer to Section 2.3.3 Population Densities and Diet of Monitor Lizards on Aguiguan.

Brahminy Blindsnake, *Ramphotyphlops braminus*

Previous studies – This species has not been reported in modern times or historic subfossil strata from Aguiguan Island, but Pregill (1998) found it in prehistoric subfossil strata.

This study – We did not search for this subterranean species, because it cannot generally be obtained by digging (perhaps these burrowing animals can retreat in burrows faster than a human can expose them), and it is more easily and often found simply through opportunistic encounters under rocks or on the surface at night. On 13 July 2008 Ernie Valdez found one near Fault Line Cave 1 (344804 E 1643215 N) in exactly this manner during our visit. We preserved (BSFS 9363) and deposited it in the Bishop Museum.

Green Anole, *Anolis carolinensis*

Previous studies – This species has not previously been reported from Aguiguan Island, but it is common on the adjacent islands of Saipan and Tinian (Rodda *et al.* 2009).

This study – We did not trap or observe this species. The camp cook saw a small green lizard (which could have been this species or *Lamprolepis smaragdina*) near the helipad, and we looked and trapped there to target this species, but turned up no further evidence. It is likely to reach Aguiguan Island in cargo brought from either Tinian or Saipan.

Emerald Skink, *Lamprolepis smaragdina*

Previous studies – This species has not been previously been reported from Aguiguan Island, though it is common on the adjacent islands of Saipan and Tinian (Rodda *et al.* 2009).

This study – We did not trap or observe this species. The camp cook saw a small green lizard (which could have been this species or *Anolis carolinensis*) near the helipad, and we trapped there to target this species, but turned up no further evidence. It is likely to reach Aguiguan Island in cargo brought from either Tinian or Saipan.

Cane Toad, *Rhinella marina*

Previous studies – This species has not previously been reported from Aguiguan Island.

This study – We did not see this species while on Aguiguan. Although common on Tinian and Saipan, it requires standing water which is absent or exceedingly rare on Aguiguan.

DISCUSSION

With the exception of the Brahminy Blindsnake, this study did not detect any new modern populations or species for the island of Aguiguan, but provided some detection rates. We accumulated additional evidence that the Pelagic Gecko does not now occur there, though it did prehistorically. Aguiguan Island is relatively free of introduced reptiles except for the *Gehyra* species. For general conclusions regarding invasive species see the accompanying report on Tinian (Rodda *et al.* 2009).

ACKNOWLEDGMENTS

The rather difficult challenge of getting staff safely to Aguiguan Island was a bureaucratic and logistic burden of the first magnitude, ably carried out on our behalf by Earl Campbell, Karl Buermeyer, Curt Kessler, Nate Hawley, Elvin Masga, Ton Castro, Peter Dunlevy, and especially Jess Omar. We are extremely grateful to all. Nate Hawley assisted with permitting. Funding was provided by the U.S. Navy (Scott Vogt) through the U.S. Fish and Wildlife Service (Earl Campbell). Lea' Bonewell assisted with stateside logistics. Ernie Valdez provided the specimen of *Ramphotyphlops braminus.* Shane Siers and Björn Lardner suggested improvements to the manuscript. Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. government.

LITERATURE CITED

Campbell, E. W., III. 1995. Report on Aguiguan Island reptile and rodent surveys conducted between 21-27 March 1995 and 30 May - 4 June 1995. Saipan MP, unpub. rep. on file with Commonwealth of Northern Mariana Islands, Div. Fish and Wildlife. 11 pp.

Cruz, J., L. Arriola, N. Johnson, and G. M. Beauprez. Wildlife and vegetation surveys; Aguiguan 2000. 2000. Saipan MP, CNMI Div. of Fish and Wildlife 2000 Tech. Rep. #2. 30 pp.

Davis, C. J. 1954. Report on the Davis expedition to Agiguan July-August, 1954: ecological studies, Island of Agiguan, Marianas Islands as related to the African snail, *Achatina fulica* Bowdich, and its introduced predator, *Gonaxis kibweziensis* (E. A. Smith). Hilo, Hawaii, Hawaii Board of Commissioners of Agriculture and Forestry. 21 pp.

Esselstyn, J. A., J. B. Cruz, L. L. Williams, and N. Hawley. Wildlife and vegetation surveys; Aguiguan 2002. 2003. Saipan MP, CNMI Div. of Fish and Wildlife 2003 Tech. Rep. #9. 55 pp.

Rodda, G. H., and T. H. Fritts. 1996. Reptiles and amphibians: reptiles of the Ordnance Annex, Naval Activities Guam, p. 43-79. *In* US Fish and Wildlife Service [ed.], Faunal survey for the Ordnance Annex, Naval Activities, Guam. US Fish and Wildlife Service, Honolulu, Hawaii.

Rodda, G. H., R. N. Reed, S. R. Siers, T. J. Hinkle, T. H. Fritts, and R. P. Reynolds. 2009. The land reptiles and amphibians of Tinian. Fort Collins CO, rep. by USGS Fort Collins Science Center to US Fish and Wildlife Service. 52 pp.

Yackel Adams, A. A., J. W. Stanford, A. S. Wiewel, G. H. Rodda, and A. F. Hambrick. Rat abundance on Aguiguan (Aguijan) Island. 2009. Fort Collins CO, rep. to US Fish and Wildlife Service from USGS Fort Collins Science Center. 15 pp.

2.3.3 POPULATION DENSITIES AND DIET OF MONITOR LIZARDS (*VARANUS INDICUS*) ON AGUIGUAN

Monitor Lizard (*Varanus indicus*). Photo by Gayle Martin

Prepared by: **Scott Vogt, Wildlife Biologist, U.S. Navy, NAVFAC Pacific, Honolulu, HI**

INTRODUCTION

The mangrove monitor lizard (*Varanus indicus*) is present on almost every island in the Marianas chain. While questions exist about if it is truly a native species or not, it has apparently inhabited the Marianas for hundreds if not thousands of years (Cota 2008).

On many of these islands the monitor lizard is the only medium or large sized predator. Feral cats are another, but are not present on some of the uninhabited islands. While some work has been done on breeding behavior, home ranges and diet on Marianas varanids population densities on any island have yet to be documented.

Aguiguan, also known as Goat Island, is 700 hectare island approximately 9km south of Tinian. It was inhabited in historic times by Japanese sugar cane growers but has been uninhabited for roughly 60 years.

Abundance index surveys for this species have been conducted on Aguiguan (Cruz *et al.* 2000, Esselstyn *et al.* 2002), but absolute population densities have not been estimated nor has there been a dietary analyses. Monitor lizard population density, demographic and dietary surveys were performed on Aguiguan during the periods of June 27-29 and July 19-August 5, 2008.

METHODS

Population Density

Population densities were measured by 2 methods, a trapping grid and distance transects.

Trapping Grid

The trapping grid consisted of 25 noose traps (Reed *et al.* 2000) in a 5 x 5 pattern with 40 meter spacing. This covered 2.56 hectares (160 meters x 160 meters). Traps were baited with squid. The grid was set up and baited on July 21, 2008 and monitored twice a day at 11:00-12:00 am and 4:00-5:00 pm, until July 26, 2008. Traps were re-baited as needed. The month before (June 2008) the trapping grid was established, a line of 4 noose traps was monitored for 3 days to test the efficacy of the squid bait. The trap line was approximately 500 meters south of the trapping grid area.

The population estimate from the trapping grid data was analyzed with the mark-recapture analyses software, program MARK.

A difficulty in trying to establish animal densities using trapping grids is determining how many animals come from outside the grid to be captured. The grid samples a larger area than its dimensions. It is often difficult to calculate the size of the sampled area and this is vital for accurate density estimates. One method for helping to mitigate for this problem is to fit animals with transmitters so that movements within and outside the grid can be ascertained. In this way, a buffer strip can then be added to the grid boundary for density calculations. For example, based on telemetry data, an extra 50 meters might be added to the grid boundary so that the true area sampled was 260m x 260m and not 160m x 160m.

Wildlife Track brand transmitters were fitted lizards as a "backpack" with brass bead chain. The transmitter was secured with bead chain around the body anterior to the hind legs and around the base of the tail posterior to the hind legs (Figure 1). All captured lizards were marked with colored duct tape wrapped around the body just anterior to hind legs (Figure 1) to ease identification. All lizards captured were measured for snout to vent length (svl) and tail length.

Because the lizards were tracked for a short period of time there were a low number of location data points. Due to the small number of data points, home ranges were calculated by multiplying the distances between the two furthest points on the east:west axis by the two furthest points on the north:south axis. Half of the largest home range size was added as a buffer strip to the trapping grid. This was calculated by taking the square root of the largest home range size (in square meters) divided by 2. The effective trapping area was used to calculate the lizard density (population estimate divided by the effective trapping area).

Figure 1. Example of transmitter attachment and marking of monitor lizards.

Distance Transects

Distance transects followed existing bird transects and covered all areas of the island. Distance sampling was performed between July 20 and August 4, 2008. Transects were slowly walked in the morning between 08:00 am and 11:00 am. When sighted the perpendicular distance from the observer to the mid-body of the monitor lizard was measured to the nearest cm with a tape measure. Since the lizard would run away, this location had to be estimated. After walking the transect in one direction, the observer waited 5 minutes and then returned along the same transect and collected data. This was added to the total transect length. Transects 1, 2, and 4 were sampled twice and transect 3 was sampled once.

Distance transect data was analyzed with the analytical software, DISTANCE.

Diet and Demographics

Monitor lizards were opportunistically shot with a .22 caliber air rifle. All lizards collected were weighed, measured (snout to vent length and tail length), sexed, and the body condition assessed Stomachs were removed and the contents identified. The snout vent lengths for the lizards collected on the trapping grid were pooled with the others for demographic analyses. For describing differences between males and females only data from those lizards that were sexed by dissection were used.

To assess local differences in diet and demographics, data from Aguiguan surveys are compared with data from the island of Sarigan. In 1998, 1999 and 2006, a total of 40 monitor lizards were opportunistically shot on Sarigan. Diet and demographic data were collected.

RESULTS

Population Density

Trapping Grid

Twelve lizards were captured on the trapping grid. There were no recaptures. Three lizards were fitted with transmitters. In addition to these, 2 lizards that were caught on the trap-line in June (4 lizards captured on 4 traps in 3 days) were fitted with transmitters.

The number of telemetry data points for each lizard was 8, 6, 7, and 6. One lizard apparently dropped the transmitter the next day. The signal was received from high up on a cliff line (transmitter not recovered) and the lizard was identified from the duct tape in a different area (within ~10 meters of capture point).

The model used to analyze the grid data in MARK was "full closed captures with heterogeneity." This model takes into account behavioral responses to being captured and individual heterogeneity (differing capture rates amongst individuals). Since there were no recaptures out of 12 animals, one assumes some type of behavioral response to being captured.

The capture rates on the inner ring of traps (0.56 lizards/trap) did not differ considerably from the outer ring (0.44 lizards/trap). If a high proportion of the captured lizards were coming off of the grid the outer ring captures would be inflated.

Program MARK produced a population estimate of 14 lizards with a 95% confidence interval of 12-37 lizards.

The telemetry data produced home range estimates of 0.18, 0.88, 0.36, and 0.62 hectares respectively. Home range size was positively correlated with body size (snout to vent length) (Chart 1). Based on size, the two smaller lizards are presumed to be female and the 2 larger ones male. Home ranges of the three lizards with transmitters on the trapping grid did not overlap for the monitored period with one exception (Chart 2). On August 3, 2008, two lizards (#s 3 and 4) were in the same tree but could not be sighted. Due to the size differences, it is probable that one was male and the other female and courtship was occurring. The male entered into the female home range.

The width of the buffer strip was estimated to be 46.5 meters (the square root of 8800 sq. meters, divided by 2). This gave an effective trapping area of 6.40 hectares (253 meters x 253 meters).

The density estimate is 2.19 lizards/ha with a 95% confidence interval of 1.88-5.78 lizards/ha. The mean body weight was 470 grams, so this gives a biomass estimate of 1029.30 grams/ha.

Chart 1. Aguiguan Monitor Lizards: Home Range Size vs Body Size

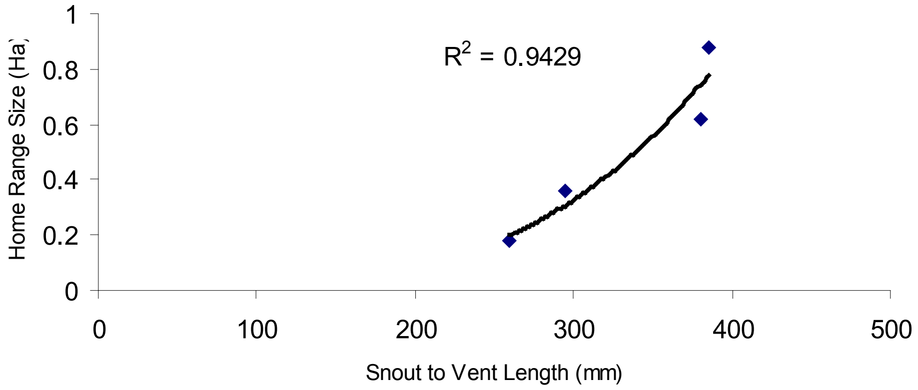
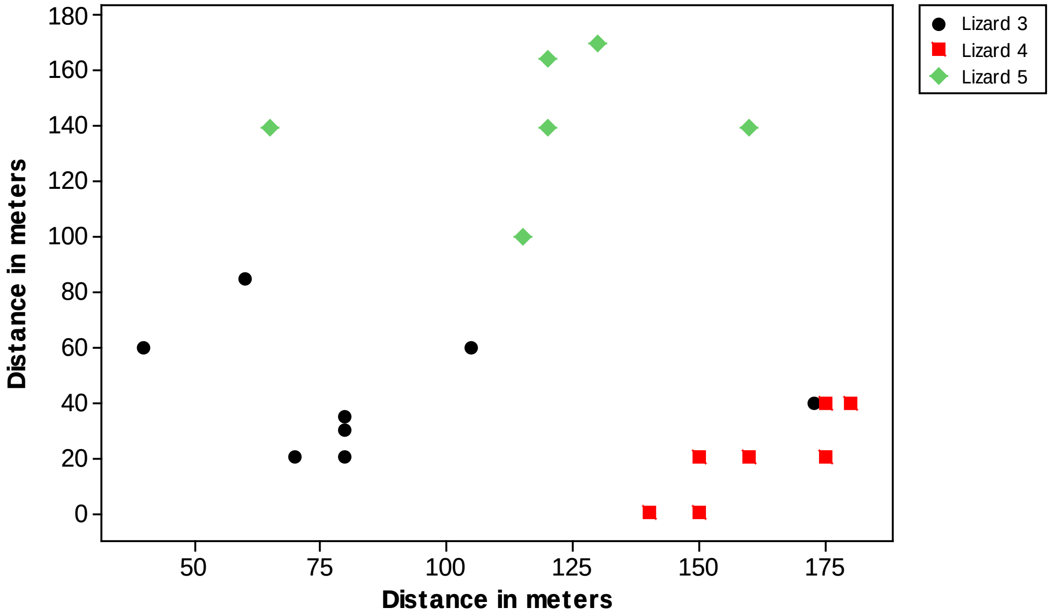


Chart 2. Home Ranges of Monitor Lizards on Aguiguan



Distance Transects

Sixty six monitor lizards were detected on 19,346 meters of transects. During the data collection it was difficult to judge the exact spot on the ground where the center of the body of the individual lizard was. For this reason, the distances were grouped in 50 cm intervals. Grouping data is recommended to improve robustness in the density estimator where the subject animal moves off before detection or heaping of distances (Buckland et al 1993). The data was analyzed using the half-normal-cosine model and right truncated at 400 cm.

The density estimate was 3.67 lizards per hectare with a 95% confidence interval of 2.55-5.29 lizards per hectare. The biomass estimate is 1,724.90 grams/ha.

There are 479 hectares of forested habitat (native, secondary and introduced) on Aguiguan. This extrapolates to a population estimate of 1,758 lizards with a 95% confidence interval of 1,221-2,534. There are an additional 158 hectares of open fields dominated by the introduced plant lantana (*Lantana camara*). Varanids were documented in this habitat, but densities were not established. Distance data was collected but was not used for analyses because of detection differences. Lizards were much harder to detect in this habitat due to the lantana occurring in very dense stands. Goats have greatly reduced the understory in the forests making monitor lizard detections much easier. The sighting rate was 1 lizard/587 meters in open fields while in the forested habitat it was 1 lizard/293 meters. If the densities are similar to the forested habitat then the island wide population estimate is 2,338 lizards with a 95% confidence interval of 1,624-3,370.

Diet

Twenty one lizards were sampled for stomach contents. Three stomachs were empty. The remaining stomachs contained roaches, rats, centipedes, grasshoppers, hermit crab parts, snails, small eggs (gecko or bird), and one stomach had food from the biologists camp (Appendix 1).

The most common prey item was a roach species, *Pycnoscelus indicus*, which was present in 14 stomachs (67%) and was 88% of the prey items documented (Chart 3). The mean number of roaches per stomach was 6 with a range of 2-14. On Sarigan, rats and lizards (skinks and geckos) constituted a significantly higher proportion of prey items (Chart 4).

Chart 3. Stomach Contents of Aguiguan Varanids

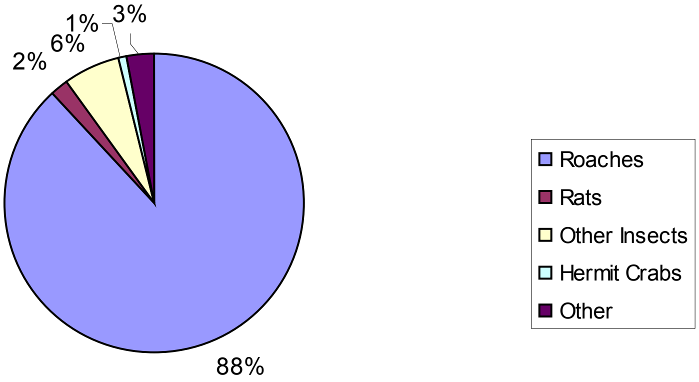
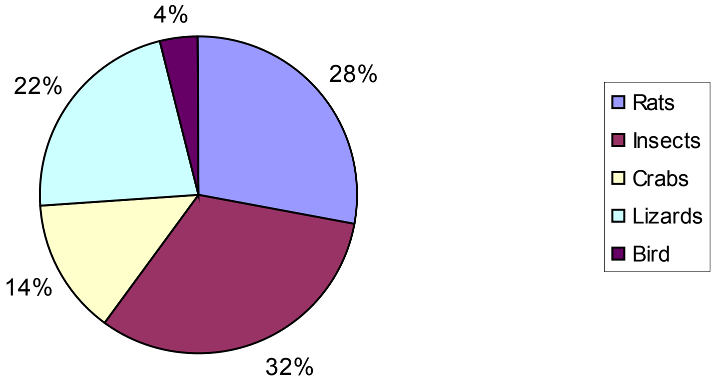


Chart 4. Stomach Contents of Sarigan Varanids



All lizards were in good body condition with large fat deposits. One female had 3 shelled eggs.

Demographics

Of the dissected lizards, 14 were male and 7 were female.

The sizes of lizards from Aguiguan were smaller than those of Sarigan (Charts 5 and 6) (T test, p= 0.01). The histogram differs from a normal bell curve and shows an abrupt drop off after 400mm snout vent length.

There is a pronounced sexual dimorphism with females being smaller than males (Chart 6). This is consistent on both islands even though the overall lizard size is smaller on Aguiguan. The males on Aguiguan are similar in size to the females on Sarigan and the females on Aguiguan are smaller still. The mean size of females on Sarigan is 18% smaller than the males. The mean size of the females on Aguiguan is 12% smaller than the males.

Chart 5. Varanid Size Classes on Sarigan and Aguiguan

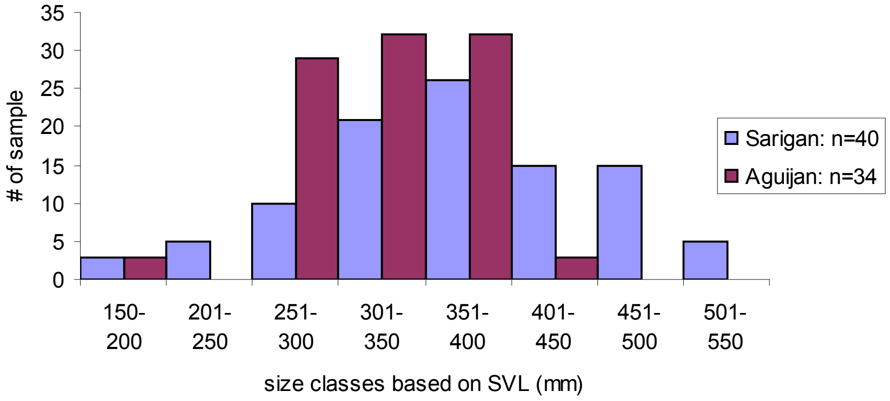
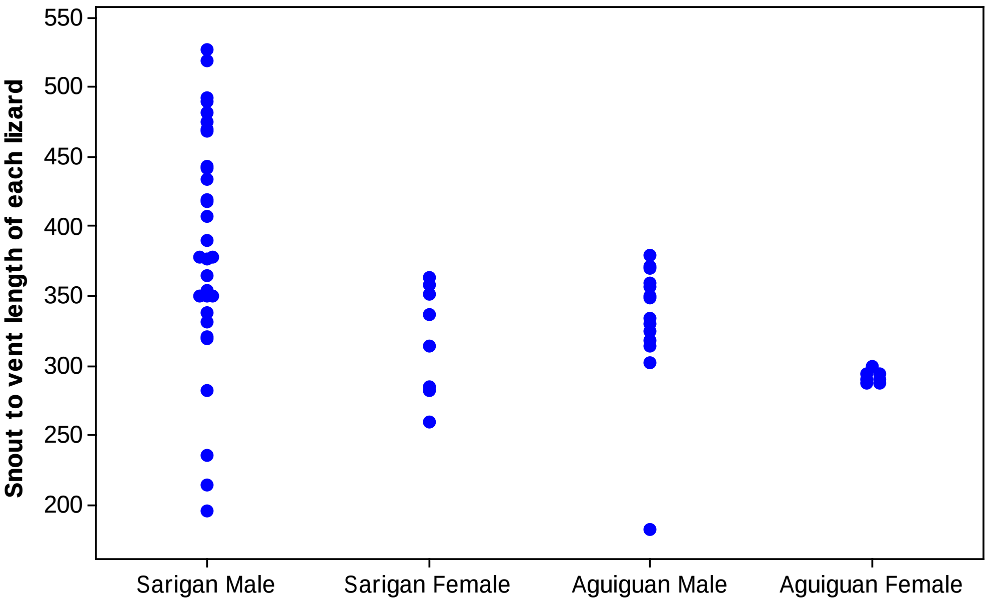


Chart 6. Varanid Sizes on the Islands of Sarigan and Aguiguan



DISCUSSION

The varanid densities documented on Aguiguan are high for this family, with home ranges being accordingly small. These estimates are most likely conservative as only the adult lizards were sampled. What percentage of the population is juvenile or hatchling is unknown.

The density estimate on the distance transects was higher than the trapping grid but the confidence intervals were almost identical. It is possible that the density in the trapping grid area was lower than the island average.

This is a small sample size, but male biased sex ratios are not uncommon in varanid studies (De Lisle 1996). This is usually caused by males having larger ranges and being more active, causing a higher probability of being sampled (De Lisle 1996). The Aguiguan home range data support this.

In 2000, the Tinian Mayors office, through the Tinian Division of Fish and Wildlife (DFW) implemented a “monitor lizard control program” on Aguiguan. Monitor lizards were opportunistically shot by DFW personnel. In one week an estimated 150 lizards were shot (T. Castro, DFW, pers. comm.) It is not clear if this continued, but, if so, this is a possible cause for the apparent smaller size classes compared with those on Sarigan in that larger animals are more apt to be shot. However, the sexual dimorphism documented on Aguiguan argues against this. If only large lizards were shot then the females should have been closer in size to the males but this was not the case. The female to male size ratio was similar for both islands. If the population was similar in 2000 to the 2008, estimate then 150 lizards represented only about 7% of the population. Also, eight years should have been enough time for the growth of the larger size class.

The CNMI Division of Fish and Wildlife performed varanid trapping and visual transects in 2000 (Cruz *et al*. 2000) and 2002 (Esselstyn *et al*. 2002). In 1995 visual surveys were conducted for varanids but without any trapping (Esselstyn *et al*. 2002). Between 2000 and 2002, trapping rates fell from a mean of 34 lizards/100 trap days to a mean of 14 lizards/100 traps days. There was a corresponding drop in sighting rates from a mean of 10 lizards/hr of search time to a mean of 6 lizards/hr of search time. This drop was attributed to the monitor lizard control program. For the present study, the trapping rate on the grid was 10 lizards/100 trap days and the sighting rate on the transects was 5 lizards/hr of search time. Both of these rates are consistent with the 2002 rates and, if the trapping and sighting rates are a valid index of the population abundance, show a stable population for this time period. One would expect an increase in the population between 2002 and 2008 if the monitor control program in 2000 was in fact a onetime sharp drop in the population.

On Sarigan vertebrates were 54% of the prey items while they were only 2% on Aguiguan. Skink densities appear to be low on Aguiguan and this is probably the reason none were found in varanid stomachs. On Aguiguan, roaches are an important prey item which is surprising given the abundance of rats on the island. Given the large fat deposits and high population density, roaches appear to have a high nutritional value.

This roach species does well in loose soil (C. Campora, pers. comm.). Feral goats, by over grazing, could be enhancing the habitat for this roach species and indirectly fueling the high varanid densities.

The emphasis on vertebrate prey on Sarigan could explain the size differences with Aguiguan, with rats and skinks supporting larger varanids. The invertebrate prey on Aguiguan could support higher numbers of smaller lizards while the higher skink densities on Sarigan negatively affect invertebrate densities and in turn reduce the importance of invertebrates as a varanid prey item.

Monitor lizards are the only large predator on Aguiguan and occur there in high densities. Top level predators can substantially affect ecosystems, both directly and indirectly. What effects the species exerts on the ecology of Aguiguan is difficult to say. If the native Marianas birds did not in fact evolve with varanid predators, then removing the lizards should be ecologically beneficial for those birds. The importance of insects in the Aguiguan varanid diet puts them in direct competition with the endangered Micronesian megapode (*Megapodius laperouse laperouse*). Losos and Greene (1988) speculated that in terms of ecological effects, varanids (excepting the largest species) most closely mimic small foxes or some civet cat species.

REFERENCES

Buckland, S. T., Anderson, D.R., Burnham, K.P. and J.L. Laake. 1993. Distance sampling, estimating abundance of biological populations. Chapman and Hall. 446 pp.

Cota, M. 2008. *Varanus indicus* and its presence on the Mariana Islands: natural geographic distribution vs. introduction. Biawak, 2008. 2(1): 18-27.

Cruz, J., L. Arriola, N. Johnson and G. Beauprez. 2000. Wildlife and vegetation surveys Aguiguan 2000. Technical Report #2 CNMI-DFW, unpublished.

De Lisle, H.F. 1996. The natural history of monitor lizards. Krieger publishing. 1996. 201 pp.

Esselstyn, J. Cruz, J.B. Williams, LL. and N. Hawely. 2003. Wildlife and vegetation surveys, Aguiguan 2002. Technical report #9. Commonwealth of the Northern Marianas Islands, Division of Fish and Wildlife, Wildlife Section.

Losos, J.B. and H.W. Greene. 1988. Ecological and evolutionary implications of diet in monitor lizards. Journal of the Linnean Society (1988), 35: 379-407.

Reed, R. N., Morton, J. M. and G.E. Desy. 2000. Use of monofilament snare traps for capture of varanid lizards. Micronesica 33(1/2): 99-104, 2000.

APPENDIX

Stomach Contents of Aguiguan Varanids

	Prey item and # in stomach					
Specimen #	Roaches	Centipedes	Rats	Grasshopper	Hermit Crab	Other
1	7					
2					1	
3	empty					
4	13					
5	empty					
6	7					
7	3					Unknown animal tissue ~5mm x5 mm
8	3					1 snail
9		1				
10						Chicken and fish from camp
11	3					2 small eggs, ~1cm long (gecko or bird)
12	8					
13	14					
14	10	1				
15	empty					
16	3		1			
17	6	1		1		
18	2	2				
19	4					
20			1			
21	3					

2.4 AVIAN SURVEYS

2.4.1 GENERAL LAND BIRD SURVEYS ON TINIAN AND AGUIGUAN

Clockwise: Golden White-eye, Bridled White-eye, and Mariana Fruit-dove. All photos by Scott Vogt.

Prepared by: **Richard Camp (U.S. Geological Survey), Thane Pratt (U.S. Geological Survey), Fred Amidon (U.S. Fish and Wildlife Service), Ann Marshall (U.S. Fish and Wildlife Service), Shelly Kremer (U.S. Fish and Wildlife Service), and Megan Laut (U.S. Fish and Wildlife Service).** (Summarized from from Camp *et al.* (2009; Appendix 3.1) by Fred Amidon, U.S. Fish and Wildlife Service)

INTRODUCTION

The first island-wide surveys of terrestrial bird species on Tinian and Aguiguan were conducted in 1982 by Engbring *et al.* (1986). In 1995 and 1996, the surveys developed by Engbring *et al.* (1986) were repeated on Tinian, however, only the Tinian monarch data were analyzed (USFWS 1996, Lusk *et al.* 2000). On Aguiguan, the Engbring *et al.* (1986) surveys were repeated in 1992, 1995, 2000, and 2002 (Craig *et al.* 1992, Cruz *et al.* 2000, Esselstyn *et al.* 2003, USFWS Unpubl. data). Unfortunately, the data collected in these surveys on Aguiguan were not collected or analyzed using the same methods or were not analyzed.

In 2008, the Department of Defense contracted the U.S. Fish and Wildlife Service's Pacific Islands Fish and Wildlife Office to conduct terrestrial and marine surveys on Tinian and Aguiguan. The following report is a summary of Camp *et al.* (2009; Appendix 1) which outlines the survey results from June 2008 forest bird point-transect surveys on Tinian and Aguiguan and assesses population trends on Tinian and Aguiguan using point-transect data collected in 1982 on Tinian and Aguiguan by Engbring *et al.* (1986) and 1996 on Tinian by the U.S. Fish and Wildlife Service (unpublished data).

METHODS

Between 27 April and 8 May 1982 Engbring *et al.* (1986) sampled a total of 216 stations on 10 transects during a survey of Tinian (Figure 1, Attachment 1). All transects were at least 300 meters apart and all stations along each transect were 150 meters apart. These transects were resurveyed during both the 1996 (28 August – 1 September) and 2008 (14 – 19 June) surveys. An additional 4 transects were sampled during the 2008 survey for a total of 254 stations (Figure 1, Attachment 1). The 4 additional transects were included to increase the number of stations in native limestone forest to improve density estimates for Tinian monarchs (see Tinian Monarchs for additional information). These transects were also at least 300 m from the nearest transect and all stations were 150 meters apart.

On Aguiguan an island-wide survey consisting of 66 stations on 4 transects was conducted on 2 and 3 June 1982, and a partial survey (transects 1 and 2 only) was conducted on 10 and 11 March (Engbring *et al*. 1986; Figure 2, Attachment 2). Data from only the June survey were used in this study. All 4 transects were resurveyed during the 2008 (25 – 27 June) survey. An additional transect of 14 stations was sampled during the 2008 survey for a total of 80 stations (Figure 2, Attachment 2). This additional transect was added to sample secondary forest and open field habitats and increase the survey coverage of the island. The additional transect was at least 300 meters away from the nearest transect and all stations along all transects were 150 meters apart.

All surveys followed standard point-transect methods, consisting of 8-minute counts where horizontal distances to all birds heard and/or seen were measured and recorded (see Engbring *et al*. 1986 for details). Sampling conditions recorded included cloud cover, rain, wind, noise level, and habitat type, and these were later used as covariates in density calculations (see below). Counts commenced at sunrise and continued up to 1100 hours and were conducted only under favorable conditions. Two observers surveyed each station in 1982, and one observer surveyed the stations in 1996 and 2008. On Tinian, only data from one counter was used for each station from the 1982 surveys, and the primary counters were identified based on their experience and survey proficiency. Engbring *et al*. (1986) analyzed bird detections from all observers to estimate bird densities. For our analysis, we used detections from only one observer to recalculate densities for the 1982 Tinian survey, thus matching the 1996 and 2008 survey effort. Calculating densities from only one of the counters is a conservative approach and ensures sampling independence. This approach approximately halved the number of birds detected; however, our density estimates were generally greater than, but otherwise similar to, those of Engbring *et al*. (see Table 8; 1986). On Tinian the 95% confidence intervals bracketed Engbring *et al*.'s estimates for all but five birds—Mariana Fruit-Dove, Micronesian Honeyeater, Tinian Monarch, Rufous Fantail, and Bridled White-eye. Differences may have resulted from analytical procedures such as selecting different truncation distances, selecting different models to estimate densities, and analytical advances in distance sampling (see Johnson *et al*. 2006), in addition to estimating densities using detections from only one of the counts (Tinian only). Data from both counters were used to estimate 1982 densities on Aguiguan and the sampling effort was adjusted appropriately.

Population status was calculated as densities (birds/km^2) and number of birds (density by habitat type multiplied by habitat type area). Densities were calculated using the program DISTANCE (Thomas *et al*. 2006) from species-specific global detection functions where data were post-stratified by survey in the stratum layer. Data were right-truncated to facilitate model fitting (Buckland *et al*. 2001), and the model with the lowest Akaike Information Criterion (AIC) was used to select the detection function that best approximated the data. Candidate models included half-normal and hazard-rate detection functions with expansion series of order two (Buckland *et al*. 2001). Sampling covariates were modeled in the multiple-covariate distance engine of DISTANCE (Thomas *et al*. 2006, Marques *et al*. 2007). Covariates (sampling conditions, habitat types, and survey year.) were used to generate the global detection function when the best approximating model was improved by four or more AIC units. Variances and confidence intervals were derived by log-normal based methods. Survey-specific, density-by-station values were generated for the population trends analyses (see below) from the global detection function using the post-stratification-by-sample option in the stratum layers annual estimates and regional estimates. Area of habitat types came from Engbring *et al*. (1986) and recent vegetation cover estimates (see 2.1 Vegetation Surveys). The area of habitat types was not available for the 1996 Tinian survey; therefore, we used the area by habitat types from Engbring *et al*. to calculate the 1996 numbers of birds. This may slightly underestimate the population size if there was more secondary forest in 1996 than 1982. Agriculture habitat type (combined agroforestry and cultivated habitat type classifications) was not used to calculate numbers of birds because the area of this habitat is very small relative to the island (< 2%), the area of the agriculture habitat type has declined (190 ha in 1982 to 174 ha in 2008; see 2.1 Vegetation Surveys), and only two stations were located in the agriculture habitat type, thus it was under-sampled. On Aguiguan, the 1982 estimates of the area of habitat types were not reliable; therefore, numbers of birds were calculated only for the 2008 survey.

Change in bird densities among the three annual estimates on Tinian was assessed with repeated measures analysis of variance (ANOVA: PROC MIXED; SAS Institute Inc., Cary, NC). Repeated measures ANOVA was also used to asses change in bird density within regions among the three annual estimates. To stabilize the error variance, densities by station were log transformed after a constant of 1 was added (to avoid ln(0)). Stations were treated as the random factor, and because the number of repeated measures was too small to fit a covariance model, we assumed the variance-covariance structure was a compound symmetry, homogeneous variance model (Littell *et al*. 1996). Degrees of freedom were adjusted using the Kenward-Roger adjustment statement and a Tukey's adjustment was used to control alpha = 0.05 for multiple-comparison procedures.

End-point comparisons of the Aguiguan bird densities were compared using a two-sample z-test. Comparing density estimates using z-tests is the recommended method (L. Thomas, pers. comm. 2008) and is an extension of the method listed in Buckland *et al*. (2001).

RESULTS AND DISCUSSION

Tinian

On Tinian, a total of 18 species were detected during one or more of the three surveys (Table 1). Sufficient numbers of individuals were detected to calculate density and abundance estimates for the collared kingfisher, island-collared dove (previously known as Philippine turtle-dove), white-throated ground-dove, Mariana fruit-dove, white tern, rufous fantail, Tinian monarch, Micronesian honeyeater, Micronesian starling, bridled white-eye, yellow bittern, and Eurasian tree sparrow. Bridled white-eyes and rufous fantails were the most abundant birds, whereas the white-throated ground-dove and yellow bittern were the least abundant bird (Table 2). Collared kingfisher, white-throated ground-dove, rufous fantail, Micronesian starling, and yellow bittern abundance increased since 1982 while Mariana fruit-dove, Tinian monarch, and Micronesian honeyeater abundance decreased since 1982 (Table 3). Although these declines were not linear, the overall changes between 1982 and 2008 were significant (Table 3). Trends for the white tern and bridled white-eye were considered relatively stable. The introduced island collared-dove and Eurasian tree sparrow both increased since 1982 (Tables 2 and 3).

Only five birds; the white-throated ground-dove, Mariana fruit-dove, rufous fantail, Tinian monarch, and Eurasian tree sparrow, showed significant differences among regions by year (Table 4). Between 1982 and 2008 white-throated ground-dove densities increased in the Diablo and Hagoi regions, and rufous fantail densities increased in the Carolinas and Masalog regions (Figure 3). Over the 27-year period Mariana fruit-dove and Tinian monarch densities declined in the Carolinas and Diablo regions, respectively.

The increase in rufous fantail and Micronesian starling abundance may be related to the decline in open field and increase in secondary forest habitats on the island between the 1980s and 2008 (see 2.1 Vegetation Surveys). Both these species primarily utilize forest habitats, including secondary forest, for foraging and breeding. However, Tinian monarch and Micronesian honeyeater abundance should have increased as well since these species also utilize secondary forest. Therefore, other factors, like a decline in foraging resources, may also be influencing bird populations. For example, the increase in white-throated ground-doves may be related to an outbreak of the introduced vine *Coccina grandis* (scarlet gourd) which is believed to be a factor in the abundance of this species on Saipan (Camp *et al.*, in review).

Aguiguan

A total of 19 species were detected on one or both of the Aguiguan surveys (Table 5). Sufficient numbers of individuals were detected for nine native and one introduced species to calculate density and abundance estimates. Bridled white-eyes were the most abundant bird at over 44,000 birds and collared kingfisher and island collared-dove were the least abundant birds (Table 6). Densities for seven of the nine native birds; collared kingfisher, white-throated ground-dove, Mariana fruit-dove, rufous fantail, Micronesian starling, bridled white-eye, and golden white-eye, were significantly greater in 2008 than 1982 (Table 6, Figure 4). No differences in densities were detected between the two surveys for white tern and Micronesian honeyeater. Densities of the introduced island collared-dove increased significantly between 1982 and 2008.

The increased densities of collared kingfishers, white-throated ground-doves, Mariana fruit-doves, rufous fantails, Micronesian starlings, bridled white-eyes, and golden white-eyes may be related to an increase in forest cover on Aguiguan since the 1982 survey. Recent land cover surveys indicate that the amount of secondary forest and tangantangan has increased since the 1982 survey (see 2.1 Vegetation Surveys). Both of these habitat types could be utilized by these avian species. Interestingly, the densities of Micronesian honeyeater did not increase. However, as noted above for Tinian, this may be related to other factors beside habitat availability.

The magnitude of the increases in rufous fantail, bridled white-eye and golden white-eye densities was surprising. However, when the detections for these species from 1982 were compared to detections along the same four transects in 1992, 1995, 2000, 2002, and 2008 the detections were much lower in 1982 than the other years (Table 7). This could, in part, be related to survey conditions and the quality of the habitat. Goat populations on the island were reduced to very low numbers from 1989 to 1990 (Rice 1991) which may have affected the available habitat for these species and their primary prey, insects. For example the amount of understory vegetation may have increased and as well as the amount of secondary forest and tangantangan habitats (see 2.1 Vegetation Surveys). However, goat populations have since increased and intense browsing is evident throughout the forest. Therefore, a decline in these species would be expected. This was not observed, however, so other factors are also likely at play.

REFERENCES

Buckland, S.T., D.R. Anderson, K.P. Burnham, J.L. Laake, D.L. Borchers, and L. Thomas. 2001. Introduction to Distance Sampling: Estimating abundance of biological populations. Oxford: Oxford University Press.

Camp, R.J., T. K. Pratt, F. Amidon, A. P. Marshall, S. Kremer, and M. Laut. 2009. Draft status and trends of the land bird avifauna on Tinian and Aguiguan, Mariana Islands. Appendix 3.1 *in* Terrestrial Resource Surveys of Tinian and Aguiguan, Mariana Islands, 2008. Working Draft. U.S. Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office, Honolulu. HI.

Camp, R.J., T.K. Pratt, A.P. Marshall, F. Amidon, and L.L. Williams. In review to Bird Conservation International. Status and trends of the land bird avifauna on Saipan, Mariana Islands, with emphasis on the endangered nightingale reed-warbler (*Acrocephalus luscinia*).

Craig, R.J., R. Chandran, and A. Ellis. 1992. Bird populations of Aguiguan: a ten year update. Pages 8-15 *in* The Aguiguan Expedition (R.J. Craig, editor). Proceedings of the Mariana Research Symposium, Volume 1. College of the Northern Mariana Islands, Saipan, MI.

Cruz, J., L. Arriola, N. Johnson, and G. Beauprez. 2000. Wildlife and vegetation surveys Aguiguan 2000: Technical Report # 2. Commonwealth of the Northern Mariana Islands, Division of Fish and Wildlife, Saipan, MI. 30 pp.

Engbring, J., F.L. Ramsey, and V.J. Wildman. 1986. Micronesian forest bird survey, 1982: Saipan, Tinian, Aguijan, and Rota. U.S. Fish and Wildlife Service Report. 143pp.

Esselstyn, J., J.B. Cruz, L.L. Williams, and N. Hawley. 2003. Wildlife and vegetation surveys Aguiguan 2002: Technical Report #9. Commonwealth of the Northern Mariana Islands, Division of Fish and Wildlife, Saipan, MI. 55 pp.

Johnson, L., R.J. Camp, K.W. Brinck, and P.C. Banko. 2006. Long-term population monitoring: lessons learned from an endangered passerine in Hawai`i. Wildlife Society Bulletin 34:1055-1063.

Littell, R.C., G.A. Milliken, W.W. Stroup, R.D. Wolfinger. 1996. SAS system for mixed models. SAS Institute Inc., Cary, NC.

Lusk, M., S. Hess, M. Reynolds, and S. Johnston. 2000. Population status of the Tinian Monarch (*Monarcha takatsukasae*) on Tinian, Commonwealth of the Northern Mariana Islands. Micronesica 32:181-190.

Marques, T.A., L. Thomas, S.G. Fancy, and S.T. Buckland. 2007. Improving estimates of bird density using multiple-covariate distance sampling. Auk 124:1229-1243.

Rice, C.G. 1991. Goat removal from Aguijan Island: lessons for future efforts. Western Section of the Wildlife Society 27:42-46.

Thomas, L., J.L. Laake, S. Strindberg, F.F. C. Marques, S.T. Buckland, D.L. Borchers, D.R. Anderson, K.P. Burnham, S.L. Hedley, J.H. Pollard, J.R.B. Bishop, and T.A. Marques. 2006. Distance 5.0. Release 2. Research Unit for Wildlife Population Assessment, University of St. Andrews, U.K. http://www.ruwpa.st-and.ac.uk/distance/.

U.S. Fish and Wildlife Service. 1996. Wildlife research report for Navy-leased lands on the island of Tinian, CNMI. Prepared for Department of the Navy, PACNAVFACENGCOM, Pearl Harbor, Hawaii. 100 pp.

Wiles, G. J. 2005. A checklist of the birds and mammals of Micronesia. Micronesica 38:141-189.

Table 1. List of birds detected from three different point-transect surveys on Tinian. In 1982 and 1996 216 stations were sampled on 10 transects, and 254 stations were sampled on 14 transects in 2008. The number of stations occupied (# Stns Ocpd) and bird detected (# Dect), and indices of percent occurrence (% Occ) and birds per station (BPS) were calculated. Nomenclature follows Wiles (2005). Density estimates were produced for birds in bold.

		1982				1996				2008			
Species	Scientific Name	# Stns Ocpd	# Dect	% Occ	BPS	# Stns Ocpd	# Dect	% Occ	BPS	# Stns Ocpd	# Dect	% Occ	BPS
Red Junglefowl	*Gallus gallus*	45	105	20.8	0.49	0	0	0.0	0.00	45	77	17.7	0.30
White-tailed Tropicbird	*Phaethon lepturus*	0	0	0.0	0.00	0	0	0.0	0.00	3	5	1.2	0.02
Yellow Bittern	*Ixobrychus sinensis*	10	10	4.6	0.05	16	18	7.4	0.08	34	38	13.3	0.15
Pacific Reef-Egret	*Egretta sacra*	1	1	0.5	<0.01	1	1	0.5	<0.01	0	0	0.0	0.00
Pacific Golden-Plover	*Pluvialis fulva*	1	1	0.5	0.00	0	0	0.0	0.00	3	11	1.2	0.04
Ruddy Turnstone	*Arenaria interpres*	0	0	0.0	0.00	0	0	0.0	0.00	1	1	0.4	<0.01
Brown Noddy	*Anous stolidus*	0	0	0.0	0.00	0	0	0.0	0.00	1	1	0.4	<0.01
White Tern	*Gygis alba*	128	344	59.3	1.59	22	52	10.2	0.24	122	322	48.0	1.27
Island Collared-Dove	*Streptopelia bitorquata*	51	66	23.6	0.31	136	256	63.0	1.19	79	116	31.1	0.46
White-throated Ground-Dove	*Gallicolumba xanthonura*	13	16	6.0	0.07	23	23	10.6	0.11	64	82	25.2	0.32
Mariana Fruit-Dove	*Ptilinopus roseicapilla*	189	623	87.5	2.88	150	240	69.4	1.11	212	462	83.4	1.82
Collared Kingfisher	*Todiramphus chloris*	150	294	69.4	1.36	124	285	57.4	1.32	190	374	74.8	1.47
Micronesian Honeyeater	*Myzomela rubratra*	131	236	60.6	1.09	60	96	27.8	0.44	87	125	34.3	0.49
Tinian Monarch	*Monarcha takatsukasae*	187	539	86.6	2.50	173	500	80.1	2.31	178	361	70.1	1.42
Rufous Fantail	*Rhipidura rufifrons*	202	786	93.5	3.64	188	502	87.0	2.32	235	686	92.5	2.70
Bridled White-eye	*Zosterops conspicillatus*	216	2,222	100.0	10.29	216	1,770	100.0	8.19	253	2,024	99.6	7.97

Micronesian Starling	*Aplonis opaca*	177	513	81.9	2.38	106	226	49.1	1.05	215	614	84.7	2.42
Eurasian Tree Sparrow	*Passer montanus*	1	1	0.5	<0.01	3	13	1.4	0.06	13	62	5.1	0.24

Table 2. Population density and abundance estimates for land birds on Tinian from three point-transect surveys. Data from 10 Engbring *et al.* (1986) transects only. First row: mean density (birds/km^2 ± SE, with 95% CI). Second row: bird abundance (sum of density by habitat type times the area of habitat types) with 95% CI. Agriculture habitat type was dropped for calculating bird abundance due to small sample size; only 2 survey stations were sampled.

Species	1982	1996	2008
Yellow Bittern	1.5 ± 0.89 (0.5–4.4)	7.4 ± 2.49 (3.9–14.1)	18.2 ± 4.56 (11.2–29.6)
	127 (30–550)	764 (270–2,302)	1,695 (835–3,575)
White Tern	144.1 ± 17.24 (113.9–182.2)	25.3 ± 7.01 (14.8–43.2)	169.9 ± 19.66 (135.4–213.2)
	13,980 (9,349–21,512)	2,846 (1,121–7,300)	15,147 (10,067–23,041)
Island Collared-Dove	12.4 ± 2.04 (9.0–17.1)	34.3 ± 3.67 (27.8–42.3)	23.9 ± 3.24 (18.4–31.2)
	1,093 (642–2,024)	3,291 (2,296–4,777)	2,198 (1,374–3,648)
White-throated Ground-Dove	4.1 ± 1.45 (2.0–8.0)	4.6 ± 1.30 (2.7–8.0)	20.2 ± 3.91 (13.8–29.5)
	434 (136–1,421)	440 (174–1,147)	1,827 (1,045–3,226)
Mariana Fruit-Dove	42.6 ± 2.64 (37.7–48.1)	15.8 ± 1.23 (13.6–18.4)	33.1 ± 1.96 (29.4–37.1)
	3,909 (3,185–4,826)	1,539 (1,155–2,065)	3,029 (2,506–3,677)
Collared Kingfisher	7.0 ± 1.46 (4.7–10.5)	22.9 ± 3.28 (17.3–30.3)	61.3 ± 4.33 (53.3–70.4)
	570 (305–1,130)	2,268 (1,329–3,883)	5,439 (4,212–7,090)
Micronesian Honeyeater	77.2 ± 6.79 (64.9–91.7)	31.2 ± 4.26 (23.9–40.8)	41.3 ± 4.86 (32.8–52.0)
	7,859 (5,877–10,700)	2,847 (1,684–4,838)	3,716 (2,458–5,667)
Tinian Monarch	634.5 ± 37.88 (564.3–713.4)	705.7 ± 43.96 (624.3–797.6)	431.3 ± 30.75 (374.9–496.2)
	60,898 (49,484–75,398)	62,863 (50,476–78,758)	38,449 (29,992–49,849)
Rufous Fantail	641.2 ± 39.30 (568.4–723.3)	766.3 ± 40.85 (690.1–851.0)	975.0 ± 48.26 (884.6–1,074.6)
	58,336 (48,119–71,134)	67,191 (55,510–82,000)	86,112 (72,786–102,594)
Bridled White-eye	3,190.9 ± 101.79 (2,996.8–3,397.6)	2,731.9 ± 81.96 (2,575.5–2,897.8)	2,997.2 ± 105.80 (2,795.8–3,213.0)
	302,477 (270,218–338,821)	253,407 (225,258–286,044)	270,785 (239,579–306,772)
Micronesian Starling	133.9 ± 13.53 (109.8–163.3)	125.1 ± 13.34 (101.5–154.2)	349.5 ± 22.47 (308.0–396.6)
	11,543 (7,994–17,041)	10,841 (7,270–16,296)	30,088 (23,633–38,565)

Eurasian Tree Sparrow	2.1 ± 2.07 (0.4–10.7)	26.7 ± 16.42 (8.7–81.5)	110.2 ± 40.54 (54.7–222.2)
	155 (29–817)	1,244 (232–6,662)	2,111 (429–10,666)

Table 3. Repeated measures analysis of variance results for trends in Tinian bird densities among years. Trends are denoted as increasing (▲), decreasing (▼), or stable (▬). Significant changes are marked in **bold**. Degrees of freedom for the differences of least squares means (Diff LSM) are 431.

				Diff LSM								
		Fixed Effects		82-96			82-08			96-08		
Species	Trend	$F_{2,398}$	*p*	Est (SE)	*t*	Adj-*p*	Est (SE)	*t*	Adj-*p*	Est (SE)	*t*	Adj-*p*
Yellow Bittern	▲	13.57	<0.001	-0.04 (0.02)	-1.86	0.153	-0.10 (0.02)	-5.14	<0.001	-0.07 (0.02)	-3.29	0.003
White Tern	▬	43.18	<0.001	0.47 (0.06)	7.55	<0.001	-0.06 (0.06)	-0.91	0.634	-0.53 (0.06)	-8.46	<0.001
Island Collared-Dove	▲	16.22	<0.001	-0.14 (0.03)	-5.66	<0.001	-0.09 (0.03)	-3.38	0.002	0.06 (0.03)	2.28	0.060
White-throated Ground-Dove	▲	27.87	<0.001	<0.01 (0.02)	-0.42	0.906	-0.12 (0.02)	-6.67	<0.001	-0.11 (0.02)	-6.24	<0.001
Mariana Fruit-Dove	▼	64.54	<0.001	0.19 (0.02)	10.92	<0.001	0.05 (0.02)	2.73	0.018	-0.14 (0.02)	-8.19	<0.001
Collared Kingfisher	▲	87.05	<0.001	-0.11 (0.03)	-3.79	<0.001	-0.36 (0.03)	-12.84	<0.001	-0.26 (0.03)	-9.05	<0.001
Micronesian Honeyeater	▼	31.76	<0.001	0.27 (0.04)	7.59	<0.001	0.20 (0.04)	5.90	<0.001	-0.06 (0.04)	-1.69	0.209
Tinian Monarch	▼	10.65	<0.001	-0.09 (0.09)	-0.97	0.597	0.31 (0.09)	3.42	0.002	0.40 (0.09)	4.39	<0.001
Rufous Fantail	▲	19.55	<0.001	-0.24 (0.09)	-2.75	0.017	-0.54 (0.09)	-6.24	<0.001	-0.30 (0.09)	-3.49	0.002
Bridled White-eye	▬	5.26	0.006	0.16 (0.05)	3.24	0.004	0.07 (0.05)	1.42	0.330	-0.09 (0.05)	-1.81	0.166
Micronesian Starling	▲	67.87	<0.001	0.04 (0.07)	0.57	0.836	-0.64 (0.07)	-9.79	<0.001	-0.68 (0.07)	-10.36	<0.001
Eurasian Tree Sparrow	▬	0.96	0.384	-0.02 (0.02)	-0.78	0.713	-0.03 (0.02)	-1.38	0.352	-0.01 (0.02)	-0.60	0.822

Table 4. Repeated measures analysis of variance results for year, region and year-region interaction fixed effects in Tinian bird densities. Data from 10 Engbring *et al.* (1986) transects only. Dash indicates interaction test not conducted because one or both main effects results were non-significant. Differences of least squares means for the significant fixed effects (bold for interaction, italics for region) are summarized in Figure 3.

	Fixed Effects					
	Year		Region		Interaction	
Species	$F_{2,392}$	P	$F_{3,196}$	P	$F_{6,392}$	P
Yellow Bittern	10.17	<0.001	0.20	0.899	—	—
White Tern	40.78	<0.001	4.15	0.007	1.71	0.116
Island Collared-Dove	19.67	<0.001	1.47	0.224	—	—
White-throated Ground-Dove	16.98	<0.001	5.19	0.002	6.60	<0.001
Mariana Fruit-Dove	66.10	<0.001	5.99	<0.001	3.76	0.001
Collared Kingfisher	81.67	<0.001	2.17	0.093	—	—
Micronesian Honeyeater	25.99	<0.001	10.89	<0.001	1.73	0.113
Tinian Monarch	8.94	<0.001	7.61	<0.001	3.10	0.006
Rufous Fantail	28.31	<0.001	5.23	0.002	6.63	<0.001
Bridled White-eye	9.29	<0.001	6.04	<0.001	11.58	<0.001
Micronesian Starling	62.05	<0.001	3.60	0.014	1.43	0.200
Eurasian Tree Sparrow	1.29	0.276	1.36	0.256	—	—

Table 5. List of birds detected from the 1982 and 2008 point-transect surveys on Aguiguan. In 1982 66 stations were sampled on 4 transects (88 counts; several stations were counted more than once), and 80 stations were sampled in 5 transects in 2008. The number of stations occupied (Stns Ocpd), birds detected (# Dect), indices of percent occurrence (% Occ) and birds per station (BPS) were calculated. Nomenclature follows Wiles (2005). Density estimates were produced for birds in bold. Scientific names are provided in superscript.

	1982				2008			
Species	# Stns Ocpd	# Dect	% Occ	BPS	# Stns Ocpd	# Dect	% Occ	BPS
Micronesian Megapode	8	14	9.1	0.16	11	15	13.8	0.19
White-tailed Tropicbird	1	1	1.1	0.01	—	—	—	—
Red-tailed Tropicbird[1]	8	13	9.1	0.15	—	—	—	—
Great Frigatebird[2]	1	2	1.1	0.02	—	—	—	—
Yellow Bittern	1	1	1.1	0.01	—	—	—	—
Brown Noddy	14	20	15.9	0.23	—	—	—	—
Black Noddy[3]	31	75	35.2	0.85	1	1	1.2	0.01
White Tern	54	218	61.4	2.48	34	84	42.5	1.05
Sooty Tern[4]	1	1	1.1	0.01	—	—	—	—
Island Collared-Dove	9	16	10.2	0.18	28	50	35	0.63
White-throated Ground-Dove	10	18	11.4	0.20	25	37	31.2	0.46
Mariana Fruit-Dove	87	757	98.9	8.60	75	240	93.8	3.00
Guam Swiftlet	26	157	29.6	1.78	9	27	11.2	0.34
Collared Kingfisher	56	154	63.6	1.75	53	101	66.2	1.26
Micronesian Honeyeater	87	745	98.9	8.47	74	174	92.5	2.18
Rufous Fantail	84	453	95.5	5.15	77	219	96.2	2.74
Golden White-eye	83	444	94.3	5.05	74	268	92.5	3.35
Bridled White-eye	88	823	100.0	9.35	77	758	96.2	9.48
Micronesian Starling	71	207	80.7	2.35	69	167	86.2	2.09

[1] = *Megapodius laperouse*
[2] = *Aerodramus bartschi*
[3] = *Sterna fuscata*
[4] = *Anous minutus*
[5] = *Phaethon rubricauda*
[6] = *Fregata minor*
[7] = *Cleptornis marchei*

Table 6. Population density and abundance estimates for native and alien Aguiguan land birds from two point-transect surveys. First row: mean density (birds/km^2 ± SE, with 95% CI). Second row: 2008 bird abundance (density by habitat times the habitat area) with 95% CI. Significance was assessed at the alpha 0.05 level using two-sample *z*-test (highlighted in bold). Change was defined as increasing (▲), decreasing (▼), or not significantly different (▬).

Species	1982	2008	*z* Value	*P*	Change
White Tern	169.6 ± 27.0 (124.2–231.6)	218.8 ± 44.2 (147.3–325.1)	-0.95	0.341	▬
		1,214 (604–3,651)			
Island Collared-Dove	4.4 ± 1.8 (2.0–9.7)	66.9 ± 16.7 (41.1–108.8)	-3.72	<0.001	▲
		307 (151–658)			
White-throated Ground-Dove	13.1 ± 4.8 (6.6–26.3)	100.2 ± 26.5 (59.9–167.6)	-3.23	0.001	▲
		484 (260–953)			
Mariana Fruit-Dove	107.5 ± 6.5 (95.4–121.1)	141.0 ± 10.8 (121.3–164.0)	-2.67	0.008	▲
		818 (604–1,170)			
Collared Kingfisher	13.1 ± 2.0 (9.7–17.8)	50.3 ± 6.6 (38.9–65.0)	-5.39	<0.001	▲
		347 (184–1,186)			
Micronesian Honeyeater	368.3 ± 19.6 (331.8–408.7)	336.2 ± 27.1 (286.7–394.1)	-0.96	0.337	▬
		2,128 (1,564–3,046)			
Rufous Fantail	568.8 ± 39.6 (496.0–652.2)	1.157.9 ± 89.3 (995.0–1,347.5)	-6.41	<0.001	▲
		6,429 (4,765–13,666)			
Golden White-eye	529.1 ± 40.6 (455.1–615.2)	1,292.6 ± 111.9 (1,089.7–1,533.4)	-6.41	<0.001	▲
		7,496 (4,983–17,387)			
Bridled White-eye	1,685.6 ± 102.3 (1,495.7–1,899.6)	6,771.2 ± 490.2 (5,867.6–7,814.1)	-10.15	<0.001	▲
		44,293 (32,246–63,031)			
Micronesian Starling	86.5 ± 10.9 (67.6–110.7)	505.2 ± 52.7 (411.5–620.3)	-7.78	<0.001	▲
		3,531 (1,902–12,374)			

Table 7. Number of birds detected (# Det) and birds per station (BPS) for 11 species recorded on 4 transects (66 stations) in 1982 (Engbring *et al.* 1986), 1992 (Craig *et al.* 1992), 1995 (USFWS, unpubl. data), 2000 (Cruz *et al.* 2000), 2002 (Esselstyn *et al.* 2003), and 2008. Eight minute counts were utilized in 1982, 1992, 1995, and 2008 while 5-minute counts were utilized in 2000 and 2002. Data from an additional transect sampled in 2008 were not included in the table.

	June 1982		May 1992		June1995		April 2000		March 2002		June 2008	
Species	# Det	BPS	# Det	BPS	# Det	BPS	# Det	BPS	# Det	BPS	# Det	BPS
Micronesian Megapode	14	0.16	11	0.17	18	0.27	12	0.20	16	0.30	13	0.20
Collared Kingfisher	56	1.75	83	1.26	89	1.35	57	0.60	40	0.90	92	1.39
Island Collared-Dove	9	0.18	11	0.17	3	0.05	3	0.02	1	0.05	17	0.26
White-throated Ground-Dove	10	0.20	8	0.12	22	0.33	16	0.20	12	0.30	22	0.33
Mariana Fruit-Dove	87	8.60	138	2.09	140	2.12	76	1.60	102	1.20	185	2.80
White Tern	64	2.48	113	1.71	86	1.30	42	0.83	52	0.67	44	0.67
Rufous Fantail	84	5.15	273	4.14	163	2.47	150	2.70	171	2.40	188	2.85
Micronesian Honeyeater	87	8.47	202	3.06	188	2.85	124	2.10	131	2.00	129	1.95
Micronesian Starling	71	2.35	127	1.92	75	1.14	74	0.90	57	1.20	139	2.11
Bridled White-eye	88	9.35	514	7.79	311	4.71	218	7.50	472	3.50	603	9.14
Golden White-eye	83	5.05	425	3.71	157	2.38	147	2.40	153	2.30	208	3.15

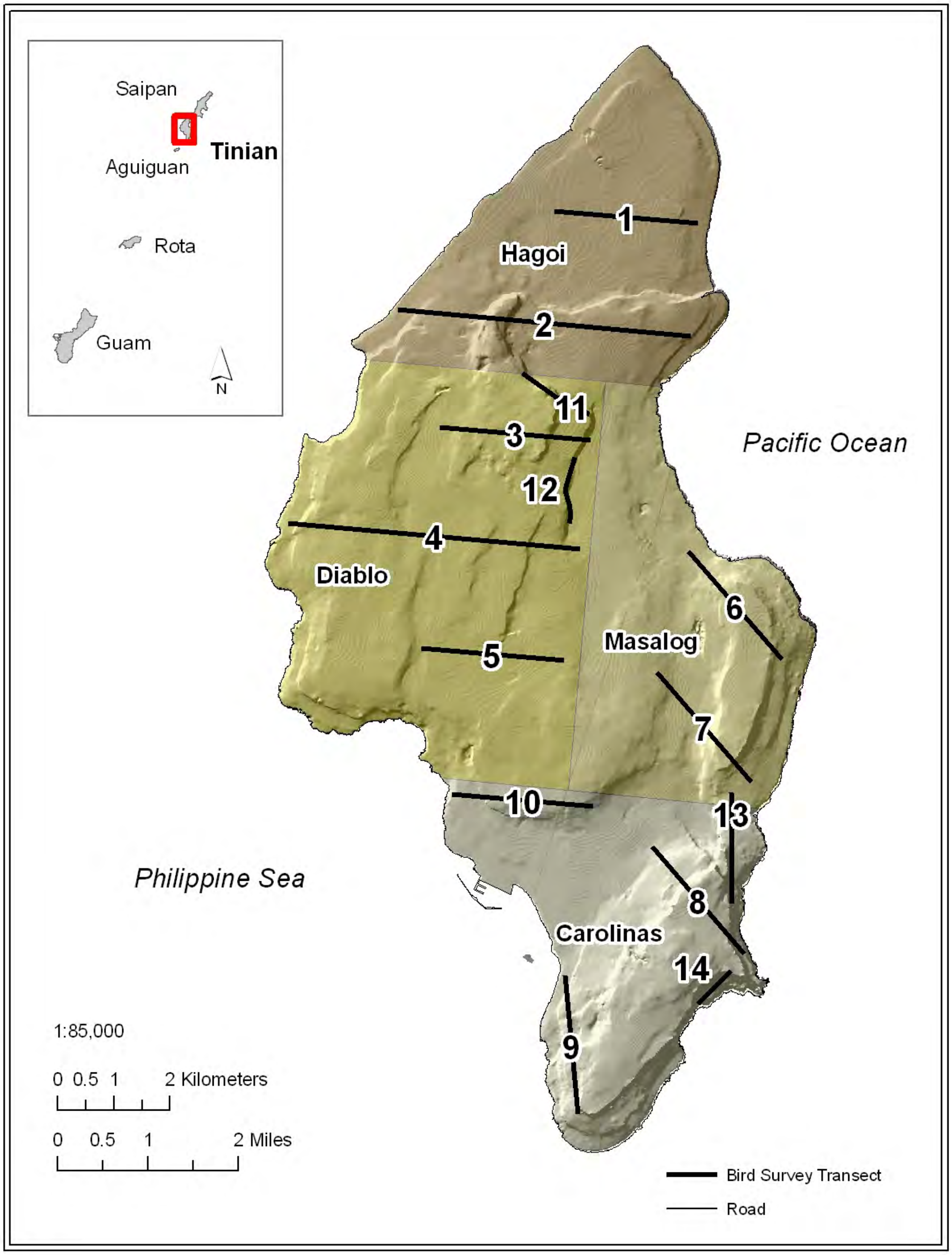


Figure 1. Island of Tinian showing the survey transects and regions (as defined by Engbring *et al.* 1986). Transects 1-10 were counted during all three surveys, and transects 11-14 were established and counted during the 2008 survey.

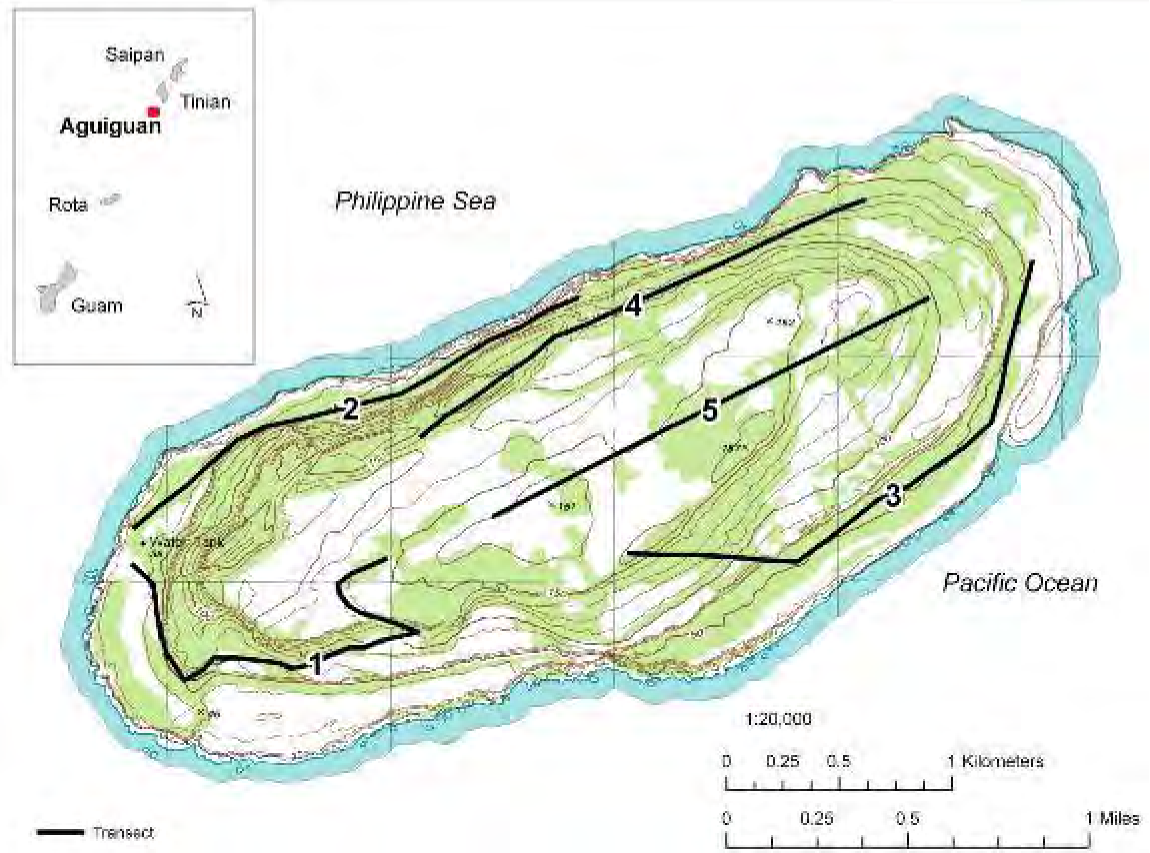
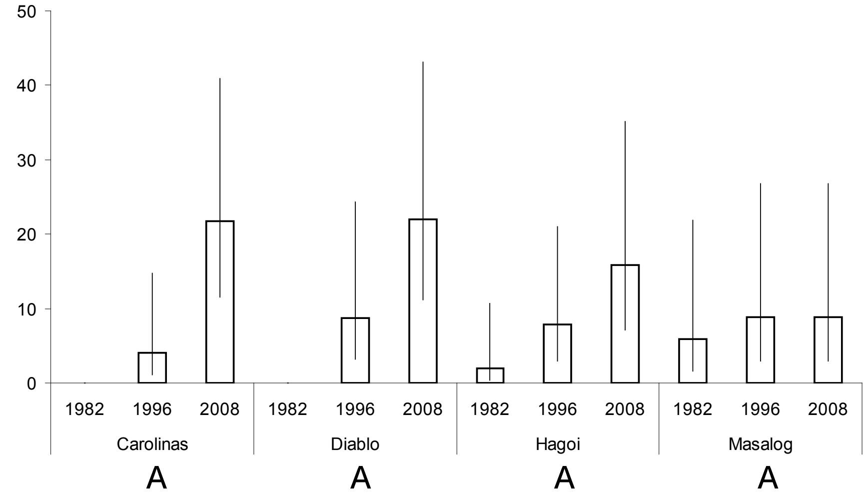
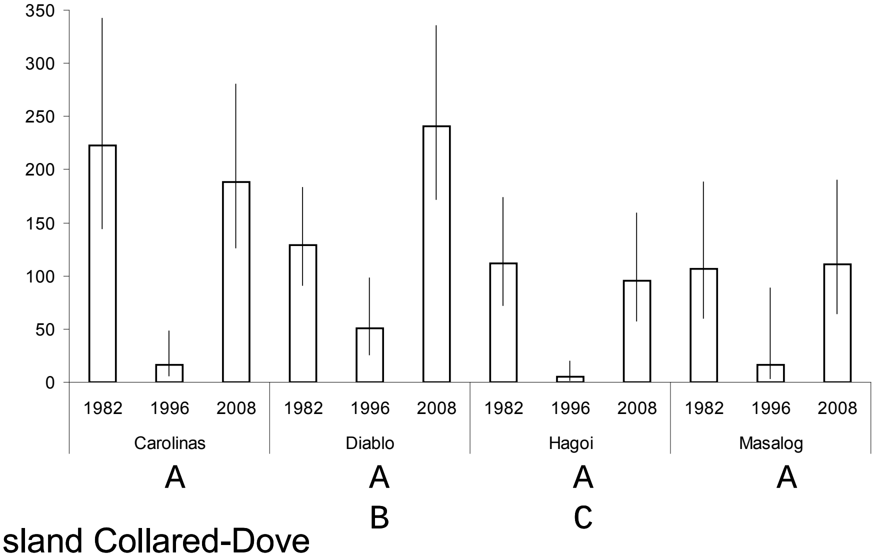
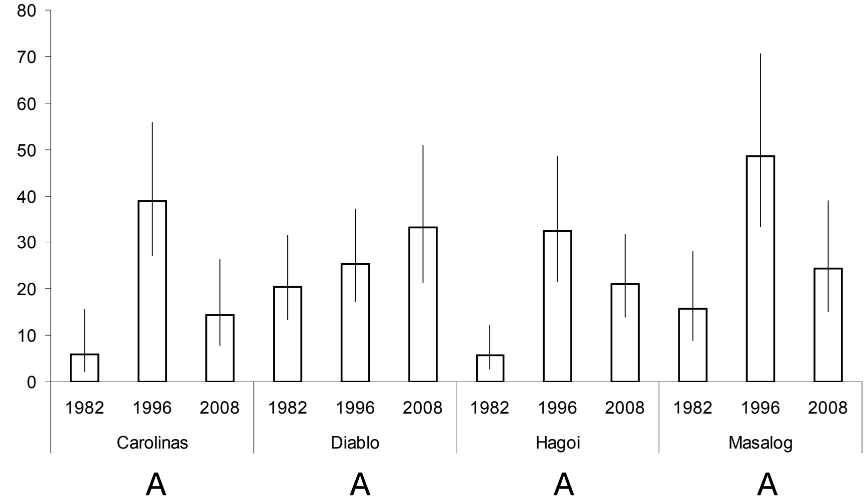
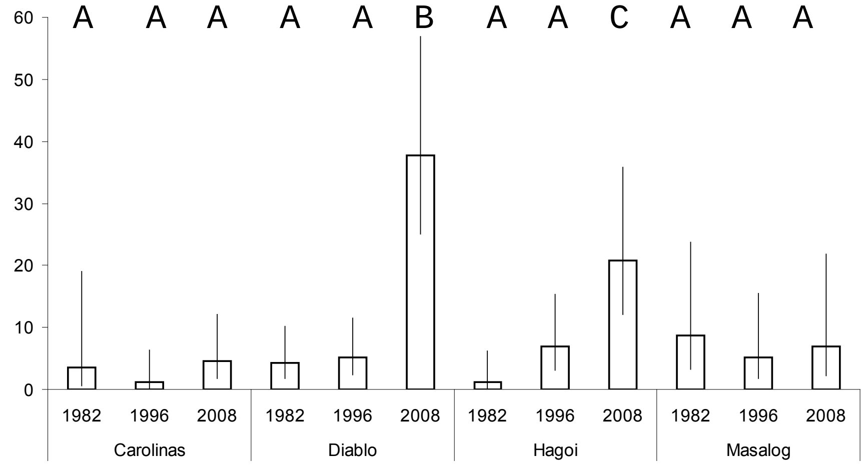


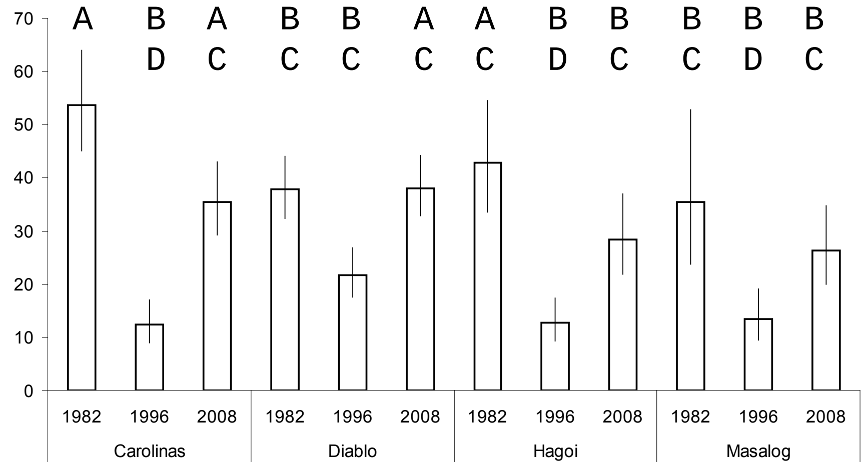
Figure 2. Island of Aguiguan showing the survey transects. Transects 1-4 were counted during both the 1982 and 2008 surveys, whereas transect 5 was established and counted during the 2008 survey.

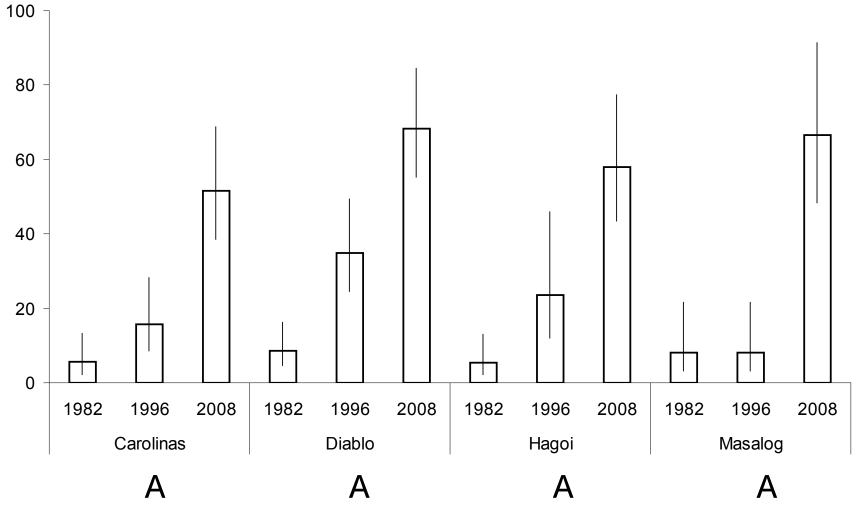

Yellow Bittern


White Tern

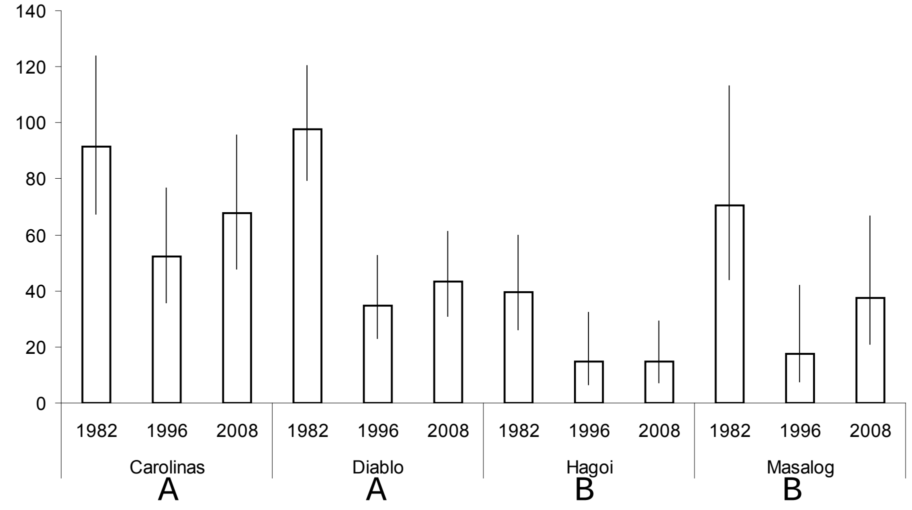

Island Collared-Dove


White-throated Ground-Dove

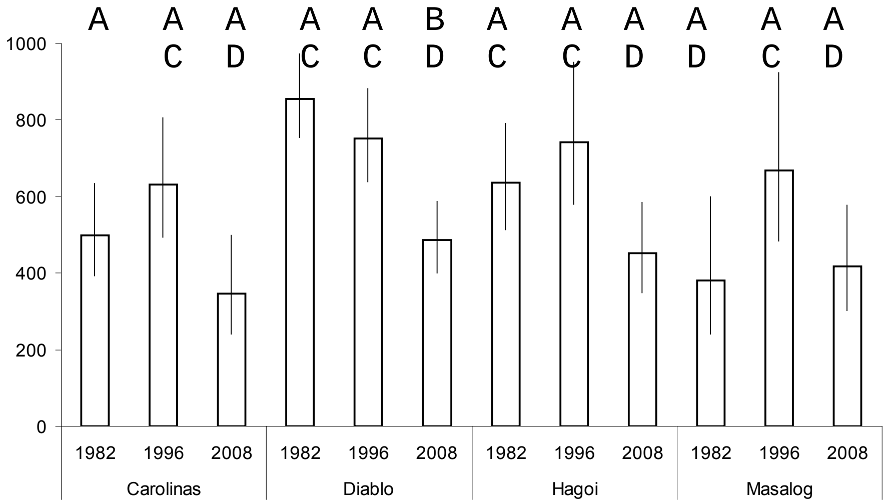

Mariana Fruit-Dove


Collared Kingfisher

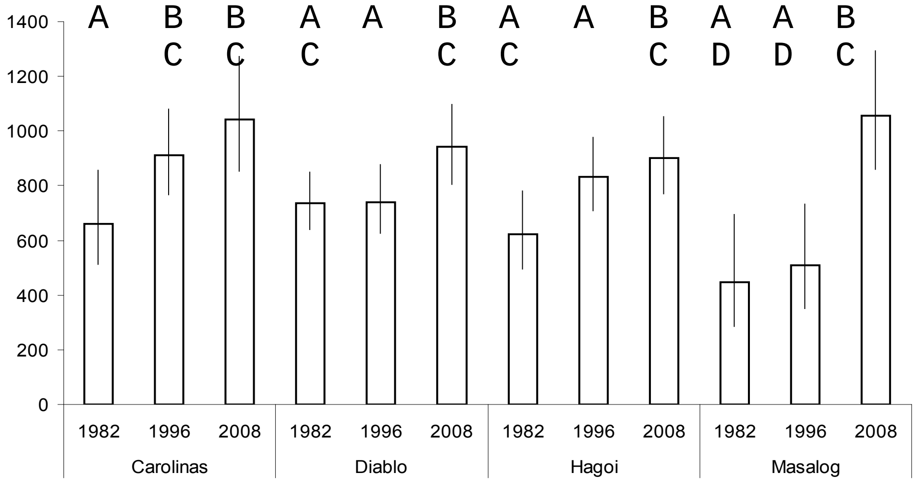
Micronesian Honeyeater



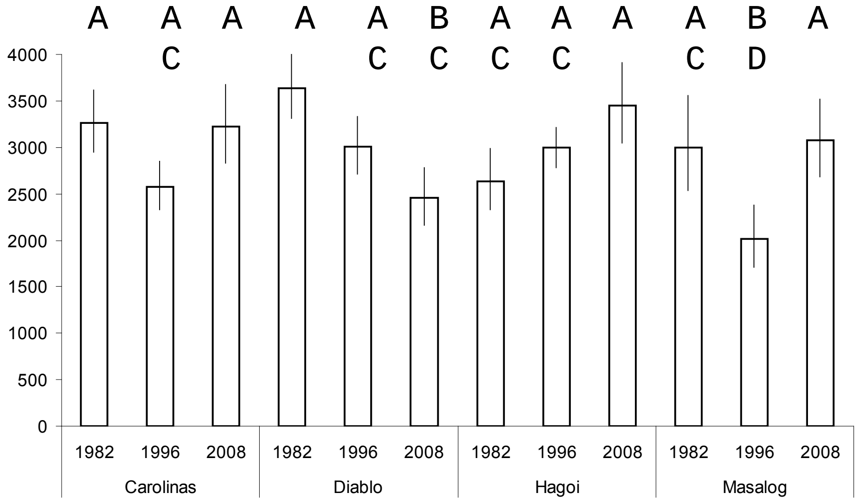
Tinian Monarch



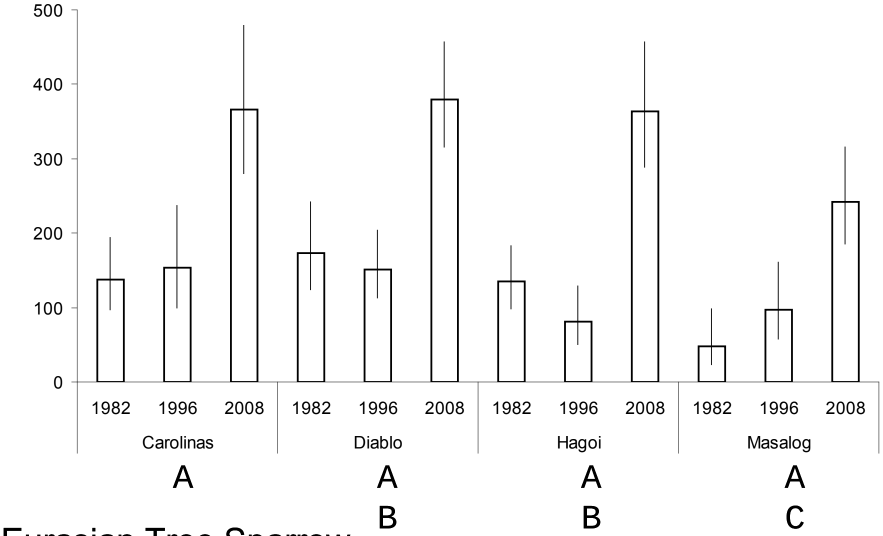
Rufous Fantail



Bridled White-eye



Micronesian Starling



Eurasian Tree Sparrow

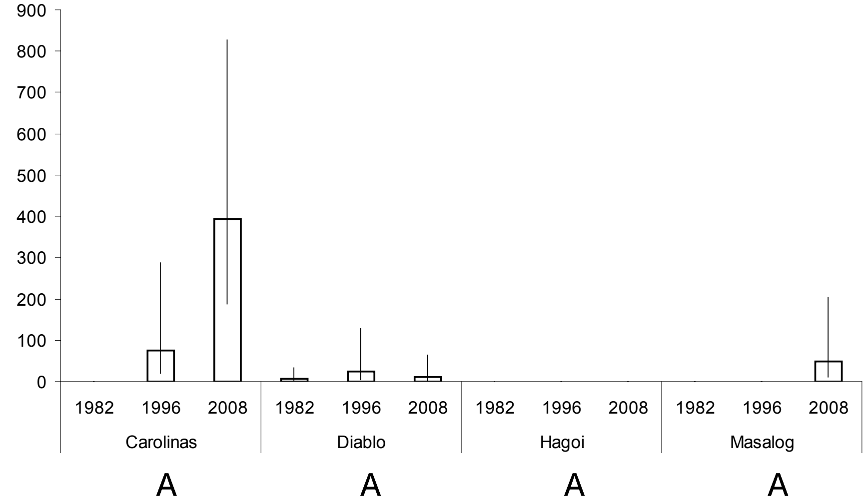


Figure 3. Density estimates (birds/km^2 and 95% CI) for Tinian land birds by region and year from three point-transect surveys. Differences of least squares means were assessed with repeated measures ANOVA. Comparisons that share the same letter are not significantly different at the 0.05 level, adjusted for multiple comparisons. Comparisons below species name are year within region results (i.e., significant year, region and interaction effects), whereas comparisons below x-axis indicate fixed effects results (*i.e.*, region or interaction effects were not significant).

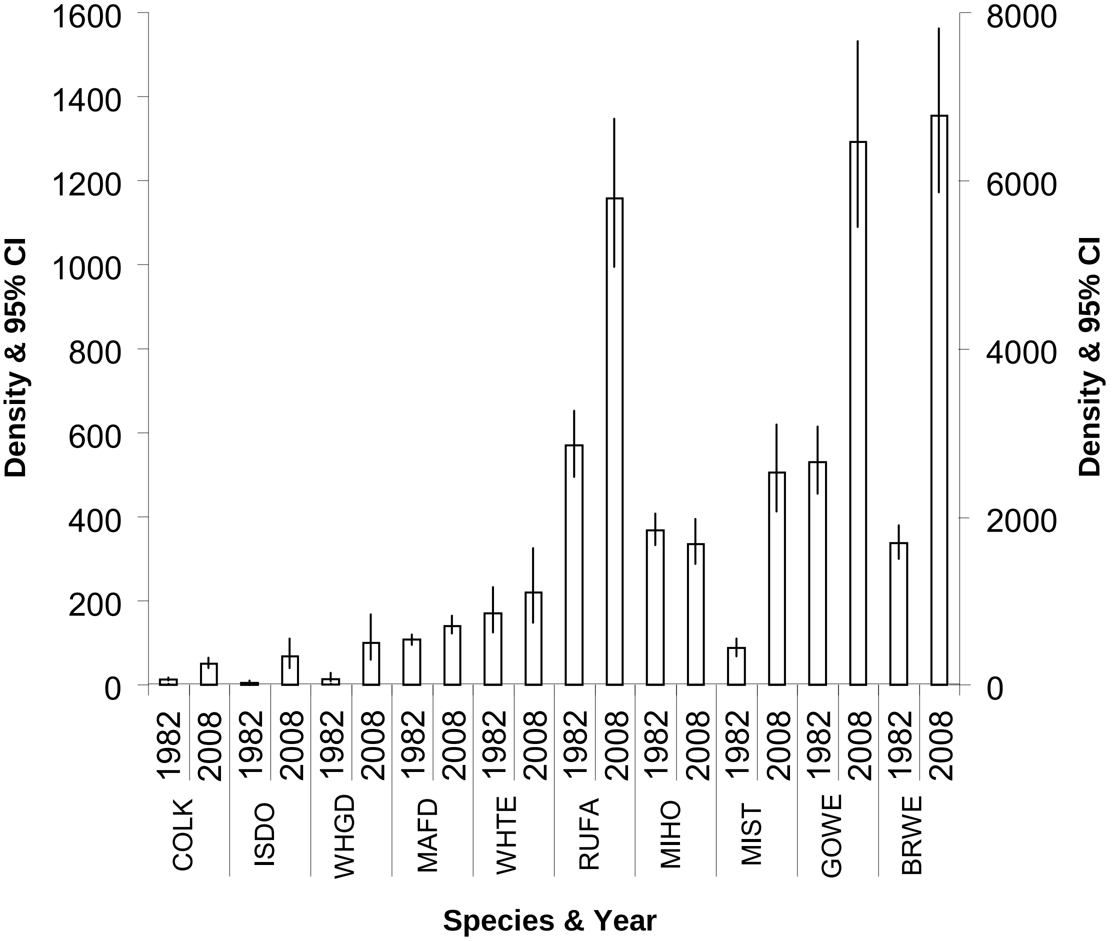


Figure 4. Density estimates (birds/km^2 and 95% CI) for native and alien Aguiguan land birds from two point-transect surveys. The primary y-axis is for the first nine species, and the secondary y-axis is for Bridled White-eye. Species codes are COLK – Collared Kingfisher; ISDO – Island Collared-Dove; WHGD – White-throated Ground-Dove; MAFD – Mariana Fruit-Dove; WHTE – White Tern; RUFA – Rufous Fantail; MIHO – Micronesian Honeyeater; MIST – Micronesian Starling; GOWE – Golden White-eye; and BRWE – Bridled White-eye.

Attachment 1. UTM coordinates for the point-transect or variable circular plot survey transects on the island of Tinian. All coordinates are in WGS84 UTM Zone 5 North. Transects 1 through 10 were established by Engbring *et al.* (1986) in 1982 and transects 11 through 14 was established in 2008.

Transect	Station	Latitude	Longitude
1	1	352759.43	1667002.60
1	2	352908.86	1666989.54
1	3	353058.29	1666976.47
1	4	353207.72	1666963.40
1	5	353357.15	1666950.34
1	6	353506.58	1666937.27
1	7	353656.01	1666924.20
1	8	353805.44	1666911.14
1	9	353954.87	1666898.07
1	10	354104.30	1666885.00
1	11	354253.73	1666871.94
1	12	354403.16	1666858.87
1	13	354552.59	1666845.80
1	14	354702.02	1666832.74
1	15	354851.45	1666819.67
1	16	355000.88	1666806.60
1	17	355150.31	1666793.54
1	18	355299.74	1666780.47
2	1	349965.19	1665235.33
2	2	350114.64	1665222.50
2	3	350264.09	1665209.68
2	4	350413.54	1665196.85
2	5	350562.99	1665184.02
2	6	350712.44	1665171.19
2	7	350861.89	1665158.36
2	8	351011.34	1665145.53
2	9	351160.77	1665132.46
2	10	351310.20	1665119.40
2	11	351459.63	1665106.33
2	12	351609.06	1665093.26
2	13	351758.49	1665080.20
2	14	351907.92	1665067.13
2	15	352057.35	1665054.06
2	16	352206.78	1665041.00
2	17	352356.21	1665027.93
2	18	352505.64	1665014.86
2	19	352655.07	1665001.80
2	20	352804.50	1664988.73
2	21	352953.93	1664975.66
2	22	353103.36	1664962.60
2	23	353252.79	1664949.53
2	24	353402.22	1664936.46
2	25	353551.65	1664923.40
2	26	353701.08	1664910.33
2	27	353850.51	1664897.26
2	28	353999.94	1664884.20

Transect	Station	Latitude	Longitude
2	29	354149.37	1664871.13
2	30	354298.80	1664858.06
2	31	354448.23	1664845.00
2	32	354597.66	1664831.93
2	33	354747.09	1664818.86
2	34	354896.52	1664805.80
2	35	355045.95	1664792.73
2	36	355195.38	1664779.66
3	1	350720.82	1663173.65
3	2	350870.25	1663160.58
3	3	351019.68	1663147.52
3	4	351169.11	1663134.45
3	5	351318.54	1663121.38
3	6	351467.97	1663108.32
3	7	351617.40	1663095.25
3	8	351766.83	1663082.18
3	9	351916.26	1663069.12
3	10	352065.69	1663056.05
3	11	352215.12	1663042.98
3	12	352364.55	1663029.92
3	13	352513.98	1663016.85
3	14	352663.41	1663003.78
3	15	352812.84	1662990.72
3	16	352962.27	1662977.65
3	17	353111.70	1662964.58
3	18	353261.13	1662951.52
3	19	353410.56	1662938.45
4	1	347996.48	1661425.61
4	2	348145.93	1661412.79
4	3	348295.38	1661399.96
4	4	348444.83	1661387.13
4	5	348594.28	1661374.30
4	6	348743.73	1661361.47
4	7	348893.18	1661348.64
4	8	349042.63	1661335.81
4	9	349192.08	1661322.98
4	10	349341.53	1661310.16
4	11	349490.98	1661297.33
4	12	349640.43	1661284.50
4	13	349789.88	1661271.67
4	14	349939.33	1661258.84
4	15	350088.78	1661246.01
4	16	350238.23	1661233.18
4	17	350387.69	1661220.35
4	18	350537.14	1661207.53
4	19	350686.59	1661194.70
4	20	350836.02	1661181.63
4	21	350985.45	1661168.56
4	22	351134.88	1661155.50
4	23	351284.31	1661142.43
4	24	351433.73	1661129.36

Transect	Station	Latitude	Longitude
4	25	351583.16	1661116.30
4	26	351732.59	1661103.23
4	27	351882.02	1661090.16
4	28	352031.45	1661077.10
4	29	352180.88	1661064.03
4	30	352330.31	1661050.96
4	31	352479.74	1661037.90
4	32	352629.17	1661024.83
4	33	352778.60	1661011.76
4	34	352928.03	1660998.70
4	35	353077.46	1660985.63
4	36	353226.89	1660972.56
5	1	350389.08	1659209.01
5	2	350538.53	1659196.18
5	3	350687.96	1659183.11
5	4	350837.39	1659170.04
5	5	350986.82	1659156.98
5	6	351136.25	1659143.91
5	7	351285.68	1659130.84
5	8	351435.11	1659117.78
5	9	351584.54	1659104.71
5	10	351733.97	1659091.64
5	11	351883.40	1659078.58
5	12	352032.83	1659065.51
5	13	352182.26	1659052.44
5	14	352331.69	1659039.38
5	15	352481.12	1659026.31
5	16	352630.55	1659013.24
5	17	352779.98	1659000.18
5	18	352929.41	1658987.11
6	1	356813.25	1658982.58
6	2	356716.59	1659097.29
6	3	356619.94	1659212.00
6	4	356523.29	1659326.71
6	5	356426.64	1659441.42
6	6	356329.99	1659556.13
6	7	356233.33	1659670.84
6	8	356136.68	1659785.55
6	9	356040.03	1659900.26
6	10	355943.38	1660014.97
6	11	355846.73	1660129.68
6	12	355750.08	1660244.39
6	13	355653.42	1660359.10
6	14	355556.77	1660473.81
6	15	355460.12	1660588.52
6	16	355363.47	1660703.23
6	17	355266.82	1660817.94
6	18	355170.16	1660932.65
7	1	354606.71	1658786.90
7	2	354703.36	1658672.19
7	3	354800.02	1658557.48

Transect	Station	Latitude	Longitude
7	4	354896.67	1658442.77
7	5	354993.32	1658328.06
7	6	355089.97	1658213.35
7	7	355186.62	1658098.64
7	8	355283.28	1657983.93
7	9	355379.93	1657869.22
7	10	355476.58	1657754.51
7	11	355573.23	1657639.80
7	12	355669.88	1657525.09
7	13	355766.53	1657410.38
7	14	355863.19	1657295.67
7	15	355959.84	1657180.96
7	16	356056.49	1657066.25
7	17	356153.14	1656951.54
7	18	356249.79	1656836.82
8	1	354504.87	1655695.61
8	2	354601.34	1655580.75
8	3	354697.81	1655465.89
8	4	354794.28	1655351.02
8	5	354890.75	1655236.16
8	6	354987.22	1655121.29
8	7	355083.69	1655006.43
8	8	355180.16	1654891.57
8	9	355276.62	1654776.70
8	10	355373.09	1654661.84
8	11	355469.56	1654546.97
8	12	355566.03	1654432.11
8	13	355662.50	1654317.25
8	14	355758.97	1654202.38
8	15	355855.44	1654087.52
8	16	355951.91	1653972.65
8	17	356048.38	1653857.79
8	18	356144.85	1653742.93
9	1	353177.60	1650850.99
9	2	353164.17	1651000.39
9	3	353150.75	1651149.79
9	4	353137.33	1651299.19
9	5	353123.90	1651448.58
9	6	353110.48	1651597.98
9	7	353097.06	1651747.38
9	8	353083.63	1651896.78
9	9	353070.21	1652046.18
9	10	353056.78	1652195.57
9	11	353043.36	1652344.97
9	12	353029.94	1652494.37
9	13	353016.51	1652643.77
9	14	353003.09	1652793.17
9	15	352989.67	1652942.57
9	16	352976.24	1653091.96
9	17	352962.82	1653241.36
9	18	352949.39	1653390.76

Transect	Station	Latitude	Longitude
10	1	350928.74	1656597.01
10	2	351078.19	1656584.18
10	3	351227.64	1656571.35
10	4	351377.09	1656558.52
10	5	351526.52	1656545.45
10	6	351675.95	1656532.39
10	7	351825.38	1656519.32
10	8	351974.81	1656506.25
10	9	352124.24	1656493.19
10	10	352273.67	1656480.12
10	11	352423.10	1656467.05
10	12	352572.53	1656453.98
10	13	352721.96	1656440.92
10	14	352871.38	1656427.85
10	15	353020.81	1656414.78
10	16	353170.24	1656401.72
10	17	353319.67	1656388.65
10	18	353469.10	1656375.58
11	1	353452.77	1663336.82
11	2	353320.05	1663398.28
11	3	353210.41	1663431.53
11	4	353150.44	1663475.13
11	5	353082.78	1663531.79
11	6	352954.55	1663600.14
11	7	352863.57	1663671.57
11	8	352750.76	1663742.48
11	9	352674.14	1663846.96
12	1	353122.99	1662596.49
12	2	353078.71	1662466.68
12	3	353007.30	1662332.00
12	4	353006.92	1662176.79
12	5	352938.08	1662044.09
12	6	352949.16	1661885.87
12	7	353025.32	1661739.87
12	8	353026.24	1661586.31
12	9	352988.91	1661442.87
13	1	355905.97	1656624.23
13	2	355905.97	1656461.84
13	3	355909.15	1656312.18
13	4	355905.97	1656162.53
13	5	355905.97	1656012.87
13	6	355905.97	1655866.40
13	7	355909.15	1655708.78
13	8	355909.15	1655559.12
13	9	355909.15	1655409.47
13	10	355909.15	1655262.99
13	11	355909.15	1655110.15
13	12	355912.34	1654960.50
13	13	355909.15	1654804.79
13	14	355915.52	1654664.69
14	1	355909.15	1653461.06

Transect	Station	Latitude	Longitude
14	2	355756.31	1653314.59
14	3	355606.66	1653158.57
14	4	355457.00	1653012.09
14	5	355310.53	1652862.44

Attachment 2. UTM coordinates for the point-transect or variable circular plot survey transects on the island of Aguiguan. All coordinates are in WGS84 UTM Zone 5 North. Transects 1 through 4 were established by Engbring *et al.* (1986) in 1982 and transect 5 was established in 2008.

Transect	Station	Latitude	Longitude
1	1	342821.64	1642070.78
1	2	342908.96	1641962.30
1	3	342940.71	1641827.36
1	4	342969.81	1641668.61
1	5	343067.71	1641549.55
1	6	343205.29	1641626.28
1	7	343369.33	1641631.57
1	8	343512.21	1641599.82
1	9	343655.08	1641631.57
1	10	343790.02	1641684.49
1	11	343922.31	1641716.24
1	12	344078.42	1641750.63
1	13	343914.38	1641811.49
1	14	343779.44	1641890.86
1	15	343803.25	1642009.93
1	16	343951.42	1642070.78
2	1	342832.23	1642210.31
2	2	342948.64	1642305.56
2	3	343062.42	1642408.74
2	4	343173.54	1642511.93
2	5	343292.60	1642609.83
2	6	343416.96	1642675.97
2	7	343570.42	1642707.72
2	8	343721.23	1642747.41
2	9	343872.04	1642779.16
2	10	344022.86	1642824.14
2	11	344152.50	1642898.22
2	12	344287.70	1642974.95
2	13	344409.41	1643049.04
2	14	344552.29	1643115.18
2	15	344684.58	1643194.56
2	16	344819.78	1643249.86
3	1	345028.80	1642109.50
3	2	345187.55	1642104.21
3	3	345338.37	1642093.62
3	4	345481.24	1642090.98
3	5	345639.99	1642072.46
3	6	345782.87	1642075.10
3	7	345909.87	1642154.48
3	8	346031.58	1642244.44
3	9	346155.93	1642326.46
3	10	346282.93	1642405.83
3	11	346401.46	1642490.79
3	12	346531.11	1642578.10
3	13	346644.88	1642697.16

Transect	Station	Latitude	Longitude
3	14	346684.57	1642834.75
3	15	346711.03	1642980.27
3	16	346748.07	1643128.43
3	17	346800.99	1643276.60
3	18	346822.15	1643414.18
4	1	344099.58	1642623.12
4	2	344216.00	1642710.43
4	3	344343.00	1642803.03
4	4	344472.65	1642898.28
4	5	344594.36	1642980.30
4	6	344713.42	1643067.62
4	7	344853.65	1643125.82
4	8	344996.52	1643197.26
4	9	345126.17	1643255.47
4	10	345271.69	1643318.97
4	11	345403.98	1643371.89
4	12	345544.21	1643440.68
4	13	345681.80	1643501.53
4	14	345816.20	1643567.70
4	15	345953.79	1643628.56
4	16	346094.02	1643689.41
5	1	344422.01	1642262.89
5	2	344562.43	1642335.11
5	3	344718.89	1642411.33
5	4	344855.29	1642487.55
5	5	344983.66	1642551.74
5	6	345148.15	1642623.95
5	7	345288.56	1642704.19
5	8	345449.03	1642780.41
5	9	345609.50	1642856.64
5	10	345765.96	1642936.87
5	11	345910.39	1643001.06
5	12	346074.87	1643081.30
5	13	346235.34	1643165.54
5	14	346375.75	1643241.77

2.4.2 SEABIRD SURVEYS


Sooty Terns. Photo by Curt Kessler.

Prepared by: **Curt Kessler, U.S. Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office, Honolulu, HI**

INTRODUCTION

Fifteen seabird species have been recorded nesting in the Mariana archipelago with at least one, and more commonly a suite of species, nesting on each island (Reichel 1991). This guild of species is an important segment of the sea-land-sea nutrient cycle. Disruption of “safe haven” nesting sites could have significant impacts on this group of important birds.

Seabird surveys are systematically conducted on Farallon de Medinilla (FDM) by the U.S. Navy, and on the island of Rota at the Sagua’ gaga (I Chenchon) Seabird Sanctuary by the Commonwealth of the Northern Mariana Islands (CNMI) – Division of Fish & Wildlife (DFW). Based on a review of 10 years of monthly surveys on FDM (Vogt 2005), the month of October has the lowest numbers of nesting seabirds on average. Therefore, seabirds found nesting at this time of year are expected to be the minimum number nesting

In 2008, the United States Marine Corp (USMC) contracted the U.S. Fish and Wildlife Service - Pacific Islands Fish and Wildlife Office (USFWS-PIFWO), to conduct terrestrial and marine surveys on Tinian and Aguiguan. The following report outlines seabird surveys that were conducted on the islands of Tinian, Aguiguan, and Naftan

Rock, CNMI during the 2008 survey. These surveys should be considered as a way to delineate those areas that have high concentrations of seabirds and where common colony nesting species occur. Species surveyed for include; brown booby (*Sula leucogaster*), red-footed bobby (*Sula sula*), masked booby (*Sula dactylatra*), brown noddy (*Anous stolidus*), black noddy (*Anous minutus*), sooty tern (*Onychoprion fuscatus,* previously *Sterna fuscata*), white tern (*Gygis alba*), and wedge tailed shearwater (*Puffinus pacificus*). Additionally, tattlers (*Heteroscelus sp*), white-tailed tropicbirds (*Phaethon lepturus*), red-tailed tropicbirds (*P. rubricauda*), and reef heron (*Egretta sacra*) were noted. The white tern was also recorded in the June 2008 island-wide point-transect or variable circular plot surveys on Tinian and Aguiguan. The seabird related results from those surveys are reported here and in Section 2.4.1 of this report. Refer to that section and Appendix 3.1 (Camp *et al*. 2009) for a detailed explanation of the survey methods.

METHODS

Shoreline, helicopter, and ground surveys were conducted in October 2008. A point-transect or variable circular plot survey was conducted to survey all bird species in June 2008 and results from previous point-transect surveys were reanalyzed to assess population trends. A description of each survey method and where they were utilized is outline below.

Shoreline and Helicopter Surveys

A shoreline survey was conducted along Navy leased lands on Tinian from approximately Barcinas Bay (14 59’26.38”N 145 36’10.29”E) to the eastside point at Puntan Masalok (15 1’10.66”N 45 39’53.02E) (Figure 1). Surveys were also conducted around the island of Aguiguan and Naftan Rock (Figure 2). The Tinian survey spanned two days (Oct. 10, 2008, 1700-1800 hrs; Oct 11 0800-1000 hrs, 1330-1430 hrs, Observers; C. Kessler –USFWS, and J. Omar – CNMI-DFW). The Naftan Rock survey took place on October 14, 2008 from 1500 hrs to 2130 hrs and was conducted by C. Kessler and E. Masga (Tinian DFW). Shoreline surveys took place opportunistically on Aguiguan between June and August 2008 and were conducted by C. Kessler, J. Omar, and E. Masga. All observations were conducted from a 17’boat (RIB-Apex brand) with the aide of 8 or 10 power binoculars. The boat cruised at a constant rate of approximately 8 miles per hour and stayed between 20 and 75 meters offshore as conditions permitted. Locations of all species sighted were either recorded using a Garmin 76CSx Global Positioning System unit (GPS) or marked on a map. Double counting was kept at a minimum by noting the direction of individual birds as they flew. At the right distance most birds flushed in the opposite direction of observer course. Limestone cliffs on the west side of Tinian were especially searched for black noddy nesting areas and roosts.

A helicopter survey of southwest coast of Aguiguan was conducted to map brown booby nesting areas. Nests were observed with the naked eye at 50-100 m distance. Nests were easily identified by observing ‘sitting’ birds in combination with bare dirt/cleared areas created by the nesting birds. This survey was conducted on October 13, 2008 at 1200 hrs and consisted of Observer C. Kessler and Pilot N. Kogure of Americopters, Inc.

Ground Surveys

A ground survey was conducted to determine density of nesting sooty tern pairs on Naftan Rock. Naftan Rock, for survey purposes, was delineated into three sections; north rock, south rock, and a small valley between (Figure 3). The area of the north and south rock (894 m^2 and 304 m^2, respectively) occupied by sooty terns was calculated by outlining the colony and determining the area with the aide of aerial photographs and Google Earth. The small valley between the north and south rock did not appear to have any nests at this time of the year and was not included in the acreage calculations. Both north and south areas were walked and the average distance between sooty tern eggs recorded in a field notebook and on digital film. These measurements were then used as the radius of a circle to calculate square meters occupied by one nest. This result was then divided into the total area to determine the number of eggs/nests for each area. It should be noted that the measurements used between nests was an average and actual spacing decreased toward the center of the nesting area and increased as one moved away from the center and approached the edges.

In addition, night vision goggles (NVG) (3rd generation ATN corp. model NVM14-3A) were utilized to record observations of wedge-tailed shearwaters using the grasslands on the north half of Naftan Rock. These observations recorded by C. Kessler on October 14, 2008 between 1900 - 2100 hrs on Naftan Rock.

Point Transect Surveys

Point-transect surveys were conducted on Tinian in 1982 (27 April – 8 May), 1996 (28 August – 1 September), and 2008 (14 – 19 June) on a total of 216 stations along 10 transects (Figure 1). All transects were at least 300 meters apart and all stations along each transect were 150 meters apart. An additional 4 transects were sampled during the 2008 survey for a total of 254 stations. On Aguiguan, 66 stations along 4 transects were sampled on June 2 and 3, 1982, and June 25-27, 2008 (Figure 2). An additional transect of 14 stations was also sampled during the 2008 survey for a total of 80 stations. This additional transect was at least 300 meters away from the nearest transect and all stations along all transects were 150 meters apart. All surveys followed standard point-transect methods, consisting of 8-minute counts where horizontal distances to all birds heard and/or seen were measured and recorded (see Engbring *et al*. 1986 for details). Population status was calculated as densities (birds/km^2) and number of birds (density by habitat type multiplied by habitat type area). Densities were calculated using the program DISTANCE (Thomas *et al*. 2006). Please refer to Section 2.4.1 of this report for a detailed explanation of the methods for the point-transect surveys.

RESULTS

Shoreline and Helicopter Surveys

A total of 36 tattlers, 11 reef herons, 28 black noddies, and 15 white terns were recorded during shoreline surveys of Navy lands on Tinian. In addition, a large colony of white terns numbering 30 plus was observed at 15 2’ 23.50”N 145 35’ 42.91”E roosting in old growth *Barringtonia asiatica* trees just below the cliff line.

No black noddy nesting areas were observed on Tinian during the survey although small groups were noted to be roosting at the north end of Barcina's Bay on the limestone cliffs that overhang the water. The coastline along the west side of the Tinian consists of flat coralline shelves along the water with large boulders in the bays along the shore. This side is protected from the prevailing winds and hosted most of the birds observed. The east side of Tinian has jagged limestone karst and rough seas due to the prevailing winds, and had significantly fewer birds. It is possible that at different seasons this trend could be reversed.

A south side helicopter survey of the cliff edge on Aguiguan recorded 44 brown booby nests (Figure 4). Nests were situated along the edges of cliffs on level ground. Five nests were also observed on large limestone boulders that had broken away from the cliff and now rested along the shore. The helicopter survey also recorded approximately 10 red-footed boobies nesting in the trees at 14 50' 41.48"N 145 33' 33.19"E (Figure 4).


Red footed bobby (Sula sula). Photo S. Vogt

A family group of 4-6 red-tailed tropicbirds were observed in the area of the boat landing on the north side of Aguiguan (Figure 4). In addition, tropicbirds were observed using the caves on the inland cliff face on the south side. Brown boobys were observed on the cliff face on the north east side (Figure 4; 14 51 50.41"N 145 34' 0.34"E). It is estimated that 10 pairs were nesting in this area; however this area was not visited by helicopter.

Black noddies were numerous and inhabited all large sea caves on the north and south west sides of Aguiguan (Figure 4). One of the caves on the north side, called black noddy cave, is known to be used by black noddies for nesting. A rough estimate of black noddies on the island for the month of October would be 400-500 individuals.

In addition to the shoreline and helicopter surveys, large mixed flocks of hundreds of seabirds consisting primarily of shearwaters, noddies, and white terns were observed offshore from Tinian and Aguiguan feeding with schools of tuna. Observations were from a boat in transit between the islands during the June to August survey period. Certain areas consistently had these concentrations of fish and seabirds and are mapped in Figure 5. These feeding areas should be viewed as a significant resource to seabirds and important to local fisherman.

Ground Surveys
Sooty terns were the most numerous nesting birds on Naftan Rock on October 14, 2008. They occupied the entire flat area of the South Rock and about one half of the flat area on the North Rock. Their nesting areas can be clearly seen on aerial photos due to the altered state of the vegetation in the areas where they nest (Figure 3). Sooty tern eggs were on average 0.3 meters apart on the North Rock and 0.46 meters apart on the South Rock. The total estimated number of eggs on Naftan Rock was therefore 3,647 eggs; 454 and 3,193 eggs on the south and north rock, respectively. This means there were approximately 7,294 individual adult sooty terns nesting on this small islet.

Brown boobies were not found to be nesting on Naftan Rock, but 15-25 individuals were observed roosting. Masked boobies were not observed during this survey, however four individuals were recorded on the islet in August 2008, and two individuals were observed in May 2007.

Brown noddies were nesting (feathered nestlings were observed) on the island and are considered to be year round nesters in the CNMI. This species was not as plentiful as observed in May 2007, probably due to seasonal fluctuations. Brown noddies are ground nesters, and were observed nesting around the periphery of the sooty terns along the cliff face and steeper parts of Naftan Rock. For this survey it was estimated 200-300 pairs were nesting on the island. Due to the steep nature of their nesting sites, this species was not rigorously surveyed for and the pair estimate given is only a rough estimate.

Wedge tailed Shearwaters use the grassy area on the North Rock and middle valley for nesting. During this survey, approximately 5-10 shearwaters were observed at night coming in off the ocean and landing in the grass, however no nests were found. However, in May 2007, more than five nests with eggs were observed in burrows in the grassy area and one shearwater was observed under a boulder in the middle valley.

Wedge-tailed shearwater (*Puffinus pacificus*). Photo C. Kessler

Ruddy Turnstone (*Arenaria interpres*) were present and approximately twenty individuals of this migratory species were found to be roosting and feeding on Naftan Rock.

Point Transect Surveys

Three seabird species were detected during the point-transect survey on Tinian (Table 1). However, only sufficient numbers of white terns were detected to calculate density and abundance estimates. The white tern population was estimated to be approximately 15,000 individuals (Table 2) and no significant difference in density was found between the 1982 and 2008 estimates (Repeated Measures ANOVA, $F_{2,398} = 43.18$, $p < 0.001$; Least Square Means, $t = -0.91$, $p = 0.634$). However, there was a significant difference between the 1996 and both the 1982 (Repeated Measures ANOVA, $F_{2,398} = 43.18$, $p < 0.001$; Least Square Means, $t = 7.55$, $p < 0.001$) and 2008 (Repeated Measures ANOVA, $F_{2,398} = 43.18$, $p < 0.001$; Least Square Means, $t = -8.46$, $p < 0.001$) estimates. No significant difference in white tern densities among regions by year was detected (Repeated Measures ANOVA, $F_{6,392} = 1.71$, $p = 0.116$). However, a significant difference in regions was detected (Repeated Measures ANOVA, $F_{3,196} = 4.15$, $p = 0.007$) and the Hagoi region had fewer white terns than the Carolinas, Diablo, and Masalog regions (Least Square Means, $p \leq 0.05$). Please refer to Section 2.4.1 of this report for a detailed explanation of the results for the VCP survey.

Three seabird species were detected during the point-transect survey on Aguiguan (Table 1). However, only sufficient numbers of white terns were detected to calculate density and abundance estimates (Table 2). The white tern population was estimated to be approximately 1,200 individuals and was not significantly different from the 1982 estimate (z value = -0.95, $p = 0.341$). Please refer to Section 2.4.1 of this report for a detailed explanation of the results for the VCP survey.

DISCUSSION

The 44 brown booby nests observed on Aguiguan in October represent a minimum number of nesting pairs on the island. In Nov 2006 between 50 and 100 nests were observed but not fully surveyed (C. Kessler, pers. obs.). In 1984, 250-300 brown boobies were recorded to be nesting along the south cliff line during the summer (DFW 1985). More observations are required to fully understand the extent of nesting in this area.

Brown Bobbies (*Sula leucogaster*) nesting on the southside of Aguiguan Island, CNMI. Nov. 2006. Photo C. Kessler

The estimated density of sooty terns on Naftan Rock (2.7 nests/m^2) falls within the range of densities reported for five other sites across the Pacific (1.3 - 4.5 nests/m^2; Schreiber *et. al.* 2002). The estimate of almost 7,300 adult birds is slightly higher than 6,000 reported in 1984 (Table 3). However in 1984 there are reports of large numbers of eggs being taken by local hunters from Naftan Rock (DFW 1985; E. Masga, pers. comm.). This practice is believed to have slowed down or stopped in recent years and could be one factor in the population increase. Another potential factor is that sooty tern nesting densities appear to be dependent on ground cover and substrate (Schreiber *et. al.* 2002). This was observed on Naftan Rock and is reflected in the densities recorded for the separate halves of the island. The terrain on the north rock was more even than on the south rock and the ground cover was also less.


Sooty Terns (*Onychoprion fuscatus*) nesting on Naftan Rock, Mariana Islands. Oct 2008. Photo by C. Kessler.

Sooty tern nests were primarily at the egg stage although a few downy chicks were recorded and one egg was observed to hatch. The observation of sooty terns breeding in October appears unusual although breeding records are not complete. Previous breeding records from the CNMI indicate this species breeds from January to September (Table 3). However monthly surveys of FDM show concentrations of sooty terns on FDM for all month except August. The FDM surveys also reveal that they are not present on the island every year and vary widely between years (Figure 3; S. Vogt, pers. com. 2008). Sooty terns are known to alter their nesting dates by region and weather patterns (Scheiber *et. al*. 2002). For the period June – August 2008 it appears that some environmental factor had changed as evidenced by noticeable die off's of brown and red-footed boobies on Aguiguan (Attachment 1), wedge-tailed shearwaters on Managaha (S. Kremer, pers. comm.) and anecdotal accounts of unusual mackerel species being caught. More research is needed to explore the relationship between weather/ocean patterns, fisheries, and the nesting of sooty terns.

Wedge tailed shearwaters were documented to nest on Naftan Rock in June (2 nests) and August (4 nests) 1984 (DFW 1985) and were observed nesting under large boulders in the central valley area that is primarily covered with the ground hugging seaside succulent *Sesuvium portulacastrum*. In May 2007, one shearwater was observed roosting under a boulder in the central area but nests with eggs were located upon the north rock in the level area adjacent to the sooty terns. This area is covered by thick grass and burrows were formed under the matted grass using the grass as tunnels. No occupied burrows were recorded, but adults were observed landing in the grassy area after dark. October is outside the reported nesting time of May-July. In January 1985 no active burrows were reported, but chewed bones lead the biologists to suspect that *Rattus exulans* might be on the island. No mammals were observed on the October 2008 survey, but a medium size coconut crab (*Birgus latro*; thoracic length = 31 mm) was recorded in the grassy area used by the shearwaters. This large land crab should be considered a predator on seabirds.

Brown noddies are ground nesters and are thought to nest year round (Chardine and Morris 1996). Nesting has been recorded for the Kastiyo area of Tinian and 340 nesting pairs were recorded on Aguiguan (DFW 1988). Naftan rock was reported to have 500 in June 1984 and 2000 in July 1983 (DFW 1988). The low estimate of 200 - 300 nesting pairs recorded during the 2008 survey reflects that these surveys were conducted outside the peak of the breeding season. Brown noddies occupied the steeper parts of the island in October but were distributed throughout the sooty tern core areas in May 2007.

The Tinian shoreline survey was intended to document black noddy cliff line nesting sites. None were recorded which mimics with similar survey in July and August of 1984 (CNMI 1985). However, black noddies are known to nest in Black noddy cave (hence the name) on Aguiguan with 120-130 individuals in July 1983 and 20-30 active nests in February 1984 (CNMI 1988). During this survey it can only be reported that black noddys occupied that cave and others along the coast. This species was also reported nesting in the Masalok area of Tinian in 1986 (DFW 1988).


White-tailed tropicbird (Phaethon lepturus). Photo S. Vogt.

Both white tailed and red tailed tropic birds are known to nest in small numbers on Aguiguan and Tinian. These species use inaccessible cliff ledges and are consistently present in small numbers about Aguiguan throughout the year. Nesting activity was observed along the north cliff face in the vicinity of the boat landing and along the southeast inland cliff face (Figure 4).

Consistently high densities of white terns were recorded during the 1982 and 2008 surveys on Aguiguan and Tinian indicating that populations appear stable. The 1996 white tern estimate on Tinian was lower than the estimates from 1982 and 2008. It is

White tern (Gygis alba). Photo S. Vogt

likely that the low tern estimate was an artifact of when the survey was conducted and not an actual change in the tern population. The original survey in 1982 and the most recent 2008 survey occurred early in the year and early in the breeding season (although terns can breed in all months of the year; Niethammer and Patrick-Castilaw 1998), whereas the 1996 survey was conducted in late August and after the peak breeding season. When not nesting, most individuals spend extended periods at sea (Niethammer and Patrick-Castilaw 1998); therefore portions of the population in 1996 were outside the sampling frame.

The results of this survey and previously reported observations that October typically has the lowest seabird breeding activity in the Mariana Islands (Vogt 2005) indicates that breeding by seabirds occurs throughout the year and that some species are always nesting regardless of month. To fully understand and manage for seabirds a standardized monthly census needs to be conducted, archipelago-wide, over a number of years. Congruently, a banding study should also be conducted to better understand seabird patterns and interactions with local and Pacific-wide fisheries. The association between seabird and fish populations in the region needs study.

REFERENCES

Camp, R.J., T. K. Pratt, F. Amidon, A. P. Marshall, S. Kremer, and M. Laut. 2009. Draft status and trends of the land bird avifauna on Tinian and Aguiguan, Mariana Islands. Appendix 3.1 *in* Terrestrial Resource Surveys of Tinian and Aguiguan, Mariana Islands, 2008. Working Draft. U.S. Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office, Honolulu. HI.

.

Chardine, J.W. and R.D. Morris. 1996. Brown Noddy *Anous stolidus*); *In* The Birds of North America, No. 220 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

DFW. 1985. Yearly progress report for fiscal year 1984; for the Pittman-Robertson Federal Aid in Wildlife Restoration Program. Unpublished report to US Fish & Wildlife Service, Honolulu, HI.

DFW. 1988. Five year progress report for years 1983-1987; for the Pittman-Robertson Federal Aid in Wildlife Restoration Program. Unpublished report to US Fish & Wildlife Service, Honolulu, HI.

DFW. 1993. Five year progress report for years 1988-1992; for the Pittman-Robertson Federal Aid in Wildlife Restoration Program. Unpublished report to US Fish & Wildlife Service, Honolulu, HI.

Diamond, A. W. and Devlin, C. M. 2003. Seabirds as Indicators of Changes in Marine Ecosystems: Ecological Monitoring on Machias Seal Island. Environmental Monitoring and Assessment 88, 153175.

Engbring, J., Ramsey, F.L. and Wildman, V.J. 1986. Micronesian forest bird survey, 1982: Saipan, Tinian, Aguijan, and Rota. U.S. Fish and Wildlife Service Report. 143pp.

Global security website. 2008. http://www.globalsecurity.org/military/facility/farallon-de medinilla.htm.

Kessler, C.C. 2006. Anatahan feral ungulate eradication project for Micronesian megapode habitat improvement for the years 2002 to 2006. Unpublished USFWS Rerpot to the U.S. Navy, COMNAVMARIANAS, Guam.

Lusk, M.R., P. Bruner, and C. Kessler. 2000. The avifauna of Farallon de Medinilla, Mariana Islands. J. Field Ornithology, 71(1):22-33.

Niethammer, K.R. and Patrick-Castilaw, L.B. 1998. White Tern (*Gygis alba*). *In* The Birds of North America, No. 371 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Reichel, J.D. 1991. Status and Conservation of Seabirds in the Mariana Islands. ICBP Technical Publication No. 11. *In* CNMI-DFW 5 year progress report 1987 to 1997 to U.S. Fish & Wildlife Service, Honolulu, HI.

Schreiber, E.A., C.J. Feare, B.A. Harrington, B.G. Murray, Jr., W.B. Robertson, Jr., M.J. Robertson, and G.E. Woolfenden. 2002. Sooty Tern *(Sterna fuscata*); *In* The Birds of North America, No. 665 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Thomas, L., Laake, J.L., Strindberg, S., Marques, F.F.C., Buckland, S.T., Borchers, D.L., Anderson, D.R., Burnham, K.P., Hedley, S.L., Pollard, J.H., Bishop, J.R.B. and Marques, T.A. 2006. Distance 5.0. Release 2. Research Unit for Wildlife Population Assessment, University of St. Andrews, U.K. http://www.ruwpa.st-and.ac.uk/distance/.

Vogt, S. 2005. Wildlife surveys on Military leased lands, Farallon de Medinilla, CNMI. Unpublished Navy report for Fiscal Years 1999-2003, Five year report for 61755NR410. Pearl Harbor, HI.

Table 1. List of seabirds detected from three different point-transect surveys on Tinian and Aguiguan. In 1982 and 1996, 216 stations were sampled on 10 transects in Tinian, and in 2008 254 stations were sampled on 14 transects in Tinian. In 1982, 66 stations were sampled on 4 transects (88 counts; several stations were counted more than once), and in 2008, 80 stations were sampled in 5 transects on Aguiguan. The number of birds detected (# Dect), and indices of percent occurrence (% Occ) and birds per station (BPS), were calculated.

				1982			1996			2008	
Species	Island	Year	# Dect	% Occ	BPS	# Dect	% Occ	BPS	# Dect	% Occ	BPS
White-tailed Tropicbird	Tinian	1982	0	0	0	0	0	0	5	1.2	0.02
(*Phaethon lepturus*)	Aguiguan	1982	1	1.1	0.01	-	-	-	0	0	0
Red-tailed Tropicbird	Tinian	1982	0	0	0	0	0	0	0	0	0
(*Phaethon rubricauda*)	Aguiguan	1982	13	9.1	0.15	-	-	-	0	0	0
Great Frigatebird	Tinian	1982	0	0	0	0	0	0	0	0	0
(*Fregata minor*)	Aguiguan	1982	2	1.1	0.02	-	-	-	0	0	0
Brown Noddy	Tinian	1982	0	0	0	0	0	0	1	0.4	<0.01
(*Anous stolidus*)	Aguiguan	1982	20	15.9	0.23	-	-	-	0	0	0
Black Noddy	Tinian	1982	0	0	0	0	0	0	0	0	0
(*Anous minutus*)	Aguiguan	1982	75	35.2	0.85	-	-	-	1	1.2	0.01
White Tern	Tinian	1982	344	59.3	1.59	52	10.2	0.24	322	48.0	1.27
(*Gygis alba*)	Aguiguan	1982	218	61.4	2.48	-	-	-	84	42.5	1.05
Sooty Tern	Tinian	1982	0	0	0	0	0	0	0	0	0
(*Sterna fuscata*)	Aguiguan	1982	1	1.1	0.01	-	-	-	0	0	0

Table 2. Population density and abundance estimates for white terns on Tinian and Aguiguan from point-transect surveys. First row: mean density (birds/km^2 ± SE, with 95% CI). Second row: 2008 bird abundance (density by habitat times the habitat area) with 95% CI. Agriculture habitat type was dropped on Tinian for calculating bird abundance due to small sample size; only 2 survey stations were sampled on Tinian.

Island	1982	1996	2008
Tinian	144.1 ± 17.24 (113.9–182.2)	25.3 ± 7.01 (14.8–43.2)	169.9 ± 19.66 (135.4–213.2)
	13,980 (9.349–21,512)	2,846 (1,121–7,300)	15,147 (10,067–23,041)
Aguiguan	169.6 ± 27.0 (124.2–231.6)	-	218.8 ± 44.2 (147.3–325.1)
	-	-	1,214 (604–3,651)

Table 3. Sooty tern (*Onychoprion fuscatus*) numbers recorded in the Mariana Islands by month and year. Sooty tern population estimates for Farallon de Medinilla are not available.

Location	Month	Year	Population	Citation
Naftan Rock	April	1982	4,500	Engbring et al 1986
Naftan Rock	June	1984	6,000	Pratt 1984
Naftan Rock	August	1984	6,000	Pratt 1984
Naftan Rock	January	1985	1,500	Lemke and Pratt 1985
Naftan Rock	February	1987	Several thousand	Reichel 1987
Naftan Rock	January	1987	0	Reichel 1987
Naftan Rock	November	2008	7,300	This survey
Guguan	July-August	1979	20,000	Clapp pers comm./ CNMI 1988
Guguan	May-June	1983	28,000	Lemke 1983a
Guguan	September	1986	25,900	Glass and Villagomez 1986
Guguan	May-June	1987	35,000	Reichel 1988
Guguan	May	1992	25,000-30,000	Rice and Stinson 1992
Asuncion	June	1992	> 2,500	Rice and Stinson 1992
Uracas	August	1979	10,000-20,000	Clapp pers comm./ CNMI 1993
Uracas	August	1983	4,000-6,000	Lemke 1983
Uracas	February	1984	7,000	Pratt and Lemke 1984b
Uracas	June	1987	250,000	Reichel 1987/CNMI 1993
Uracas	June	1992	206,128	CNMI 1993

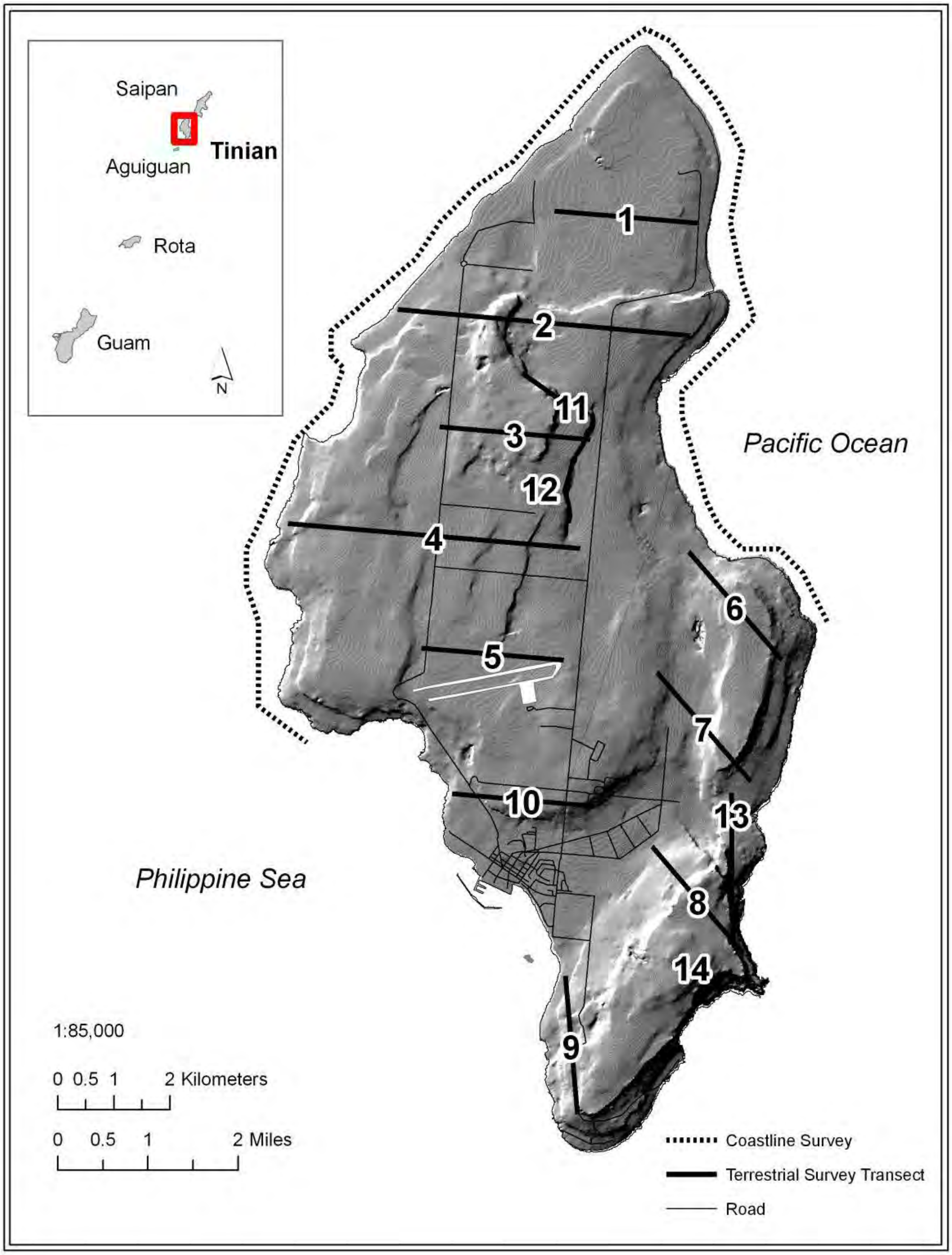


Figure 1. Island of Tinian showing the coastline survey route and terrestrial survey transects. Transects 1-10 were counted during all three surveys, and transects 11-14 were established and counted during the 2008 survey.

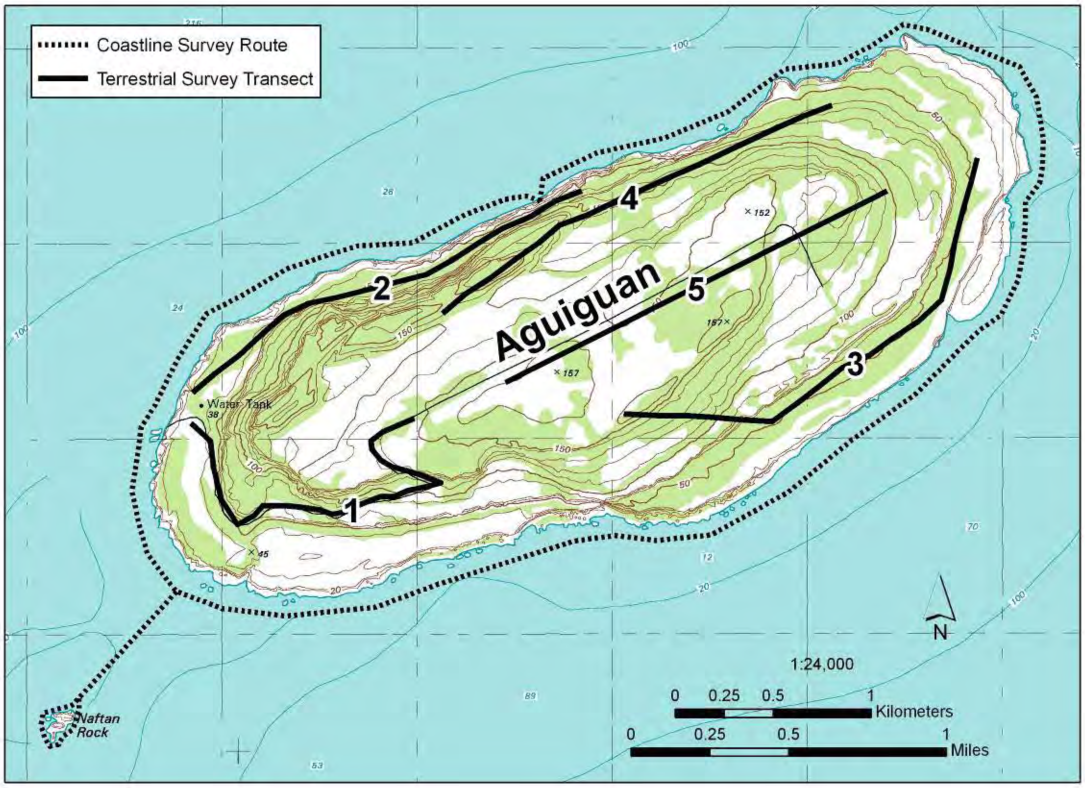
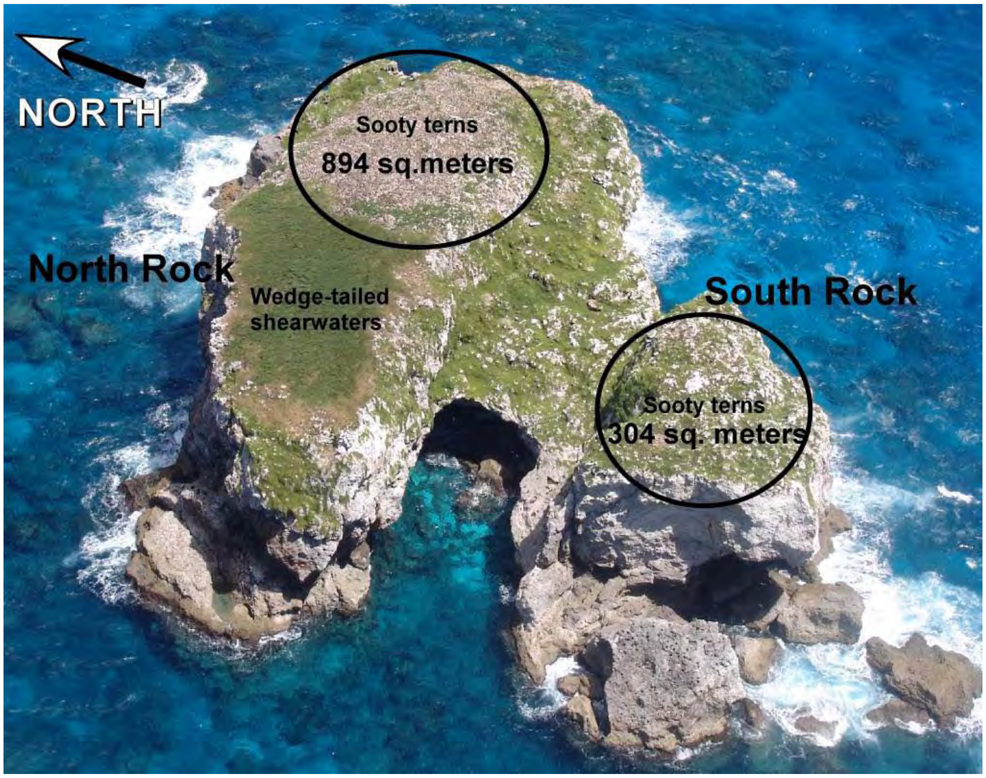


Figure 2. Island of Aguiguan and Naftan Rock showing the coastline survey route and terrestrial survey transects. Transects 1-4 were counted during both the 1982 and 2008 surveys, whereas transect 5 was established and counted during the 2008 survey.



Naftan Rock, Mariana Islands. Sept. 2004. Aerial view outlining areas calculated for Sooty Tern (*Onychoprion fuscatus*) density estimates.

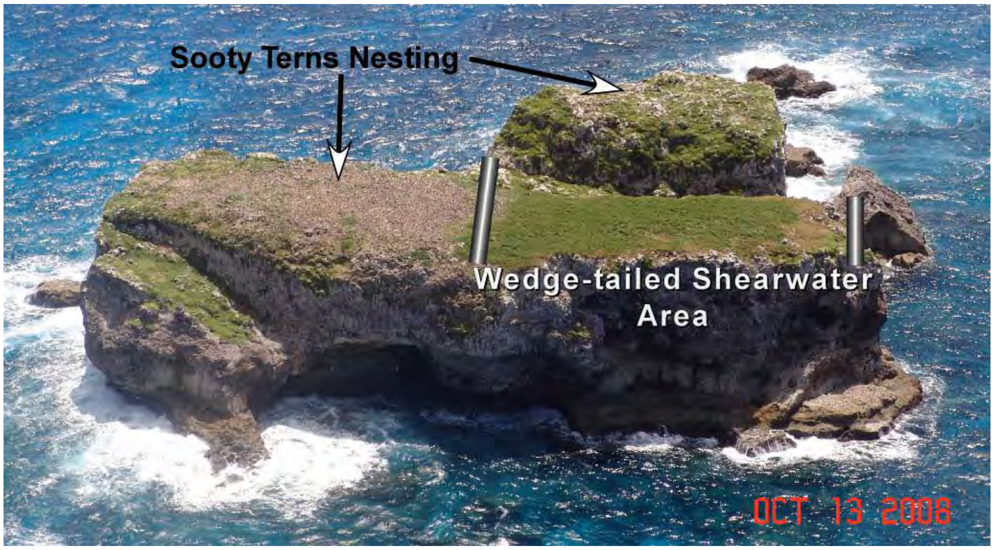


Figure 3. Naftan Rock, Mariana Islands. Oct. 2008. Areas of different species use are clearly discernable due to changes in vegetation

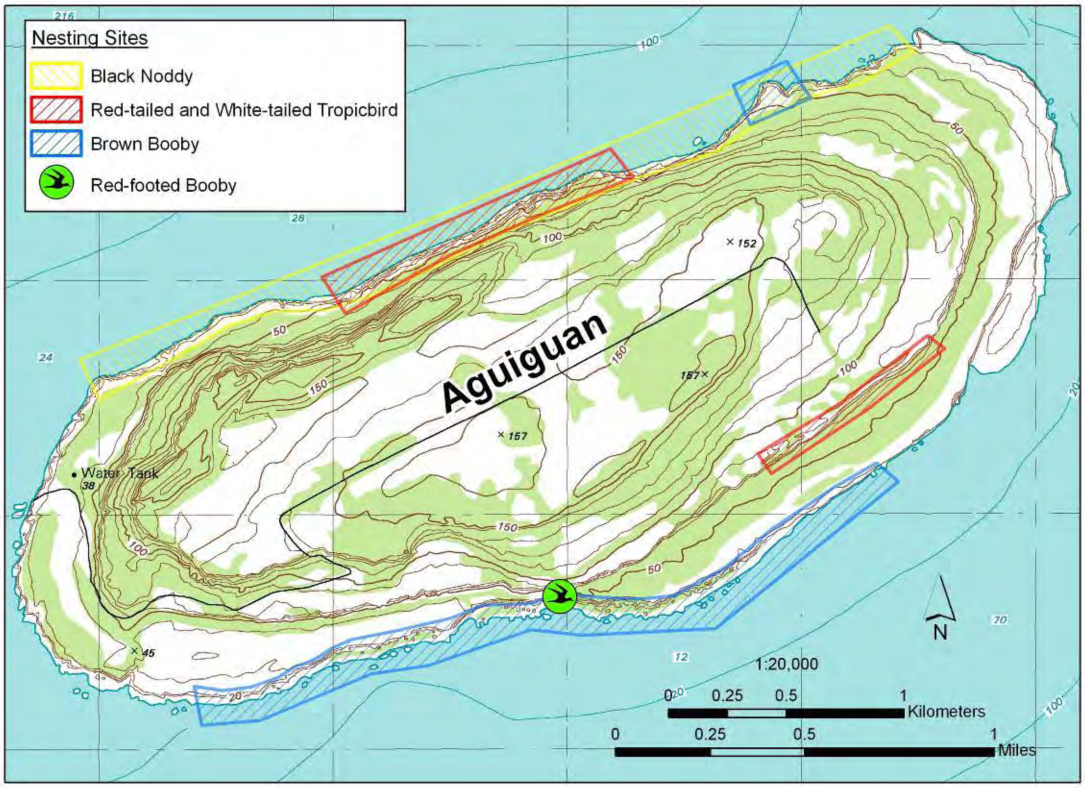


Figure 4. Location of seabird nesting sites observed on the island of Aguiguan during shoreline and helicopter surveys in 2008.

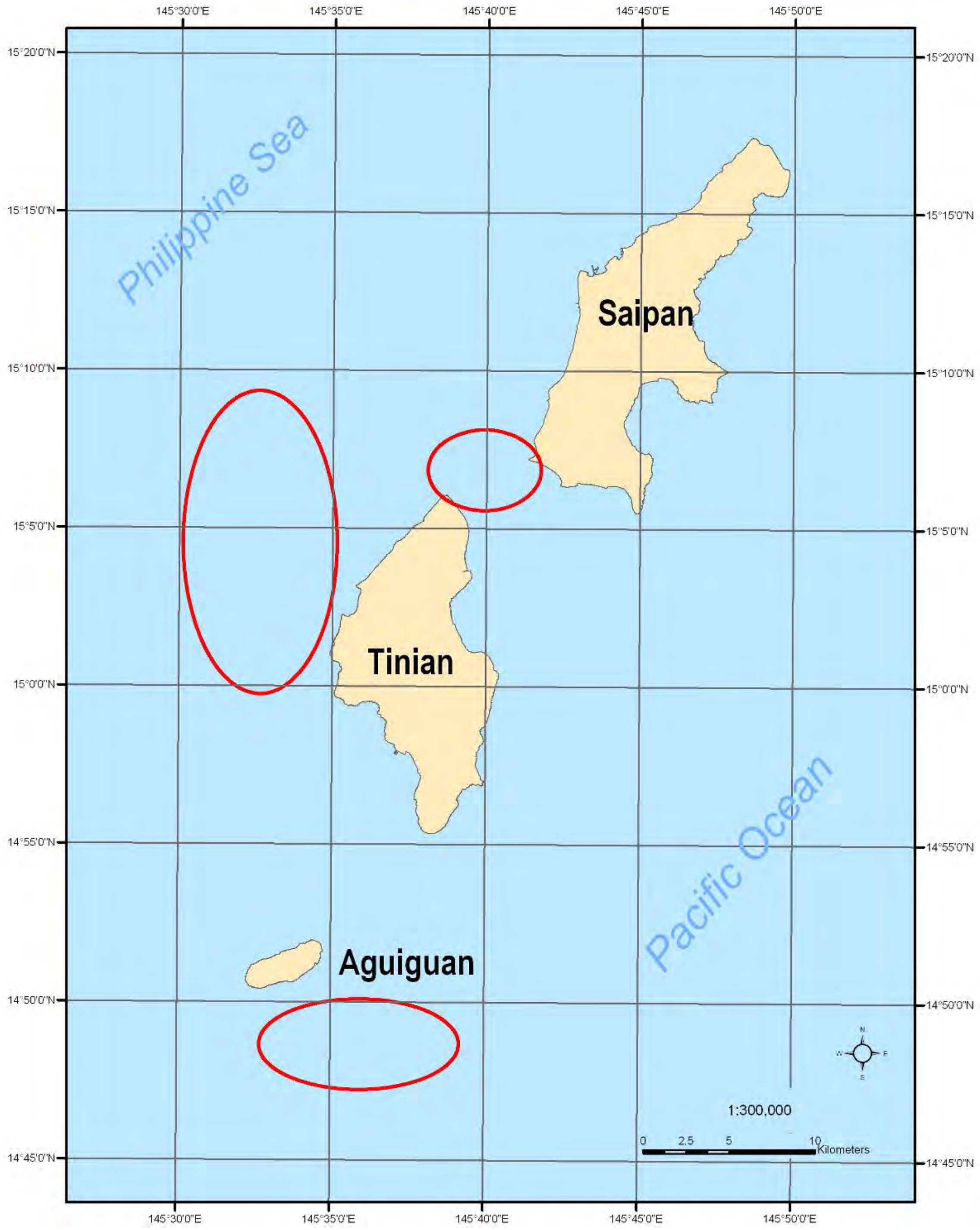


Figure 5. Areas of high seabird and tuna interaction near the islands of Aguiguan, Tinian, and Saipan in 2008.

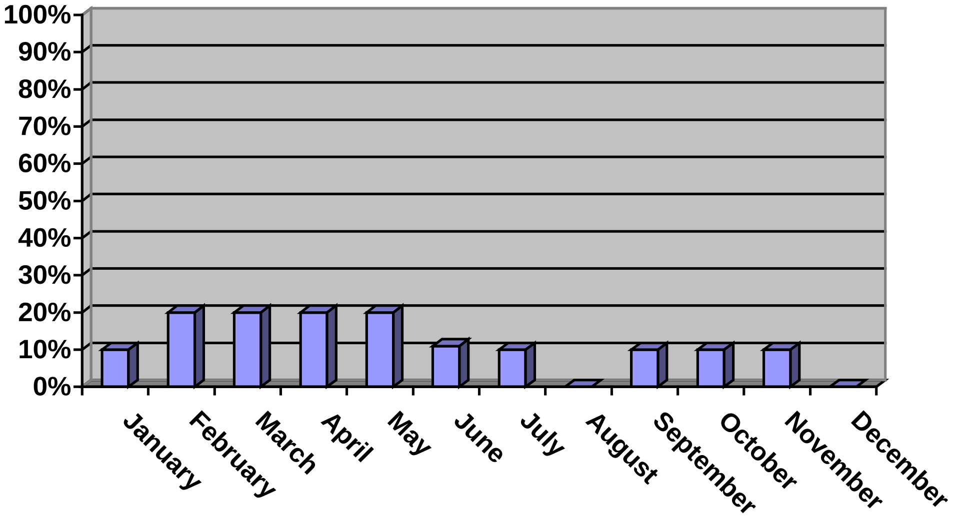


Figure 6. Percentage likelihood of sooty terns being present on Farallon de Medinilla based on monthly surveys from 1998 to 2007 (S. Vogt, pers. comm.). Surveys were not conducted in June 1999.

Attachment 1

Observers - Curt Kessler, Wildlife Biologist, USFWS, Honolulu, HI, Jess Omar, Conservation Officer, Saipan, CNMI, Joshua Fisher, Biologist, USFWS, Honolulu, HI.

Field note; Dead seabirds floating off Aguiguan (Goat Island), Commonwealth of the Northern Mariana Islands.

- Brown booby (*Sulu leucogaster*) – August 11-15, 2008 – Collected one dead adult female on August 11 and noted two others (both adult females) floating in the waters near Aguiguan Island, CNMI. Birds appeared to have been dead for two to three days judging from the smell. Another 3 sick and 8 dead brown boobies were observed by the boat crew during that week. Two or three sick brown boobies were observed on rocks along the Aguiguan coast on August 15. These birds appeared lethargic and unable to fly. Brown boobies do nest on Aguiguan Island (est. min. 50 pairs and could be over 100 pairs). Nearest islands with colonies: Rota – 50 miles, FDM – 85 miles.

- Red-footed booby (*Sula sula*) – August 26, 2008 - Observed 10-20 dead birds floating on the ocean on the west side of Aguiguan Island, CNMI. At least 10 birds were approached in order to identify and collect specimens. Most birds appeared to have been dead for at least three days based on smell. Only one bird was collected. Both adults and juveniles were observed dead. Red footed boobies do nest on Aguiguan Island in small numbers (est. 20 pairs). Nearest islands with colonies : Rota – 50 miles, FDM – 85 miles.

2.4.3 MICRONESIAN MEGAPODE ON TINIAN AND AGUIGUAN

Micronesian Megapode (*Megapodius laperouse*). Photo by Scott Vogt.

Prepared by: **Curt Kessler and Fred Amidon, U.S. Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office, Honolulu, HI**

INTRODUCTION

The Micronesian megapode (*Megapodius laperouse*) is a pigeon-sized bird in the family Megapodiidae, an old-world family restricted to the Australasian region and best known for its unusual reptile-like behavior of burying its eggs rather than incubating them as do all other birds. The Mariana Islands subspecies (*M. laperouse laperouse*), called sasangat in Chamorro and sasangal in Carolinian, was once found throughout the Mariana Islands but has since been extirpated or reduced in numbers particularly on limestone islands with human populations. The reasons for the disappearance from these islands are not entirely understood, but it is suspected that alien predators, loss of habitat, past egg-collecting, and over-hunting are factors (USFWS 1998). Currently, Micronesian megapodes are known or believed to occur on Aguiguan, Tinian, Saipan, Farallon de Medinilla (FDM), Sarigan, Guguan, Alamagan, Pagan, Agrihan, Asuncion, Maug, and possibly Uracus in the Mariana archipelago. They were extirpated from Guam and Rota around the turn of the century (USFWS 1998) and may have been extirpated from Anatahan due to volcanic activity in 2005. The U.S. Fish and Wildlife Service (USFWS) listed this species as endangered throughout its range in 1970 (USFWS 1970). Populations appear to be stable or possibly increasing in the unpopulated volcanic northern islands of the CNMI (Division of Fish and Wildlife 2000a-f, Martin *et al.* 2008, Vogt 2008). Currently a cooperative effort between the Commonwealth of the Northern

Mariana Islands – Division of Fish and Wildlife (DFW), Northern Islands Mayor's Office (NIMO), USFWS, and the U.S. Navy (USN) is underway to restore habitat in the northern islands and assess status for this species as outlined in the Micronesian Megapode Recovery Plan (USFWS 1998). For additional information on life history and recovery objectives, see the Micronesian Megapode Recovery Plan.

The original type specimen for *Megapodius laperouse* was collected on Tinian during the Uranie expedition in 1820. At the time, megapodes were reported as uncommon, and they seem to have declined steadily until 1945, when no megapodes were reported on the island (Baker 1951). Megapodes were observed on Tinian again in the late 1970's and have been observed periodically since then (Wiles *et al*. 1987, O'Daniel and Krueger 1999, Witteman 2001). No breeding activity has been observed on Tinian, and the birds are thought to have migrated from Saipan or Aguiguan; however, Tinian has not been thoroughly searched for nests (Kessler, pers. observation). Megapode observations on Tinian are usually associated with limestone forest and cliff-line habitat in the Maga and Mt. Laso areas on Navy leased lands. Therefore, it is believed that this is an important habitat for the species on the island.

Megapodes have been found in consistently low numbers on Aguiguan based on reports from the 1930's, 1950's, 1980's, and 1990's (Takatsukasa 1932-1938, Owen 1974, Engbring *et al.* 1986). In 1982, Engbring *et al.* (1986) estimated that a population of at least 11 megapodes existed on Aguiguan. In 2000, Cruz *et al.* (2000) estimated that there were 51 megapodes on Aguiguan, and Esselstyn *et al.* (2003) estimated a population of 72 (range 34-149) megapodes in 2002.

The breeding biology of megapodes in the southern limestone islands is still a mystery, although both Aguiguan and Saipan are assumed to have breeding populations. Megapodes utilize burrow-nesting at sun-exposed beaches, cinder fields, geothermal sites, and between the roots of trees (decompositional heat) and mound-building (decompostional heat) for incubating their eggs (Glass and Aldan 1988, Elliott 1994, Wiles and Conry 2001). However, sandy beach habitat on Tinian and Saipan are very limited and heavily used for recreation, and non-existent on Aguiguan. Also, cinder fields and geothermal sites are only available in the northern islands. Therefore, rotting trees and mound-building are the likely egg incubation sites on these islands. However, this has not been confirmed, and further research is needed to identify this important aspect of the species' biology to conserve and protect the species.

Effective methods for surveying Micronesian megapodes have also not been identified. Traditionally, point-transect or variable circular plot methodology has been used to survey for birds on most of the Mariana islands (*e.g.*, Engbring *et al.* 1986). However, this method is not as effective for secretive species, like the Micronesian megapode, or rare species. Therefore several survey methods were undertaken in 2008 in an effort to compare methods and identify a standard survey methodology.

METHODS

Point-Transect Surveys

We conducted island-wide point-transect or variable circular plot surveys on Tinian and Aguiguan between June 14 and 19 and June 25 and 27, 2008, respectively. On Tinian, we sampled a total of 254 stations on 14 transects (Figure 1). Ten of the transects and 216 of the stations were surveyed previously in 1982 (Engbring *et al*. 1986) and 1996 (USFWS, unpublished data). Four additional transects were included in the 2008 surveys to increase the survey coverage of limestone forest habitat on the island. On Aguiguan, we sampled 80 stations on 5 transects (Figure 2). Four of the transects were previously surveyed in 1982 (Engbring *et al*. 1986), 1992 (Craig *et al*. 1992), 1995 (USFWS unpublished data), 2000 (Cruz *et al*. 2000), and 2002 (Esselstyn *et al*. 2003). An additional transect of 14 stations was sampled during the 2008 survey for a total of 80 stations. This additional transect was added to sample secondary forest and open field habitats and increase the areal coverage of the island. All stations along all transects on both islands were 150 meters apart.

All surveys were conducted by one observer and followed standard point-transect methods, consisting of 8-minute counts and estimation of horizontal distances to all birds heard and/or seen (see Reynolds *et al.* 1980 or Engbring *et al*. 1986 for details). Sampling conditions recorded included cloud cover, rain, wind, noise level, and habitat type, and these were later used as covariates in density calculations (see 2.4.1 General Land Birds for additional information). Counts commenced at sunrise and continued until 1100 hours and were conducted only under favorable weather conditions.

The point-transect technique requires 75-100 detections to model the detection function for each species effectively (Buckland *et al*. 2001). If sufficient detections were recorded, densities were calculated using the program DISTANCE (Thomas *et al*. 2006). For additional information on point-transect sampling and data analysis see 2.4.1 General Land Birds.

Playback Surveys

Playback surveys were conducted for Micronesian megapodes on Tinian and Aguiguan between August 13 and 18 and August 20 and 22, 2008, respectively. A total of 21 stations along 3 transects were sampled on Tinian (Figure 3 Attachment 1). Two of the transects were previously sampled during the point-transect survey in June 2008 (transects 11 and 12; Figure 1 and an additional transect was established for this survey. All stations were 150 meters apart and all transects were at least 300 meters apart. In order to maximize the likelihood of detections, all transects were established in limestone forest in the Mount Lasu and Maga areas where megapodes had been previously recorded (USFWS 1998, Vogt 2008). On Aguiguan, the transects and stations used for the point-transect sampling were also sampled for Micronesian megapode playback surveys.

All stations were sampled by a single observer. During the survey, digitally recorded Micronesian megapode calls obtained on Sarigan were broadcast. Pair duet and alert calls were played on an electronic game caller (Foxpro FX5 ™) for one minute at each station. The observer then measured and recorded the horizontal distances of all Micronesian megapodes heard and/or seen during a four minute survey period (one minute of playbacks and three minutes of observation). Leupold 9x32 mm RXB-IV Range Finding Binoculars (Leupold, Beaverton, OR) were used to assist with distance estimation. However, not all distance estimates were derived from range-finder estimates. Weather and habitat conditions were recorded at each station. Counts commenced at sunrise and continued until completed (typically prior to 1100 hours) and were conducted only under favorable weather conditions.

Similar to the analysis of point-transect data, densities would be calculated from playback data using the program DISTANCE (Thomas *et al*. 2006) if sufficient detections (75 – 100) were recorded during the survey and if the movement in response to the playback calls could be accounted for during population estimation. Responsive movements were not available, therefore, estimates from point-transect methods were unreliable. Following methods outlined by Reynolds and Snetsinger (2001), we calculated the likelihood of detecting a small population of Micronesian megapodes in the Mount Lasu and Maga regions of Tinian regions (the areas where megapodes were last observed on Tinian) to determine survey effectiveness (see 2.4.4 Nightingale Reed-warbler for additional information on this technique). The effective survey area was approximated by calculating the area around each survey station using the effective detection radius of the megapode using the program DISTANCE (Thomas *et al.* 2006). The expected range of the Micronesian megapode in the Mount Lasu and Maga regions was estimated using native limestone forest estimates from 2006 Forest Service data (Forest Service 2006).

Territory Mapping

Between August 12 and 21, 2008, four study plots were established on Aguiguan and sampled to estimate Micronesian megapode territory densities (Figure 2). All four plots were established in native limestone forest in areas where megapodes were observed previously to maximize the likelihood of recording megapodes. Each study plot consisted of a grid of points at 50 meter intervals developed using Hawth's Analysis Tools© 3.27 in ArcMap 9.2 (ESRI, Redlands, CA). All points were downloaded into Garmin 76CSx (Garmin International, Inc., Olathe, KS) Global Positioning System (GPS) units as waypoints to assist with mapping. GPS tracks from each unit were also recorded during each survey and downloaded into ArcMap 9.2 to identify the effective search area.

Each plot was surveyed in the morning (0600-1200) and afternoon (1500-1800) over several days by one or several surveyors. All Micronesian megapode detections were marked on a map of the study plot using symbols for movements and activities outlined by Bibby *et al.* (2000) and/or recorded in a field notebook with GPS waypoints. These locations were then transferred to a master map of each study plot in ArcMap 9.2.

Micronesian megapode territory densities were determined by counting the number of pairs within each study plot and dividing the total number of territories by the size of the plot. Territories which overlapped the edge of the plot were included as half territories (Bibby *et al.* 2000). The presence of a territory was determined through a combination of visual observations of pairs and territorial behaviors. In addition, efforts were made to capture, band, and collect feather and blood samples from Micronesian megapodes in and outside of the study plot. We used unique combinations of color bands to facilitate territory mapping efforts and future efforts to obtain survival estimates. Feather and blood samples were collected for potential genetic analysis. The boundary of each study plot was defined as the outer points in the study plot grid. These were typically associated with cliff lines and forest edges.

RESULTS

Tinian

No Micronesian megapodes were detected during point-transect or playback surveys on Tinian. Search effort for point-transect surveys totaled 90 hours with approximately 26 percent of the time (23 hours) spent surveying native limestone forest. Search effort for the playback survey totaled 11 hours, all in native limestone forest. The effective detection radius of the megapode was 38 m ± 4m, based on an analysis of the Aguiguan playback survey data, while the effective search area and expected range of the megapode in the Mount Lasu and Maga regions was 9 ha and 71 ha, respectively. Therefore, the likelihood of detecting a megapode if the population in the Mount Lasu and Maga regions was two megapodes was estimated to be 24 percent. However, we believe this may be an underestimate. Based on our observations during playback surveys and color-banding of megapodes on Aguiguan, playbacks typically elicited a response from the majority of the territory holders in the area (see Aguiguan below). Therefore, it is likely that we would have detected megapodes in the area if they were present.

Aguiguan

One hundred four person-hours were spent territory mapping in the four study plots, which translated to approximately 2 person hours per hectare. In addition, 16 Micronesian megapodes were captured and color-banded in and outside the study plots (Table 2). Approximately 80 percent of the birds captured were paired. A total of 15 territories was identified. The average territory density per hectare was 0.27 (± 0.03 Standard Deviation (SD)), and territory size in limestone forest 3.76 ha (± 0.40 SD), respectively (Table 3). If we assume densities in our sampling plots are representative of megapode densities in all native forest areas occupied by megapodes (280 ha; see Table 3 in 2.1 Vegetation Surveys), we estimate there could be up to 75 Micronesian megapode territories on Aguiguan in this habitat. This estimate excluded the limestone forest along the southeast coast of Aguiguan where no megapodes were detected during the 1982, 1995, 2000, 2002, and 2008 point-transect and 2008 playback surveys, as well as secondary forest habitats, which are used by Micronesian megapodes, but were not sampled during the territory mapping.

Fifteen Micronesian megapodes were detected on Aguiguan during the point-transect surveys in June 2008 (Table 1). Unfortunately, there were insufficient detections to calculate densities. To estimate the proportion of megapodes present that were detected during the point-transect survey, we compared detections at stations that overlapped with our study plots (n = 17). If the megapodes detected during the August territory mapping were present in the same areas during the June point-transect survey, then the point-transect survey detected approximately 17 percent (number of birds detected per station/ number of birds territory mapped per station) of the population.

Forty Micronesian megapodes were detected on Aguiguan during the playback surveys in August 2008 (Table 1). The number of detections and stations occupied by megapodes was more than double the number of detections during the June 2008 point-transect survey (Table 1), indicating that playback surveys may be more effective than point-transect surveys for megapodes. Unfortunately, insufficient detections were recorded to estimate megapode densities using this method. In addition, observations reported by the playback surveyors indicate that Micronesian megapodes were moving in response to the playbacks which may bias detection distance estimates (Buckland *et al.* 2006). This movement will need to be assessed to properly calibrate future playback surveys (P.M. Gorresen, pers. comm. 2008). Also, comparing the playback detections at stations which overlapped the territory mapping plots (n = 17) indicates approximately 50 percent of the megapodes present were detected. We believe that this is an underestimate because prior to the survey three of the birds in the plots were captured using playbacks and banded which may have reduced their subsequent responsiveness to the playbacks. If we assume these birds would have responded and were detected, approximately 67 percent of the megapodes present would have been detected.

No evidence of Micronesian megapode breeding was recorded during the surveys despite efforts to locate potential nest sites. In addition, no juveniles or recently hatched birds were observed during the surveys. Finally, morphometrics, band numbers, and color-band combinations for all megapodes caught and banded are summarized in Table 2. Blood and feather samples of each individual caught during banding were collected and are stored at the Burke Museum of Natural History and Culture, University of Washington, Seattle, Washington.

DISCUSSION

In conclusion, we estimate that there could be up to 75 megapode territories on Aguiguan. However, because this study was intended to be a pilot study the megapode study plots were placed in areas known to be occupied by megapodes. Therefore, densities in these study plots may not reflect megapode densities across the island and extrapolating from these estimates should be done with caution. Previous estimates for the island by Engbring *et al.* (11 megapodes; 1986), Cruz *et al.* (51 megapodes; 2000), and Esselstyn *et al.* (72 megapodes; 2003) were much lower than the 75 territories estimated from the study plots in this study. In general, the number of detections per station recorded during the point-transect survey on Aguiguan was also similar to those reported in previous surveys (Table 1). This may indicate that the number of detections, and potentially the population, has been relatively stable since 1982. If the population has been stable over this time period then the megapode population may be less than 75 territories.

No Micronesian megapodes were recorded on Tinian, in spite of extensive surveys. This is consistent with previous surveys where detections were sporadic (Wiles et al. 1987, O'Daniel and Krueger 1999, Witteman 2001, Vogt 2008). Due to the sporadic nature of these detections we cannot rule out the possibility that megapodes may yet be present on Tinian in low numbers and may utilize the native forest habitats elsewhere on the island.

One potential reason for the sporadic nature of the detections on Tinian is inter-island movements and factors limiting population growth on the island. Movement between islands is documented for megapodes in Palau (Pratt *et al*. 1980), and the Micronesian megapode seems capable of crossing the water gaps between the islands in the Mariana archipelago (the maximum distance between islands is 60 miles, minimum distance is 3 miles, average is 36 miles). This is especially true for Tinian which is only 3 miles from Saipan and 6 miles from Aguiguan. If megapodes dispersed from Aguiguan or Saipan, they would likely end up on Tinian. However, a population on Tinian may not be able to persist on the island due to predation (*e.g.*, feral cats (*Felis cattus*)) or some other potential limiting factor (*i.e.*, limited breeding habitat).

Micronesian megapode home range size and territoriality are poorly known. It is obvious that they defend some type of area as evidenced by a pair's quick and agitated response to played-back recordings of duetting pairs during this study. However, we believe they may be defending food resources and not nesting habitat based on the lack of observations of nesting activity. Our average territory size estimate from the territory mapping was 3.8 ha compared with reported territory sizes ranging from 1 ha (Glass and Aldan 1988) to <10 ha (Lemke 1984) in the Mariana Islands. We believe that territory size likely varies with habitat conditions. Our observations on Aguiguan and reports from other islands in the archipelago (Lemke 1984, Glass and Aldan 1988, Vogt 2008; Kessler, pers. observation) indicate that closed canopy forest with a moist or wet substrate is probably richer foraging habitat than open forest with a dry substrate; territories in the former habitat are likely to be smaller than territories in the latter. This was especially evident on Aguiguan where megapodes were more common in areas with well developed canopies and wetter conditions (Kessler, pers. observation)

Survey efforts for Micronesian megapodes yielded mixed results. The point-transect survey method yielded few detections and was not found to be effective on Tinian and Aguiguan. This result could reflect low numbers of birds and infrequent vocalizations. However, if megapode detection data can be obtained and pooled from point-transect surveys that were conducted under the same survey conditions it may be possible to develop detection models to estimate densities for surveys with insufficient detections (P.M. Gorresen, pers. comm. 2008).

Playback surveys were found to substantially increase the number of detections. However, the results may be biased by megapodes moving closer to the station, in response to the playback, before being detected. In addition, not all individuals present may have been detected and a minimum number of detections are needed to effectively estimate densities. Therefore, it may not be applicable for estimating densities in small populations where detections are expected to be low. Biases associated with playbacks can be corrected using data on bird movements in response to playbacks (*e.g.*, Klavitter and Marzluff 2007). However, this may prove challenging for megapodes without radio telemetery work due to the difficulty in detecting individuals even with playbacks.

Territory mapping, in conjunction with color-banding and playbacks, could be an effective survey tool. However, establishing and surveying plots is difficult and time consuming, and territory densities can vary between and within habitat types. Multiple study plots are needed to account for this variability which increases the survey effort. This may prove to be difficult on remote islands without sufficient logistical support and time to establish and conduct the surveys.

Mark-resight estimates were not tested during this study. However, megapode responses to banding and playbacks during this study do provide some insight into the effectiveness of this technique. In general, banded birds were shy and difficult to resight after being caught and handled. If sufficient time is available to allow the birds to recover from being handled, this technique could be utilized.

Clearly, more work is needed to develop an effective megapode survey tool that can be used to compare populations across the species' entire range. Playbacks may prove to be the most effective tool if the response of megapodes to playbacks can be fully evaluated. Alternatively, a combination of intensive territory mapping and playback surveys may prove to be an effective method.

No evidence of megapode breeding on Aguiguan was recorded during this study. Megapodes use a variety of nesting strategies (Glass and Aldan 1988, Wiles and Conry 2001) and it is speculated that mound building or burrowing in decomposing trees might be the methods utilized on Aguiguan. However, mounds have yet to be identified despite unverified reports of small mound-like structures on the island (USFWS 1998). In the absence of nesting mounds, we hypothesize that decomposing trees and/or tree roots are the mostly likely method of incubation on the island. Efforts to locate nests in these substrates in August 2008 were not successful. However, the birds may not have been breeding at that time. Therefore, additional work is still needed to identify the breeding strategy of Micronesian megapodes on Aguiguan and the other southern islands.

Although important questions remain unanswered, the following information will assist land managers concerned with the development of the U.S. Marines Corp (USMC) training area on Tinian:

- No megapodes pairs were detected on Navy leased lands in 2008.
- Megapodes may immigrate into the area based on past survey observations.
- Megapodes will most likely be found in limestone forest areas and cliff-line habitat.
- Their territories, once they become established, would be approximately 3.5 ha.
- They would likely seek old growth limestone forest or forest strand beach habitat for nesting.
- The most practical way to detect if a pair is present is through the use of playbacks.

The areas on Navy lands that were used in the past by megapodes are the tracts of native forest at Mt. Lasu and Maga (USFWS 1998, Vogt 2008), and these areas could again be occupied by megapodes. This cliff-line forest habitat will be in direct conflict with USMC Safety Danger Zone's (SDZ's) which will require the backdrop protection of the cliffs for various weapon ranges. Unfortunately there are few areas of this native habitat left on Tinian (Figure 1), and these are found primarily on the land leased by the Navy. Those areas of cliff line will be the contention point of any weapons range proposals due to the rarity of this habitat and its importance to native species.

REFERENCES

Baker, R.H. 1951. The avifauna of Micronesia, its origin, evolution, and distribution. Univ. Kansas Publ., Mus. Nat. Hist. 3:1-359.

Bibby, C., N.D. Burgess, and D.A. Hill. 1992. Bird census techniques. London: Academic Press.

Buckland, S.T., D.R. Anderson, K.P. Burnham, J.L. Laake, D.L. Borchers, and L. Thomas. 2001. Introduction to Distance Sampling: Estimating abundance of biological populations. Oxford: Oxford University Press.

Craig, R. J., R. Chandran, and A. Ellis. 1992. Bird populations of Aguiguan: a ten year update. Pages 8-15 *in* The Aguiguan Expedition (R.J. Craig, editor). Proceedings of the Mariana Research Symposium, Volume 1. College of the Northern Mariana Islands, Saipan, MI.

Cruz, J., L. Arriola, N. Johnson, and G. Beauprez. 2000. Wildlife and Vegetation surveys Aguiguan 2000. Technical report #2 CNMI-DFW. Unpublished report by CNMI Division of Fish and Wildlife Wildlife Section and Tinian Department of Lands and Natural Resources Fish and Wildlife and Conservation Sections. 30 pp.

Division of Fish and Wildlife. 2000a. Wildlife and Vegetation Surveys, Sarigan 2000. Technical Report # 5. CNMI-DFW, unpubl. 51 pp.

Division of Fish and Wildlife. 2000b. Wildlife and Vegetation Surveys, Guguan 2000. Technical Report #3. CNMI-DFW, unpubl. 42 pp.

Division of Fish and Wildlife. 2000c. Wildlife and Vegetation Surveys, Alamagan 2000. Technical Report #4. CNMI-DFW, unpubl. 39 pp.

Division of Fish and Wildlife. 2000d. Wildlife and Vegetation Surveys, Pagan 2000. Technical Report #7. CNMI-DFW, unpubl. 66 pp.

Division of Fish and Wildlife 2000e. Wildlife and Vegetation Surveys, Agrihan 2000. Technical Report #8. CNMI-DFW, unpubl. 43 pp.

Division of Fish and Wildlife. 2000f. Wildlife and Vegetation Surveys, Anatahan 2000. Technical Report #6. CNMI-DFW, unpubl. 48 pp.

Elliott, A. 1994. Family Megapodiidae (Megapodes). Pages 278-309 *in* Handbook of the Birds of the World. Volume 2, New World Vultures to Guineafowl (del Hoyo, J. Elliott, and J. Sargatal, eds). Lynx Edicions, Barcelona.

Engbring, J., F.L. Ramsey, and V.J. Wildman. 1986. Micronesian forest bird survey, 1982: Saipan, Tinian, Aguijan, and Rota. U.S. Fish and Wildlife Service Report. 143pp.

Esselstyn, J., J.B. Cruz, L.L. Williams, and N. Hawley. 2003. Wildlife and Vegetation surveys, Aguiguan 2002. Technical report #9 CNMI-DFW. Unpublished report by Division of Fish and Wildlife Wildlife Section, Saipan, CNMI. 55 pp.

Glass, P.O. and D. Aldan. 1988. Micronesian Megapode Surveys and Research. Pp 131-153, *In* Division of fish and Wildlife Progress report: 1982-1987. CNMI Division of Fish and Wildlife, Saipan.

Klavitter, J.L. and J.M. Marzluff. 2007. Methods to correct for density inflation biases in Hawaiian hawk surveys using attractant calls. Journal of Raptor Research 41:81-89.

Lemke, T.O. 1985. Micronesian megapode surveys and inventories. Pp. 106-112, *In* Commonwealth of the Northern Mariana Islands Division of Fish and Wildlife Annual Report FY 1984.

Martin, G, L.L. Williams, J.B. de Cruz, N.B. Hawley, S. Vogt, B. D. Smith, O. Borquin, S. Kremer, and C. Kessler. 2008. Wildlife and vegetation surveys of Sarigan Island, April 13-25, 2006. Unpublished Technical Report 14, CNMI Division of Fish and Wildlife.

O'Daniel, D. and S. Krueger. 1999. Recent sightings of the Micronesian Megapode on Tinian, Mariana Islands. Micronesica 31(2): 301-307.

Pratt, H.D., J. Engbring, P.L. Bruner, and D.G. Berrett. 1980. Notes on the taxonomy, natural history, and status of the resident birds of Palau. Condor 82:117-131.

Reynolds, M.H. and T.J. Snetsinger. 2001. The Hawaii rare bird search 1994-1996. Studies in Avian Biology 22: 133-143.

Reynolds, R.T., J.M. Scott, and R.A. Nussbaum. 1980. A variable circular-plot method for estimating bird numbers. Condor 82:309-313.

Thomas, L., J.L. Laake, S. Strindberg, F.F.C. Marques, S.T. Buckland, D.L. Borchers, D.R. Anderson, K.P. Burnham, S.L. Hedley, J.H. Pollard, J.R.B. Bishop, and T.A. Marques. 2006. Distance 5.0. Release 2. Research Unit for Wildlife Population Assessment, University of St. Andrews, U.K. <http://www.ruwpa.st-and.ac.uk/distance/>

Vogt, S. 2008. Micronesian megapode (*Megapodius laperouse laperouse*) surveys on Farallon de Medinilla, Commonweath of the Northern Mariana Islands. Unpublished report for U.S. Navy. 8pp.

Wiles, G.J. and P.J. Conry. 2001. Characteristics of Nest Mounds of Micronesian Megapodes in Palau. Journal of Field Ornithology 72(2):267-275.

Wiles, G.J., R.E. Beck, and A.B. Amerson. 1987. The Micronesian Megapode on Tinian, Mariana Islands. Elepaio 47:1-3.

Witteman, G. 2001. A Quantitative Survey and Inventory of the Micronesian Megapode and its Habitat on Tinian, CNMI. Unpublished report prepared for: URS Corporation, Honolulu, HI.

U.S. Fish and Wildlife Service. 1970. Conservation of endangered species and other fish or wildlife. Federal Register 35 (106):8491-8498.

U.S. Fish and Wildlife Service. 1998. Recovery Plan for the Micronesian Megapode (*Megapodius laperouse laperouse*). U.S.Fish and Wildlife Service. Portland., Or. 65+pp.

U.S. Forest Service. 2006. CNMI Veg Data Package (7.5 mb). Retrieved online, November 25, 2008, from http://www.fs.fed.us/r5/spf/fhp/fhm/landcover/islands/index.shtml

Table 1. Number of stations sampled, number of stations where Micronesian megapodes were detected, number of megapodes detected, percent occupancy of stations, and megapodes per station for point-transect (PT) and playback (PB) surveys conducted on Aguiguan. Number of stations occupied were not available for 1992, 2000, and 2002.

Year	Month	Survey Type	Stations Sampled	Stations Occupied	Number Detected	Percent Occupancy	Birds per Station
1982	June	PT	66	8	14	9.1	0.16
1992	May	PT	66	UNK	11	UNK	0.17
1995	June	PT	66	12	16	18.2	0.24
2000	April	PT	66	UNK	12	UNK	0.18
2002	March	PT	66	UNK	16	UNK	0.24
2008	June	PT	80	11	15	13.8	0.19
2008	August	PB	80	24	40	30.0	0.50

Table 2. Band number, color band combination, morphology, and pair status for Mirconesian megapodes banded on Aguiguan in August 2008. Color band combinations are read from top to bottom and include yellow (Y), green (G), black (K), red (R), purple (P), white (W), blue (B), and a U.S. Fish and Wildlife Service aluminum band (A).

Bird	Date	Band #	Left Leg	Right Leg	Weight (g)	Wing (mm)	Tail (mm)	Tarsus (mm)	Culmen (mm)	Status
1	13 Aug	1096-96501	Y/Y	Y/A	373	191	65	53	22	Single
2	14 Aug	1096-96502	Y/G	Y/A	381	192	65	57	18	Paired with Bird 3
3	14 Aug	1096-96503	Y/K	Y/A	378	179	56	58	19	Paired with Bird 2
4	14 Aug	1096-96504	R/R	R/A	446	183	64	52	19	Single
5	14 Aug	1096-96505	R/W	R/A	343	176	60	53	20	Single
6	14 Aug	1096-96506	P/P	P/A	398	182	69	65	19	Paired with Bird 7
7	14 Aug	1096-96507	P/W	P/A	436	190	61	54	24	Paired with Bird 6
8	19 Aug	1096-96508	R/Y	R/A	346	189	53	57	20	Pair with unbanded
9	19 Aug	1096-96509	G/G	G/A	428	190	62	56	19	Pair with unbanded
10	20 Aug	1096-96510	B/B	B/A	366	185	62	52	18	Paired with Bird 11
11	20 Aug	1096-96511	B/R	B/A	358	190	72	53	21	Paired with Bird 10
12	20 Aug	1096-96512	R/K	R/A	421	-	-	-	-	Pair with unbanded
13	20 Aug	1096-96513	R/G	R/A	358	186	55	57	21	Pair with unbanded
14	21 Aug	1096-96514	B/W	R/A	365	192	68	54	17	Pair with unbanded
15	21 Aug	1096-96515	B/G	R/A	330	182	59	56	22	Pair with unbanded
16	21 Aug	1096-96516	K/R	G/A	375	160	72	60	20	Pair with unbanded

Table 3. Micronesian megapode territory densities (territories/ha) in four limestone forest plots on Aguiguan in August 2008.

Plot	Plot Size (ha)	Number of Territories	Territory Size	Density
1	12.84	3	4.28	0.23
2	10.03	3	3.34	0.30
3	19.25	5	3.85	0.26
4	14.28	4	3.57	0.28
	Mean (± Standard Deviation)		3.76 (± 0.40)	0.27 (± 0.03)

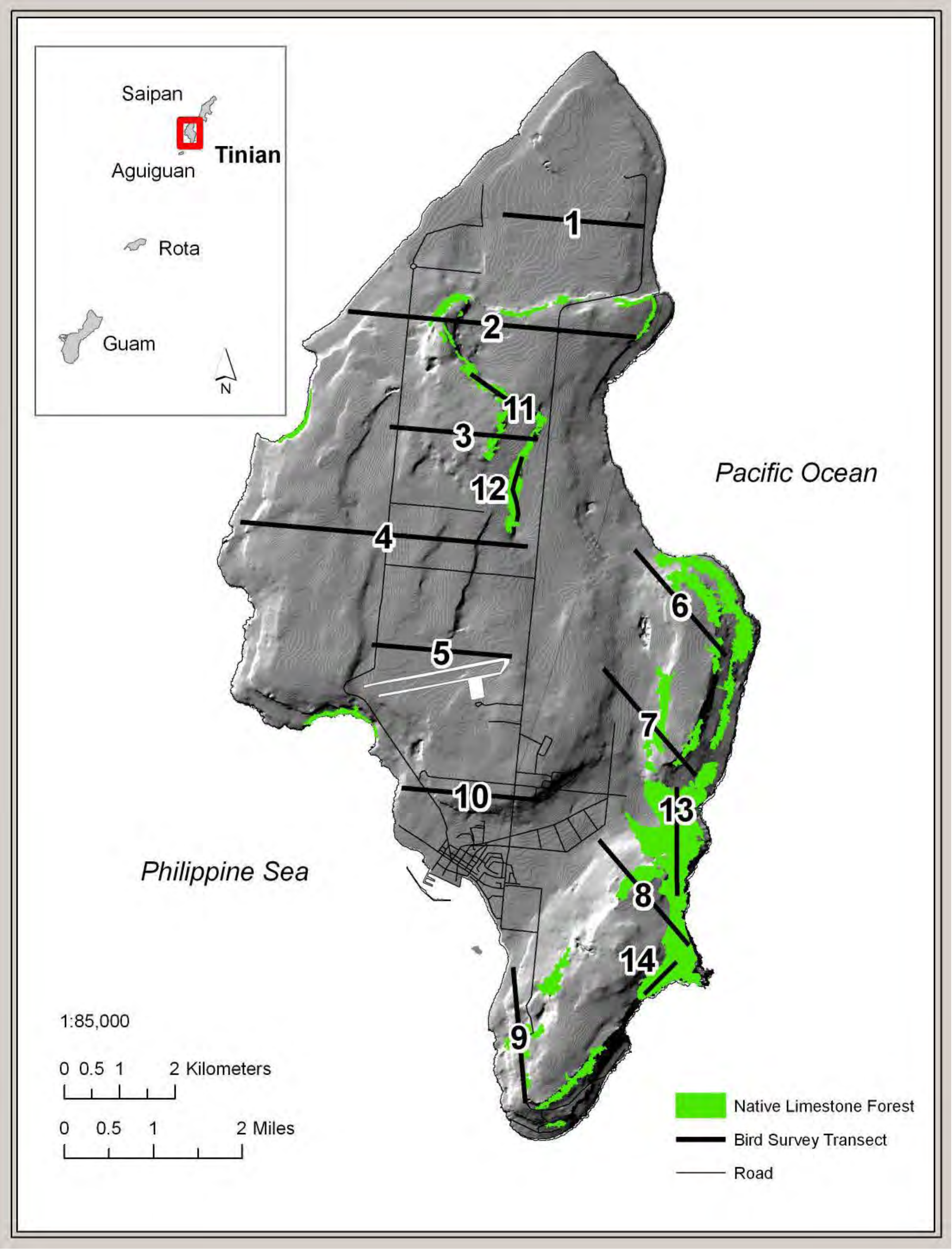


Figure 1. Island of Tinian showing the survey transects and native limestone forest.

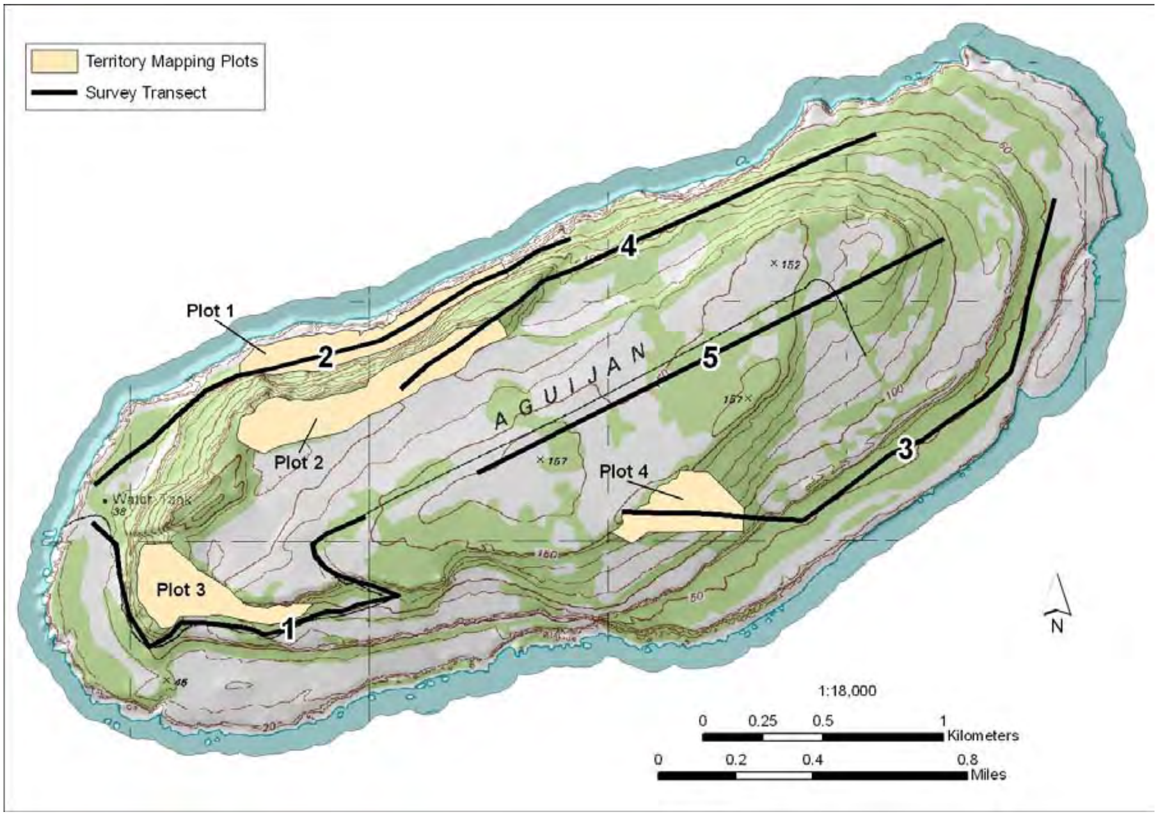


Figure 2. Survey transects and territory mapping plots surveyed on the island of Aguiguan in 2008.

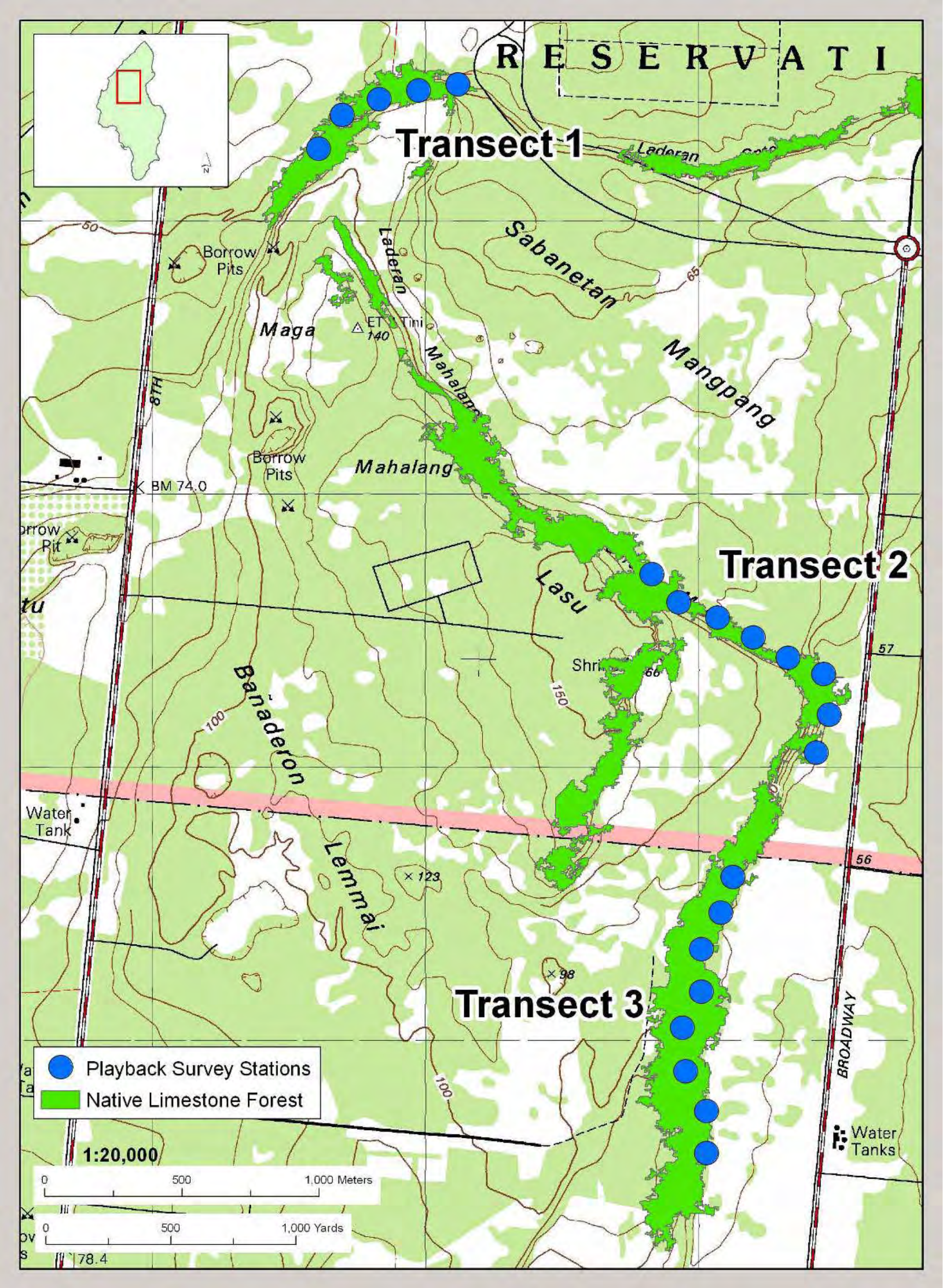


Figure 3. Playback survey stations and native limestone forest in the Mount Lasu and Maga regions of Tinian.

Attachment 1. UTM coordinates for the Micronesian megapode playback survey transects on the island of Tinian. All coordinates are in WGS84 UTM Zone 5 North. Stations along transect 3 correspond with transect 12 of the point-transect survey (see section 2.4.1 General Landbirds on Tinian and Aguiguan Attachment 1)

Transect	Station	Latitude	Longitude
1	1	352116.51	1665499.20
1	2	351972.25	1665476.71
1	3	351693.23	1665386.83
1	4	351607.22	1665263.67
1	5	351828.95	1665445.23
2	1	353474.33	1663190.50
2	2	353427.50	1663052.17
2	3	353454.51	1663339.75
2	4	353323.66	1663398.40
2	5	353196.90	1663473.75
2	6	353066.19	1663546.83
2	7	352923.87	1663602.55
2	8	352825.12	1663706.10
3	1	353122.99	1662596.49
3	2	353078.71	1662466.68
3	3	353007.30	1662332.00
3	4	353006.92	1662176.79
3	5	352938.08	1662044.09
3	6	352949.16	1661885.87
3	7	353025.32	1661739.87
3	8	353026.24	1661586.31

2.4.4 NIGHTINGALE REED-WARBLER ON AGUIGUAN

Male Nightingale Reed-warbler.
Photo by Scott Vogt.

Prepared by: **Ann P. Marshall (U.S. Fish and Wildlife Service), Fred Amidon (U.S. Fish and Wildlife Service), Paul Radley (CNMI Division of Fish and Wildlife), Gayle Martin (CNMI Division of Fish and Wildlife), and Rick Camp (U.S. Geological Survey)**

INTRODUCTION

The endangered nightingale reed-warbler (*Acrocephalus luscinia*), known in the Chamorro language as *ga'ga'karisu* (bird of the reeds) on Saipan, is a medium-sized, yellowish, long-billed passerine (USFWS 1998). It was federally listed as endangered in 1970 (USFWS 1970).

The nightingale reed-warbler belongs to the Old World reed-warbler group (Sylviinae: *Acrocephalus*), which is widespread from Europe through Australasia (Watson *et al*. 1986). The nightingale reed-warbler is endemic to the Mariana Islands and is known historically from five islands in the archipelago: Guam, Aguiguan, Saipan, Alamagan, and Pagan. In addition, the nightingale reed-warbler occurred prehistorically on Tinian (Steadman 1995). Currently, three subspecies of the nightingale reed-warbler are recognized: (1) *A. l. luscinia* on Guam, Saipan, and Alamagan; (2) *A. l. nijoi* on Aguiguan; and (3) *A. l. yamashinae* on Pagan (Pratt *et al*. 1987, Watson *et al*. 1986). Previously, Yamashina (1942) recognized four subspecies: (1) *A. l. luscinia* on Guam; (2) *A. l. hiwae* on Saipan and Alamagan; (3) *A. l. nijoi* on Aguiguan; and (4) *A. l. yamashinae* on Pagan. Mitochondrial DNA analysis provides some evidence that nightingale reed-warblers from Guam and Saipan do not fall out as sister taxa and that Guam birds fall outside the clade (a group of living organisms including all descendants sharing specific genetic traits of a common ancestor) of other Pacific Island *Acrocephalus* and therefore may be descended from a different continental ancestor (Beth Slikas, *in litt*. 2000).

The nightingale reed-warbler is believed extirpated from 3 islands. It has been extirpated, by unknown factors, from Guam since the late 1960s (Engbring *et al.* 1986, Reichel *et al.* 1992, Tenorio and Associates 1979). On Aguiguan, Engbring *et al*. (1986) report 3 probable records of nightingale reed-warblers during their 1982 surveys, though none were recorded on the counts. Several incidental observations of nightingale reed-warblers were also made during the survey team's visit 1-4 June 1982; and therefore, based on this information, they made a liberal estimation of 15 birds on the island (Engbring *et al*. 1986). Glass (1987) reported on 10 observations of nightingale reed-warblers, one made in 1984 and the other nine in February 1987. Following that report, the species was thought to be extirpated from Aguiguan (Reichel *et al.* 1992), but two singing males were observed in 1992 (Craig and Chandran 1992), and one was observed in 1993 (Lusk 1993). The last observation of nightingale reed-warblers on Aguiguan occurred in 1995 (USFWS 1998). Focused survey efforts in 2000 and 2002 failed to detect nightingale reed-warblers (Cruz *et al*. 2000, Esselstyn *et al*. 2003). The Pagan subspecies was extirpated from Pagan, presumably due to volcanic activity or habitat loss from overgrazing between the 1960s and 1981 (Glass 1987).

Based on the best current available information, between 2,769 to 3,596 pairs of nightingale reed-warblers are likely distributed over two islands: Saipan (2,596 pairs), and Alamagan (173-1,000 pairs) (DFW 2000; Camp *et al*. in review). Additional information on the nightingale reed-warbler can be obtained from Craig (1992), the recovery plan (USFWS 1998), and Mosher (2006).

The present surveys were conducted in June and August of 2008 to search for the nightingale reed-warbler on Aguiguan where they may have been extirpated. The information from these surveys will help us better understand the status of the species in the Mariana Islands.

METHODS

Study Area – The island of Aguiguan is part of the Commonwealth of the Northern Mariana Islands (CNMI). It is the second smallest (7 km^2), uninhabited limestone island in the archipelago and is found off the southwest coast of Tinian (Figure 1; Engbring *et al*. 1986). Because of the large number of feral goats (*Capra hircus*) on the island, Aguiguan is usually referred to locally as "Goat Island." Human activities (*e.g.*, commercial agriculture and timber harvesting) on the island have extensively altered the vegetation. In 1982, about 47 percent of native forest remained, and the remainder was about 4 percent secondary (mixed introduced and native) vegetation, 43 percent open field, and around 5 ha of tangantangan (*Leucaena leucocephala*) forest (Engbring *et al.* 1986). Based on a recent land cover assessment, the island is currently around 49 percent native forest, 14 percent secondary vegetation, 23 percent open field, 6 percent tangantangan, and 4 percent coastal strand (See 2.1 Vegetation Surveys).

Observer Training – Observers with previous experience surveying for birds in the Mariana Islands were given distance calibration and bird vocalization training in different habitats prior to the actual surveys.

Figure 1. The island of Aguiguan, CNMI, showing the survey transects 1 -5.

Survey Methodology – Island-wide point surveys (or variable circular plot (VCP) surveys) were conducted for forest birds 25 to 27 June, 2008, and directed surveys using playbacks for the nightingale reed-warbler were conducted 20 to 22 August, 2008, on Aguiguan. Five transects with a total of 80 stations were surveyed (Figure 1; Camp *et al*. 2009). VCP surveys followed standard point-transect methods, consisting of 8-minute counts at each station (150 m apart) and estimation of the horizontal distance to each bird heard or seen (See Reynolds *et al*. 1980, Engbring *et al*. 1986 for details). Counts commenced at sunrise and continued to 1100 hours and were conducted under favorable weather conditions.

The same transects used for the VCP surveys were used for the directed surveys. A taped vocalization of a male nightingale reed-warbler singing was played for one minute at each station. Following the recording, the observers looked and listened for nightingale reed-warblers for three minutes. Weather and habitat sampling conditions were recorded at each station during both surveys. Counts commenced at sunrise and continued until 1100 hours, and were conducted under favorable weather conditions. Survey effort was recorded in hours as the difference between the start and end times for each survey (sunrise to 1100).

Following Reynolds and Snetsinger (2001), we calculated detection probabilities to estimate the likelihood of extirpation of the nightingale reed-warbler on Aguiguan. Scott *et al*. (1986) calculated the probability (*p*) of detecting one bird from a randomly distributed population of *n* individuals as:

$$p = 1 - \left(1 - \frac{a}{A}\right)^{n}$$

The effective search area (*a*; 125.9 hectares (ha)) was approximated by calculating the area for the effective detection radius (EDR) of the nightingale reed-warbler (71 meters; R. Camp, USGS-BRD, pers. comm. 2008) using ArcMap 9.2 (ESRI, Redlands, CA) and XTools Pro 5.2.0 (Data East, LLC, Novosibirsk, Russia). *A*, the last known range of the nightingale reed-warbler on Aguiguan, was estimated as the total forested habitat (native, secondary, and tangantangan) available on the island (479 ha; see Table 3, 2.1, Vegetation Surveys). We started with 15 birds as the hypothetical population size, *n*, as 15 was the last estimated number of birds on Aguiguan (Engbring *et al*. 1986).

Using Reed's (1996) modification of Guyann *et al*.'s (1985) statistical methods to infer extinction, we also calculated the minimum number of visits,

$$N_{\min} = \frac{\ln \alpha}{\ln(1 - p)}$$

$N_{\min}$ needed for 95% (α=0.05) and 99% (α=0.01) probability of detection. *N* is the number of independent visits made to search for the species. We defined one visit as 10 hours of search effort.

RESULTS

No nightingale reed-warblers were detected during VCP counts or during the directed nightingale reed-warbler searches. One vocalization recorded during the directed surveys and thought perhaps to be a nightingale reed-warbler was later definitively identified as a golden white-eye (*Cleptornis marchei*). Search effort totaled 47 hours for the June and August 2008 surveys on Aguiguan. We determined the probability of detecting nightingale reed-warblers for various population sizes (n), starting with a population of 15, the population estimated by Engbring *et al*. (1986) in 1982. The likelihood of detecting 1 nightingale reed-warbler if the population was 15 during the 2008 surveys was 99 percent (Table 1) while the likelihood of detecting one nightingale reed-warbler if the population on Aguiguan was two was 46 percent (Table 1). In addition, if the population was two nightingale reed-warblers, 5 visits (95 percent probability) would be needed to achieve a 95 percent likelihood of detecting a reed-warbler compared to our 4.7 visits made, therefore, we cannot infer extirpation based on our 2008 survey effort.

One observer noted that several endemic species apparently responded (*i.e.*, approach observer playing tape or to sing) to playbacks of the nightingale reed-warbler, in particular the golden white-eye, whose song sounds similar (Glass 1987). Rufous fantails (*Rhipidura rufifrons*), bridled white eyes (*Zosterops conspicillatus*), Micronesian starlings (*Aplonis opaca*), and Micronesian myzomela (*Myzomela rubratra*) were also noted by this observer to occasionally respond to the playbacks. The CNMI Division of Fish and Wildlife (DFW; Esselstyn *et al*. 2003) also noted that golden white-eyes and Micronesian starlings respond to nightingale reed-warbler playbacks during surveys.

Table 1. Detection probability (DP) for one nightingale reed-warbler from a population of *n* birds randomly distributed across the known range.

Nightingale Reed-warbler Population (*n*)	DP	N_{min} for DP =95%	N_{min} for DP =99%
15	0.99	.65	1.01
10	0.95	.98	1.51
5	0.78	1.96	3.02
2	0.46	4.91	7.55

DISCUSSION

Although we cannot infer extirpation of the nightingale reed-warbler on Aguiguan at this time, the population would have to have been two or less birds in order not to have been detected during the 2008 surveys. It should be noted that some rare Hawaiian birds have been rediscovered after they were presumed extinct (Reynolds and Snetsinger 2001). Additional search effort could increase our confidence in the possibility of extirpation of this species or could lead to detections if it is still present. For example, additional searches could be established in areas of Aguiguan that were not covered during the 2008 surveys.

It should be noted that the DFW conducted surveys along the 4 Engbring *et al*. (1986) transects on Aguiguan in 2000 and again in 2002 (Cruz *et al*. 2000, Esselstyn *et al*. 2003) and also failed to detect nightingale reed-warblers. It has been suggested that the birds detected during the Craig and Chandran (1992) surveys were actually young birds colonizing from Saipan, and that the subspecies designation of nightingale reed-warblers on Aguiguan is not valid (Craig and Chandran 1992, Esselstyn *et al*. 2003). The additional search effort needed should be made, and if the species is extant, efforts should be made to preserve what may be a unique island subspecies. If however, nightingale reed-warblers have been extirpated on Aguiguan, the opportunity then exists to reintroduce birds there from either Saipan or Alamagan. In fact, one of the objectives in the nightingale reed-warbler recovery plan is the establishment of at least three additional populations, with emphasis on islands where they used to occur (USFWS 1998). Reintroductions to Aguiguan, will increase population numbers and species distribution and contribute to the delisting goals for this species. A plan to reintroduce nightingale reed-warbler should first be developed and should include information on habitat restoration and preservation as well as limiting factors for the birds.

REFERENCES

Bibby, C., N.D. Burgess, and D.A. Hill. 1992. Bird census techniques. London: Academic Press.

Camp, R.J., T.K. Pratt, F. Amidon, A.P. Marshall, S. Kremer, and M. Laut. 2009. Draft status and trends of the land bird avifauna on Tinian and Aguiguan, Mariana Islands. Appendix 3.1 *in* Terrestrial Resource Surveys of Tinian and Aguiguan, Mariana Islands, 2008. Working Draft. U.S. Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office, Honolulu. HI.

Camp, R.J., T.K. Pratt, A.P. Marshall, F. Amidon, and L.L. Williams. In review (to Bird Conservation International). Status and trends of the land bird avifauna on Saipan, Mariana Islands, with emphasis on the endangered nightingale reed-warbler (*Acrocephalus luscinia*).

Craig, R.J. 1992. Territoriality, habitat use, and ecological distinctness of an endangered Pacific island reed-warbler. Journal of Field Ornithology 63:436-444.

Craig, R.J., and R. Chandran. 1992. Wildlife species recorded during the Aguiguan expedition. Pp. 1-7 in The Aguiguan Expedition, Proceedings of the Marianas Research Symposium 1 (R. Craig, ed.). Published by Northern Marianas College. 65 pp.

Cruz, J., L. Arriola, N. Johnson, and G. Beauprez. 2000. Wildlife and Vegetation surveys Aguiguan 2000. Technical report #2 CNMI-DFW. Unpublished report by CNMI Division of Fish and Wildlife Wildlife Section and Tinian Department of Lands and Natural Resources Fish and Wildlife and Conservation Sections. 30 pp.

Division of Fish and Wildlife (DFW). 2000. Summary of wildlife surveys Alamagan Island. Unpublished report by the CNMI Division of Fish and Wildlife and Northern Mayor's Office, Saipan, CNMI, 10-17 June 2002. 39 pp.

Engbring, J., F.L. Ramsey, and V.J. Wildman. 1986. Micronesian forest bird survey, 1982: Saipan, Tinian, Aguijan, and Rota. U.S. Fish and Wildlife Service Report. 143pp.

Esselstyn, J., J.B. Cruz, L.L. Williams, and N. Hawley. 2003. Wildlife and Vegetation surveys, Aguiguan 2002. Technical report #9 CNMI-DFW. Unpublished report by Division of Fish and Wildlife Wildlife Section, Saipan, CNMI. 55 pp.

Glass, P.O. 1987. Nightingale reed-warbler surveys and inventories. Pp. 154-158, *in* CNMI, Division of Fish and Wildlife Progress Report, Department of Land and Natural Resources, Saipan, MP.

Guyann, D.C., Jr., R.L. Downing, and G.R Askew. 1985. Estimating the probability of non-detection of low density population. Cryptozoology 4: 55-60.

Lusk, M. 1993. Field trip report Aguiguan, August 23-26. Unpublished report by the CNMI, Division of Fish and Wildlife Report, Saipan, MP.

Mosher, S. 2006. Ecology of the endangered nightingale reed-warbler (*Acrocephalus luscinia*) on Saipan, Micronesia. M.S. thesis, University of Idaho, Boise, ID. 111 pp.

Pratt, H.D., P.L. Bruner, and D.G. Berrett. 1987. A field guide to the birds of Hawaii and the tropical Pacific. Princeton, N.J.: Princeton University Press.

Reed, J.M. 1996. Using statistical probability to increase confidence of inferring species extinction. Conservation Biology 10: 1283-1285.

Reichel, J.D., G.J. Wiles, and P.O. Glass. 1992. Island extinctions: the case of the endangered nightingale reed-warbler. Wilson Bulletin 104:44-54.

Reynolds, M.H. and T.J. Snetsinger. 2001. The Hawaii rare bird search 1994-1996. Studies in Avian Biology 22: 133-143.

Reynolds, R.T., J.M. Scott, and R.A. Nussbaum. 1980. A variable circular-plot method for estimating bird numbers. Condor 82: 309-313.

Scott, J.M., S. Mountainspring, F.L. Ramsey, and C.B. Kepler. 1986. Forest bird communities of the Hawaiian Islands: their dynamics, ecology, and conservation. Studies in Avian Biology 9: 1-431.

Steadman, D.W. 1995. Determining the natural distribution of resident birds in the Mariana Islands. Preliminary report to U.S. Fish and Wildlife Service, Honolulu, HI.

Tenorio and Associates. 1979. Ornithological survey of wetlands in Guam, Saipan, Tinian, and Pagan. U.S. Army Corps of Engineers, Pacific Ocean Division. 202 pp.

U.S. Fish and Wildlife Service (USFWS). 1970. Conservation of endangered species and other fish and wildlife. Federal Register 35:18319-18322.

U.S. Fish and Wildlife Service (USFWS). 1998. Recovery plan for the nightingale reed-warbler, *Acrocephalus luscinia*. Portland, Oregon. 62pp.

Watson , G.E., M.A. Traylor, Jr., and E. Mayr. 1986. Family Sylviidae. *In* E. Mayr and G.W. Cottrell (eds.), Check-list of birds of the world, Vol. 11: 3-294. Cambridge, MA: Museum of Comparative Zoology. 638 pp.

Yamashina, Y. 1942. A new subspecies of *Conopoderas luscinia* from the Mariana Islands. Bull. Biogeogr. Soc. Japan 12: 81-83.

2.4.5 TINIAN MONARCH SURVEYS

Tinian Monarch (*Monarcha takatsukasae*). Photo by Eric VanderWerf.

Prepared by: **Fred Amidon, U.S. Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office, Honolulu, HI. Point-transect methods, results, tables, and figures, with some modifications, from Camp *et al.* (2009) (Appendix 3.1).**

INTRODUCTION

The Tinian monarch (*Monarcha takatsukasae*), or Chuchurican Tinian in the Chamorro language, is a small (15 centimeter) forest bird in the monarch flycatcher family (Monarchidae)(Baker 1951). The monarch currently is found only on the island of Tinian, but examination of museum specimens by Peters (1996) suggested a now extirpated population may have occurred on the island of Saipan, just north of Tinian. The monarch also was reported from the tiny island of Aguiguan just south of Tinian in the early 1950s, but some authorities discount this report as an error (Engbring *et al*. 1986).

The monarch inhabits a variety of forest types on Tinian, including native limestone forest dominated by *Ficus* spp., *Mammea odorata*, *Guamia mariannae*, *Cynometra ramiflora*, *Aglaia mariannensis*, *Premna obtusifolia*, *Pisonia grandis*, *Ochrosia mariannensis*, *Neisosperma oppositifolia*, *Intsia bijuga*, *Melanolepis multiglandulosa*, *Eugenia* spp., *Pandanus* spp., *Artocarpus* spp., and *Hernandia* spp., secondary vegetation consisting primarily of *Casuarina equisetifolia* and the non-natives *Acacia confusa*, *Albizia lebbeck*, *Cocos nucifera*, and *Delonix regia*, with some native species mixed in, like *Melanolepis multiglandulosa* and *Aidia cochinchinensis*, and nearly pure stands of introduced *Leucaena leucocephala* (tangantangan) (Engbring *et al*. 1986, Falanruw *et al.* 1989, USFWS 1996).

The first island-wide survey for the species took place in May 1982 using the point-transect or variable circular plot method. From that survey the population was estimated to be 39,338 birds and Tinian monarchs were found distributed throughout the island in all forest types (Engbring *et al*. 1986). A second survey of the Tinian monarch population took place in August and September 1996 using the same transects and methods as in 1982 (see Figure 1). The 1996 survey estimated the monarch population at 55,721 birds (Lusk *et al*. 2000), which was significantly higher than the estimate of 39,338 birds from 1982 found by Engbring *et al*. (1986). The 1996 survey also found that vegetation density had increased significantly in all forest types since 1982. Lusk *et al*. (2000) hypothesized that the increase in the monarch population was related to increases in density of vegetation in both native and introduced forest habitats, which may have been related to a decrease in grazing pressure.

The Tinian monarch was listed as endangered in 1970 (35 FR 8491) under the authority of the Endangered Species Conservation Act of 1969 (16 U.S.C. 668cc). The Tinian monarch was reclassified from endangered to threatened on April 6, 1987 (52 FR 10890), and on September 21, 2004, the monarch was removed from the Federal List of Endangered and Threatened Wildlife (69 FR 56367). A post-delisting monitoring plan was developed in 2005 (USFWS 2005) and was initiated in 2006 (USFWS 2008). Currently, the Tinian monarch is listed as threatened/endangered by Commonwealth of the Northern Mariana Islands and vulnerable by the World Conservation Union (IUCN 2008).

In 2008, the Department of Defense contracted the U.S. Fish and Wildlife Service's Pacific Islands Fish and Wildlife Office to conduct terrestrial and marine surveys on Tinian and Aguiguan. The following report outlines the survey results from June 2008 point-transect or variable circular plot surveys and Tinian monarch territory mapping in August 2008. The point-transect section is a summary of Camp *et al.* (2009; Appendix 3.1) with some modifications.

METHODS

Point-Transect Surveys

Between 27 April and 8 May 1982 Engbring *et al.* (1986) sampled a total of 216 stations on 10 transects during an island-wide survey of the island of Tinian (Figure 1). All transects were at least 300 meters apart and all stations along each transect were 150 meters apart (Engbring *et al.* 1986). These transects were resurveyed during both the 1996 (28 August – 1 September) and 2008 (14 – 19 June) surveys. An additional 4 transects were sampled during the 2008 survey for a total of 254 stations. The 4 additional transects were included to increase the number of stations in native limestone forest and to improve density estimates for the Tinian monarch (Table 1). These transects were also at least 300 m from the nearest transect and all stations were 150 meters apart.

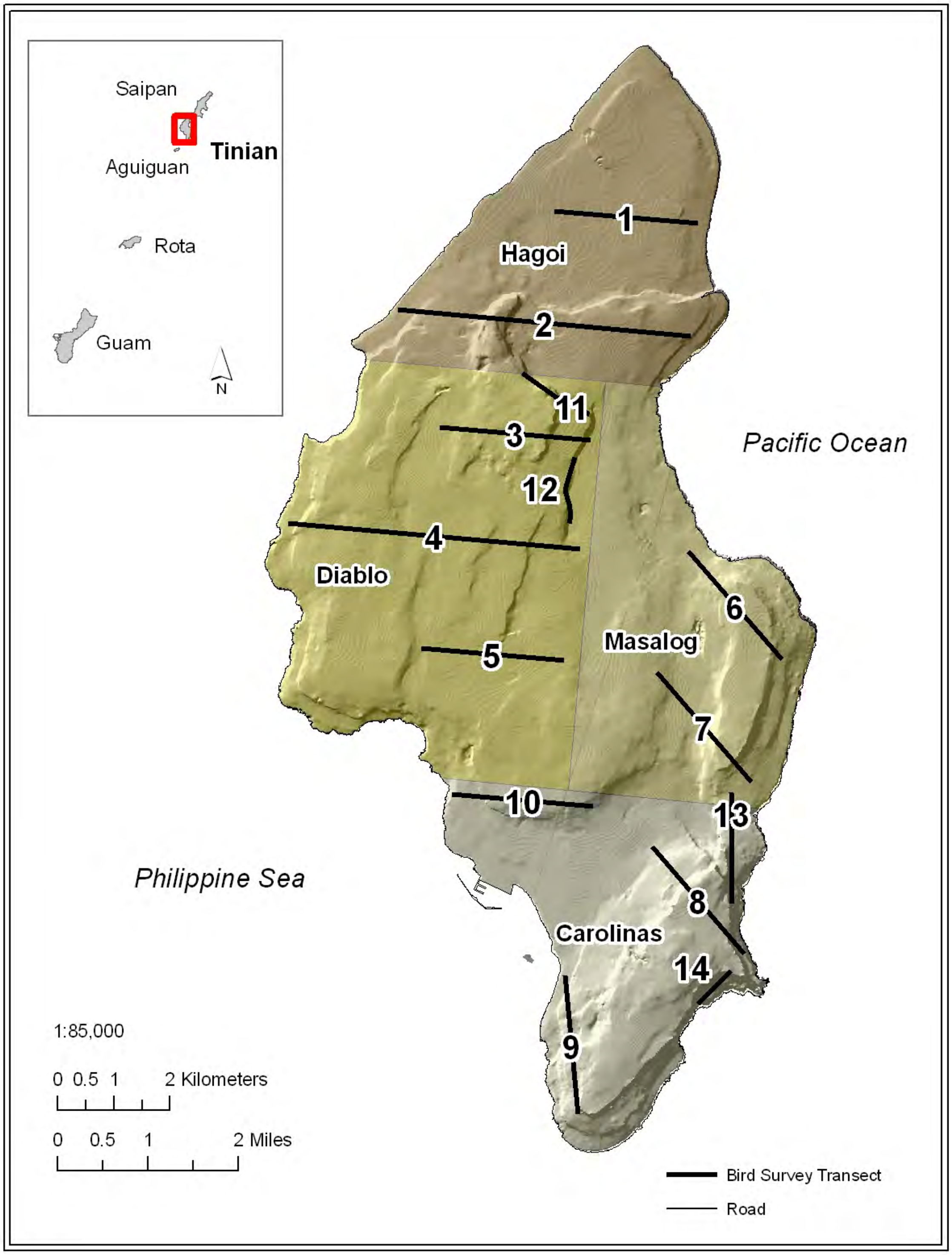


Figure 1. Island of Tinian showing the survey transects and regions (as defined by Engbring *et al.* 1986). Transects 1-10 were counted during all three surveys, and transects 11-14 were established and counted during the 2008 survey.

Table 1. Tinian monarch detections by habitat type in 1982 and estimated number of stations need to produce density estimates with 10%, 20% and 30% coefficients of variation.

			Coefficient of Variation		
Habitat	Stations	Detection	10%	20%	30%
Limestone Forest	14	35	120	30	13
Secondary Forest	67	198	102	25	11
Tangantangan	95	207	138	34	15

All surveys followed standard point-transect methods, consisting of 8-minute counts where horizontal distances to all birds heard and/or seen were measured and recorded (see Engbring *et al*. 1986 for details). Sampling conditions recorded included cloud cover, rain, wind, noise level, and habitat type, and these were later used as covariates in density calculations (see below). Counts commenced at sunrise and continued up to 1100 hours and were conducted only under favorable conditions.

Two observers surveyed each station in 1982, and one observer surveyed the stations in 1996 and 2008. Only data from one counter was used for each station from the 1982 surveys for this analysis and the counters were identified based on their experience and survey proficiency. Engbring *et al*. (1986) analyzed bird detections from all observers to estimate bird densities. For our analysis, we used detections from only one observer to recalculate densities for the 1982 Tinian survey, thus matching the 1996 and 2008 survey effort. Calculating densities from only one of the counters is a conservative approach and ensures sampling independence. This approach approximately halved the number of birds detected; however, our density estimates were generally greater than, but otherwise similar to, those of Engbring *et al*. (see Table 8; 1986). On Tinian the 95% confidence intervals bracketed Engbring *et al*.'s estimates for all but five birds—Mariana Fruit-Dove, Micronesian Honeyeater, Tinian Monarch, Rufous Fantail, and Bridled White-eye. Differences may have resulted from analytical procedures such as selecting different truncation distances, selecting different models to estimate densities, and analytical advances in distance sampling (see Johnson *et al*. 2006), in addition to estimating densities using detections from only one of the counts.

Population status was calculated as densities (birds/km^2) and number of birds (density by habitat type multiplied by habitat type area). Density was calculated using the program DISTANCE (Thomas *et al*. 2006) from species-specific global detection functions where data were post-stratified by survey in the stratum layer. Data were right-truncated to facilitate model fitting (Buckland *et al*. 2001), and the model with the lowest Akaike Information Criterion (AIC) was used to select the detection function that best approximated the data. Candidate models included half-normal and hazard-rate detection functions with expansion series of order two (Buckland *et al*. 2001). Sampling covariates were modeled in the multiple-covariate distance engine of DISTANCE (Thomas *et al*. 2006, Marques *et al*. 2007). Covariates (sampling conditions and survey year) were used to generate the global detection function when the best approximating model was improved by four or more AIC units. Variances and confidence intervals were derived by log-normal based methods. Survey-specific density by station values were generated for the population trends analyses (see below) from the global detection function using the post-stratification by sample option in the stratum layers annual estimates and regional estimates.

Area of habitat types came from Engbring *et al*. (1986) and recent vegetation cover estimates (see 2.1 Vegetation Survey). The area of habitat types was not available for the 1996 Tinian survey; therefore, we used the area by habitat types from Engbring *et al*. to calculate the 1996 numbers of birds. This may slightly underestimate the population size if there was more secondary forest in 1996 than 1982. The agriculture habitat type (combined agroforestry and cultivated habitat type classifications) was not used to calculate numbers of birds because the area of this habitat is very small relative to the island (< 2%), the area of the agriculture habitat type has declined (190 ha in 1982 to 174 ha in 2008; see 2.1 Vegetation Surveys), and only two stations were located in the agriculture habitat type, thus it was under-sampled.

Change in density among the three annual estimates on Tinian was assessed with repeated measures analysis of variance (ANOVA: PROC MIXED; SAS Institute Inc., Cary, NC). Repeated measures ANOVA was also used to assess change in bird densities within regions among the three annual estimates. To stabilize the error variances, densities by station were log transformed after a constant of 1 was added (to avoid ln(0)). Stations were treated as the random factor, and we assumed the variance-covariance structure was a compound symmetry, homogeneous variance model (Littell *et al*. 1996). The degrees of freedom were adjusted using the Kenward-Roger adjustment statement and a Tukey's adjustment was used to control alpha = 0.05 for multiple-comparison procedures. Differences by habitat for Tinian Monarch from the 2008 survey were compared using a one-way ANOVA (PROC MIXED) with the same options as those used in the repeated measures models. The agriculture habitat was dropped from this analysis because only two stations were sampled within the habitat.

Territory Mapping

Between August 4 and 29, 2008, four study plots were established and sampled to estimate Tinian monarch territory densities (Figure 2). Two study plots were established in areas designated as secondary forest and two plots were established in areas designated as tangantangan thicket. Each plot was situated so that it straddled a minimum of two survey stations on transects sampled during the June 2008 point-transect survey to compare territory density estimates with point-transect survey results. Because of the patchy nature of these forest types and the goal of establishing the plot along an established transect, the locations of each plot were not randomly selected.

Each study plot consisted of a series of cut trails and stations marked at 50 meter increments to assist with territory mapping. Each plot was surveyed in the morning (0600-1200) and afternoon (1500-1800) over several days by teams of surveyors. Initially, efforts were undertaken to survey the entire plot during one morning or afternoon survey station. However, due to the amount of bird activity and size of the study plots it was not feasible to cover the plot adequately using this method. Therefore, each survey team focused on a portion of the study plot and multiple survey teams were often employed to increase survey coverage and to obtain dual observations of territorial pairs.

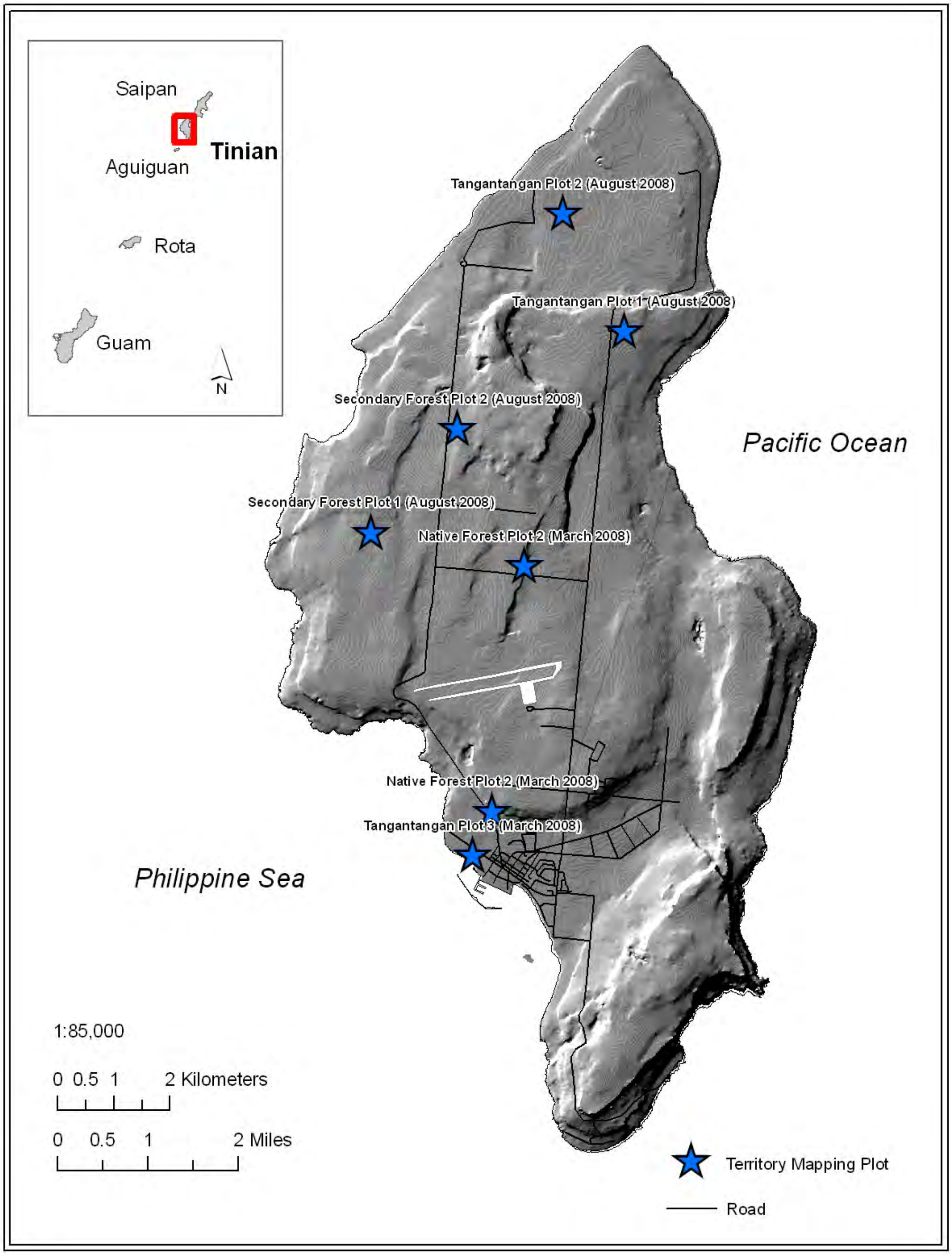


Figure 2. Island of Tinian showing the locations of the territory mapping plots sampled in March and August 2008.

All Tinian monarch detections were marked on a map of the study plot using symbols for movements and activities outlined by Bibby *et al.* (2000). These locations were then transferred to a master map of the study plot created with ArcMap 9.2 (ESRI, Redlands, CA). The locations of all nests found in the study plot were recorded with a Garmin etrex or 76CSx unit (Garmin International, Inc., Olathe, KS) Global Positioning System (GPS) unit and downloaded to ArcMap 9.2. Survey coverage and effort was also tracked using the track function of the GPS units which were downloaded to ArcMap 9.2 daily.

Tinian monarch territory densities were determined by counting the number of Tinian monarch pairs within each study plot and dividing the total number of territories by the size of the plot. Territories which overlapped the edge of the plot were included as half territories (Bibby *et al.* 2000). The presence of a territory was determined through a combination of visual observations of pairs or family groups, active nests, and territorial behaviors (*e.g.*, singing, territorial defense). In addition, simultaneous observations of birds and nesting activity by multiple observers were used whenever possible to confirm the presence of adjacent territories. The boundary of each study plot was defined as the outer east and west transects and ends of each north and south running transect. On average a buffer between 20 and 50 meters around each plot was surveyed based on the GPS tracks.

Territory density estimates for two limestone forest plots and an additional tangantangan thicket plot were obtained from the post-delisting monitoring results obtained in March 2008 (USFWS 2008). Territories in these study plots were delineated based on observations of individually marked birds in each study plot.

RESULTS AND DISCUSSION

Point-Transect Surveys

A total of 361 Tinian monarchs were detected during the June 2008 survey and the population was estimated to be approximately 38,000 individuals (Table 2). This population estimate represents a significant decline from our estimates for the 1982 and 1996 surveys (Table 2). In addition, densities of Tinian monarchs in the Diablo region declined significantly between 1982 and 2008 (Figure 3).

Table 2. Population density (birds/km^2 ± SE, with 95% CI) and abundance (density times the area of Tinian; 101.01 km^2; with 95% CI) estimates for Tinian monarchs from three point transect surveys. The 1982 and 1996 data were reanalyzed using current analysis procedures (see Methods above).

Year	Density	Abundance
1982	634.5 ± 37.88 (564.3–713.4)	60,898 (49,484–75,398)[1]
1996	705.7 ± 43.96 (624.3–797.6)	62,863 (50,476–78,758)[2]
2008	431.3 ± 30.75 (374.9–496.2)	38,449 (29,992–49,849)

[1] 39,338 (35,161–43,515), Engbring *et al.* (1986) – Estimate from original report

[2] 55,721 (48,345–63,495), Lusk *et al.* (1986) – Estimate from original report

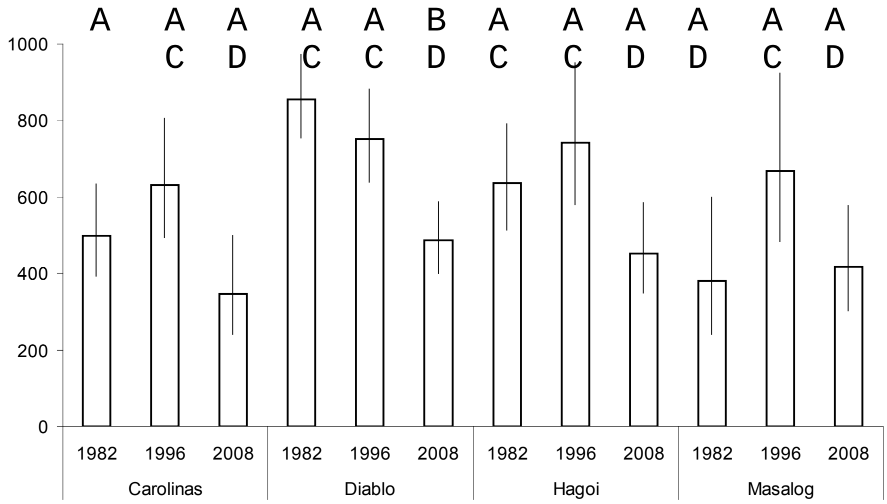


Figure 3. Density estimates (birds/km^2 and 95% CI) for Tinian monarchs by region and year from three point-transect surveys. Differences of least squares means were assessed with repeated measures ANOVA. Comparisons, within region by year, that share the same letter are not significantly different at the 0.05 level, adjusted for multiple comparisons (see 2.4.1 General Land Birds).

Tinian monarchs were recorded in all of the land cover types sampled on Tinian, but their densities were not distributed evenly among these land cover types (Table 3). Based on the 2008 survey, the greatest monarch densities were observed in limestone forest, secondary forest, and tangantangan thicket. The smallest densities were found in open field and urban/residential habitats. Monarch densities in limestone and secondary forests were greater than those in open field and urban/residential, but not different from densities in tangantangan thicket (Table 3).

Table 3. Tinian Monarch density estimates (birds/ha), standard error (SE), and 95% confidence intervals (Lower and Upper 95% CI) by habitat in 2008 based on point-transect sampling.

Habitat	Estimate	SE	L 95% CI	U 95% CI
Agriculture	1.75	1.75	*	*
Limestone Forest	6.41	0.74	5.09	8.05
Open Field	2.83	0.64	1.81	4.44
Secondary Forest	5.82	0.54	4.84	7.01
Tangantangan Thicket	4.36	0.47	3.52	5.39
Urban/Residential	1.50	1.04	0.32	6.94

*Sample size was insufficient to estimate reliable confidence intervals.

Engbring *et al.* (1986) and Lusk *et al*. (2000) both calculated lower Tinian Monarch abundance than our estimates for 1982 and 1996, respectively (Table 2). In addition, the estimate of 35,846 (± 2,211 SE) Tinian monarchs for 1982 by Lusk *et al*. (2000) is also lower than our estimate and the estimate by Engbring *et al.* (1986). These changes are due to differences between the analytical procedures, specifically differences in the model selected and advances within program DISTANCE. For example, Lusk *et al*. (2000) did not extrapolate densities to abundance for 2,375 ha of open fields, although monarchs were detected in this habitat. After dropping densities from the open fields and adjusting for this area difference, our densities resulted in 48,424 birds, an estimate that fell within their 95% CI. This difference is easily accounted for in differences between our methods, specifically differences in the model selected and advances within program DISTANCE. Lusk *et al*. (2000) calculated their density estimate from a half-normal model with polynomial adjustments and an effective detection radius (EDR) of just over 34 m. We estimated the EDR at 30.18 m from a hazard-rate detection function (without adjustments) and incorporating observers as a covariate, where the smaller EDR resulted in greater densities. Lastly, Lusk *et al*. (2000) used program VCPADJ (Fancy 1997) and a previous version of DISTANCE (Laake *et al*. 1994) to standardize the survey conditions and estimate densities. The updated version of DISTANCE (Thomas *et al*. 2006) we used incorporates all of the modeling in one program and uses an improved technique to account for differences in sampling conditions (Thomas *et al*. 2006, Marques *et al*. 2007).

Territory Mapping

A total of 680 person hours was spent territory mapping in the four study plots established in August 2008 which translated to approximately 24 person hours per hectare. Active breeding (*e.g.*, nest building, chick rearing, and family groups) was observed on all plots. A total of 65.5 territories were identified and territory densities ranged from 1.7 to 2.9 territories per hectare (Table 4). When compared with the territory mapping results in March 2008, territory densities were highest in limestone forest followed by secondary forest and then tangantangan thickets. Densities in tangantangan thickets were the most variable with densities overlapping estimates for secondary forest plots (Table 4). The lowest density was found in tangantangan plot 2 which was located off of the northfield runway (Figure 2).

Table 4. Tinian monarch territory density estimates (territories/ha) in limestone forest, secondary forest and tangantangan thickets in 2008 based on territory mapping. See USFWS (2008) for methods used to survey plots in March 2008.

Habitat	Plot	Plot Size (ha)	Number of Territories	Density	Survey Month
Limestone Forest	1	1.91	15	7.8	March
	2	2.42	18	7.5	March
Secondary Forest	1	6.47	18.5	2.9	August
	2	5.72	14	2.5	August
Tangantangan Thicket	1	8.59	21	2.4	August
	2	6.90	12	1.7	August
	3	3.26	8	2.5	March

The results from 2008 were similar to the three study plots surveyed in 1995 (USFWS 1996). That study reported 6.41 (95% CI: 5.09 – 8.05) birds per hectare in limestone forest and 5.82 (95% CI: 4.84 – 7.01) and 4.36 (95% CI: 3.52 – 5.39) birds per hectare in secondary forest and tangantangan thicket, respectively (USFWS 1996). The limestone forest plot in the 1995 study coincides with limestone forest plot 2, also known as the Airport Mitigation Plot, in the 2008 survey while the tangantangan plot in the 1995 study was near tangantangan plot 2 in this study. These results indicate that territory densities may be consistent over time as similar densities were reported in the two survey periods. However, results for different plots in the same habitat and between habitats can be variable.

One potential source of this variability is the disturbance history of each study plot. Tangantangan plot number 2 was located between two of the old runways built during World War II. This area was likely leveled and had all of its topsoil removed as part of runway construction. This site also had the lowest densities of Tinian monarch territories and the lowest recorded nesting activity. The two plots in limestone forest appear to have not been developed before, during, and after the war and had the highest Tinian monarch territory densities. The remaining plots all experienced some level of agricultural or military development based on 1945 aerial photographs. Tangantangan plot 3 is located on the site where a sugar refinery was constructed and destroyed during the war. Secondary forest plot 1 was likely an agricultural field prior to the war but had structures built on it near the end of the war. Both tangantangan plot 1 and secondary forest plot 2 were both agricultural fields prior to the war and were not developed after the war.

Another potential source of variability is the presence of broadleaved trees in the understory. Information from the 1995 Tinian monarch study indicates that nests are typically associated with native trees in the understory (USFWS 1996). Observations made during this survey also indicate that understory composition may be important to Tinian monarch breeding. In addition, habitat sampling along bird survey transects on Tinian by Vogt (2009; see Appendix 3.2) showed a positive correlation between Tinian monarch detections and tree diversity. This study also found that Tinian monarch detections were higher in mixed forest than areas dominated by tangantangan. Unfortunately, due to the nature of the survey and time limitations we were unable to assess this relationship further during the 2008 study. Further work on this subject is warranted.

The point-transect survey included the entire island and sampled a wide range of habitats. Therefore, the variability in density estimates in study plots should be incorporated within the point-transect estimates. To compare the two estimates we multiplied the territory density in each plot by two to estimate the minimum number of birds per hectare in the plot. We then compared these estimates to the point-transect density estimates and found that the confidence intervals for the point-transect estimates included the density estimates for the tangantangan and secondary forest plots (Table 3). This helps confirm the robustness of the point-transect estimates in these habitat types. However, the density estimates for the study plots were well outside the confidence intervals of the point-transect estimate for limestone forest. Both plots were located in thin stretches of native forest so Tinian monarchs may be more concentrated in these areas. To assess if the width of the native forest was a factor we classified all stations in native forest as either thin (≤ 300 meters wide, n = 30) or wide (>300 meters wide, n = 24) native forest stations. We then compared detections for each station type and found that Tinian monarch detections were significantly higher on stations in thin stretches of native forest (2.50 ± 0.26) than stations in wide stretches of native forest (0.83 ± 0.26) (two sample T-test, $T = -5.06$, $P<0.001$, $df = 51$). Therefore, Tinian monarch densities may be higher in thinner stretches of native forest which may account for the higher densities in our territory mapping plots. However, it should be noted that the wide stretches of native forest that were sampled were all along the southeast coast of the island (transect 13 and 14 and the southern end of transect 8; Figure 1). In general, the terrain in this region was very rugged and the habitat quality in this area may differ from the other patches of native forest on the island. Therefore, further work is needed to determine why densities may differ among the patches of native forest.

Conclusions and Management Recommendations

The Tinian monarch population currently consists of approximately 40,000 individuals and has experienced a population decline since 1982 and 1996. The cause of the Tinian monarch population decline is uncertain. Tinian monarchs are primarily associated with forested habitats and the availability of these habitats have increased or remained stable since 1982 (see 2.1 Vegetation Surveys). In addition, other forest bird species, like the rufous fantail, have increased since 1982 (see General Land Birds). Therefore, gross changes in forest cover seem an unlikely cause of the decline. The quality of the forested habitat may have changed, for example we noted heavy cover on invasive vines in our secondary forest study plots, which may have reduced density of monarchs these areas could sustain. The decline may also be related to non-habitat factors as well. Observations of pox-like lesions on some individuals during color-banding may indicate that disease may be playing a role in the decline. Introduced predators, like rats, could also be impacting Tinian monarch populations.

Though habitat loss does not appear to be major in the decline of Tinian monarch, the loss of important habitat to the species is expected to further endanger this endemic species. Territory mapping and point-transect estimates indicate forested areas, especially native limestone forest, are important to the long-term conservation of this species. However, the value of each habitat type may be dependent on quantity of native trees in the understory. Therefore, we recommend the following management and research activities:

1. Avoid impacts to the remaining native limestone forest areas as these provide important habitat to the Tinian monarch and serve as seed sources for native forest restoration in adjacent habitats;
2. Actively restore native forest through planting of native trees in the understory of secondary forest and tangantangan habitats;
3. Convert open fields to native forest or potentially secondary forest;
4. Evaluate Tinian monarch habitat selection in native limestone forest, secondary forest, and tangantangan habitats;
5. Evaluate the potential impact of avian disease and predation on Tinian monarch populations; and
6. Prevent the introduction of brown treesnakes to Tinian.

REFERENCES

Baker, R. H. 1951. The avifauna of Micronesia, its origin, evolution, and distribution. University of Kansas Publications, Museum of Natural History 3:1-359.

Biddy, C., N.D. Burgess, D.A. Hill, and S.H. Mustoe. 2000. Bird census techniques. Academic Press, London, UK.

Buckland, S.T., D.R. Anderson, K.P. Burnham, J.L. Laake, D.L. Borchers, and L. Thomas. 2001. Introduction to Distance Sampling: Estimating abundance of biological populations. Oxford: Oxford University Press.

Camp, R.J., T.K. Pratt, F. Amidon, A.P. Marshall, S. Kremer, and M. Laut. 2009. Draft status and trends of the land bird avifauna on Tinian and Aguiguan, Mariana Islands. Appendix 3.1 *in* Terrestrial Resource Surveys of Tinian and Aguiguan, Mariana Islands, 2008. Working Draft. U.S. Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office, Honolulu. HI

Engbring, J., F. Ramsey, and V. J. Wildman. 1986. Micronesian Forest Bird Survey, 1982: Saipan, Tinian, Agiguan, and Rota. U.S. Fish and Wildlife Service, Honolulu, Hawaii. 143 pp.

Falanruw, M.C., T.G. Cole, and A.H. Ambacker. 1989. Vegetation survey of Rota, Tinian, and Saipan, Commonwealth of the Northern Mariana Islands. USDA Forest Service Resource Bulletin PSW-27. 11 pp.

Fancy, S.G. 1997. A new approach for analyzing bird densities from variable circular-plot counts. Pacific Science 50:107-114.

Johnson, L., R.J. Camp, K.W. Brinck, and P.C. Banko. 2006. Long-term population monitoring: lessons learned from an endangered passerine in Hawai`i. Wildlife Society Bulletin 34:1055-1063.

Laake, J.L., S.T. Buckland, D.R. Anderson, and K.P. Burnham. 1994. DISTANCE user’s guide V2.1. Colorado Cooperative Fish and Wildlife Research Unit, Colorado Statue University, Fort Collins, CO.

Littell, R.C., G.A. Milliken, W.W. Stroup, and R.D. Wolfinger. 1996. SAS system for mixed models. SAS Institute Inc., Cary, NC.

Lusk, M., S. Hess, M. Reynolds, and S. Johnston. 2000. Population status of the Tinian Monarch (*Monarcha takatsukasae*) on Tinian, Commonwealth of the Northern Mariana Islands. Micronesica 32:181-190.

Marques, T.A., L. Thomas, S.G. Fancy, and S.T. Buckland. 2007. Improving estimates of bird density using multiple-covariate distance sampling. Auk 124:1229-1243.

Peters, D.S. 1996. *Monarcha takatsukasae* (Yamashina 1931) - ein Nachweis von Saipan (Aves: Monarchidae). Senckenbergiana Biologica 76:15-17.

Thomas, L., J.L. Laake, S. Strindberg, F.F.C. Marques, S.T. Buckland, D.L. Borchers, D.R. Anderson, K.P. Burnham, S.L. Hedley, J.H. Pollard, J.R.B. Bishop, and T.A. Marques. 2006. Distance 5.0. Release 2. Research Unit for Wildlife Population Assessment, University of St. Andrews, U.K. http://www.ruwpa.st-and.ac.uk/distance/.

U.S. Fish and Wildlife Service. 1996. Wildlife research report for Navy-leased lands on the island of Tinian, CNMI. Prepared for Department of the Navy, PACNAVFACENGCOM, Pearl Harbor, Hawaii. 100 pp.

U.S. Fish and Wildlife Service. 2005. Post-delisting monitoring plan for the Tinian Monarch, *Monarcha takatsukasae*. Endangered Species Division, Pacific Islands Fish and Wildlife Office, Honolulu, HI.

U.S. Fish and Wildlife Service. 2008. Tinian Monarch Post-delisting Monitoring: Progress Report and Trip Report for 27 February-9 March 2008. Endangered Species Division, Pacific Islands Fish and Wildlife Office, Honolulu, HI.

World Conservation Union (IUCN). 2008. 2008 IUCN Red List of Threatened Species. Available online at <http://www.redlist.org> (accessed 10 November 2008).

2.5 MAMMAL SURVEYS

2.5.1 RODENTS ON TINIAN

Suncus murinus. Photo by Curt Kessler.

Prepared by: **Andrew Wiewel, Amy Yackel Adams, and Gordon Rodda, U.S. Geological Survey, Biological Resources Discipline, Fort Collins Research Station, Fort Collins, CO.** (Summarized from Wiewal *et al.* 2008 (Appendix 3.3) by Curt Kessler, U.S. Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office, Honolulu, HI)

INTRODUCTION

The following report is a synopsis of a much larger report entitled: Systematic Rodent Monitoring; A Study of the Introduced Small Mammals of the Mariana Islands, by Wiewel *et. al.* 2008. Only those findings pertanent to the island of Tinian are presented here. Please reference the full document for further explanations of methods and results for the islands of Tinian, Saipan, Rota, and Guam (Appendix 3.3).

Introduced small mammals often have detrimental effects on island ecology (Atkinson 1985, Towns *et al.* 2006). Direct effects of introduced small mammals include competition with, or predation on, various amphibian (Worthy 1987, Towns and Daugherty 1994), avian (Fisher and Baldwin 1946, Wirtz 1972, Recher and Clark 1974, Atkinson 1977, Martin *et al.* 2000, Smith *et al.* 2006), invertebrate (Bremner *et al.* 1984; Kuschel and Worthy 1996; Brook 1999, 2000; Carlton and Hodder 2003; Priddel *et al.* 2003), mammalian (Daniel 1990, Goodman 1995, Pascal *et al.* 2005), and reptilian species (Whitaker 1973; Newman 1994; Towns 1994; Towns and Daugherty 1994; Cree *et al.* 1995; Hoare *et al.* 2007a,b), often resulting in population declines or even extirpation. Introduced small mammals may also suppress plant recruitment by consuming bark, flowers, foliage, fruits, seeds, or seedlings (Allen *et al.* 1994; Campbell and Atkinson 1999, 2002; McConkey *et al.* 2003; Wilson *et al.* 2003); in extreme cases this recruitment suppression can result in local extirpation (Campbell and Atkinson 1999, 2002). Less apparent but equally important indirect effects include disruption of island trophic systems (Fritts and Rodda 1998, Towns 1999) and nutrient cycling (Fukami *et al.* 2006), modification of vegetative community structure and successional patterns

(Campbell and Atkinson 1999, 2002; Athens *et al.* 2002), and creation of novel vectors and reservoirs for diseases and parasites of both animals (Pickering and Norris 1996, Martina *et al.* 2006) and humans (Chanteau *et al.* 1998, Lindo *et al.* 2002, Bitam *et al.* 2006, Jiang *et al.* 2006). However, our understanding of these effects is limited by incomplete knowledge of small mammal distribution, density, and biomass on many islands. Such information is especially critical in the Mariana Islands, where introduced small mammals are keystone prey for the introduced brown treesnake (*Boiga irregularis*) and small mammal density is inversely related to the effectiveness of brown treesnake control and management tools, such as mouse-attractant traps.

In an effort to address these concerns, we deployed mark-recapture livetrapping methodology to determine introduced small mammal distribution, density, and biomass at 3 sites on Tinian. We sampled one site each of grassland, *Leucaena* forest, and native limestone forest. In addition, we conducted snaptrapping at these sites following livetrapping, which allowed direct comparison between these sampling methods as well as estimates in indices generated from them. Livetrapping and snaptrapping occurred between April 2005 and June 2007.

Based on a review of available data the introduced small mammal community of the Mariana Islands consists of 5 or 6 species (with possible additional subspecies), ranging from the earliest introduction, *Rattus exulans*, which occurred no later than A.D. 1000–1200 (Steadman 1999) to the most recent introduction, *Suncus murinus*, first captured on Guam in 1953 (Peterson 1956). Later introductions include *Mus musculus*, first reported on Guam in 1819 (Freycinet 2003), and *R. norvegicus*, first reported on Saipan in the late 1800's (Kuroda 1938 cited by Wiles *et al.* 1990). Regarding the polytypic species *M. musculus*, it is not clear which, or how many, subspecies (*M. m. musculus*, *M. m. domesticus*, or *M. m. castaneus*; Musser and Carleton 2005) have been introduced. It is notable that Prager *et al.* (1998) found *M. m. castaneus* on Tinian, although this identification was based on genetic analysis of a single speciment. Two additional species, *R. rattus* and *R. tanezumi*, have been documented in the Mariana Islands (Baker 1946, Johnson 1962, Yosida *et al.* 1985), although their current status is unclear. The complex taxonomic history of these closely related species (Musser and Carleton 2005), which were only recently separated based on karyotypic differences (*R. rattus*: 2n = 38; *R. tanezumi*: 2n = 42) as well as biochemical and morphological features (Schwabe 1979, Baverstock *et al.* 1983), complicates the investigation of historic introductions and current distribution. Additional confusion arises from the limited hybridization observed in both laboratory (Yosida *et al.* 1971) and wild (Baverstock *et al.* 1983) populations, which led Baverstock *et al.* (1983:978) to conclude that *R. rattus* and *R. tanezumi* "…are best considered as incipient species. Where they meet, they may introgress, become sympatric without interbreeding, or one may replace the other depending upon the prevailing biological conditions."

METHODS

For a complete description of the study site selection and small mammal sampling protocols used during this research (described below), please refer to Wiewel (2005).

Study Site Selection

We sampled 3 sites on Tinian (Figure 1) between April 2005 and June 2007 (Table 1). Study sites were identified using a 1:25,000 scale topographical map (U.S. Geological Survey 1999) and 1:20,000 scale vegetation maps (Falanruw *et al.* 1989). Sites were evaluated based on habitat type, available area of relatively homogeneous habitat, and land ownership status. Selected sites represented the 3 major habitat types of the southern Mariana Islands: native limestone forest, grassland, and *L. leucocephala*-dominated secondary forest.

Table 1. Introduced small mammal sampling site coordinates and dates on Tinian, 2005–2007. Coordinates indicate the site centroid, and are presented in decimal degrees (WGS 84, UTM Zone 55 North).

Site	Habitat	Dates Sampled	Latitude	Longitude
KAST	grassland	Oct 24–28, 2005	14.951	145.651
ABLE	*Leucaena* forest	Nov 7–11, 2005	15.076	145.640
LSUS	native forest	Oct 31–Nov 4, 2005	15.043	145.629

Small Mammal Sampling

Due to the uncertainty surrounding the status of *R. rattus* and *R. tanezumi* in the Mariana Islands, we collected genetic material from all captured *Rattus* to allow determination of species identification and distribution. Until samples are processed, however, we will use the more recognized term *R. rattus* to refer to the combined sample of unidentified *Rattus* species.

At each site, mark-recapture livetrapping was conducted for 5 consecutive nights on an 11 × 11 grid with 12.5 m intervals between each trap station (grid area = 1.56 ha). A single standard-length folding Sherman live trap (229 × 89 × 76 mm; H.B. Sherman Traps, Inc., Tallahassee, FL) was placed at each trap station (*n* = 121) and a single Haguruma wire mesh live trap (approximately 285 × 210 × 140 mm; Standard Trading Co., Honolulu, HI) was placed at every other trap station (*n* = 36). Closed traps were placed on the grid a minimum of 2 nights prior to the beginning of sampling to provide an opportunity for small mammals to acclimate to their presence. Traps were placed on the ground and, whenever possible, located next to or beneath clumps of grass, downed woody debris, or rocks to provide shelter from sun and rain. Traps were baited with a mixture of peanut butter, oats, and food-grade paraffin (Wiewel 2004) and were checked beginning at 0730–0800 each day. Traps were closed during the day to minimize trap mortality. Traps were reopened at approximately 1600 and rebaited as necessary to ensure bait freshness.

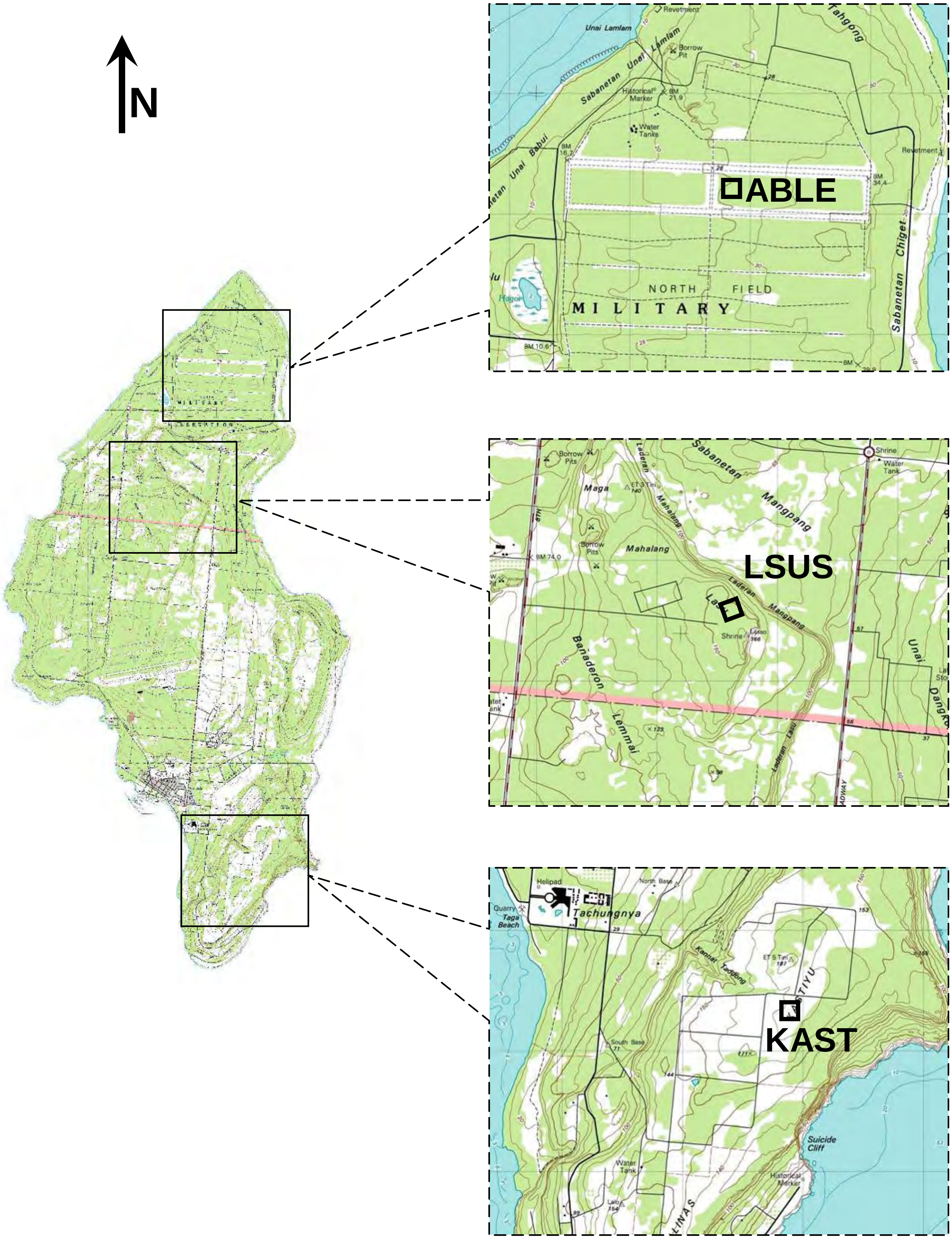


Figure 1. Introduced small mammal sampling locations on Tinian, Mariana Islands, 2005–2007. Sampling grids are delineated with bold squares, which represent an area of 125 m^2 (1.56 ha). See Table 1 for site coordinates, sampling dates, and habitat classifications.

Captured animals were examined and measured to determine species, sex, age, reproductive status, mass (g), head-body length (mm), tail length (mm), right hind foot length (mm), right ear length (mm), and testes length (mm; if applicable). Captured individuals were uniquely marked in each ear with numbered metal ear tags (*M. musculus* and *S. murinus*: small ear tags produced by S. Roestenburg, Riverton, UT; *Rattus* species: #1005-1, National Band and Tag Co., Newport, KY). Recaptured animals were examined to determine tag number. All capture, handling, and marking techniques followed guidelines approved by the American Society of Mammalogists (Gannon *et al.* 2007) and the U.S. Geological Survey Animal Care and Use Committee (Fort Collins Science Center).

Each site was also sampled with 5 consecutive nights of snaptrapping during the week following livetrapping. Results of snaptrapping are described elsewhere; however, data collected during snaptrapping were included in the calculation of both body condition index (a covariate used in mark-recapture abundance estimation) and mean maximum distance moved (used in density estimation).

Data Analysis

We estimated density and biomass separately for each species. First, we generated site-specific estimates of abundance using estimated capture and recapture probability modeled from livetrapping data. Because these estimates had no associated area component, our second step was to estimate the effective trapping area for each site with reference to each species' mean maximum distance moved between captures. Third, we estimated density as abundance/ effective trapping area. Fourth, we determined mean body mass based on measurements of captured animals at each site. Fifth, for each site we estimated biomass as the product of site-specific density and site-specific mean body mass. Finally, we created variance-covariance matrices to separately calculate the variances of density and biomass estimates. For a more detailed explanation of data analysis please refer to the original document (Appendix 3.3).

RESULTS

We captured a total of 241 *Rattus rattus*, 167 *Suncus nurinus*, 9 *Mus musculus*, and 1 *Rattus norvegicus* on Tinian (Table 2). No *Rattus exulans* were captured at the three sites sampled.

Density Estimates

R. rattus mean maximum distance moved on Tinian were 14.5 ± 1.3 m, 95% CI = 11.9–17.1; *n* = 180. When combined with the nominal grid area of 1.56 ha, these mean maximum distance moved estimates resulted in effective trapping areas of 1.95 ha and mean *R. rattus* density estimates of 73.0/ha (*n* = 3) on Tinian (Table 3).

S. murinus mean maximum distance moved varied primarily between habitats, and was greatest in grassland (29.2 ± 2.7 m, 95% CI = 23.7–34.7; *n* = 48), followed by mixed habitat (19.3 ± 3.2 m, 95% CI = 12.7–25.9; *n* = 25), *Leucaena* forest (16.3 ± 1.4 m, 95% CI = 13.6–19.0; *n* = 68), and native forest (14.2 ± 3.5 m, 95% CI = 6.4–22.0; *n* = 12).

Mean estimated density was 52.8/ha ($n = 2$) in *Leucaena* forest, 24.2/ha ($n = 2$) in native forest, 20.2/ha ($n = 2$) in mixed habitat, and 9.7/ha ($n = 2$) in grassland (Table 3).

Table 2. *Mus musculus*, *Rattus exulans*, *R. norvegicus*, *R. rattus*, and *Suncus murinus* individuals captured (M_{t+1}) and total captures ($n_{.}$) during mark-recapture livetrapping in grassland, *Leucaena* forest, mixed, and native forest habitats on Tinian, 2005–2007. Blank entries indicate zero captures.

		M. musculus		*R. exulans*		*R. norvegicus*		*R. rattus*		*S. murinus*	
Site	Habitat	M_{t+1}	$n_{.}$	M_{t+1}	$n_{.}$	M_{t+1}	$n_{.}$	M_{t+1}	$n_{.}$	M_{t+1}	$n_{.}$
KAST	grassland	9	12			1	1	106	132	11	11
ABLE	*Leucaena* forest							55	81	93	113
LSUS	native forest							80	92	43	43

Table 3. *Rattus rattus*, *Suncus murinus*, and *Mus musculus* density estimates ($\hat{D}$; animals/ha), standard errors (SE), and 95% confidence intervals (95% CI) in grassland, *Leucaena* forest, mixed, and native forest habitats on Tinian, 2005–2007. Blank entries indicate zero captures, and therefore zero estimated density.

		R. rattus			*S. murinus*			*M. musculus*		
Site	Habitat	$\hat{D}$	SE	95% CI	$\hat{D}$	SE	95% CI	$\hat{D}$	SE	95% CI
Tinian										
KAST	grassland	99.9	17.9	64.8–135.0	8.9	2.5	4.0–13.8	8.2	2.7	2.9–13.5
ABLE	*Leucaena* forest	44.0	7.3	29.7–58.3	73.7	20.1	34.3–113.1			
LSUS	native forest	75.1	13.6	48.4–101.8	32.8	9.6	14.0–51.6			

M. musculus mean maximum distance moved on Tinian were 11.7 ± 8.7 m, 95% CI = 0–28.0; $n = 3$. When combined with the nominal grid area of 1.56 ha, these mean maximum distance moved estimates resulted in effective trapping areas of 1.87 ha for Tinian. These model-averaged effective trapping areas produced mean *M. musculus* density estimates of 2.6/ha ($n = 3$) on Tinian (Table 3).

Biomass Estimates

R. rattus, *S. murinus*, and *M. musculus* varied dramatically in morphology, with mean *R. rattus* mass being much greater (121.9 ± 1.8 g, 95% CI = 118.3–125.5; $n = 707$) than mean *S. murinus* mass (25.7 ± 0.4 g, 95% CI = 25.0–26.5; $n = 298$) or mean *M. musculus* mass (12.5 ± 0.2 g, 95% CI = 12.1–12.9; $n = 154$). *R. rattus* biomass was markedly greater than *S. murinus* or *M. musculus* biomass across sampled habitats (Table 4). Mean estimated *R. rattus* biomass was 11.6 kg/ha. The mean estimated *S. murinus* biomass was 1.9 kg/ha. When evaluating biomass across habitats on Tinian, mean *R. rattus* biomass was greatest in grassland, with a maximum estimate of 11.6 kg/ha in this habitat. In other habitats, mean estimated *R. rattus* biomass was roughly half that estimated for grassland, although maximum biomass estimates exceeded 8 kg/ha in both mixed habitat and native forest. In contrast to *R. rattus*, mean estimated *S. murinus* biomass was lowest in grassland and highest in *Leucaena* forest on Tinian, with a maximum estimate of 1.9 kg/ha in this habitat. Mean estimated *M. musculus* biomass was greatest in grassland on Tinian, with a maximum estimate of 0.4 kg/ha in this habitat.

Table 4. *Rattus rattus*, *Suncus murinus*, and *Mus musculus* biomass estimates ($\hat{Biom}$; kg/ha), standard errors (SE), and 95% confidence intervals (95% CI) in grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Rota, Saipan, and Tinian, 2005–2007. Blank entries indicate zero captures, and therefore zero estimated biomass.

		R. rattus			*S. murinus*			*M. musculus*		
Site	Habitat	$\hat{Biom}$	SE	95% CI	$\hat{Biom}$	SE	95% CI	$\hat{Biom}$	SE	95% CI
Tinian										
KAST	grassland	11.57	2.11	7.43–15.71	0.16	0.05	0.06–0.26	0.11	0.04	0.03–0.19
ABLE	*Leucaena* forest	5.09	0.88	3.37–6.81	1.87	0.52	0.85–2.89			
LSUS	native forest	8.78	1.63	5.59–11.97	0.83	0.25	0.34–1.32			

DISCUSSION

Three species, *R. rattus*, *S. murinus*, and *M. musculus*, were commonly captured during this study. Two additional species, *R. exulans* and *R. norvegicus*, were captured infrequently and in very low numbers (Table 2). Of these species, *R. rattus* attains the greatest density and biomass in grasslands on Tinian when compared to the other southern limestone islands of Guam (2.42, SE = 0.58), Rota (9.80, SE = 1.62), and Saipan (4.13, SE = 0.83) (Wiewel *et al.* 2008). Maximum estimates of *R. rattus* density on Tinian are 2–3 times greater than the highest known historic values from Guam and also greater than estimates from other tropical Pacific islands, including Pohnpei (4.0–8.5/ha; Strecker 1962), Majuro (11.3/ha; Strecker 1962), Eniwetok (19.9/ha; Jackson 1967), and the Galapagos (0.2–18.9/ha; Clark 1980). Indeed, the peak densities observed during this study are suggestive of population irruptions. Conversely, the fact that high density *R. rattus* populations were observed across habitats, islands, and time is not indicative of an irruptive event, and instead suggests that high density *R. rattus* populations may be fairly common to Tinian. Comparable (and even higher) densities have been recorded for *R. exulans* on small relatively competitor- and predator-free islands. On Kure Atoll, Wirtz (1972) documented a mean *R. exulans* density of 111.2/ha during monthly sampling from March 1964 to May 1965, with monthly estimates ranging from 49.4/ha to 185.3/ha. Similarly, on Tititiri Matangi Island, New Zealand, Moller and Craig (1987) estimated peak *R. exulans* densities of 130 ± 20/ha in grassland and 101 ± 12/ha in forest during regular sampling from February 1975 to May 1977.

S. murinus is generally less common than *R. rattus* in the Mariana Islands. The low mass of *S. murinus* (in relation to *R. rattus*) resulted in *S. murinus* biomass estimates that were only 1–37% of the estimated *R. rattus* biomass for the same site. Overall, *S. murinus* density exceeded 30/ha. On Tinian, *S. murinus* density and biomass were greater in forest than grassland, with the highest values occurring in *Leucaena* forest. In general, our estimates of *S. murinus* density are comparable to historic values from Guam (25.4/ha, Barbehenn 1969, 1974; 19.1/ha, Savidge 1986) and more recent estimates from Saipan (16.7–27.3/ha, S. Vogt unpublished data). Our estimates are also similar to values obtained for the islands of Ile aux Aigrettes (29.2/ha) and Ile de la Passe (20/ha), located off the coast of Mauritius in the Indian Ocean (Varnham *et al.* 2002). However, our maximum estimated *S. murinus* density of 73.7/ha greatly exceeds known values, and could indicate an irruptive potential for this species in the Mariana Islands.

M. musculus is a relatively minor component of the introduced small mammal community from a biomass standpoint, with estimates ranging from 0.01–0.45 kg/ha. However, *M. musculus* capture probability may have been negatively influenced by *R. rattus* activity (Brown *et al.* 1996, Weihong *et al.* 1999). To investigate this possibility, we added site-specific *R. rattus* density to the top *M. musculus* model in a post-hoc MARK analysis. As anticipated, *R. rattus* density had a negative effect on *M. musculus* capture probability ($\beta = -0.008 \pm 0.006$, 95% CI = -0.019–0.003), although this effect was weak as demonstrated by the 95% CI that asymmetrically overlapped zero. Nonetheless, the trend of decreasing *M. musculus* capture probability with increasing *R. rattus* density suggests that this relationship warrants further investigation and should be considered during sampling design and data analysis. For example, the use of multiple trap types may decrease the likelihood of capture probability suppression of non-dominant species (Brown *et al.* 1996, Weihong *et al.* 1999, Gragg 2004). There was an indication of habitat specialization for *M. musculus*, as maximum density and biomass occurred at grassland and mixed habitat sites with patchy vegetative growth and exposed soil. Baker (1946:398) noted a similar preference for "open grass and brush land" and areas where "limestone soils are exposed" on Guam. Similar habitat preferences for this species have been noted for other tropical Pacific islands (Nicholson and Warner 1953, Berry and Jackson 1979).

When interpreting these (and other) density and biomass estimates, it is essential to recognize the potential for temporal variability in introduced small mammal populations. For example, annual sampling at a single site on Guam demonstrated significant temporal variation in *R. rattus* density and biomass, which increased from 2.6/ha and 0.4 kg/ha in 2005 to 15.3/ha and 2.9 kg/ha in 2006. In 2007, 10 days of livetrapping (1570 trap nights) at this site yielded zero captures. Note that this sampling occurred at the same time each year (early May–early June) and therefore represents annual temporal variability. It is also possible that introduced small mammal density and biomass exhibit intra-annual temporal variability in the Mariana Islands. One slight complication is that this site is used for an ongoing, long-term brown treesnake population study (Rodda *et al.* 2007) and is surrounded by a snake- and ungulate-proof fence (i.e., brown treesnakes can not enter or exit and ungulates are excluded), suggesting that the site is not directly comparable with other forested areas on Guam. For example, the exclusion of introduced ungulates has resulted in rapid and dramatic shifts in vegetation structure and composition compared to the surrounding landscape (M. Christy, unpublished data). Nonetheless, the temporal variability in *R. rattus* density and biomass observed at this site suggests that introduced small mammal density and biomass may fluctuate greatly over relatively short time spans in the Mariana Islands. The potential for temporal variability should always be considered when interpreting density and biomass estimates, which are merely a snapshot of a dynamic population.

Implications for Mariana Island Ecology and Brown Treesnake Control and Management

Although little direct evidence currently exists for the Mariana Islands, it seems likely that the high-density introduced small mammal populations documented during this research have negative effects on native fauna and flora, and that introduced species (including small mammals) have modified Mariana Island ecosystems and ecosystem function (Fritts and Rodda 1998). In recent years, researchers have noted apparent declines of several avian species in the Mariana Islands, including the bridled white-eye (*Zosterops rotensis*; Amidon 2000, Fancy and Snetsinger 2001) and Mariana crow (*Corvus kubaryi*; Plentovich *et al.* 2005, U.S. Fish and Wildlife Service 2005) on Rota and the Micronesian megapode (*Megapodius laperouse*) and Mariana fruit dove (*Ptilinopus roseicapilla*) on Saipan (Craig 1999). Numerous hypotheses, including predation by introduced species (e.g., *Rattus*, black drongos, and feral cats), avian diseases or parasites, pesticides, and habitat degradation associated with land-use changes or typhoon damage, have been considered (Craig 1999, Amidon 2000, Fancy and Snetsinger 2001, Plentovich *et al.* 2005, U.S. Fish and Wildlife Service 2005, Ha *et al.* in prep). While predation by black drongos, diseases, and pesticides have largely been ruled out and habitat degradation is increasingly seen as an important factor in avian declines (e.g., Fancy and Snetsinger 2001, Ha *et al.* in prep), the role of introduced small mammals remains unclear. Predation by introduced *Rattus* species is often rejected as a cause of recent avian declines because ≥1 *Rattus* species have been present in the Mariana Islands for at least 1000 years. However, this rejection does not account for differential effects of various *Rattus* species on birds (Atkinson 1985, Thibault *et al.* 2002, Towns *et al.* 2006), as *R. exulans* (the earliest introduction to the Mariana Islands; Steadman 1999) is generally considered least detrimental to avian species. Perhaps more importantly, the potential impact of *R. rattus* or *R. tanezumi* on avian species is unknown, and the uncertainty surrounding the status and distribution of *R. diardii*, *R. rattus*, and *R. tanezumi* in the Mariana Islands further complicates matters. Further, temporal shifts in the presence or abundance of *Rattus* species may obscure their role in avian declines. High-density introduced small mammal populations on Rota, Saipan, and Tinian might also impact avian species through dietary competition, especially during the dry season when certain food items may become scarce. Food competition for invertebrate and reptile foods could be especially problematic for nesting birds, as these high protein prey items are required for nestlings.

Predation by introduced small mammals may also have direct negative effects on invertebrate or reptile populations in the Mariana Islands. Although *Rattus* species are often implicated in invertebrate and reptilian declines (Whitaker 1973; Bremner *et al.* 1984; Cree *et al.* 1995; Priddel *et al.* 2003; Hoare *et al.* 2007a,b), the insectivorous *S. murinus* may be more problematic for these taxa in the Mariana Islands. *S. murinus* has been implicated in the decline of native invertebrates and reptiles on Mauritius and nearby islands (Varnham *et al.* 2002). On Guam, Barbehenn (1974) commented that no skinks were observed during hundreds of hours of small mammal trapping during the peak of the *S. murinus* irruption in the early 1960's, which contrasts with the current abundance and visibility of skinks on Guam. More recently, Fritts and Rodda (1998) noted large differences in mean skink density between Saipan, where *S. murinus* was common (2200 skinks/ha), areas on Guam with few *S. murinus* (8850 skinks/ha), and areas on Guam where both *S. murinus* and brown treesnakes were excluded (13,200 skinks/ha). Similarly, Rodda and Fritts (1992) implicated *S. murinus* in the decline of the pelagic gecko (*Nactus pelagicus*), when they found that this gecko was common on Rota, where *S. murinus* was absent, but highly localized (Guam) or rare or possibly extinct (Saipan and Tinian) on islands with high past or current *S. murinus* populations.

Recent research suggests that introduced small mammals have important impacts on the effectiveness of brown treesnake control efforts, which are highly dependent on traps using live, domestic mice (*M. musculus*) as attractants. These traps are placed around ports, airports, and other cargo-handling facilities on Guam, as well as in locations vulnerable to accidental brown treesnake introductions, such as Rota, Saipan, and Tinian. Mouse-attractant traps are also commonly deployed during the response to snake sightings in brown treesnake-free locations. However, research conducted on Guam suggests that brown treesnake trap capture rates are inversely related to introduced small mammal density. For example, Rodda *et al.* (2001) found a strong correlation ($r^2 = 0.90$) between brown treesnake trap capture rates and indices of small mammal density and documented a 7-fold increase in brown treesnake capture rates in areas of very low small mammal density on Guam. Similarly, Gragg *et al.* (2007) documented a 22–65% increase in brown treesnake trap capture probability after reducing rodent populations with localized rodenticide application. These findings suggest reduced effectiveness of mouse-attractant traps on Tinian, Rota, and Saipan. Further, the majority of brown treesnake control and eradication tools currently being developed and evaluated, such as various acetaminophen delivery devices (Savarie *et al.* 2001), also rely on mouse-based attractants and will likely be subject to the same reduction in effectiveness in areas of high introduced small mammal density.

A second, though perhaps less obvious, effect of introduced small mammals on brown treesnake control and management relates to their impact on island trophic systems and predator-prey relationships. On Guam, introduced prey species, including small mammals, skinks, and geckos, were abundant and widespread at the time of brown treesnake introduction following World War II (Baker 1946, Fritts and Rodda 1998). Because these introduced prey species evolved with various predators, they were better able to persist under brown treesnake predation than the predator-naïve native species of Guam. In so doing, introduced prey species supported a high-density brown treesnake population, even as native avian and reptilian species declined. By the time brown treesnake predation pressure began to reduce introduced prey densities and brown treesnake density also began to decline because of food limitations, many native species were already extinct. Unfortunately, the high introduced small mammal density and biomass documented on the island of Tinian during this research suggests that a similar scenario could develop on this island should a brown treesnake population become established.

REFERENCES

Allen, R.B., W.G. Lee, and B.D. Rance. 1994. Regeneration in indigenous forest after eradication of Norway rats, Breaksea Island, New Zealand. New Zealand Journal of Botany 32:429–439.

Amidon, F.A. 2000. Habitat relationships and life history of the Rota bridled white-eye (*Zosterops rotensis*). M.S. Thesis, Virginia Polytechnic Institute and State University, Blacksburg, Virginia, USA.

Athens, J.S., H.D. Tuggle, J.V. Ward, and D.J. Welch. 2002. Avifaunal extinctions, vegetation change, and Polynesian impacts in prehistoric Hawai'i. Archaeology in Oceania 37:57–78.

Atkinson, I.A.E. 1977. A reassessment of factors, particularly *Rattus rattus* L., that influenced the decline of endemic forest birds in the Hawaiian Islands. Pacific Science 31:109–133.

Atkinson, I.A.E. 1985. The spread of commensal species of *Rattus* to oceanic islands and their effects on island avifaunas. Pages 35–81 *in* P.J. Moors, editor. Conservation of island birds. International Council for Bird Preservation, Technical Publication No. 3, Cambridge, England.

Baker, R.H. 1946. A study of rodent populations on Guam, Mariana Islands. Ecological Monographs 16:393–408.

Barbehenn, K.R. 1969. Responses of rodents and shrews to patterns of removal trapping. Pages 247–252 *in* D.W. Parrack, editor. Indian rodent symposium. Calcutta, India.

Barbehenn, K.R. 1974. Recent invasions of Micronesia by small mammals. Micronesica 10:41–50.

Baverstock, P.R., M. Adams, L.R. Maxson, and T.H. Yosida. 1983. Genetic differentiation among karyotypic forms of the black rat, *Rattus rattus*. Genetics 105:969-983.

Berry, R.J. and W.B. Jackson. 1979. House mice on Enewetak Atoll. Journal of Mammalogy 60:222–225.

Bitam, I, B. Baziz, J.-M. Rolain, M. Belkaid, and D. Raoult. 2006. Zoonotic focus of plague, Algeria. Emerging Infectious Diseases 12:1975–1977.

Bremner, A.G., C.F. Butcher, and G.B. Patterson. 1984. The density of indigenous invertebrates on three islands in Breaksea Sound, Fiordland, in relation to the distribution of introduced mammals. Journal of the Royal Society of New Zealand 14:379–386.

Brook, F.J. 1999. Changes in the landsnail fauna of Lady Alice Island, northeastern New Zealand. Journal of the Royal Society of New Zealand 29:135–157.

Brook, F.J. 2000. Prehistoric predation of the landsnail *Placostylus ambagiosus* Suter (Stylommatophora: Bulimulidae), and evidence for the timing of establishment of rats in northernmost New Zealand. Journal of the Royal Society of New Zealand 30:227–241.

Brown, K.P., H. Moller, J. Innes, and N. Alterio. 1996. Calibration of tunnel tracking rates to estimate relative abundance of ship rats (*Rattus rattus*) and mice (*Mus musculus*) in a New Zealand forest. New Zealand Journal of Ecology 20:271–275.

Campbell, D.J. and I.A.E. Atkinson. 1999. Effects of kiore (*Rattus exulans* Peale) on recruitment of indigenous coastal trees on northern offshore islands of New Zealand. Journal of the Royal Society of New Zealand 29:265–290.

Campbell, D.J. and I.A.E. Atkinson. 2002. Depression of tree recruitment by the Pacific rat (*Rattus exulans* Peale) on New Zealand's northern offshore islands. Biological Conservation 107:19–35.

Carlton, J.T. and J. Hodder. 2003. Maritime mammals: terrestrial mammals as consumers in marine intertidal communities. Marine Ecology Progress Series 256:271–286.

Chanteau, S., L. Ratsifasoamanana, B. Rasoamanana, L. Rahalison, J. Randriambelosoa, J. Roux, and D. Rabeson. 1998. Plague, a reemerging disease in Madagascar. Emerging Infectious Diseases 4:101–104.

Clark, D.B. 1980. Population ecology of *Rattus rattus* across a desert-montane forest gradient in the Galapagos Islands. Ecology 61:1422–1433.

Craig, R.J. 1999. Conservation of endangered white-eyes (Zosteropidae) in the tropical Pacific. Bird Conservation Research, Inc. Contribution No. 1:1–8.

Cree, A., C.H. Daugherty, and J.M. Hay. 1995. Reproduction of a rare New Zealand reptile, the tuatara, *Sphenodon punctatus*, on rat-free and rat-inhabited islands. Conservation Biology 9:373–383.

Daniel, M.J. 1990. Greater short-tailed bat. Pages 131–136 *in* C.M. King, editor. The handbook of New Zealand mammals. Oxford University Press, Auckland, New Zealand.

Falanruw, M.V.C., T.G. Cole, and A.H. Ambacher. 1989. Vegetation survey of Rota, Tinian, and Saipan, Commonwealth of the Northern Mariana Islands. Resource

Bulletin PSW-27. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, California, USA.

Fancy, S.G. and T.J. Snetsinger. 2001. What caused the population decline of the bridled white-eye on Rota, Mariana Islands? Pages 274–280 *in* J.M. Scott, S. Conant, and C. van Riper, III, editors. Evolution, ecology, conservation, and management of Hawaiian birds: a vanishing avifauna. Studies in Avian Biology No. 22, Cooper Ornithological Society.

Fisher, H.I. and P.H. Baldwin. 1946. War and the birds of Midway Atoll. Condor 48:3–15.

Freycinet, L.C.d. 2003. An account of the Corvette L' Uranie's sojourn at the Mariana Islands, 1819; supplemented with the Journal of Rose de Freycinet. Occasional Historic Papers No. 13, CNMI Division of Historic Preservation, Saipan, Commonwealth of the Northern Mariana Islands.

Fritts, T.H. and G.H. Rodda. 1998. The role of introduced species in the degradation of island ecosystems: a case history of Guam. Annual Review of Ecology and Systematics 29:113–140.

Fukami, T., D.A. Wardle, P.J. Bellingham, C.P.H. Mulder, D.R. Towns, G.W. Yeates, K.I. Bonner, M.S. Durrett, M.N. Grant-Hoffman, and W.M. Williamson. 2006. Above- and below-ground impacts of introduced predators in seabird-dominated island ecosystems. Ecology Letters 9:1299–1307.

Gannon, W.L., R.S. Sikes, and the Animal Care and Use Committee of the American Society of Mammalogists. 2007. Guidelines of the American Society of Mammalogists for the use of wild mammals in research. Journal of Mammalogy 88:809–823.

Goodman, S.M. 1995. *Rattus* on Madagascar and the dilemma of protecting the endemic rodent fauna. Conservation Biology 9:450–453.

Gragg, J.E. 2004. Rodent reduction for enhanced control of brown treesnakes (*Boiga irregularis*). M.S. Thesis, Colorado State University, Fort Collins, Colorado, USA.

Gragg, J.E., G.H. Rodda, J.A. Savidge, G.C. White, K. Dean-Bradley, and A.R. Ellingson. 2007. Response of brown treesnakes to reduction of their rodent prey. Journal of Wildlife Management 71:2311–2317.

Ha, J.C., J. Buckley, and R.R. Ha. In Prep. Forest bird abundances and the potential for typhoon impact on bird populations on the island of Rota, Northern Mariana Islands.

Hoare, J.M., L.K. Adams, L.S. Bull, and D.R. Towns. 2007a. Attempting to manage complex predator-prey interactions fails to avert imminent extinction of a threatened New Zealand skink population. Journal of Wildlife Management 71:1576–1584.

Hoare, J.M., S. Pledger, N.J. Nelson, and C.H. Daugherty. 2007b. Avoiding aliens: behavioural plasticity in habitat use enables large, nocturnal geckos to survive Pacific rat invasions. Biological Conservation 136:510–519.

Jackson, W.B. 1967. Productivity in high and low islands with special emphasis to rodent populations. Micronesica 3:5–15.

Jiang, J., D.W. Soeatmadji, K.M. Henry, S. Ratiwayanto, M.J. Bangs, and A.L. Richards. 2006. *Rickettsia felis* in *Xenopsylla cheopis*, Java, Indonesia. Emerging Infectious Diseases 12:1281–1283.

Johnson, D.H. 1962. Rodents and other Micronesian animals collected. Pages 21–38 *in* T.I. Storer, editor. Pacific island rat ecology: report of a study made on Ponape and adjacent islands 1955–1958. Bernice P. Bishop Museum Bulletin 225, Honolulu, Hawaii, USA.

Kuroda, N. 1938. A list of the Japanese mammals. Privately printed, Tokyo, Japan.

Kuschel, G. and T.H. Worthy. 1996. Past distribution of large weevils (Coleoptera: Curculionidae) in the South Island, New Zealand, based on Holocene fossil remains. New Zealand Entomologist 19:15–22.

Lindo, J.F., C. Waugh, J. Hall, C. Cunningham-Myrie, D. Ashley, M.L. Eberhard, J.J. Sullivan, H.S. Bishop, D.G. Robinson, T. Holtz, and R.D. Robinson. 2002. Enzootic *Angiostrongylus cantonensis* in rats and snails after an outbreak of human eosinophilic meningitis, Jamaica. Emerging Infectious Diseases 8:324–326.

Martin, J.-L., J.-C. Thibault, and V. Bretagnolle. 2000. Black rats, island characteristics, and colonial nesting birds in the Mediterranean: consequences of an ancient introduction. Conservation Biology 14:1452–1466.

Martina, B.E.E., G. van Doornum, G.M. Dorrestein, H.G.M. Niesters, K.J. Stittelaar, M.A.B.I. Wolters, H.G.H. van Bolhuis, and A.D.M.E. Osterhaus. 2006. Cowpox virus transmission from rats to monkeys, the Netherlands. Emerging Infectious Diseases 12:1005–1007.

McConkey, K.R., D.R. Drake, H.J. Meehan, and N. Parsons. 2003. Husking stations provide evidence of seed predation by introduced rodents in Tongan rain forests. Biological Conservation 109:221–225.

Metteler, L.S. 1986. Pacific island names: a map and name guide to the new Pacific. Bernice P. Bishop Museum Miscellaneous Publication 34, Honolulu, Hawaii, USA.

Moller, H. and J.L. Craig. 1987. The population ecology of *Rattus exulans* on Tiritiri Matangi Island, and a model of comparative population dynamics in New Zealand. New Zealand Journal of Zoology 14:305–328.

Mueller-Dombois, D. and F.R. Fosberg. 1998. Vegetation of the tropical Pacific islands. Springer, New York, New York, USA.

Musser, G.G. and M.D. Carleton. 2005. Family Muridae. Pages 1189–1531 *in* D.E. Wilson and D.M. Reeder, editors. Mammal species of the world: a taxonomic and geographic reference. Third Edition. Smithsonian Institute Press, Washington, D.C., USA.

Newman, D.G. 1994. Effects of a mouse, *Mus musculus*, eradication programme and habitat change on lizard populations of Mana Island, New Zealand, with special reference to McGregor's skink, *Cyclodina macgregori*. New Zealand Journal of Zoology 21:443–456.

Nicholson, A.J. and D.W. Warner. 1953. The rodents of New Caledonia. Journal of Mammalogy 34:168–179.

Pascal, M., F. Siorat, O. Lorvelec, P. Yésou, and D. Simberloff. 2005. A pleasing consequence of Norway rat eradication: two shrew species recover. Diversity and Distributions 11:193–198.

Peterson, G.D., Jr. 1956. *Suncus murinus*, a recent introduction to Guam. Journal of Mammalogy 37:278–279.

Pickering, J. and C.A. Norris. 1996. New evidence concerning the extinction of the endemic murid *Rattus macleari* from Christmas Island, Indian Ocean. Australian Mammalogy 19:19–25.

Plentovich, S., J.M. Morton, J. Bart, R.J. Camp, M.R. Lusk, N.C. Johnson, and E.A. VanderWerf. 2005. Population trends of Mariana crow (Corvus kubaryi) on Rota, Commonwealth of the Northern Mariana Islands. Bird Conservation International 15:211–224.

Prager, E.M., C. Orrego, and R.D. Sage. 1998. Genetic variation and phylogeography of central Asian and other house mice, including a major new mitochondrial lineage in Yemen. Genetics 150:835-861.

Priddel, D., N. Carlile, M. Humphrey, S. Fellenberg, and D. Hiscox. 2003. Rediscovery of the ‘extinct’ Lord Howe Island stick-insect (*Dryococelus australis* (Montrouzier)) (Phasmatodea) and recommendations for its conservation. Biodiversity and Conservation 12:1391–1403.

Recher, H.F. and S.S. Clark. 1974. A biological survey of Lord Howe Island with recommendations for the conservation of the island's wildlife. Biological Conservation 6:263–273.

Rodda, G.H. and T.H. Fritts. 1992. The impact of the introduction of the brown tree snake, *Boiga irregularis*, on Guam's lizards. Journal of Herpetology 26:166–174.

Rodda, G.H., K. Dean-Bradley, and C.P. Qualls. 2001. The effect of prey density on brown treesnake capture probability. Brown Treesnake 2001: Research & Management. Micronesica 37:181–182.

Rodda, G.H., J.A. Savidge, C.L. Tyrrell, M.T. Christy, and A.R. Ellingson. 2007. Size bias in visual searches and trapping of brown treesnakes on Guam. Journal of Wildlife Management 71:656–661.

Savarie, P.J., J.A. Shivik, G.C. White, J.C. Hurley, and L. Clark. 2001. Use of acetaminophen for large-scale control of brown treesnakes. Journal of Wildlife Management 65:356–365.

Savidge, J.A. 1986. The role of disease and predation in the decline of Guam's avifauna. Dissertation, University of Illinois, Urbana-Champaign, Illinois, USA.

Schwabe, H.W. 1979. Vergleichend-allometrische untersuchungen an den schädeln Europäischer und Asiatischer hausratten (*Rattus rattus* L.). Zeitschrift fur Säugetierkunde 44:354–360.

Smith, D.G., E.K. Shiinoki, and E.A. VanderWerf. 2006. Recovery of native species following rat eradication on Mokoli'i Island, O'ahu, Hawai'i. Pacific Science 60:299–303.

Steadman, D.W. 1999. The prehistory of vertebrates, especially birds, on Tinian, Aguiguan, and Rota, Northern Mariana Islands. Micronesica 31:319–345.

Strecker, R.L. 1962. Population levels. Pages 74–79 *in* T.I. Storer, editor. Pacific island rat ecology: report of a study made on Ponape and adjacent islands 1955–1958. Bernice P. Bishop Museum Bulletin 225, Honolulu, Hawaii, USA.

Thibault, J.-C., J.-L. Martin, A. Penloup, and J.-Y. Meyer. 2002. Understanding the decline and extinction of monarchs (Aves) in Polynesian islands. Biological Conservation 108:161–174.

Towns, D.R. 1994. The role of ecological restoration in the conservation of Whitaker's skink (*Cyclodina whitakeri*), a rare New Zealand lizard (Lacertilia: Scincidae). New Zealand Journal of Zoology 21:457–471.

Towns, D.R. 1999. Interactions between geckos, honeydew scale insects and host plants revealed on islands in northern New Zealand, following eradication of introduced rats and rabbits. Pages 329–335 *in* C.R. Veitch and M.N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, U.K.

Towns, D.R. and C.H. Daugherty. 1994. Patterns of range contractions and extinctions in the New Zealand herpetofauna following human colonisation. New Zealand Journal of Zoology 21:315–339.

Towns, D., I. Atkinson, and C. Daugherty. 2006. Have the harmful effects of introduced rats on islands been exaggerated? Biological Invasions 8:863–891.

U.S. Census Bureau. 2003. Population and housing profile, 2000: The Commonwealth of the Northern Mariana Islands. Government Printing Office, Washington, D.C., USA.

U.S. Fish and Wildlife Service. 2005. Draft revised recovery plan for the aga or Mariana crow (*Corvus kubaryi*). Portland, Oregon, USA.

U.S. Geological Survey. 1999. Tinian [map]. 1:25000. 7.5 Minute Series. Washington, D.C., USA.

Varnham, K., S.S. Roy, A. Seymour, J. Mauremootoo, C.G. Jones, and S. Harris. 2002. Eradicating Indian musk shrews (*Suncus murinus*, Soricidae) from Mauritian offshore islands. Pages 342–349 *in* C.R. Veitch and M.N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, U.K.

Weihong, J, C.R. Veitch, and J.L. Craig. 1999. An evaluation of the efficiency of rodent trapping methods: the effect of trap arrangement, cover type, and bait. New Zealand Journal of Ecology 23:45–51.

Whitaker, A.H. 1973. Lizard populations on islands with and without Polynesian Rats, *Rattus exulans* (Peale). Proceedings of the New Zealand Ecological Society 20:121–130.

Wiewel, A.S. 2004. Systematic rodent monitoring project: WAPA Sherman trap comparison. Report to USGS, Fort Collins, Colorado, USA.

Wiewel, A.S. 2005. Systematic rodent monitoring protocol. Report to USGS, Fort Collins, Colorado, USA.

Wiewel, A.S., A.M. Yackel Adams, and G.H. Rodda. 2008. Systematic rodent monitoring: study of the introduced small mammals of the Mariana Islands. Report to USGS, Fort Collins, Colorado, USA.

Wiles, G.J., A.B. Amerson, Jr., and R.E. Beck, Jr. 1990. The mammals of Tinian, Mariana Islands. Micronesica 23:167–180.

Wilson, D.J., W.G. Lee, R.A. Webster, and R.B. Allen. 2003. Effects of possums and rats on seedling establishment at two forest sites in New Zealand. New Zealand Journal of Ecology 27:147–155.

Wirtz, W.O., II. 1972. Population ecology of the Polynesian rat, *Rattus exulans*, on Kure Atoll, Hawaii. Pacific Science 26:433–464.

Worthy, T.H. 1987. Palaeoecological information concerning members of the frog genus *Leiopelma*: Leiopelmatidae in New Zealand. Journal of the Royal Society of New Zealand 17:409–420.

Yosida, T.H., H. Kato, K. Tsuchiya, and K. Moriwaki. 1971. Karyotypes and serum transferring patterns of hybrids between Asian and Oceanian black rats, *Rattus rattus*. Chromosoma 34:40–50.

Yosida, T.H., T. Udagawa, M. Ishibashi, K. Moriwaki, T. Yabe, and T. Hamada. 1985. Studies on the karyotypes of the black rats distributed in the Pacific and south Pacific islands, with special regard to the border line of the Asian and Oceanian type of black rats in the Pacific Ocean. Proceedings of the Japanese Academy, Series B 61:71–74.

2.5.2 RODENTS ON AGUIGUAN

Prepared by: Amy A. Yackel Adams, James W. Stanford, Andrew S. Wiewel, Gordon H. Rodda, and Allen F. Hambrick. U.S. Geological Survey, Biological Resources Discipline, Fort Collins Research Station, Fort Collins, CO

INTRODUCTION

Aguiguan (spelled Aguijan on USGS maps) is a small (720 ha) currently-uninhabited island south of Tinian, Mariana Islands. *Rattus exulans* arrived on inhabited Aguiguan around 1000 AD, as evidenced by numerous subfossil deposits of rat bones in cultural strata (Steadman 2006). The island had a small and perhaps intermittent human population until all residents were removed by Spanish authorities in 1695, after which the island remained uninhabited until 1936 (Butler 1992). For four years prior and during World War II, the Japanese developed all tillable parts of the island for sugarcane production, but farming ceased and all inhabitants were repatriated to Japan at the conclusion of the war (Bowers 2001). Presumably, the resident rats would have benefitted from the abandoned sugarcane and cessation of rat control efforts. However, in the decade following the war four scientific expeditions specifically noted the scarcity of rats on the island. The first three expeditions (Enders 1949; Owen 1952 cited in Davis 1954; Peterson 1954 cited in Davis 1954) found no evidence of rats, although it is unclear how much effort they put into the search. Over a three-week period Davis (1954) conducted both visual and trap searches, using a variety of baits, yet captured zero rats and observed only two despite high visibility in the goat-overbrowsed understory. Scientists did not visit the island again until 1983 when the rats were seen frequently (Kosaka et al. 1983). A panoply of recent studies provided trap capture rates for the relatively numerous rats, but these neither quantified the absolute abundance of the rats nor obtained genetic samples with which the species of rat could be verified (Campbell 1995, Cruz et al. 2000, Esselstyn et al. 2003).

Wiewel et al. (2009, in press) recently developed robust mark-recapture methods for quantifying absolute population densities of rats in the Mariana Islands. The analytical protocol used by Wiewel et al. (in press) accommodates neophobia (the tendency of rats to avoid novel objects such as traps) sex, reproductive status, size, body condition, trap shyness, a variety of weather covariates such as rain and wind, and unique local factors such as island identity. We sought to replicate Wiewel et al.'s methodology (with minor modifications) on Aguiguan, and thereby quantify the absolute density of a population that had become legendary over the past three decades for high rat abundance. Furthermore, genetic material obtained during the Wiewel et al. study of large Mariana Islands (Guam, Rota, Tinian, and Saipan) showed that the local large rat species was neither of the two species suspected (*R. rattus* or *R. tanezumi*), but an unexpected clade of rats originating in insular southern Asia (*R.* cf. *diardii*; sensu Robins et al. 2007). However, the rat on Aguiguan is considerably smaller than those on the adjacent islands of Guam, Rota, Tinian, and Saipan, and is believed to be *R. exulans*. Because recent morphological and genetic examination of the nominal *R. exulans* population elsewhere in the Pacific (Wake Island) revealed the presence of both *R. exulans* and a stunted form of *R. tanezumi* (P. Dunlevy, WS, pers. comm.), we collected morphological and genetic material from all rats captured during mark-recapture sampling on Aguiguan to confirm species identity. The technical details of the genetic work will be reported elsewhere. In this report we will provide the genetic findings and the absolute population density estimate obtained by mark-recapture.

MATERIALS AND METHODS

Between 22 July and 1 August 2008, we conducted rodent mark-recapture sampling on Aguiguan for 11 consecutive nights on an 11 × 11 grid with 12.5 m intervals between each trap station (nominal grid area = 1.56 ha). Sampling occurred in native forest with an understory of *Guamia mariannae*; other vegetation present included *Pisonia grandis, Ficus prolixa, and Leucaena leucocephala* with a trapping substrate of soil and limestone. The trap grid was located near the western end of the island on the top plateau (grid centriod: N latitude 14.854 and E longitude 145.552).

We placed a single standard-length folding Sherman live trap (229 × 89 × 76 mm; H.B. Sherman Traps, Inc., Tallahassee, FL) at each trap station (*n* = 121). In addition, a single Haguruma wire mesh live trap (approximately 285 × 210 × 140 mm; Standard Trading Co., Honolulu, HI) was placed at every other trap station (*n* = 36); thus the Hagarumas were spaced 25 m apart in a regular grid overlaying the Sherman grid. Closed and unbaited traps were placed on the grid 21 July 2009, one night prior to the beginning of sampling to provide an opportunity for rodents to acclimate to their presence.

We placed traps on the ground and, whenever possible, positioned them to provide shelter from sun and rain. To enhance trap success, we primarily baited traps with 1) a mixture of peanut butter, oats, and food-grade paraffin, or 2) coconut meat. Approximately equal amounts of these bait items were offered on the grid each night.

We checked traps beginning at 0730–0800 each day and closed them during the day to minimize trap mortality. We reopened traps at approximately 1600 and re-baited as necessary to ensure bait freshness. We used the method described by Nelson and Clark (1973) to account for sprung traps when calculating sampling effort.

We examined and measured captured animals to determine species, sex, age, reproductive status, mass (g), head-body length (mm), tail length (mm), right hind foot length (mm), right ear length (mm), and testes length (mm; if applicable). Captured individuals were uniquely marked in each ear with numbered metal ear tags (#1005-1, National Band and Tag Co., Newport, KY). Recaptured animals were examined to determine tag number. All capture, handling, and marking techniques followed guidelines approved by the American Society of Mammalogists (Gannon et al. 2007) and the U.S. Geological Survey Animal Care and Use Committee (USGS Fort Collins Science Center). We collected DNA samples by pulling several guard hairs (and their associated follicles) from all captured rats. Genetic materials were stored in a dry envelope. Five randomly-selected follicles were analyzed following the cytochrome oxidase I procedure outlined in Robins et al. (2007).

Abundance estimates were generated in Program MARK 4.3 (White and Burnham 1999) using the conditional likelihood closed capture-recapture model developed by Huggins (1989, 1991). Our analysis followed an information-theoretic approach involving model selection and multi-model inference. Model selection was based on Akaike's Information Criterion corrected for small sample size (AIC_c; Burnham and Anderson 2002). Models were considered competitive with the top-ranked model when $\Delta AIC_c \leq 2.0$ (Burnham and Anderson 2002). To provide a robust abundance estimate, we model-averaged abundance estimates based on Akaike weights (w_i; Burnham and Anderson 2002) and included the entire model set except for models with nonsensical standard errors for β estimates (e.g., $\beta = -11.6$, $SE(\beta) = 42.6$), which were removed prior to model averaging abundance estimates.

We initiated our modeling efforts by evaluating six specific time structures to address suspected trap neophobia. Our neophobia models allowed capture probability to vary during the first three (Step3), five (Step5), and seven (Step7) sampling occasions, while holding capture probability constant for remaining sampling occasions. We also allowed neophobia to linearly diminish (i.e., capture probability increase linear) over the first three (Ramp3), five (Ramp5), and seven (Ramp7) sampling occasions, while holding capture probability constant for remaining sampling occasions. Our motivation for these models came from literature accounts of neophobia for introduced *Rattus* spp. (Temme and Jackson 1979, Inglis et al. 1996, and Clapperton 2006), neophobia lasting four days (two days trap exposure plus two days trapping) for *R.* cf. *diardii* in the southern Mariana Islands (Wiewel et al. 2009), and from trap results of *R. exulans* on Rota (rats were not trapped until five days of trap experience; Wiewel et al. 2008). We used the neophobia structure with the greatest support, along with individual and environmental covariates, and behavior (b) to define the global model. Using the global model, we then proceeded through a series of more parsimonious models. Covariates under consideration included sex, age (adult or juvenile), body condition index, head-body length (length), body size, and rain amount (during the past 24 hour period [rain24] and the cumulative rainfall effects over the past 48 hour period [rain48]). Rain amount was a quantitative measure of total rainfall (mm) measured at the trap grid center. We calculated body condition index as the ratio between the observed and expected mass of an individual, where expected mass was determined from a linear regression of ln mass vs. ln length. Body size was a species-specific composite variable created from a principal components analysis (Proc FACTOR, SAS Institute, 2003) of mass, length, tail length, hind foot length, and ear length measured for each captured individual. We evaluated this variable only in the global model in place of length. We used the variable (size or length) with the greatest support to build subsequent models. We assessed covariate importance by evaluating their slope estimates and 95% confidence intervals, where covariates with 95% confidence intervals not overlapping zero were considered influential on capture probability. Burnham and Anderson (2002:167) recommend the use of summed Akaike weights to evaluate the relative importance of covariates when a balanced model set is used (e.g., in our analysis each variable appeared in 11 models). We computed a relative importance measure for each variable by summing Akaike weights over every model in which that variable appeared. All estimates are presented as mean ± 1 SE.

We calculated *R. exulans* density by dividing the model-averaged abundance estimates by effective trapping area (ETA), where ETA equaled the total area encompassed by the trapping grid (1.56 ha) plus a boundary strip of ½ the mean maximum distance moved (MMDM) between captures for individuals captured two or more times (Wilson and Anderson 1985).

RESULTS

Rats were specifically uniform in morphology from Aguiguan and genetic analysis (*n* = 5) of the mtDNA cytochrome oxidase I gene region indicated that all hair follicle samples were that of *R. exulans* (S. J. Oyler-McCance and J. St. John, unpublished data).

We captured 48 *R. exulans* (33 females and 15 males) in 1,668 adjusted trap nights (1,727 total trap nights). Of these, 46 individuals were included in a mark-recapture analysis (2 escaped before they could be marked). We had 14 recaptures of 12 individuals. Of the 46 individuals used in the analysis, 42 were adults and 4 were juveniles. Average mass was 63.3 ± 2.54 g (95% CI = 58.4–68.3, *n* = 46).

R. exulans capture and recapture probability were best explained by models allowing neophobia to diminish linearly (i.e., capture probability increased linearly) until occasion 7 with additive effects of cumulative rainfall over the past 48 hours (rain48) and sex (Table 1). Capture probability increased slightly with increasing rainfall over a two day period ($\beta = 0.04 \pm 0.02$, 95% CI = -0.01, 0.08; Figure 1). The covariates ordered by estimated importance are sex, length, condition index, and age, as portrayed by the summed weights of 0.70, 0.34, 0.29, and 0.28, respectively (an importance value ≥ 0.5 indicates that the variable is important to the process being investigated [Barbieri and Berger 2004]). Females were much more likely to be captured than males after initial trap occasions ($\beta = -1.74 \pm 1.03$, 95% CI = -3.77, 0.30; Figure 2).

Mean maximum distance moved was 35.2 ± 5.8 m (95% CI = 23.8–46.7; *n* = 12). When combined with the nominal grid area of 1.56 ha, these MMDM estimates resulted in an effective trap area of 2.57 ha. The model-averaged *R. exulans* abundance estimate generated from our models equaled 141 ± 106 rats (95% CI 46-350). Average *R. exulans* density was calculated to be 55 individuals/ha.

DISCUSSION

Unlike the rat population on Wake Island, morphological examination indicated that only one species was present, and the DNA testing confirmed that all were *R. exulans*. This agrees with the subfossil material (Steadman 2006), and thus it is unlikely that temporal fluctuations in Aguiguan rat abundance were due to changing rat species composition, though a double replacement (*R. exulans* > unknown *Rattus* spp. > *R. exulans*) cannot be excluded with the data available.

The best estimate of rat density for Aguiguan was 55/ha, a value that is high without being exceptional. There are no firm *R. exulans* population density estimates for other localities in the Mariana Islands, and for western Micronesia we have located only the relative abundances of Strecker (1962) on Pohnpei Island. Using removal trapping Strecker (1962) obtained trapping success rates of 13-79% (mostly around 45%), which far exceeds the 3% mean capture rates we observed. Atkinson and Moller (1990) give absolute *R. exulans* population densities for various Pacific islands (none within Micronesia) of 6-170/ha in grassland and10-80/ha in forest. Thus our point estimate (55/ha) would appear to be near the middle of the potential range.

The observed density is also moderate in comparison to values Wiewel et al. (2009) obtained for *R.* cf. *diardii* collected in the Marianas. The mean body mass for *R.* cf. *diardii* on nearby Tinian (130 g) was roughly twice that of the *R. exulans* we sampled on Aguiguan (63 g). Despite this, the range of best density estimates for the larger rat on Tinian ranged 44/ha (introduced forest) to 99.9/ha (grassland), with native forest being near the middle (75.1/ha). Thus the observed absolute population density of *R. exulans* on Aguiguan (55/ha) is best characterized as moderate, at least with reference to nearby islands in the Marianas chain.

This finding does not comport well with the subjective impression most biologists have regarding the apparent superabundance of rats on Aguiguan. For example, Rodda et al. (2009) reported an extraordinary rat sighting rate of 16.8 rats per hour when conducting visual surveys for lizards on Aguiguan. Rodda et al. (2009) suggest that high visibility associated with heavy goat grazing of understory vegetation was responsible for elevated visual detection rates of the relatively moderate number of rats present.

It is also possible that the rats on Aguiguan are less wary as a consequence of reduced human persecution (the island has not been permanently inhabited for >60 years). Lack of wariness might lead to a relatively high sighting rate. If the rats on Aguiguan were generally less wary as a consequence of generations without human persecution, however, one might expect them to exhibit little fear of novel human objects such as traps. Instead, we observed eight days (one day of trap exposure plus seven days of trapping) of neophobia, a duration without precedent in our studies of rats in the Mariana Islands (Wiewel et al. 2009, in press). The causes of high rat visibility on Aguiguan in the face of only moderate rat density remain to be elucidated.

ACKNOWLEDGMENTS

The rather difficult challenge of getting staff safely to Aguiguan was a bureaucratic and logistic burden of the first magnitude, ably carried out on our behalf by Earl Campbell, Karl Buermeyer, Curt Kessler, Nate Hawley, Elvin Masga, Ton Castro, Peter Dunlevy, and especially Jess Omar. We are extremely grateful to all. Nate Hawley assisted with permitting. Funding was provided by the U.S. Navy (Scott Vogt) through the U.S. Fish and Wildlife Service (Earl Campbell). Lea' Bonewell assisted with stateside logistics. Sarah Oyler-McCance conducted the DNA analysis of the Aguiguan rat follicles. Any

use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

LITERATURE CITED

Atkinson, I.A.E., and Moller, H. 1990. Kiore, in C. M. King, eds., The handbook of New Zealand mammals: Auckland, New Zealand, Oxford Univ. Press, p. 175-192.

Barbieri, M.M., and Berger J.O. 2004. Optimal predictive model selection. The Annals of Statistics 32: 879-897.

Bowers, N.M. 2001. Problems of resettlement on Saipan, Tinian and Rota Mariana Islands. Occasional Historical Papers Series No. 7. CNMI Division of Historic Preservation, Saipan, Commonwealth of the Northern Mariana Islands.

Burnham, K.P., and Anderson D.R. 2002. Model selection and multimodel inference: a practical information-theoretic approach. New York, N.Y., Second Edition, Springer, 488 p.

Butler, B.M. 1992. An archaeological survey of Aguiguan (Aguijan), Northern Mariana Islands. Saipan MP, Micronesian Archaeological Survey Rep. No. 29. 260p.

Campbell, E.W., III. 1995. Report on Aguiguan Island reptile and rodent surveys conducted between 21-27 March 1995 and 30 May - 4 June 1995. Saipan MP, unpub. rep. to CNMI Div. Fish Wildlife.

Clapperton, B.K. 2006. A review of the current knowledge of rodent behaviour in relation to control devices. Science for Conservation 263: Wellington, New Zealand, New Zealand Department of Conservation.

Cruz, J., L. Arriola, Johnson N., and Beauprez G.M. 2000. Wildlife and vegetation surveys; Aguiguan 2000. Saipan MP, CNMI Div. of Fish and Wildlife Tech. Rep. No. 2.

Davis, C.J. 1954. Report on the Davis expedition to Agiguan July-August, 1954: ecological studies, Island of Agiguan, Marianas Islands as related to the African snail, Achatina fulica Bowdich, and its introduced predator, Gonaxis kibweziensis (E. A. Smith). Hilo, Hawaii, Hawaii Board of Commissioners of Agriculture and Forestry. 21 p.

Enders, R.K.1949. A field study of the rats in the Marianas and Palaus. Rep. No. 3, Sci. Invest. Micronesia, Pac. Sci. Board, Nat. Res. Council. N7-onr-291.

Esselstyn J.A., Cruz J.B., Williams L.L., and Hawley N. 2003. Wildlife and vegetation surveys; Aguiguan 2002. Saipan MP, CNMI Div. of Fish and Wildlife Tech. Rep. No. 9, 55 p.

Gannon, W.L. and Sikes R.S. 2007. The Animal Care and Use Committee of the American Society of Mammalogists. Guidelines of the American Society of Mammalogists for the use of wild mammals in research. Journal of Mammalogy 88:809–823.

Huggins, R.M. 1989. On the statistical analysis of capture-recapture experiments. Biometrika 76:133-140.

Huggins, R.M. 1991. Some practical aspects of a conditional likelihood approach to capture experiments. Biometrics 47:725-732.

Kosaka, E.F., Lemke T.O., Pratt T.K., Aldan D.T., and Falig C. 1983. Agiguan field trip report, 11-14 July 1983. Saipan MP, rep. to CNMI Div. Fish Wildlife.

Inglis, I.R., Shepherd D.S., Smith P., Haynes P.J., Bull D.S., Cowan D.P., and Whitehead D. 1996. Foraging behaviour of wild rats (Rattus norvegicus) towards new foods and bait containers. Applied Animal Behaviour Science 47:175–190.

Nelson, L., Jr., and Clark F.W. 1973. Correction for sprung traps in catch/effort calculations of trapping results. Journal of Mammalogy 54:295–298.

Robins, J.H., Hingston M., Matisoo-Smith E., and Ross H.A. 2007. Identifying Rattus species using mitochondrial DNA. Molecular Ecology Notes 7:717-729.

Rodda, G.H., Reed R.N., and Stanford J.W. 2009. The land reptiles of Aguiguan (Aguijan) Island. rep. to US Fish and Wildlife Service by USGS Fort Collins Science Center, Fort Collins, Colorado. 11 p.

SAS Institute. 2003. SAS/STAT software. Version 9.1. SAS Institute, Inc., Cary, North Carolina, USA.

Steadman, D.W. 2006. Extinction and biogeography of tropical Pacific birds. Univ. Chicago Press, Chicago.

Strecker, R.L. 1962. Population levels in T.I. Storer ed., Pacific Island rat ecology; report of a study made on Ponape and adjacent islands 1955 - 1958. Honolulu Hawaii, Bernice P. Bishop Museum Bull. 225, p. 74-79.

Temme, M., and W.B. Jackson. 1979. Criteria for trap evaluation. American Society of Testing Materials Special Technical Publication 680:58–67.

Wiewel, A.S., Yackel Adams A.A., and Rodda G.H., 2008. Systematic rodent monitoring: a study of the introduced small mammals of the Mariana Islands. Fort Collins Colorado, rep. to USGS Fort Collins Sci. Center. 201 pp.

Wiewel, A.S., Yackel Adams A.A., and Rodda G.H. 2009. Density, distribution, and biomass of introduced small mammals in the southern Mariana Islands. Pacific Science 63:207-224.

Wiewel, A.S., Yackel Adams A.A., and Rodda G.H. in press. Evaluating abundance estimate precision and the assumptions of a count-based index for small mammals. Journal of Wildlife Management 73.

Wilson, K.R., and Anderson D.R. 1985. Evaluation of two density estimators of small mammal population size. Journal of Mammalogy 66:13-21.

White, G.C., and Burnham K.P. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46(Supplement):120–138.

Table 1. Model selection results for mark-recapture modeling of capture (*p*) and recapture (*c*) probability for *R. exulans* data collected on Aguiguan, 2008. Results include the relative Akaike's Information Criterion corrected for small sample size (ΔAIC_c), and Akaike weight (w_i), and number of model parameters (*K*). See text for abbreviations used in model names.

Model	ΔAIC_c	w_i	*K*
Ramp7 + rain48 + sex	0.00	0.206	4
Ramp7 + rain48 + sex + length	1.13	0.117	5
Ramp7 + rain48 + sex + age	1.79	0.084	5
Ramp7 + rain48 + sex + body condition	1.85	0.082	5
Ramp7 + rain48	2.56	0.057	3
Ramp7 + rain48 + sex + length + body condition	2.78	0.051	6
Ramp7	2.98	0.046	2
Ramp7 + rain48 + sex + age + length	3.06	0.045	6
Ramp7 + b	3.28	0.040	3
Ramp7 + rain48 + sex + age + body condition	3.73	0.032	6
Ramp7 + rain48 + sex + age + length + body condition	4.09	0.027	7
Ramp7 + b + rain48	4.21	0.025	4
Ramp7 + rain48 + body condition	4.26	0.025	4
Ramp7 + b + rain48 + sex + age + length + body condition	4.32	0.024	8
Ramp7 + rain24	4.37	0.023	3
Ramp7 + sex + age + length + body condition	4.49	0.022	6
Ramp7 + rain48 + age	4.57	0.021	4
Ramp7 + rain48 + length	4.58	0.021	4
Ramp7 + rain24 + sex + age + length + body condition	5.91	0.011	7
Ramp7 + rain48 + length + body condition	6.30	0.009	5
Ramp7 + rain48 + age + body condition	6.30	0.009	5
Ramp7 + rain48 + age + length	6.53	0.008	5
Step5	7.39	0.005	2
Ramp7 + rain48 + age + length + body condition	8.35	0.003	6
Ramp5	8.39	0.003	2
Step5 + b	9.29	0.002	3
Ramp5 + b	9.83	0.002	3
Step7	10.98	0.001	2
Step3	19.48	0.000	2
Ramp3	19.76	0.000	2

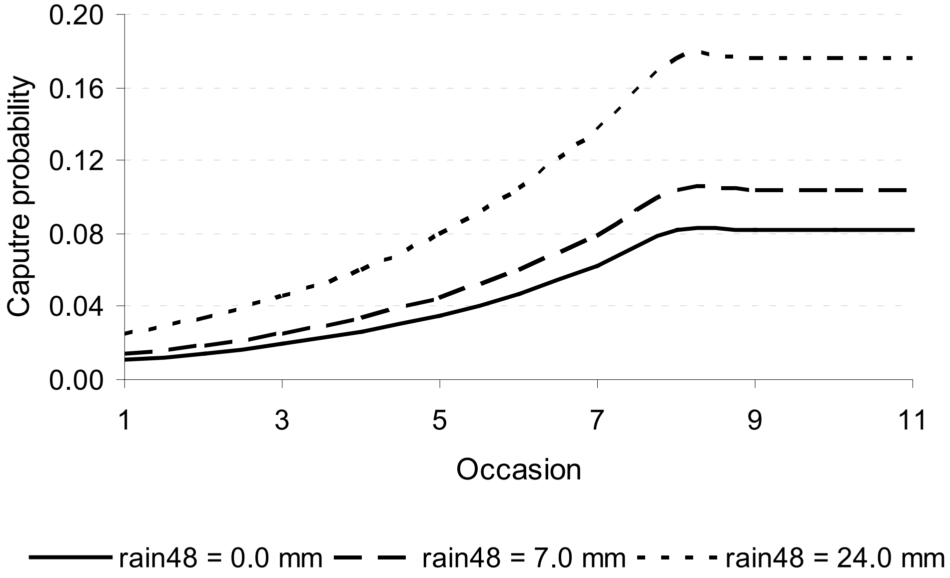


Figure 1. Cumulative rainfall effects over the past 48 hours (rain48) for female *Rattus exulans* on capture probability under three scenarios of no rainfall, the average cumulative rainfall value of 7.0 mm specified for each occasion, and the maximum cumulative rainfall of 24.0 mm specified for each occasion. A similar additive effect was seen in males but is not illustrated in this figure.

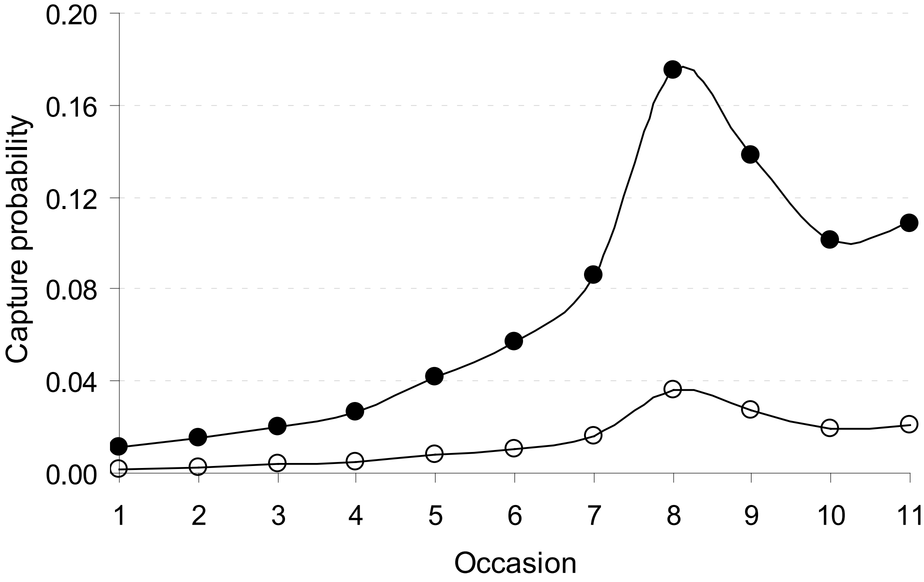


Figure 2. Effect of neophobia (reduced capture probability during occasions 1 through 7) and sex (female = •, male = ○) on *Rattus exulans* capture probability. Model also includes cumulative rainfall effects over the last 48 hours, which has a slightly positive effect on capture probability.

2.5.3 MARIANA FRUIT BAT ON TINIAN AND AGUIGUAN

Prepared by: Anne Brooke. **U.S.Navy, NAVFACMAR, Guam**

INTRODUCTION

Surveys for the Mariana fruit bat or fanihi (*Pteropus mariannus mariannus*) were conducted on the islands of Aguiguan and Tinian in 2008. Once common throughout the Mariana archipelago, fruit bats in the southern islands continue to be hunted and impacted by foraging habitat loss and numbers remain low (reviewed in U.S. Fish and Wildlife Service 2005). Bats have been occasionally sighted on Tinian and a small number of bats are resident on Aguiguan. This report provides an estimate of current numbers on Aguiguan and Tinian and reviews the findings of earlier surveys.

METHODS

Estimations of island-wide bat numbers were made using direct colony counts and station counts. Bats sleep during the day in canopy emergent trees solitarily or in colonial aggregations that may be spread over several acres. Colonial roosts are typically in locations that are difficult for people to reach, such as on cliffs or in remote forest areas. Direct colony counts were made during the day at a single colonial roost and at dusk when bats were leaving to forage. Station counts for solitary bats were conducted at dawn or dusk as bats depart or return from foraging (Utzurrum *et al.* 2003). Locations for station counts were selected for wide and unimpeded views of forests that would likely serve as roost sites for bats.

Tinian surveys were conducted February-August 2008 at sites used in earlier surveys and in new locations (Wheeler 1980, Wiles *et al.* 1989, Krueger andO'Daniel 1999, Cruz *et al.* 2000). Surveys were conducted on Aguiguan July 19-23, 2008. The minimum number of animals observed at each site was recorded. Low light, lack of distinctive markings, and the observers distance from the animals make individual recognition difficult. For each bat seen, the direction it flew and the location where it was lost from sight was noted. If a bat was subsequently seen that could have been the same individual returning to the site or leaving a tree where it had roosted, it was noted but not included in the tally.

RESULTS

Tinian

No bat colonies were observed on Tinian so no direct colony counts were conducted. Eight separate station counts were conducted at seven locations. No bats were observed during station counts or opportunistically.

Aguiguan

Thirteen station counts were conducted at eight different sites and a single colonial roost was counted, yielding an island-wide estimate of 40-60 bats.

The combined total from the station and roost count was 39 to 47 individual bats. Taking into consideration bats observed in counts as well as those encountered in the forest, a minimum estimate for the island is 40 bats. Given the areas not covered during the surveys, it is reasonable to assume that an additional 20 bats may be present, giving an island-wide estimate of 40 to 60 bats.

DISCUSSION

In the last 29 years, few bats have been observed during surveys on Tinian although island residents report occasionally seeing bats (T. Castro, E. Masaga and F. Muna, pers. comm.). During surveys in 1979 two bats were observed in the Kastiyu forest and an island-wide estimate of 25 to 100 was calculated based on available forest habitat (Wheeler 1980). In 1983-1984 bats were sighted three times on Tinian and the number estimated island-wide was less than 25 individuals (Wiles *et al.* 1989). Surveys in 1994-1995 recorded no bats, but two incidental sightings were reported from other locations on Tinian (Krueger and O'Daniel 1999). No bats were sighted during two surveys in 2000 and 2001 (Cruz *et al.* 2000, Johnson 2001). In June 2005, approximately five bats were seen in the cliff-line forest during a routine forest bird survey of the Maga bird transect (S. Vogt, pers. comm.).

Bats occasionally have been seen in flight between Saipan, Tinian and Aguiguan. A group of approximately 50 bats was seen flying over the ocean toward Tinian from the southern part of Saipan in 2001 (L. Bulgrin, pers. comm.). On two occasions in 2008 single bats were watched as they flew from Aguiguan over the channel towards Tinian (C. Kessler, pers. comm.). One bat was seen during the day (0930), the other was seen at 2200 using nightvision goggles, and both were lost from sight when far over the channel. Travel between the islands may be natural dispersal movements or the result of disturbances caused by hunting (Wiles and Glass 1990).

Surveys on Aguiguan have shown a small but widely fluctuating number of bats in the past 54 years. The amount of time spent on the island, knowledge of likely colonial roost sites, survey locations, methods and analysis have differed among these surveys. In spite of the varied methods, it is clear that the number of bats on Aguiguan has remained low.

Early surveys in 1954 reported bats flushing on several occasions from the forest on the northeast lower terrace (Davis 1954). Surveys in 1983 and 1984 reported only one or two bats seen each trip, and less than 10 bats were thought to be present (Pratt and Lemke 1984). A 1987 survey located a colony of approximately 24 bats and estimated roughly 40 bats on island (Reichel *et al.* 1987). A minimum of 200 bats were estimated in 1988 based on a colony of approximately 60 and sighting of at least 136 bats foraging one evening (Reichel *et al.* 1988). The increase was attributed to immigration from another island. A maximum of 30 bats was estimated in 1989 and in 1992 (Rice and Reichel 1989, Craig and Chandran 1992). In 1995, a colonial roost of approximately 100 bats was located and added to solitary bats in flight for an island-wide estimate of 200 (Worthington and Taisacan 1995). In 2000, an island-wide estimate of 150 to 200 was based on approximately 63 bats observed, 23 solitary and two colonial roosts each with roughly 20 individuals (Cruz *et al.* 2000). The 2002 survey estimated 40-60 bats based on observations of 29 solitary individuals and no colonial roosts (Esselstyn *et al.* 2003).

During 1975 -1981, the Commonwealth of the Northern Mariana Islands (CNMI) legally exported 16,495 bats to the Guam to supply market demand (Wiles 1992). During these six years, 1,366 bats were exported from Tinian (50-433 annually). It is likely that many of these bats originated on Aguiguan (Wiles *et al.* 1989). The legal exporting of bats was curtailed in 1989 when fruit bats were included in the Convention of International Trade in Endangered Species (CITES). Fruit bats were listed as a threatened/endangered species by the CNMI government in 2000 (Berger *et al.* 2005), and were Federally listed as threatened in the CNMI in 2005 (USFWS 2005).

Ample time has passed since the CNMI government listed bats as a protected species (8 years ago and 19 years since exportation was banned) for bats to have recovered such that they could be legally hunted. This has not happened; poaching has continued unchecked. When hunting pressure has been reduced on other islands, fruit bat numbers have rapidly increased. Bats were extensively hunted in Palau during the 1980s when an estimated 2,000-5,000 *P. mariannus pelewensis* were killed annually (Wiles *et al.* 1997). Within ten years of cessation of commercial hunting for export to Guam, bats became common (Wiles *et al.* 1997). Fruit bats on Tutuila (American Samoa) declined from ca 12,000 to

1,500-2,500 from overhunting after cyclones in the early 1990s (Craig *et al.* 1994). Hunting was restricted, and within ten years, bats had increased to 7,000-8,000 (Utzurrum *et al.* 2006). As Saipan is the most comparable of the southern CNMI islands to Tutuila in size and habitat. It seems reasonable that Saipan could support several thousand bats, Tinian and Aguiguan could support hundreds.

Consumer demand for fruit bats remains the driving force for illegal hunting, preventing the recovery of bats in the southern islands of the CNMI. Bats were reported to sell for $50 on Tinian in 2008 and $140 on Saipan in 2006. The value of bats on Guam is beyond a monetary value with payment made by in-kind favors. Without immediate support from leading government officials and law enforcement, fruit bats will be extirpated from the southern Mariana Islands.

ACKNOWLEDGEMENTS

Elvin Masga, Fabrious Muna, and Julia Boland assisted with counts on Aguiguan, Vanessa Pepi assisted with counts on Tinian.

2.5.4 SHEATH-TAILED BAT ON AGUIGUAN

Compiled by: Thomas J. O'Shea and Ernest W. Valdez. U.S. Geological Survey, Fort Collins Science Center, Colorado

Note: This is the Executive summary section of Assessment for Pacific Sheath-tailed Bats (*Emballonura semicaudata rotensis*) on Aguiguan, Commonwealth of the Northern Mariana Islands. For the entire document please refer to Appendix 3.3

Executive Summary

The subspecies of the Pacific sheath-tailed bat that once occurred throughout the Mariana Islands (*Emballonura semicaudata rotensis*) has not been well studied biologically, despite its declining status. It is a small insectivorous bat, and in the Mariana Islands it is known only to roost in caves. All available data indicate that it now occurs only as a single remnant population on Aguiguan. Overall the species is categorized as Endangered by the International Union for the Conservation of Nature and Natural Resources. The subspecies is protected by the Commonwealth of the Northern Mariana Islands (CNMI) law, and is considered a Category 3 candidate for listing under the U.S. Endangered Species Act. This categorization under U.S. law is based on the imminence and magnitude of threats, but further actions have not had the highest priority possible in part because the remaining population on Aguiguan has been considered to be a subspecies of a more widely found species. However, a thorough, modern quantitative morphometric and molecular genetic analysis is needed to verify if the subspecific level in the taxonomic hierarchy is accurate or if full species designation may be warranted for the population in the Marianas Islands.

In this report we document results from a biological assessment for Pacific sheath-tailed bats carried out in 2008 on Aguiguan and Tinian, CNMI. The field work was done by a team consisting of a former Guam Division of Aquatic and Wildlife Resources biologist with past experience surveying for this species and four bat biologists from the U.S. Geological Survey (USGS) Fort Collins Science Center and the USGS Pacific Island Ecosystems Research Center. The assessment consisted of determining present abundance and use of caves on Aguiguan by these bats and interpreting these data in comparison with a synthesis of the literature and past unpublished data; establishing baseline site occupancy models of spatial foraging habitat use through monitoring of ultrasonic echolocation calls; determining basic aspects of diet through analysis of fecal material; sampling bats through capture to obtain new data on reproduction and body size, as well as to collect samples for future genetic analysis; and determining characteristics of temperature and humidity in caves. We conducted a review of specimens available in research museums, and obtained samples from guano deposits that may be useful in analysis for contaminants in comparison with analysis of guano from other islands where these bats have become extinct. We also conducted a limited survey for the presence of these bats on Tinian.

Our report summarizes previously unpublished results on numbers of Pacific sheath-tailed bats roosting in caves on Aguiguan in 1995 and 2003, and compares past results with findings from new surveys conducted in 2008. Overall, we examined the abundance, roosting behavior, and distribution of Pacific sheath-tailed bats on Aguiguan by searching caves and hollow trees for roosting bats during the day. Counts of bats at caves show that a small population of Pacific sheath-tailed bats continues to exist on Aguiguan, with a range of 359 - 466 individuals counted at five of 41 caves in 2008. Comparison with past counts suggests that this population has increased over the last 13 years. Bats appeared to prefer roosting in larger caves and displayed fidelity toward five of the seven roosts found occupied in the study. Occupied caves were larger than most unoccupied caves but had similar conditions of temperature and humidity. In 2008 one cave consistently housed the largest colony, with a range of 308–382 bats counted, whereas counts at other occupied caves on Aguiguan yielded 1–64 individuals. Slight variability occurred in replicate counts on different dates during the 2008 survey. We found no evidence of hollow tree trunks being used as roosts. It is possible that a small number of colonies of these bats may remain undiscovered at inaccessible caves on Aguiguan.

Evaluation of trends in colony sizes of cave bats throughout the world generally relies on count data that are uncalibrated index values, which can be difficult to interpret. Therefore this assessment also sought to utilize a recently developed quantitative approach to establish a baseline site occupancy model of spatial occurrence of foraging Pacific sheath-tailed bats on Aguiguan. This method uses detection of bat ultrasonic calls to assess presence absence of foraging bats at night in relation to various habitat attributes. Thirty one echolocation stations were deployed across Aguiguan between 25 June and 14 July 2008. Twenty one of the 31 stations recorded ultrasonic pulses from sheath tailed bats over a period of 19 days, with 35,858 calls recorded. Ten percent of the calls were characterized as peak activity, 40% as moderate activity, and 50% as brief passes. Analyses show that peak activity and occurrence is related to canopy cover, vegetation stature, and distance to known roosts. Native limestone forest is preferred foraging habitat. Echolocation calls of Pacific sheath-tailed bats were characterized for the first time, and search phase calls were similar to those of other emballonurid bats that use a narrow bandwidth and short pulse duration to forage in cluttered vegetation.

There has been no prior information on the food habits of the Pacific sheath-tailed bat anywhere in the species' range. Herein we reported on new findings from analysis of fecal material from this bat on Aguiguan. We collected and analyzed 200 fecal pellets of bats from two roosts (Guano Cave and Crevice Cave). The diet of the Pacific sheath-tailed bat was diverse, but mostly consisted of small-sized prey ranging from 1.7 to 6.4 mm in length. Overall hymenopterans (ants, wasps, and bees), lepidopterans (moths), and coleopterans (beetles) were the three major food items in the diet of bats from both roosts. However, the ranking of volumes of each insect order consumed varied between roosts. At Guano Cave, hymenopterans made up 64% of the diet, followed by coleopterans (10%), and lepidopterans (8%). At Crevice Cave, lepidopterans made up 45% of the diet, followed by hymenopterans (41%), and coleopterans (10%). Within Hymenoptera, most of the prey items belonged to ichneumondoidea (parasitoid wasps), followed by formicids (ants belonging to Formicinae and Ponerinae; i.e., trap jaw ants). Because alates (= winged adults) of ants and termites (isopterans) found in fecal samples generally have wings only when they are reproductive or establishing new colonies, it is likely that Pacific sheath-tailed bats take advantage of seasonal food sources. In other areas the occurrences of these winged forms are often present during the onset of rains; we sampled guano at the onset of the rainy season on Aguiguan (late June to early July). Lepidopterans, specifically microlepidopterans, likely were another seasonally abundant prey item. Silken fungus beetles and leaf beetles identified in the guano appear to be forest dependent species and were a consistent component of the bats' diet. Not only do these and other prey items indicate that these bats forage mainly in forest habitat during late June and early July, but that they also capture prey near (above and below) the canopy. From these diet analyses, we categorize the Pacific sheath-tailed bat as an aerial insectivore or hawker, similar to other emballonurids around the world.

We also collected various other samples and obtained information on the biology and natural history of Pacific sheath-tailed bats on Aguiguan. We used standard means to capture Pacific sheath-tailed bats in mist nets while they dispersed or foraged through the forest, but these attempts were largely unsuccessful because these bats were highly maneuverable and easily avoided mist nets on close approach. We successfully captured 12 adult bats and one attached suckling young by using hand nets on bats in flight in the forest, or mist nets set in or near caves used as roosts. Both methods have logistical problems and limitations: in addition to the high maneuverability of the bats precluding use of mist nets in standard configurations, considerable time is required to accrue multiple captures using hand nets. Caves where bats roost are co-occupied by endangered Mariana swiftlets. Thus capturing bats at caves has the potential to disturb both the bats and the swiftlets. We found that these bats can be very sensitive to initial handling, but stress can be reduced by placing bats individually in cloth bags promptly after capture and before examining them. We determined body mass, length of forearm, and reproductive condition of the 12 adult bats. In addition to qualitative features of skull morphology, length of forearm has been given as a characteristic distinguishing between some subspecies of *E. semicaudata*. However, these new forearm measurements show that there is considerable overlap in body size between *E. semicaudata rotensis* and the other three subspecies of Pacific sheath-tailed bats. We also collected small wing biopsies from12 bats prior to release for some basic preliminary genetic analyses to ascertain genetic diversity of the population on Aguiguan and the depth of division of this subspecies based on comparison with published data on genetics of *E. s. semicaudata* from Fiji. This work will be carried out by USGS geneticists in 2009. We also prepared two museum voucher specimens of *E. s. rotensis*, increasing the number of known specimens from the Mariana Islands available in United States museums from two to four. We reviewed the literature and queried a limited number of online databases to compile updated information on specimens of Pacific sheath-tailed bats that might be available for taxonomic study. Considerable numbers of specimens including other subspecies are available worldwide (over 380), and about 22 additional specimens from the Marianas Islands (including Guam) are housed in museums in France and Japan. Expanded study of museum specimens and comparative genetic analyses are needed to fully ascertain the systematic status of the Pacific sheath-tailed bat population on Aguiguan.

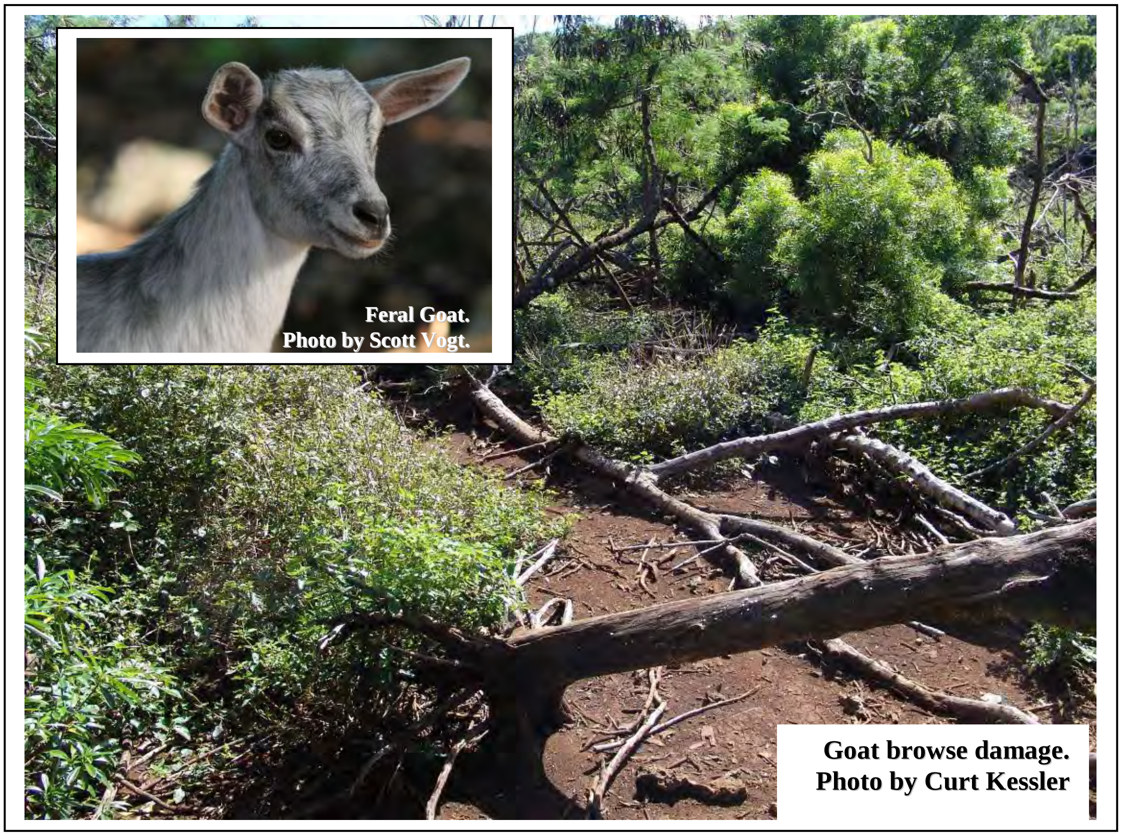
There is limited information on reproduction in Pacific sheath-tailed bats in the CNMI or elsewhere. Six female bats captured by Wiles and others on Aguiguan late in the rainy season of 2003 were apparently not reproductive. In contrast, seven of the eight female bats we captured in June and July 2008 were either pregnant or lactating. We also observed 11 pups at roosts in caves during June and July 2008; all were singletons. None of the bats we captured were volant young of the year. The presence of reproductive females and pups or embryos in June and July but no volant young suggests the hypothesis that Pacific sheath-tailed bats on Aguiguan may have a diffuse seasonality in reproduction, such that the period of late gestation, lactation, and maturation of young coincides with the late June to early November rainy season. We observed one large embryo in a female dissected in June 2008, as was also observed in a female dissected by Lemke in June 1984. These dissections and the observations of 11 apparent singleton pups suggest a litter size of one. If reproduction occurs only once per year and litter size is one, then the capacity for population growth in Pacific sheath-tailed bats will be very limited. All bats that we captured at caves in 2008 and by others in years past were females, whereas 4 bats captured at dusk dispersing along a steep rocky hillside, not near any known colony, were males. This suggests that perhaps males may form bachelor colonies apart from roosts occupied primarily by females, as is known for other Old World species in the genus Emballonura. Elaborate social behavior patterns were also suggested by the audible communication sounds produced by bats that we observed foraging and dispersing through the forest and flying into caves.

The scientific literature includes speculation that the extinction of Pacific sheath-tailed bats on other islands may have been attributable at least in part to past use of organochlorine insecticides. However, there is no chemical or toxicological evidence that bears directly on this speculation. Analyses based on other species of insectivorous bats have shown that concentrations of organochlorine insecticides in bat guano can provide diagnostic evidence of mortality and population declines. Aguiguan has been mostly uninhabited since the use of organochlorines became widespread elsewhere in the world. Thus guano samples from sheath-tailed bats on Aguiguan could provide comparative baselines with which to compare contamination of guano from islands where these bats have become extinct (e.g. Guam). Therefore we used contaminant free sampling approaches to obtain guano at 3 different depth levels (i.e., surface, 10 and 20 cm below surface) from two areas of a guano pile beneath roosting bats at Guano Cave. These samples are stored in the USGS laboratory at the Fort Collins Science Center and can be made available for future chemical analysis. However, because this guano was deposited over many years, the material also likely includes particles of guano from Mariana swiftlets. The degree of mixing of guano from these two sources should be estimated using microscopic techniques prior to chemical analysis.

Pacific sheath-tailed bats are only known from Tinian based on prehistoric deposits in caves. During the last 4 days and nights of our study we made an effort to document the presence of Pacific sheath-tailed bats on Tinian using echolocation detectors. We also queried knowledgeable individuals, and watched for bats and listened for audible calls during the echolocation surveys. We felt that our best chance for success in documenting bats on Tinian would be echolocation-based sampling in limestone forest areas because of their heavy use of this habitat for foraging on Aguiguan. We deployed two monitoring stations that sampled continuously all night long, both set out for one night in a forest in the Mount Lasso area and for a second night in the Kastiyu Forest. We also sampled for one night at each of these sites using ad hoc walking transects and echolocation detectors during the first part of the night, corresponding to peak times of bat echolocation activity on Aguiguan. No bats were detected. However, this survey was far from exhaustive and additional effort using echolocation detectors over wider areas of forest and searches of caves will be needed to rule out the possibility that a small remnant population of these bats may still exist on Tinian. Similar echolocation detector based surveys would also be useful on two other islands in the CNMI (Anatahan and Maug) where tentative sightings were reported in the early 1980s but never subsequently confirmed.

A number of considerations for future activities stem from the findings of this assessment. These are best characterized as activities related to management for conservation, monitoring, and research. Considerations for management for conservation include limiting disturbance of and access to caves used by roosting bats; and increasing the extent of native limestone forest, decreasing existing stands of invasive plants, and eliminating or avoiding actions that would reduce the amount of native limestone forest on Aguiguan. Considerations for future monitoring of sheath-tailed bats on Aguiguan include periodic monitoring of numbers of bats utilizing key caves, and monitoring the use of foraging habitat with echolocation detectors and site occupancy models. Considerations for research include searching the more inaccessible areas on Aguiguan for the presence of additional colonies that may occupy caves requiring technical climbing and caving skills to reach; increasing the foundation of ecological knowledge of this species pertinent to its conservation and management, including investigations into seasonal aspects of reproduction, roosting, and foraging biology; conducting a modern analysis of the taxonomic status of *Emballonura semicaudata* and its subspecies using combined quantitative morphometric and molecular genetic approaches; and further assessing the possible occurrence of Pacific sheath-tailed bats on Tinian and other islands.

2.5.5 FERAL GOATS ON TINIAN AND AGUIGUAN



Feral Goat.
Photo by Scott Vogt.

Goat browse damage.
Photo by Curt Kessler

Prepared by: **Curt Kessler, U.S. Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office, Honolulu, HI.**

INTRODUCTION

Browsing by feral goats (*Capra hircus*) on Aguiguan is altering the spectacular old growth limestone forest leading to replacement with invasive plant species, primarily lantana (*Lantana camara*). This process has two major components that compound the problem over time thus accelerating the forest loss. First, the goats eat all seedlings from most native trees which halts the natural regeneration of limestone forest. This also allows non-palatable invasive plant species such as lantana to colonize and form a monoculture. Second, as canopy trees die and are not replaced, the open canopy allows more solar heating of the forest floor, drying out the soil and destroying the forest floor microclimate. This drying stresses the trees and reduces the trees' ability to survive the dry seasons. As more trees die over time the dry areas increase, which accelerates the cycle until a drought period when massive tree die-offs can occur (Kessler 2002a). Though currently uninhabited, the forest on Aguiguan has been significantly altered by human activities (Engbring *et al*. 1986) and recovery of the forest is unlikely to occur without the removal of the goats. Between 1989 and 1990, an effort was made to eradicate goats from Aguiguan. During that time, 158 goats were removed leaving an estimated 40 goats on the island (Rice 1991). Unfortunately the eradication program did not continue and the goat population has increased since that time. Until recently, there was no feral goat population established on Tinian.

METHODS AND RESULTS

Aguiguan is approximately 720 hectares (ha). Based on site visits during November 2006 and June-September 2008 (C. Kessler, USFWS, pers. comm.), previous surveys of Aguiguan (Lemeke 1984, Esselstyn *et al*. 2003), the current condition of the vegetation on the island, as well as findings from Sarigan and Anatahan (Kessler 2002b, Worthington *et al.* 2001), Aguiguan is estimated to have about 1,440 goats or 2 per hectare. Recent hunting might have reduced this number slightly, but the population is still likely over 1,000 goats.

DISCUSSION

The Mariana Islands possess a very fertile ecology. Introduced, feral, and invasive species usually thrive on these islands due to the mild climate, plentiful rain, and continual growing season. The island of Tinian was known in the early 1800's as an island overrun by feral livestock. However that changed in the early 1900's and until recently Tinian has had no feral ungulates. The neighboring island of Aguiguan however, is known as Goat Island and has had feral goats for the last 50 years and maybe much longer. In 1989, an attempt to remove the goats was undertaken by CNMI-DFW which reduced the goats to under 100 (Rice 1991) but the effort was halted due to political concerns. Since that time the goat population has recovered and is at or near the capacity of the island (Kessler pers. obs.).

Feral goats in the Marianas have an average density of 2 goats/ha. (DFW 1985, Esselstyn *et al*. 2003, Kessler 2001). On Aguiguan, Lemke estimated 1,000 goats in 1984 and Esselstyn estimated 1,143 individuals (range 943 to 2,117) in 2002; both surveys used transect survey methods. The current estimate of the Aguiguan goat population is still within these ranges and evidence supports that goats are severely impacting the native forest.

Aguiguan would be a relatively easy island to eradicate goats from due to its small size and close proximity to Tinian. It is estimated that with a budget of $500,000 that the goats could be removed in 2-3 months. Currently there is opposition to the eradication of goats from the Tinian Mayor's Office. For the past 60 years, Aguiguan has been a place to hunt and gather resources for the residents of Tinian, and goats are considered one of these resources. However, it is suspected that if Aguiguan was leased, that this opposition would evaporate. Especially if the island would begin to produce cash revenue through tourism as well as provide traditionally cultural native species for consumption. Aguiguan is the only limestone island with most of the original species intact including old growth forest. Based on a recent land cover assessment, the island is currently around 49 percent native forest (See Section 2.1, Vegetation Surveys on Tinian and Aguiguan).

Goats have recently been transported from Aguiguan to Tinian as per instructions of the Mayor of Tinian. These goats, which anecdotal accounts put a total at 200 individuals, were released into the native forest on Tinian public lands to propagate. A survey around the coast on October 11, 2008 confirmed at least 20 goats at Puntan Kastiyu (14°56'53.90"N 145°39'53.38"E). It appeared that this herd is already creating trails, accelerating erosion, and impacting the native vegetation on the hillside. If public hunting of goats was allowed on Tinian then the threat of overpopulation might be negated.

REFERENCES

DFW. 1985. Yearly progress report for fiscal year 1984; for the Pittman-Robertson Federal Aid in Wildlife Restoration Program. Unpublished report to US Fish & Wildlife Service, Honolulu, HI.

Engbring, J., F.L. Ramsey, and V.J. Wildman. 1986. Micronesian forest bird survey, 1982: Saipan, Tinian, Aguijan, and Rota. U.S. Fish and Wildlife Service Report. 143pp.

Esselstyn, J., J.B. Cruz, L.L. Williams, and N. Hawley. 2003. Wildlife and Vegetation surveys, Aguiguan 2002. Technical report #9 CNMI-DFW. Unpublished report by Division of Fish and Wildlife Wildlife Section, Saipan, CNMI. 55 pp.

Kessler, C.C. 2002a. Forest loss on Anatahan Island, CNMI; A Photo Comparison 1944-1999. Unpublished report In; Anatahan Feral Ungulate Project 2002-2006 to the US Navy from US Fish and Wildlife Service.

Kessler, C.C., 2002b. Eradication of feral goats and pigs and consequences for other biota on Sarigan Island, Commonwealth of the Northern Mariana Islands. Pp 132-140 in Turning the tide: the eradication of invasive species, Proceedings to the International Conference on Eradication of Island Invasives (C.R. Veitch and M.N. Cloud, eds). Occasional Paper of the IUCN Species Survival Commission 27. 414 pp.

Rice, C.G.. 1991. Goat removal from Aguijan Island: lessons for future efforts. Trans. West. Sect. Wildl. Soc. 27: 42-46.

Worthington, D.J., A.P. Marshall, G.J. Wiles, and C.C. Kessler. 2001. Abundance and management of Mariana fruit bats and feral ungulates on Anatahan, Mariana Islands. Pacific Conserv. Biol. 7(2): 134-142.

3.0 APPENDICES

3.1 STATUS AND TRENDS OF THE LAND BIRD AVIFAUNA ON TINIAN AND AGUIGUAN, MARIANA ISLAND

3.2 SYSTEMATIC RODENT MONITORING: A STUDY OF THE SMALL MAMMALS IN THE MARIANA ISLANDS

3.3 ASSESSMENT FOR PACIFIC SHEATH - TAILED BATS (*EMBALLONURA SEMICAUDATA ROTENSIS*) ON AGUIGUAN, COMMONWEALTH OF THE NORTHERN MARIANA ISLANDS

APPENDIX 3.1

STATUS AND TRENDS OF THE LAND BIRD AVIFAUNA ON TINIAN AND AGUIGUAN, MARIANA ISLANDS

Richard J. Camp[1], Thane K. Pratt[2], Fred Amidon[3], Ann P. Marshall[3], Shelly Kremer[3], and Megan Laut[3]

[1]U.S. Geological Survey, Hawai`i Cooperative Studies Unit,
Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, Hawai`i

[2]U.S. Geological Survey, Pacific Island Ecosystems Research Center,
Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, Hawai`i

[3] U.S. Fish and Wildlife Service Pacific Islands Office
300 Ala Moana Boulevard, Honolulu, Hawai`i

CITATION
Camp, R. J., T. K. Pratt, F. Amidon, A. P. Marshall, S. Kremer, and M. Laut. (2009). Status and trends of the land bird avifauna on Tinian and Aguiguan, Mariana Islands. Appendix 3.1 *in* Terrestrial Resource surveys of Tinian and Aguiguan, Mariana Islands, 2008. U.S. Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office, Honolulu, HI.

Keywords: abundance estimation; Aguiguan; bird counts; Mariana Islands; point-transect sampling; Tinian; trends

March 2009

Table of Contents

Abstract 3
Introduction 4
Methods 4
Results 8
Discussion 10
Acknowledgements 14
References 15

List of Figures

Figure 1. Island of Tinian 17
Figure 2. Island of Aguiguan. 18
Figure 3. Density estimates for native and alien Tinian land birds by year 20
Figure 4. Density estimates for native and alien Tinian land birds by region and year... 26
Figure 5. Density estimates for the Tinian Monarch. 27
Figure 6. Density estimates for native and alien Aguiguan land birds by year. 28
Figure 7. Vegetation changes in central Aguiguan. 30

List of Tables

Table 1. List of birds detected on Tinian. 31
Table 2. Population density and abundance estimates for Tinian land birds. 33
Table 3. Trends in Tinian land bird densities among years. 35
Table 4. Results for fixed effects measures in Tinian land bird densities. 37
Table 5. Results of Tinian Monarch densities by habitat 38
Table 6. Measures of precision in Tinian Monarch densities. 39
Table 7. List of birds detected on Aguiguan. 40
Table 8. Population density and abundance estimates for Aguiguan land birds. 42
Table 9. Comparison of density and change in native land bird populations from Tinian, Aguiguan, and Saipan. 43

List of Appendices

Appendix 1. Species data and models 44
Appendix 2. Results from region and year analyses for Tinian land birds. 49
Appendix 3. Break down of the Tinian Monarch population by habitat and year 63

Abstract

Avian surveys were conducted on the islands of Tinian and Aguiguan, Marianas Islands, in 2008 by the U.S. Fish and Wildlife Service to provide current baseline densities and abundances and assess population trends using data collected from previous surveys. On Tinian, during the three surveys (1982, 1996, and 2008), 18 species were detected, and abundances and trends were assessed for 12 species. Half of the 10 native species—Yellow Bittern (*Ixobrychus sinensis)*, White-throated Ground-Dove (*Gallicolumba xanthonura*), Collared Kingfisher (*Todiramphus chloris*), Rufous Fantail (*Rhipidura rufifrons*), and Micronesian Starling (*Aplonis opaca*)—and one alien bird—Island Collared-Dove (*Streptopelia bitorquata*)—have increased since 1982. Three native birds—Mariana Fruit-Dove (*Ptilinopus roseicapilla*), Micronesian Honeyeater (*Myzomela rubratra*), and Tinian Monarch (*Monarcha takatsukasae*)—have decreased since 1982. Trends for the remaining two native birds—White Tern (*Gygis alba*) and Bridled White-eye (*Zosterops saypani*)—and one alien bird—Eurasian Tree Sparrow (*Passer montanus*)—were considered relatively stable. Only five birds—White-throated Ground-Dove, Mariana Fruit-Dove, Tinian Monarch, Rufous Fantail, and Bridled White-eye—showed significant differences among regions of Tinian by year. Tinian Monarch was found in all habitat types, with the greatest monarch densities observed in limestone forest, secondary forest, and tangantangan (*Leucaena leucocephala*) thicket and the smallest densities found in open fields and urban/residential habitats. On Aguiguan, 19 species were detected on one or both of the surveys (1982 and 2008), and abundance estimates were produced for nine native and one alien species. Densities for seven of the nine native birds—White-throated Ground-Dove, Mariana Fruit-Dove, Collared Kingfisher, Rufous Fantail, Bridled White-eye, Golden White-eye (*Cleptornis marchei*), and Micronesian Starling—and the alien bird—Island Collared-Dove—were significantly greater in 2008 than 1982. No differences in densities were detected between the two surveys for White Tern and Micronesian Honeyeater. Three native land birds—Micronesian Megapode (*Megapodius laperouse*), Guam Swiftlet (*Collocalia bartschi*), and Nightingale Reed-Warbler (*Acrocephalus luscinia*)—were either not detected during the point-transect counts or the numbers of birds detected were too small to estimate densities for either island. Increased military operations on Tinian may result in increases in habitat clearings and the human population, which would expand human dominated habitats, and declines in some bird populations would be likely to continue or be exacerbated with these actions. Expanded military activities on Tinian would also mean increased movement between Guam and Tinian, elevating the probability of transporting the Brown Tree Snake (*Boiga irregularis*) to Tinian.

Introduction

The Department of Defense (DOD) has proposed expanding military operations in the Mariana Islands. To determine the future impacts of military operations on bird populations on these islands, the DOD contracted the U.S. Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office, to coordinate avian surveys on the islands of Tinian and Aguiguan in the Commonwealth of the Northern Mariana Islands (CNMI). The survey data will be used to establish population baseline information to compare with any later change in status and distribution of the birds.

Current avian population estimates were calculated for the whole island for both Tinian and Aguiguan, and by regions for Tinian Island. These estimates were compared with results from a previous survey of both islands that was undertaken in 1982 by Engbring *et al*. (1986), yielding trends spanning 27 years. On Tinian, trends in bird populations across the island and within regions were compared from three surveys: the 1982 Engbring *et al*. survey, a survey in 1996 by the U.S. Fish and Wildlife Service (unpublished data, Lusk *et al*. 2000), and again in 2008. Aguiguan was surveyed in 1982 and 2008, and end-point comparisons were used to assess population changes. Particular attention was given to assess the status of the Tinian Monarch. Formerly listed as an endangered species, the monarch was delisted on September 21, 2004 (69 FR 65367) and is being monitored by the Fish and Wildlife Service through field surveys of distribution and abundance and tracking of land use and development on Tinian.

Methods

Survey area

Tinian

Tinian is the second largest of the CNMI islands at 101.01 km^2 (15^o 00` N, 145$^o$ 35` E). The island consists of low-lying plateaus and a gentle limestone ridge dominated by Puntan Carolinas (elevation 196 m). The vegetation of Tinian currently consists of mixed second-growth forests, grassy savannas, and introduced forests, most of which are tangantangan (*Leucaena leucocephala*) thickets (Engbring *et al*. 1986). The little native vegetation that remains on Tinian (5%; Engbring *et al*. 1986) has been greatly altered by centuries of human use and non-native species and is basically confined to a few cliffs and adjacent steep limestone slopes (Engbring *et al*. 1986).

Aguiguan

Aguiguan is a small, uninhabited island located 8 km southwest of Tinian (7.09 km^2; 14^o 51` N, 145$^o$ 33` E). It is made up of several concentric plateaus bounded by steep scarps, and the top most plateau is about 150 m in elevation. Like other CNMI islands, the vegetation on Aguiguan has been extensively altered by human activity, so the available native forest is limited. In addition, the island has a large feral goat (*Capra hircus*) population which continues to alter the native forest.

Bird surveys

On Tinian, the baseline survey conducted between 27 April and 8 May 1982 sampled a total of 216 stations on 10 transects with representative island-wide coverage across geography and habitats (Engbring *et al*. 1986; Figure 1). Placement of transects was random-systematic (Engbring *et al*. 1986). These transects were located and resurveyed during both the 1996 (28 August – 1 September) and 2008 (14 – 19 June) surveys. An additional four transects were sampled during the 2008 survey for a total of 254 stations (transect 11 – 9 stations; transect 12 – 9 stations; transect 13 – 14 stations; and transect 14 – 5 stations). The four transects were added to increase the sampling of native limestone forest and improve density estimates for Tinian Monarch.

On Aguiguan, an island-wide survey consisting of 66 stations on four transects (random-systematic placement) was conducted on 2 and 3 June 1982, and a partial survey (transects 1 and 2 only) was conducted on 10 and 11 March 1982 (Engbring *et al*. 1986; Figure 2). Data from only the June survey were used in this study because all stations were sampled and the survey month coincides with the 2008 survey. All four transects were located and resurveyed during the 2008 (25 – 27 June) survey. An additional transect of 14 stations was sampled during the 2008 survey for a total of 80 stations. This transect was added to increase the numbers of bird detected and to sample the top most plateau; however, placement of this transect on the plateau was random.

All surveys followed standard point-transect methods, consisting of 8-minute counts, where horizontal distances to all birds heard and/or seen were measured and recorded (see Engbring *et al*. 1986 for details). Sampling conditions recorded included cloud cover, rain, wind, noise level, and habitat type, and these were later used as covariates in density calculations (see *Population status* below). Counts commenced at sunrise and continued up to four hours and were conducted only under prescibed conditions.

Stations were surveyed by two observers in 1982 and one observer in 1996 and 2008. Data from only one counter were used for each station from the 1982 Tinian surveys, and the best counters were identified based on their experience and survey proficiency. Engbring *et al*. (1986) analyzed bird detections from all observers to estimate bird densities. For our analysis, we used detections from only one observer to recalculate densities for the 1982 Tinian survey, thus matching the 1996 and 2008 survey effort. Calculating densities from only one of the counters is a conservative approach and ensures sampling independence. This approach approximately halved the number of birds detected; however, our density estimates were generally greater than, but otherwise similar to, those of Engbring *et al*. (see their Table 8; 1986). On Tinian the 95% confidence intervals bracketed Engbring *et al*.'s estimates for all but four birds—Mariana Fruit-Dove, Tinian Monarch, Rufous Fantail, and Bridled White-eye. Differences may have resulted from analytical procedures such as selecting different truncation distances, selecting different models to estimate densities, and analytical advances in distance sampling (see Johnson *et al*. 2006), in addition to estimating densities using detections from only one of the counts (Tinian only). Data from both counters were used to estimate 1982 densities on Aguiguan because it was a small data set, and the sampling effort was adjusted appropriately.

Population status

Population status was calculated as density (birds/km^2) and number of birds (density by habitat type multiplied by habitat type area). Density was calculated using the program DISTANCE, version 5.0, release 2 (Thomas *et al*. 2006) from species-specific global detection functions, where data were post-stratified by survey. Data were right-truncated to facilitate model fitting (Buckland *et al*. 2001:16). Candidate models included half-normal and hazard-rate detection functions with expansion series of order two (Buckland *et al*. 2001:361, 365). Sampling covariates were modeled in the multiple-covariate distance engine of DISTANCE (Thomas *et al*. 2006, Marques *et al*. 2007). The model with the lowest Akaike Information Criterion (AIC) was used to select the detection function that best approximated the data. Covariates (sampling conditions, habitat types, and survey year) were used to generate the global detection function when the best approximating model was improved by four or more AIC units (Appendix 1). Variances and confidence intervals were derived by log-normal based methods. Survey-specific, density-by-station values were generated for the population trends analyses (see *Population trends* below) from the global detection function using the post-stratification-by-sample option. Area of habitat types came from Engbring *et al*. (1986) and U.S. Fish and Wildlife Service (2008). The area of habitat types was not available for the 1996 Tinian survey; therefore, we used the area by habitat types from Engbring *et al*. to calculate the 1996 numbers of birds. This may slightly underestimate the population size if there was more secondary forest in 1996 than 1982. Agriculture habitat type (combined agroforestry and cultivated habitat type classifications) was not used to calculate numbers of birds because the area of this habitat is very small relative to the island (< 2%), the area of the agriculture habitat type has declined (190 ha in 1982 to 174 ha in 2008; U.S. Fish and Wildlife Service 2008), and insufficient numbers of stations were established in the agriculture habitat type to produce reliable density estimates (1 in 1982, 4 in 1996, and 2 in 2008), thus it was under-sampled. In addition, coastal and urban/residential habitat types were inconsistently and under-sampled (coastal: 3 stations in 1982, 1 in 1996, and 0 in 2008; urban/residential: 0 stations in 1982 and 1996, and 7 in 2008), and were not used in calculating population estimates. On Aguiguan, the 1982 estimates of the area of habitat types were not reliable; therefore, numbers of birds were calculated only for the 2008 survey.

Population trends

Change in bird density among the three annual estimates on Tinian was assessed with repeated measures analysis of variance (ANOVA: PROC MIXED; SAS Institute Inc., Cary, NC). To stabilize the error variance, density-by-station values were ln(density+1) transformed. Repeated measures ANOVA was also used to assess change in bird density within regions among the three annual estimates. Stations were treated as the random factor, and because the number of repeated measures was too small to fit a covariance model, we assumed the variance-covariance structure was a compound symmetry, homogeneous-variance model (Littell *et al*. 1996). Degrees of freedom was adjusted using the Kenward-Roger adjustment statement, and a Tukey's adjustment was used to control experiment-wise alpha = 0.05 for multiple-comparison procedures. A further analysis was conducted to assess differences by habitat type for Tinian Monarch from the 2008 survey using a one-way ANOVA (PROC MIXED) with the same options as those used in the repeated measures models. The agriculture habitat was dropped from this analysis because only two stations were sampled within the habitat during the 2008 survey.

End-point comparisons of the Aguiguan bird densities were compared using a two-sample *z*-test. Comparing density estimates using *z*-tests is the recommended method (L. Thomas, pers. comm.) and is an extension of the method listed in Buckland *et al*. (2001:353).

Results

Tinian

A total of 18 species was detected during one or more of the three surveys on Tinian (Table 1). Sufficient numbers of individuals were detected for 10 native and two alien species to calculate density and abundance estimates. Bridled White-eye and Rufous Fantail were the most abundant birds, whereas White-throated Ground-Dove and Yellow Bittern were the least abundant birds (Table 2). Half of the 10 native species—Yellow Bittern, White-throated Ground-Dove, Collared Kingfisher, Rufous Fantail, and Micronesian Starling—have increased since 1982 (Table 3, Figure 3). Three native birds—Mariana Fruit-Dove, Micronesian Honeyeater, and Tinian Monarch—have decreased in the same period. Although these declines were not linear (Figure 3), the overall changes between 1982 and 2008 were significant (Table 3). Trends for the remaining two native birds—White Tern and Bridled White-eye—were considered relatively stable. The alien bird—Island Collared-Dove—increased since 1982 or remained relatively stable, respectively (Tables 2 and 3, Figure 3). Although Eurasian Tree Sparrow densities increased 98% from 2 to 110 birds/km^2 between 1982 and 2008, their densities were not estimated well enough to make strong conclusions, and we conclude they have remained relatively stable.

Only 5 birds—White-throated Ground-Dove, Mariana Fruit-Dove, Tinian Monarch, Rufous Fantail, and Bridled White-eye—showed significant differences among regions by year (Table 4, Appendix 2). Between 1982 and 2008, White-throated Ground-Dove densities increased in the Diablo and Hagoi regions, and Rufous Fantail densities increased in the Carolinas and Masalog regions (Figure 4). Mariana Fruit-Dove densities declined in the Carolinas, and Tinian Monarch and Bridled White-eye densities declined in the Diablo region. In addition, densities of 3 birds—White Tern, Micronesian Honeyeater, and Micronesian Starling—differed by year and region but the year-region interaction was insignificant (Table 4, Figure 4, Appendix 2). White Tern densities were greater in Diablo than in Hagoi, but densities in those regions were not different from densities in the Carolinas and Masalog. Densities of Micronesian Honeyeater were greater in the Carolinas and Diablo regions than in the Hagoi and Masalog regions. Micronesian Starling densities were lower in Masalog than in the other regions.

Tinian Monarch densities have declined both temporally (survey year comparisons) and spatially (regional comparisons). We also tested for differences in Tinian Monarch densities among the different habitat types. Tinian Monarchs were found in all habitat types, but their densities were not distributed evenly among the habitats (Figure 5). Based on the 2008 survey, the greatest monarch densities were observed in limestone forest, secondary forest, and tangantangan thicket. The smallest densities were found in open field and urban/residential habitats. Monarch densities in limestone and secondary forests were greater than those in open field and urban/residential habitat but not different from densities in tangantangan thicket (Table 5, Appendix 3).

We used the coefficient of variation (CV=SE/density) to evaluate Tinian Monarch estimator certainty by comparing the variability in densities calculated with and without the newly established transects. During the 2008 survey, 37 stations were sampled on four new transects. All of the stations were in limestone forest habitat, except that two stations on transect 13 were located in tangantangan thicket habitat. Both of these habitats contain high densities of Tinian Monarch (Table 5). Incorporating the new transects increased the precision of monarch estimates in limestone forest habitat by more than 50% compared to estimates from just the original transects (Table 6). Sampling the new transects helped to improve precision in monarch densities by 15% in the Carolinas and Diablo regions, and most of the improvement was in estimates from the Carolinas Region. Overall, the precision of the island-wide monarch estimate was increased by almost 9%.

Aguiguan

A total of 19 species was detected on the Aguiguan surveys (Table 7). Sufficient numbers of individuals were detected for nine native and one alien species to calculate density and abundance estimates. Bridled White-eye was the most abundant bird at over 44,000 birds on the small 7 km^2 island, and Collared Kingfisher and Island Collared-Dove were the least abundant birds (Table 8). Densities for seven of the nine native birds—White-throated Ground-Dove, Mariana Fruit-Dove, Collared Kingfisher, Rufous Fantail, Bridled White-eye, Golden White-eye, and Micronesian Starling—were significantly greater in 2008 than 1982 (Table 8, Figure 6). No differences in densities were detected between the two surveys for White Tern and Micronesian Honeyeater. Densities of the alien Island Collared-Dove had increased significantly between 1982 and 2008.

Trends Across Islands

Densities have increased or remained stable for 84% (21 of 25 populations) of the nine native land bird species shared between Saipan (Camp *et al*., in press) and one or both of the islands covered in this study (Table 9). White-throated Ground-Dove and Micronesian Starling populations increased on all three islands. Yellow Bittern, Collared Kingfisher, and Bridled White-eye populations either increased or remained stable. Change in the status of the Mariana Fruit-Dove, Micronesian Honeyeater, Rufous Fantail, and Golden White-eye populations was mixed among the islands.

Discussion

Island Trends

Abundances of half of the 10 native birds on Tinian— Yellow Bittern, White-throated Ground-Dove, Collared Kingfisher, Rufous Fantail, and Micronesian Starling—and seven of nine native birds on Aguiguan—White-throated Ground-Dove, Mariana Fruit-Dove, Collared Kingfisher, Rufous Fantail, Bridled White-eye, Golden White-eye, and Micronesian Starling—have increased since the 1982 survey. In addition, three native birds on both islands have remained stable—White Tern on both islands, Bridled White-eye on Tinian, and Micronesian Honeyeater on Aguiguan. Large increases in densities of Yellow Bittern, Rufous Fantail, and Micronesian Starling on Tinian, and Rufous Fantail on Aguiguan support increasing their status classification. Changes in the other birds were not sufficient to warrant reclassification. Reichel and Glass (1991) listed Yellow Bittern as rare, and now at more than 1,600 birds the species can be considered uncommon—observing them in representative habitat is not certain but likely. Rufous Fantail and Micronesian Starling on Tinian may be considered abundant. Abundances of about 86,000 and 30,000 birds, respectively, make finding them in large numbers within representative habitat a certainty. Likewise, Rufous Fantail on Aguiguan may be considered abundant at more than 6,400 birds. Alien birds—Island Collared-Dove and Eurasian Tree Sparrow—densities increased on both islands and Tinian, respectively, and both species may be categorized as common or abundant.

No species had declined on Aguiguan, whereas Mariana Fruit-Dove, Micronesian Honeyeater, and Tinian Monarch declined on Tinian. Relatively large numbers of these birds remain on Tinian (> 3,000 individuals), and changes to their abundance status are unwarranted. However, declines for these native species are a concern, especially for the Tinian Monarch, which is endemic to Tinian and listed as threatened by the CNMI and vulnerable by the IUCN. Likely causes for these declines include predation and habitat loss/degradation. One possible explanation for increases in Aguiguan birds has been extensive expansion of secondary forest and brush habitats. About half of the island was cleared for agriculture during the 1930s and 1940s, and those fallow fields are now dominated by *Lantana camara* and other alien plants, and secondary forest (Figure 7). Forests currently cover about 70% of the island, and an additional 20% of the island is occupied primarily by *L. camara* fields, providing habitat for birds.

Trends Across Islands

The U.S. Fish and Wildlife Service conducted a land bird survey on Saipan in 2007 and assessed population trends (Camp *et al*., in press). Comparing trends among the neighboring Mariana Islands of Tinian, Aguiguan, and Saipan provides an index of the species' regional trends. The carnivorous birds—Yellow Bittern and Collared Kingfisher—increased or remained stable. Densities of Yellow Bittern have increased on Tinian and Saipan, but the species is found in very low numbers on Aguiguan. In fact, no birds were detected on count during the 2008 Aguiguan survey, although one was seen along a transect (APM, pers. obs.), and only one bird was detected during the 1982 survey. Yellow Bittern inhabit swamps, marshes, and other grassy habitats, and secondary forest, and bittern may be absent from Aguiguan because very little grass-dominated habitat now occurs on this island. In contrast, bittern may be increasing on Tinian and Saipan where grassy and open habitats have increased.

Trends among the fruit-eating birds—White-throated Ground-Dove and Mariana Fruit-Dove—were mixed, and the pattern does not appear to correspond to increases in human populations. Micronesian Starling, a largely frugivorous species, increased on all three islands. Camp *et al*. (in press) speculated that fruit-eating birds on Saipan may have benefited from the expansion of scarlet gourd (*Coccinia grandis*). This alien, smothering vine, also occurs on Tinian but only locally and has not formed dense canopies. Scarlet gourd is not reported from Aguiguan. Thus, it is likely that scarlet gourd is does not account for much of the increases in the fruit-eating bird populations on Tinian and Aguiguan. Another explanation is that there may be different patterns of hunting across the islands that account for the mixed trends. For example, people have traditionally hunted White-throated Ground-Dove and Mariana Fruit-Dove; however, it is not legal to hunt these doves but current hunting prevalence is unknown.

The insectivorous Rufous Fantail increased on Tinian and Aguiguan but decreased on the more densely human-populated Saipan. Trends for birds with diets including insects, nectar, and fruits were mixed. The Aguiguan population of Bridled White-eye may have increased in response to expansion of secondary forest and lantana field habitats. Habitat change and increased human populations may not be strong enough drivers to effect Bridled White-eye populations on Saipan and Tinian. Golden White-eye is known from the recent fossil record to have formerly occurred on Tinian, where it is now extinct (Craig 1999). The species was detected in large numbers on Aguiguan, and the population there has more than doubled (529 to 1,293 birds/km^2) between 1982 and 2008. Craig (1996, as cited in Craig 1999) estimated Golden White-eye densities on Saipan at about 1,200 birds/km^2, an estimate that roughly matches the 1997 point-transect density (Camp *et al*., in press). The current Golden White-eye densities on Aguiguan were almost twice that reported from Saipan (1,300 and 700 birds/km^2, respectively), and their trends were in opposite directions—increasing on Aguiguan and decreasing on Saipan (Camp *et al*., in press). The Golden White-eye decline on Saipan may be a result of increasing human populations and habitat loss/degradation, whereas these factors are not affecting the population on uninhabited Aguiguan.

Generally, the birds on Tinian, Aguiguan, and Saipan are doing comparatively well for insular species. This is surprising given that nearly all of the native forests on Tinian and Saipan have been lost and that all habitats on Aguiguan suffer from heavy browsing by feral goats and forest regeneration is thus severely selective. Recent surveys on Rota showed that seven of eight bird trends have declined (Amar *et al*. 2008). The only bird to increase on Rota was the Micronesian Starling, which has also increased on the other three islands. Similar to our findings, Amar *et al*. conclude that the loss of forests or the spread of scarlet gourd does not fully explain bird population trends on Rota. Likewise, large-scale climate change, increases in human populations on Rota, Saipan, and Tinian, and Malathion insecticide spraying do not appear to be consistent drivers of bird trends. The status of Brown Tree Snake on Rota, Tinian, and Aguiguan is unknown, but reports of sightings are very rare. Brown Tree Snakes have been frequently sighted on Saipan

(Rodda and Savidge 2007). However, declines in the bird populations do not follow the geographic pattern of snakes spreading across an island, as they did on Guam (Savidge 1987). Further research is needed to identify the causative agents of population change in these four islands.

Rare Species and Those Not Appropriate for Point-Transect Sampling

Three native land birds—Micronesian Megapode, Guam Swiftlet, and Nightingale Reed-Warbler—were either not detected during the point-transect counts or the numbers of birds detected were too few to estimate densities. Point-transect methods may not be appropriate for the very rare megapode and reed-warbler, and the behavior of the swiftlet violates modeling assumptions. A remnant population of a few Micronesian Megapode may persist on Tinian (Wiles *et al*. 1987, U.S. Fish and Wildlife Service 1998a), although no individuals were detected during any of the three point-transect surveys. Wiles *et al*. (1987) speculated that the megapode population on Tinian may originate from birds being brought in by humans or possibly dispersing from nearby populations on Aguiguan or Saipan. Aguiguan supports a small Micronesian Megapode population (U.S. Fish and Wildlife Service 1998a), and about equal numbers of birds were detected during the 1982 and 2008 surveys (14 and 15 birds, respectively). During the 1982 survey on Aguiguan, four Nightingale Reed-Warbler incidental sightings were recorded, but not during the 8-min counts (Engbring *et al*. 1986). The reed-warbler has not been observed on Aguiguan since the mid-1990s and may be extirpated on Aguiguan (U.S. Fish and Wildlife Service 1998b, Esselstyn *et al*. 2003). The Nightingale Reed-Warbler was not detected by the 2008 survey, either during counts or incidentally. The Guam Swiftlet historically occurred on Tinian but is extinct on the island (U.S. Fish and Wildlife Service 1991, Cruz *et al*. 2008); no swiftlets were detected during the three point-transect surveys. Cruz *et al*. (2008) noted that the Aguiguan swiftlet population has probably remained fairly stable between 1987 and 2002; however, it is notable that the numbers of birds detected in 2008 were only 17% of those detected in 1982 (27 and 157 birds, respectively). This apparent decline was further supported by the drop in numbers of birds detected at roosting cave counts between 1985 and 1997-2002 (Cruz *et al*. 2008).

The 1996 White Tern estimate on Tinian was markedly lower than from the other surveys. It is likely that the low tern estimate was an artifact of when the survey was conducted and not an actual change in the tern population. The original survey in 1982 and the most recent 2008 survey occurred early in the year and early in the breeding season (although terns can breed in all months of the year; Niethammer and Patrick-Castilaw 1998), whereas the 1996 survey was conducted in late August and after the breeding season. When not nesting, most individuals spend extended periods at sea (Niethammer and Patrick-Castilaw 1998); therefore portions of the population in 1996 were outside the sampling frame. In addition, the 1996 survey focused on passerines, and not all tern detections may have been recorded (FAA, pers. comm.).

Tinian Monarch concerns

Lusk *et al*. (2000) calculated the 1996 Tinian Monarch abundance at about 55,700 birds, which is 11% less than our estimate of 62,900 birds. This change is due to differences between the analytical procedures. For example, Lusk *et al*. (2000) did not extrapolate densities to abundance for 2,375 ha of open fields, although monarchs were detected in this habitat. After dropping densities from the open fields and adjusting for this area difference, our densities resulted in 48,424 birds, an estimate that fell within their 95% CI. This difference is easily accounted for in differences between our methods, specifically differences in the model selected and advances within program DISTANCE. Lusk *et al*. (2000) calculated their density estimate from a half-normal model with polynomial adjustments and an effective detection radius (EDR) of just over 34 m. We estimated the EDR at 30.18 m from a hazard-rate detection function (without adjustments) and incorporating observers as a covariate, where the smaller EDR resulted in greater densities. Lastly, Lusk *et al*. (2000) used program VCPADJ (Fancy 1997) and a previous version of DISTANCE (Laake *et al*. 1994) to standardize the survey conditions and estimate densities. The updated version of DISTANCE (Thomas *et al*. 2006) we used incorporates all of the modeling in one program and uses an improved technique to account for differences in sampling conditions (Thomas *et al*. 2006, Marques *et al*. 2007).

Estimator certainty usually declines with decreasing density estimates; however, this pattern was not observed for the 2008 Tinian Monarch estimate. There was an almost 3-fold decrease in estimator certainty for the 2008 estimate than that observed for either the 1982 or 1996 estimates. Variability in monarch densities on the new transects was substantially less than that observed on the entire set of original transects and the subset of original transects within the same regions. In the two regions where additional transects were sampled—Carolinas and Diablo—variability in the Tinian Monarch density diverged (see Appendix 2). Variability in the monarch density in the Diablo region remained low even though densities declined. In contrast, uncertainty increased 4-fold in the Carolinas region. The additional stations sampled during the 2008 survey in the Carolinas region reduced variability to the Tinian Monarch estimate, but estimator certainty was poorer than in previous surveys. Adding stations to the limestone forest habitat improved estimator certainty by 50%. Thus, additional stations may be needed to further improve estimator certainty. Allocation of stations for monitoring Tinian Monarch should consider additional sampling in habitats with uncertain estimates including agriculture (CV>100%), urban/residential (CV=69%), and lastly in open field habitat where 23% CV is adequate for trends monitoring. Also, additional sampling could be allocated in the Carolinas region to help reduce the almost 50% CV.

The U.S. Fish and Wildlife Service (2005) post-delisting plan for the Tinian Monarch identified the loss of habitat as a primary threat. The USFWS identified limestone and secondary forests and tangantangan thicket as quality habitat for the monarch (densities of 30.7, 7.7, and 6.0 birds/ha, respectively). Monach densities in 2008 declined dramatically by 79% in limestone forests and substantially by 24% and 27% in secondary forest and tangantangan thicket, respectively, from those reported by U.S. Fish and Wildlife Service (2005). We also show that the monarch population declined over the 27-year period, and the decline between 1996 and 2008 may be attributed to reduced bird density in open field habitat. Continued monitoring of the Tinian Monarch will be necessary to track its long-term survival, especially when the species is faced with population declines, threats such as the potential invasion of the Brown Tree Snake, and habitat lost to the increasing development of Tinian Island.

Bird Monitoring for Conservation on Tinian

The current status of the Brown Tree Snake on Tinian is unknown, but there have been several reports of snakes from Tinian and other CNMI islands (Colvin *et al*. 2005). Interdiction measures to prevent the introduction and establishment of snakes are crucial for the survival of CNMI land birds. If established, the Brown Tree Snake will decimate the avifauna (Savidge 1987, Wiles *et al*. 2003). Military operations are likely to increase traffic between Guam and Tinian, increasing the probability of transporting Brown Tree Snake to Tinian.

Military operations are likely to result in increases in the human population and land use conversion, which will expand human dominated habitats. Between 1980 and 2000, the human population on Tinian increased 309 % from 866 to 3,540 people, respectively (CNMI Department of Commerce 2001). Human increases were concentrated in and around the main settlement, San Jose, and not in the northern two-thirds of the island leased by the military. Humans have predominantly increased in the Carolinas region (which includes much of San Jose), where both alien birds and four native birds—Yellow Bittern, Collared Kingfisher, Rufous Fantail, and Micronesian Starling—increased. In contrast, Tinian Monarch, a native bird typically associated with forests, especially limestone forests, declined in the Carolinas region where housing, roads, and services have expanded. These bird trend patterns could well continue or be exacerbated by increasing military actions.

Acknowledgements

Analyses of the bird monitoring data from Tinian and Aguiguan were conducted by the Hawaii Forest Bird Interagency Database Project, a project of the U.S. Geological Survey-Pacific Island Ecosystems Research Center (PIERC) and the cooperating agencies Pacific Islands Office of the U.S. Fish and Wildlife Service and University of Hawaii Pacific Cooperative Studies Unit. We thank the field biologists who organized and collected the data: C. Aguon, F. Amidon, P. Ashman, J. Engbring, S. Kremer, M. Laut, A. Marshall, and P. Pyle. Also, we thank everyone who helped cut transects over the years, the CNMI Division of Fish and Wildlife for their assistance and permits, and the Mayor and people of Tinian for their support. The following reviewers helped improve the manuscript: K. Brinck, L. Mehrhoff, and L. Williams. We also thank H. Kozuba for preparation of data described here and assistance with generating tables and figures. This study was funded by the U.S. Navy, Pacific Islands Office of the Fish and Wildlife Service, and by PIERC. Any use of trade, product, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the U.S. Government.

References

Amar, A., Amidon, F., Arroyo, B. Esselstyn, J.A., and Marshall, A.P. (2008) Population trends of the forest bird community on the Pacific Island of Rota, Mariana Islands. Condor 110:421-427.

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L. and Thomas, L. (2001) Introduction to Distance Sampling: Estimating abundance of biological populations. Oxford: Oxford University Press.

Camp, R.J., T.K. Pratt, A.P. Marshall, F. Amidon, and L.L. Williams. (In press). Status and trends of the land bird avifauna on Saipan, Mariana Islands, with emphasis on the endangered Nightingale Reed-Warbler (*Acrocephalus luscinia*). Bird Conservation International.

Colvin, B.A., Fall, M.W., Fitzgerald, L.A. and Loope, L.L. (2005) *Review of brown treesnake problems and control programs: report of observations and recommendations*. Unpublished report to the U.S. Department of Interior, Office of Insular affairs, March 2005.

CNMI Department of Commerce. (2001) Census 2000 update. CNMI Department of Commerce website accessed 11 August 2008: http://www.commerce.gov.mp/new/PDF/central_statistic/Cen/2000%20Cen%20Update.pdf.

Craig, R.J. (1999) Conservation of endangered White-eyes (Zosteropidae) in the tropical Pacific. Published by Bird Conservation Research, Inc., Putnam, CT. 8pp.

Cruz, J.B., Kremer, S.R., Martin, G., Williams, L.L. and Camacho, V.A. (2008) Relative abundance and distribution of Mariana Swiftlets (Aves: Apodidea) in the Northern Mariana Islands. Pacific Science 62:233-246.

Esselstyn, J., Cruz, J.B., Williams, L.L., and Hawley, N. (2003) Wildlife and vegetation surveys Aguiguan 2002. CNMI Division of Fish and Wildlife Technical Report #9.

Engbring, J., Ramsey, F.L. and Wildman, V.J. (1986) Micronesian forest bird survey, 1982: Saipan, Tinian, Aguijan, and Rota. U.S. Fish and Wildlife Service Report. 143pp.

Fancy, S.G. (1997) A new approach for analyzing bird densities from variable circular-plot counts. Pacific Science 50:107-114.

Johnson, L., Camp, R.J., Brinck, K.W. and Banko, P.C. (2006) Long-term population monitoring: lessons learned from an endangered passerine in Hawai`i. Wildlife Society Bulletin 34:1055-1063.

Laake, J.L., Buckland, S.T., Anderson, D.R. and Burnham, K.P. (1994) DISTANCE user's guide V2.1. Colorado Cooperative Fish and Wildlife Research Unit, Colorado Statue University, Fort Collins, CO.

Littell, R.C., Milliken, G.A., Stroup, W.W. and Wolfinger, R.D. (1996) SAS system for mixed models. SAS Institute Inc., Cary, NC.

Lusk, M., Hess, S., Reynolds, M. and Johnston, S. (2000) Population status of the Tinian Monarch (*Monarcha takatsukasae*) on Tinian, Commonwealth of the Northern Mariana Islands. Micronesica 32:181-190.

Marques, T.A., Thomas, L., Fancy, S.G. and Buckland, S.T. (2007) Improving estimates of bird density using multiple-covariate distance sampling. Auk 124:1229-1243.

Niethammer, K.R. and Patrick-Castilaw, L.B. (1998) White Tern (Gygis alba). *In* The Birds of North America, No. 371 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Reichel, J.D. and Glass, P.O. (1991) Checklist of the birds of the Mariana Islands. `Elepaio 51:3-11.

Rodda, G.H., and Savidge, J.A. (2007) Biology and impacts of Pacific island invasive species. 2. *Bioga irregularis*, the Brown Tree Snake (Reptilia: Columbridae). Pacific Science 61:307-324.

Savidge, J.A. (1987) Extinction of an island forest avifauna by an introduced snake. Ecology 68:660-668.

Thomas, L., Laake, J.L., Strindberg, S., Marques, F.F.C., Buckland, S.T., Borchers, D.L., Anderson, D.R., Burnham, K.P., Hedley, S.L., Pollard, J.H., Bishop, J.R.B. and Marques, T.A. (2006) Distance 5.0. Release 2. Research Unit for Wildlife Population Assessment, University of St. Andrews, U.K. http://www.ruwpa.st-and.ac.uk/distance/.

U.S. Fish and Wildlife Service. (1998a) Recovery plan for the Micronesian Megapode (*Megapodius laperouse laperouse*). Portland, OR.

U.S. Fish and Wildlife Service. (1998b) Recovery plan for the Nightingale Reed-warbler, *Acrocephalus luscinia*. Portland, OR.

U.S. Fish and Wildlife Service. (1991) Recovery plan for the Mariana Islands population of the Vanikoro Swiftlet, *Aerodramus vanikorensis bartschi*. Portland, OR.

U.S. Fish and Wildlife Service. (2005) Post-delisting monitoring plan for the Tinian Monarch, *Monarcha takatsukasae*. Endangered Species Division, Pacific Islands Fish and Wildlife Office, Honolulu, HI.

U.S. Fish and Wildlife Service. (2008) Terrestrial resource surveys of Tinian and Aguiguan, Mariana Islands. U.S. Fish and Wildlife Service, Pacific Islands Fish and Wildlife Office, Honolulu, HI.

Wiles, G.J., Bart, J., Beck, R.E., Jr. and Aguon, C.F. (2003) Impacts of the brown tree snake: patterns of decline and species persistence in Guam's avifauna. Conservation Biology 17:1350-1360.

Wiles, G.J., Beck, R.E. and Amerson, A.B. (1987) The Micronesian Megapode on Tinian, Mariana Islands. `Elepaio 47:1-3.

Figure 1. Island of Tinian showing the survey transects and regions (as defined by Engbring et al. 1986). Transects 1-10 were counted during all three surveys, and transects 11-14 were established and counted during the 2008 survey.

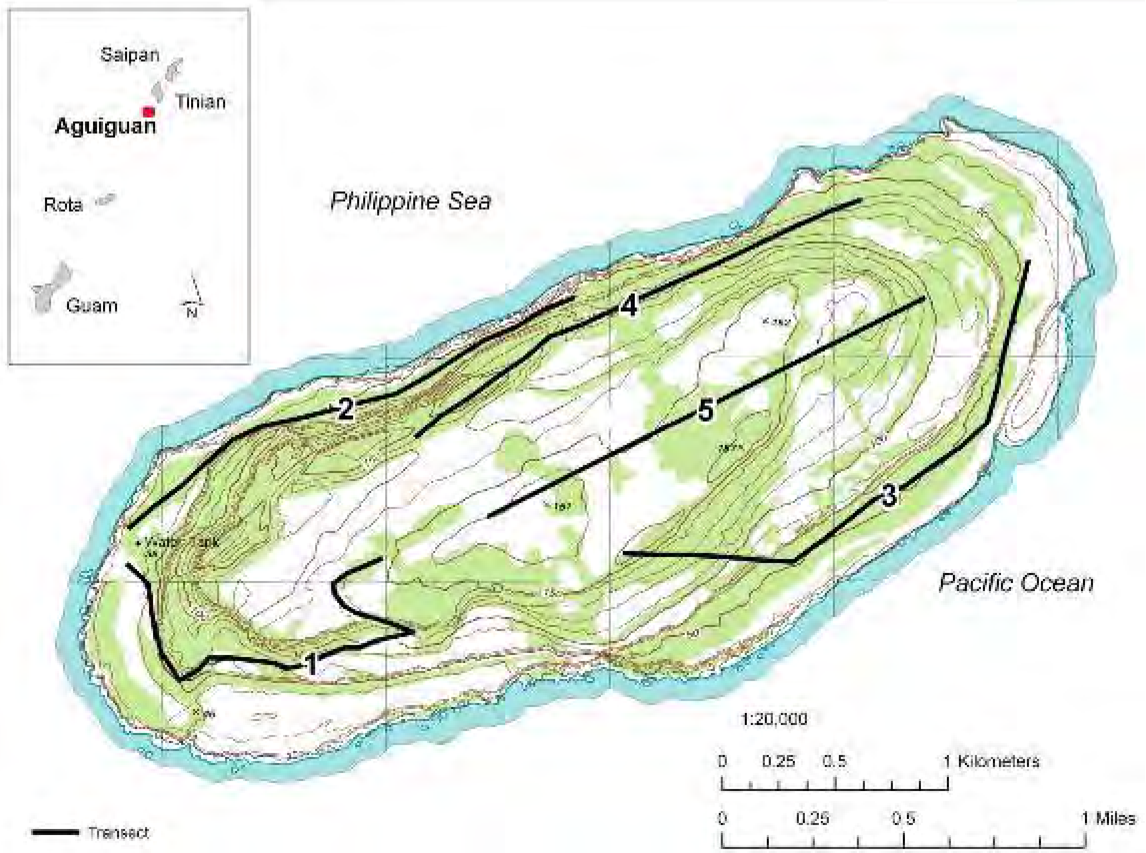
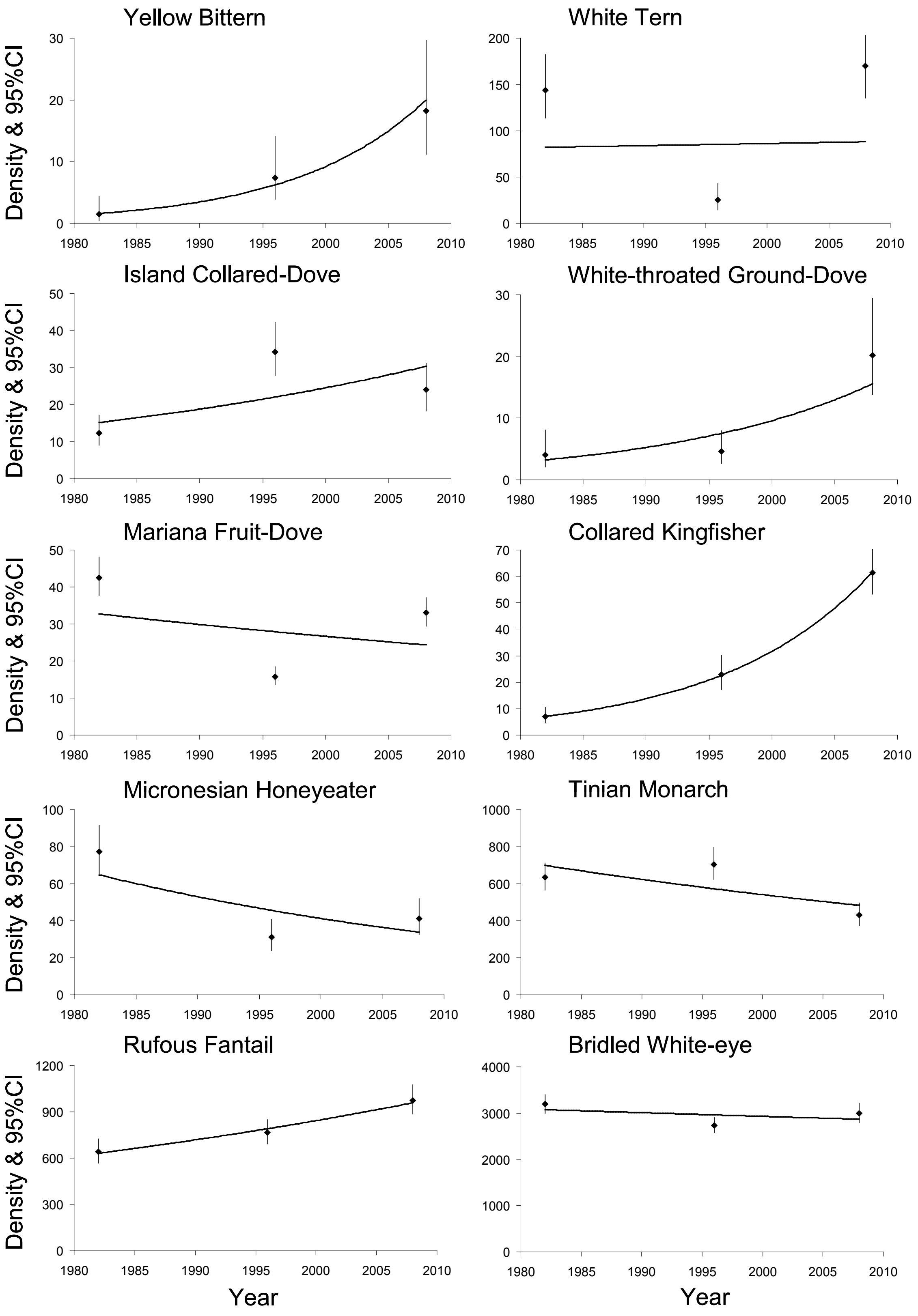


Figure 2. Island of Aguiguan showing the survey transects. Transects 1-4 were counted during both the 1982 and 2008 surveys, whereas transect 5 was established and counted during the 2008 survey.


Yellow Bittern
White Tern
Island Collared-Dove
White-throated Ground-Dove
Mariana Fruit-Dove
Collared Kingfisher
Micronesian Honeyeater
Tinian Monarch
Rufous Fantail
Bridled White-eye
Density & 95%CI
Year
1980
1985
1990
1995
2000
2005
2010
0
10
20
30
40
50
60
70
80
100
150
200
300
400
600
800
900
1000
1200
2000
3000
4000

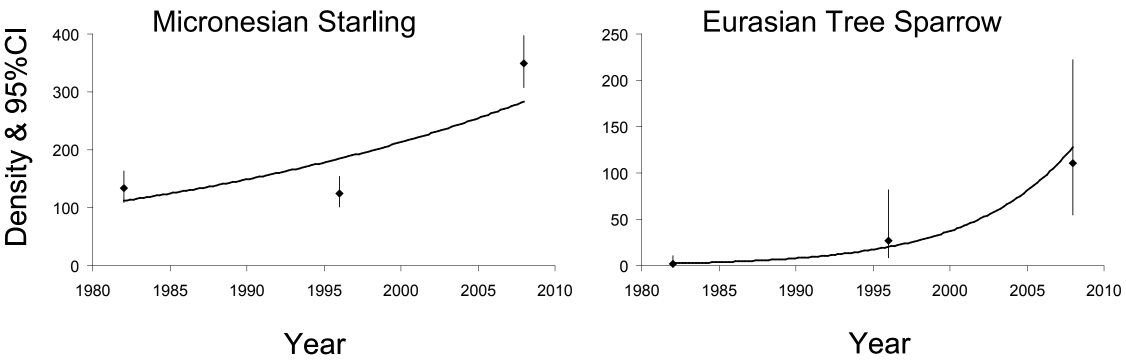
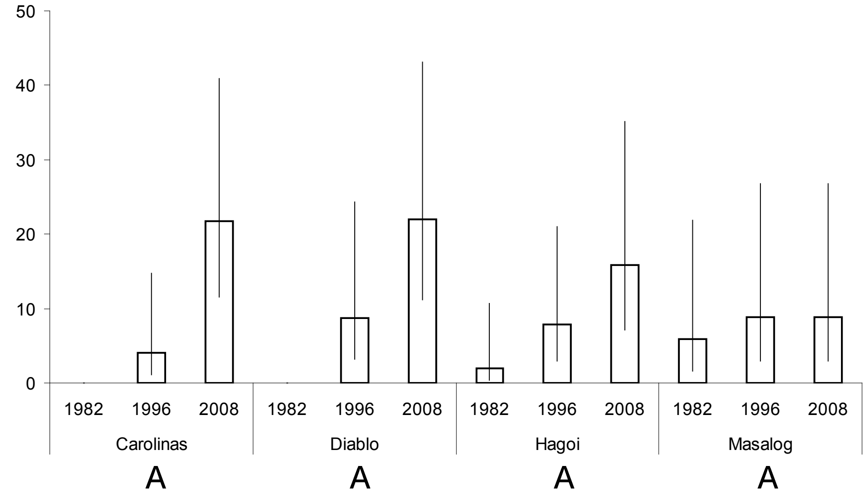
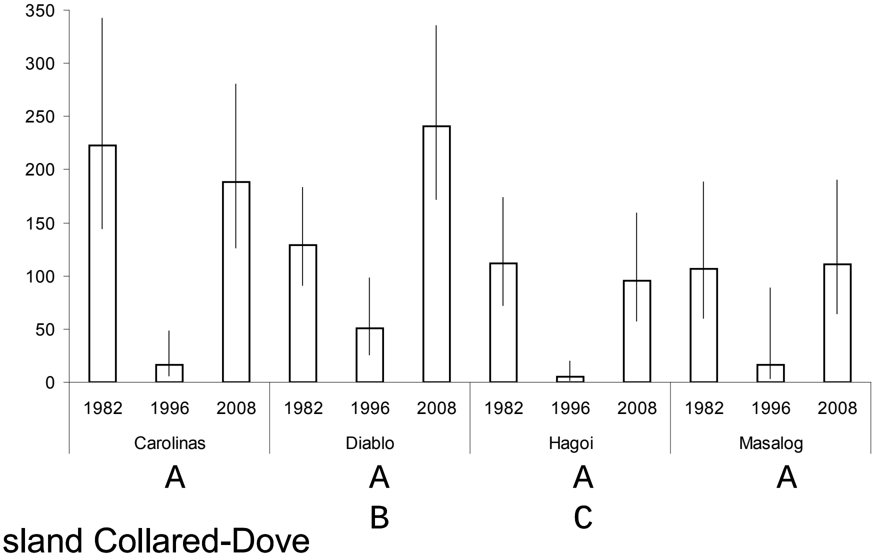
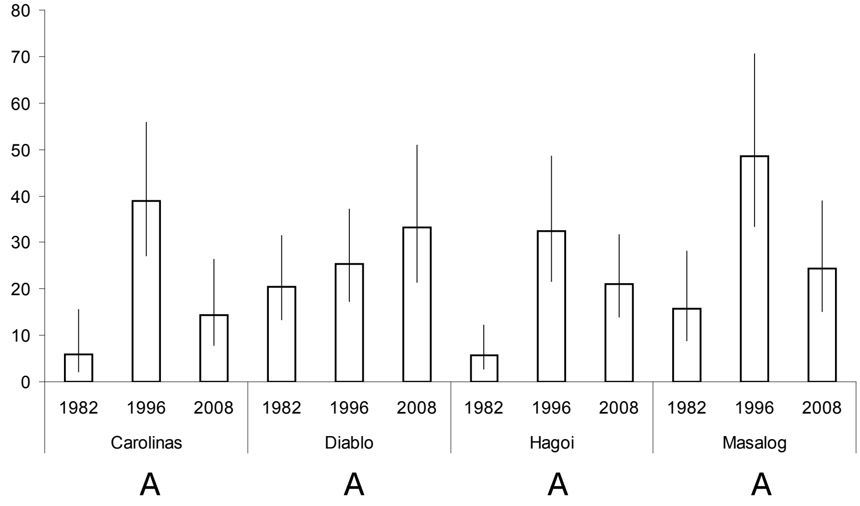


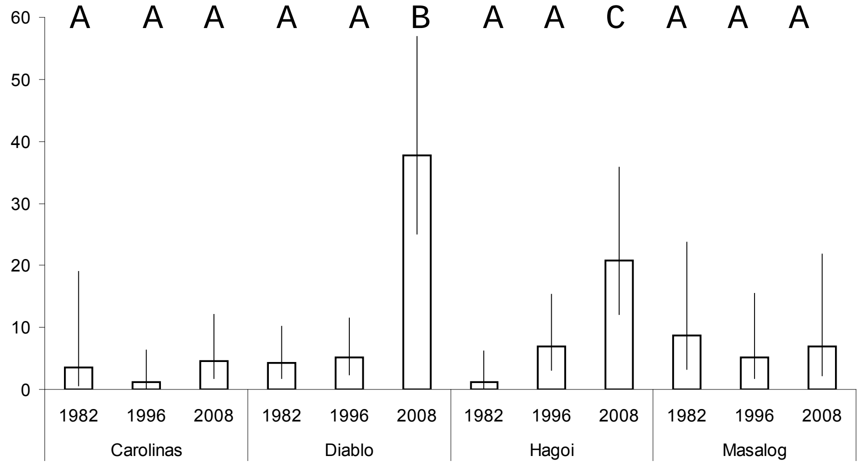
Figure 3. Density estimates (birds/km2 and 95% CI) for native and alien Tinian land birds from three point-transect surveys. Densities were fitted with a line from an exponential model to illustrate population trends.


Yellow Bittern

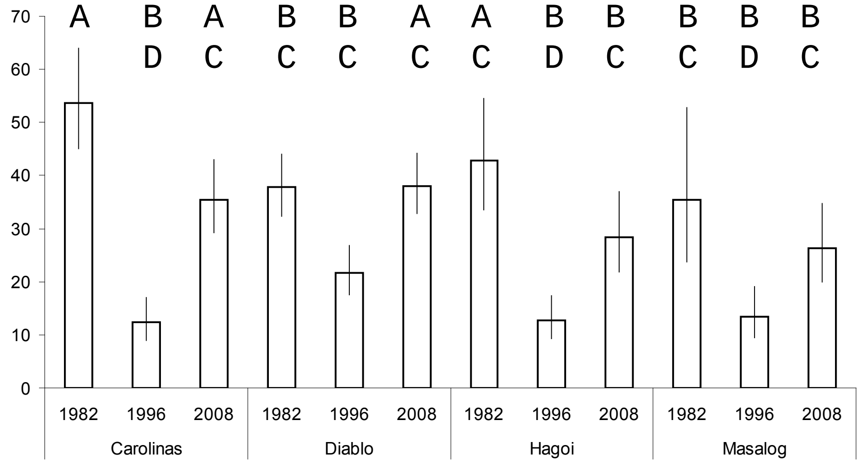

White Tern


Island Collared-Dove

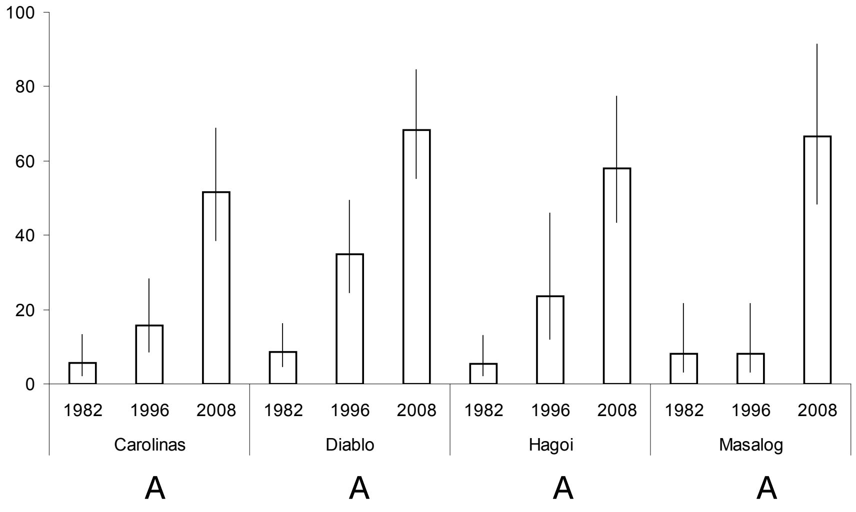
White-throated Ground-Dove



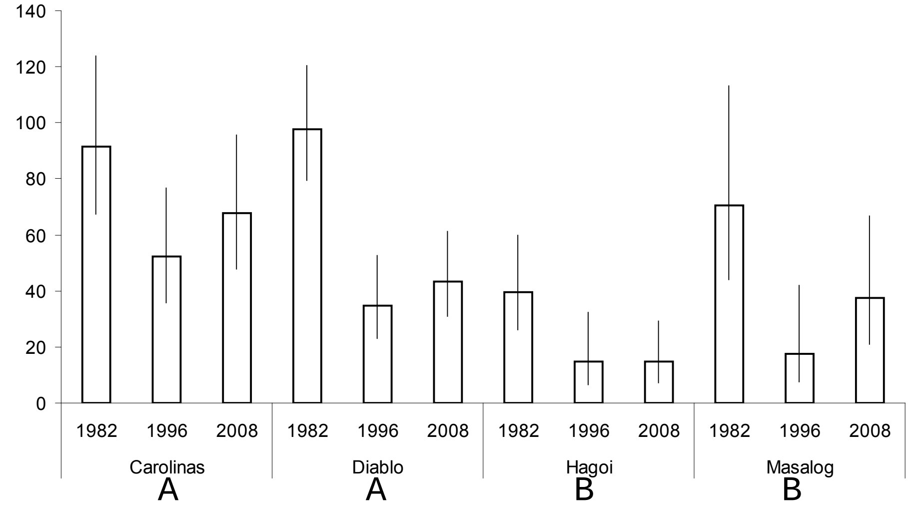
Mariana Fruit-Dove



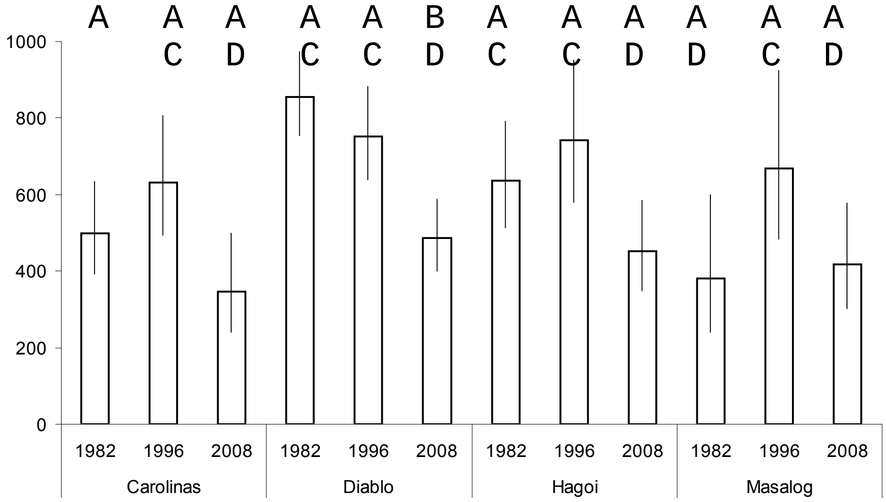
Collared Kingfisher



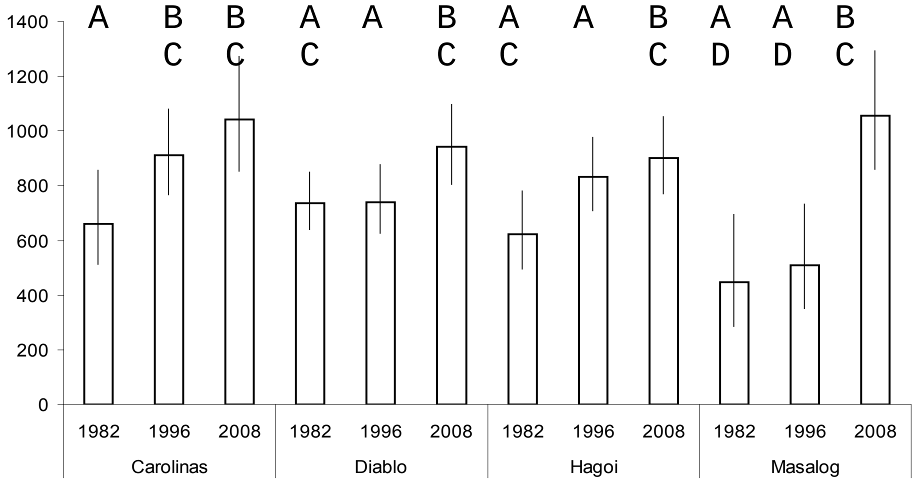
Micronesian Honeyeater



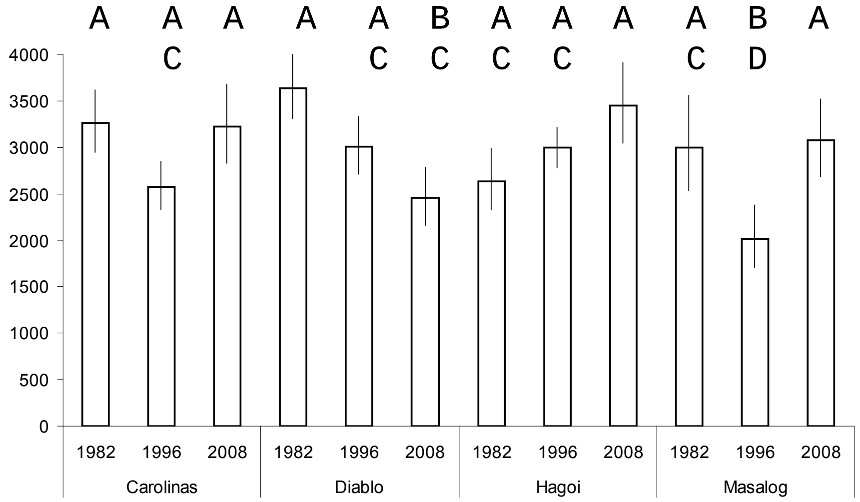
Tinian Monarch



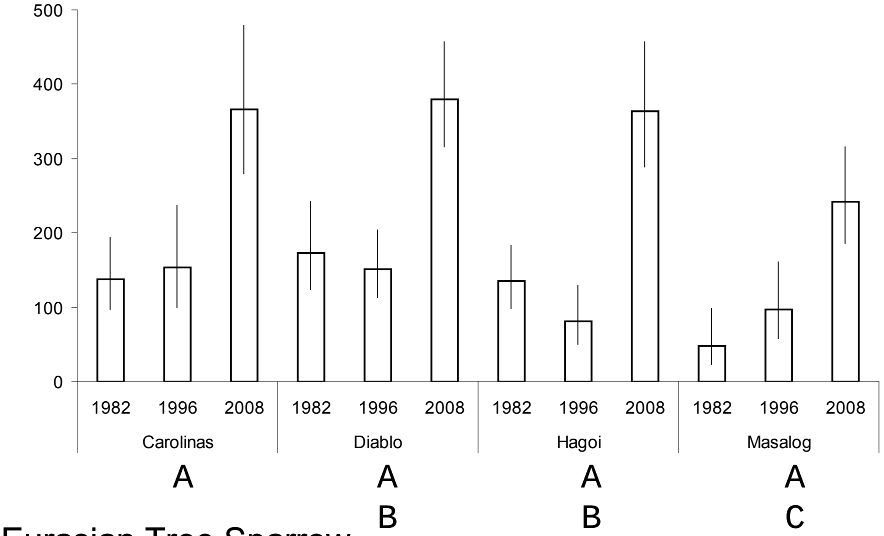
Rufous Fantail



Bridled White-eye



Micronesian Starling



Eurasian Tree Sparrow

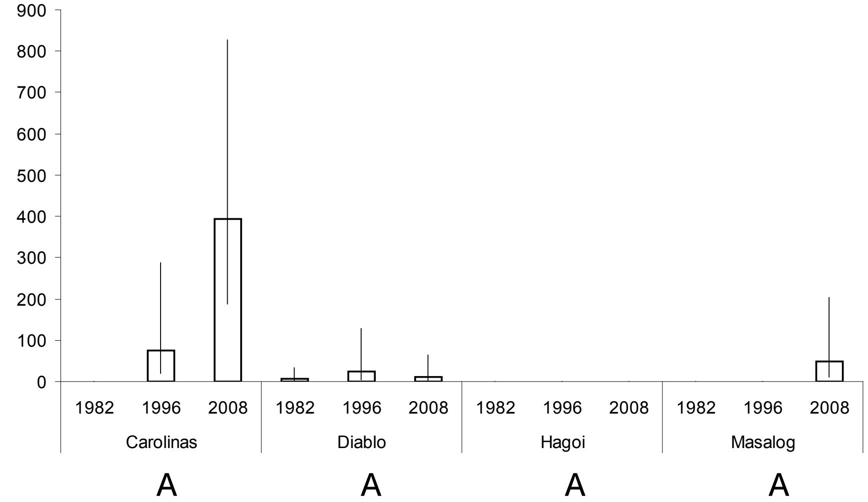


Figure 4. Density estimates (birds/km^2 and 95% CI) for native and alien Tinian land birds by region and year from three point-transect surveys. Differences of least squares means were assessed with repeated measures ANOVA (see Appendix 2 for details). Comparisons that share the same letter are not significantly different at the 0.05 level, adjusted for multiple comparisons. Comparisons below species name are year within region results (i.e., significant year, region and interaction effects), whereas comparisons below x-axis indicate fixed effects results (i.e., region or interaction effects were not significant).

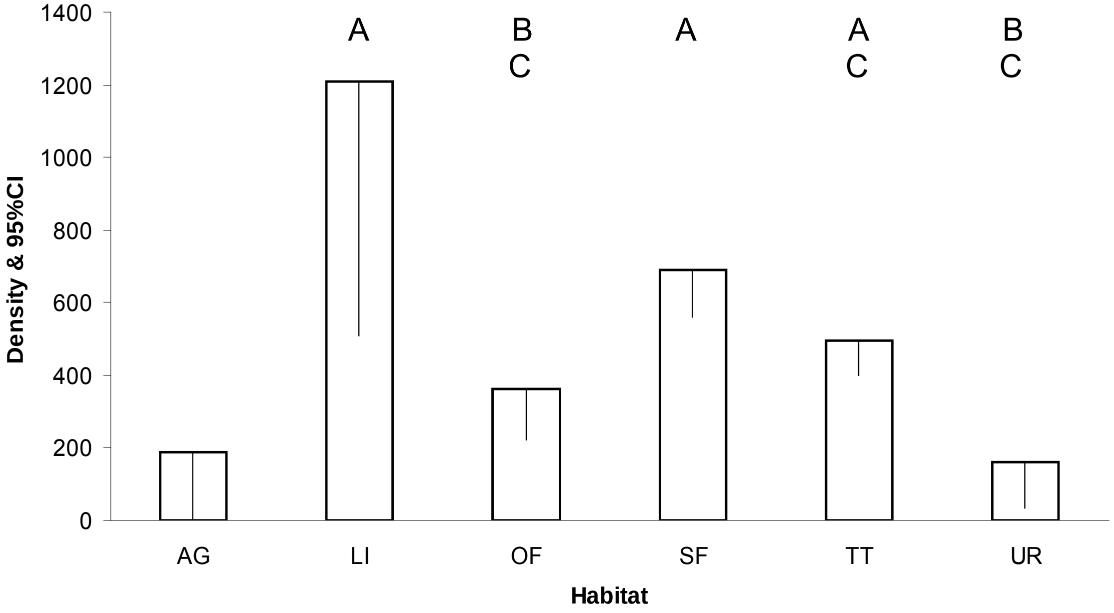


Figure 5. Density estimates (birds/km^2 and lower 95% CI) for the Tinian Monarch from all 14 transects sampled during the 2008 point-transect survey (data from all 14 transects). Habitat types are AG – agriculture, LI – limestone forest, OF – open field, SF – secondary forest, TT – tangantangan thicket, and UR – urban/residential. Differences of least squares means were assessed with a 1-way ANOVA. Agriculture habitat was dropped from this analysis and coastal habitat was not sampled in 2008 (see Methods). Comparisons that share the same letter are not significantly different at the 0.05 level, adjusted for multiple comparisons. Monarch densities in limestone and secondary forests were greater than those in open field and urban/residential.

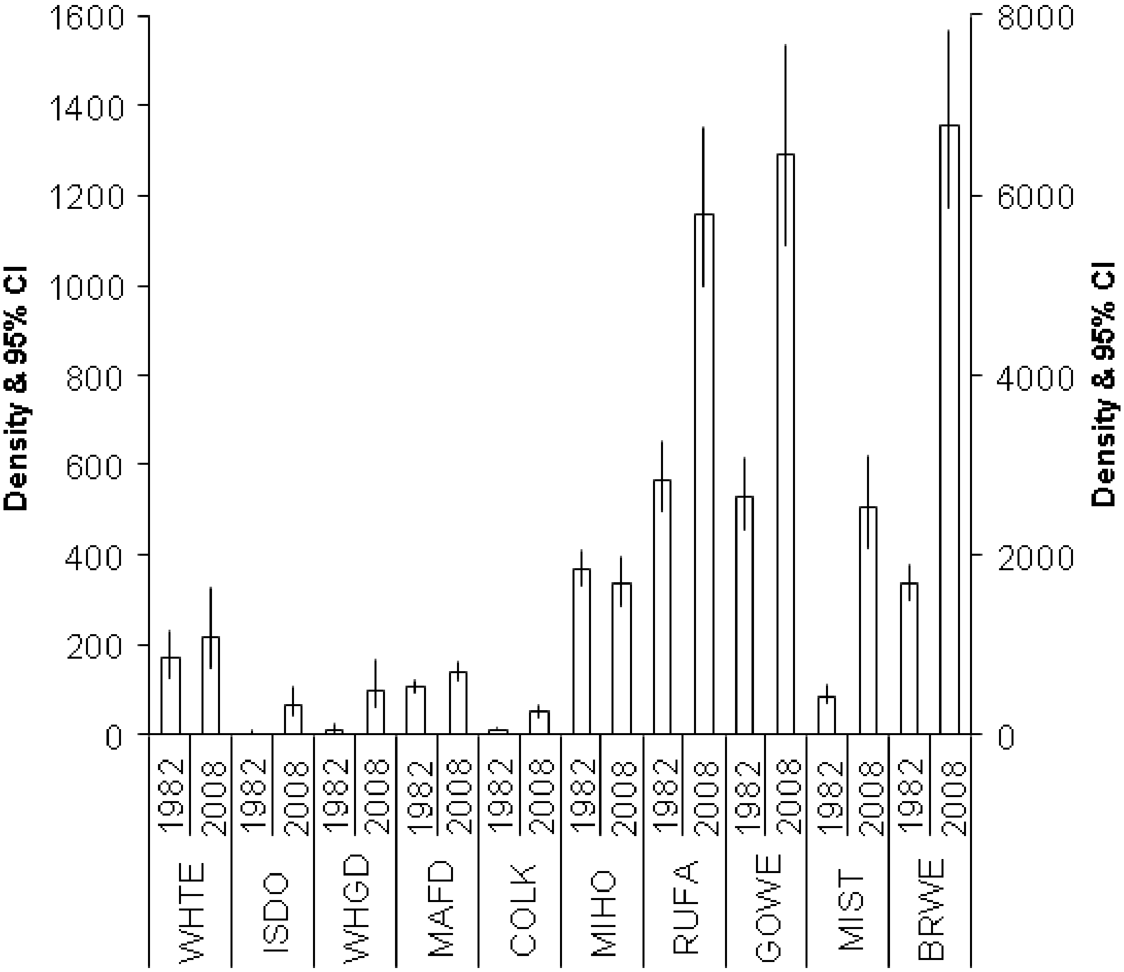


Figure 6. Density estimates (birds/km^2 and 95% CI) for native and alien Aguiguan land birds from two point-transect surveys. The primary y-axis is for the first nine species, and the secondary y-axis is for Bridled White-eye. Species codes are WHTE – White Tern; ISDO – Island Collared-Dove; WHGD – White-throated Ground-Dove; MAFD – Mariana Fruit-Dove; COLK – Collared Kingfisher; MIHO – Micronesian Honeyeater; RUFA – Rufous Fantail; GOWE – Golden White-eye; MIST – Micronesian Starling; and BRWE – Bridled White-eye.

Vegetation Changes in Central Aguiguan

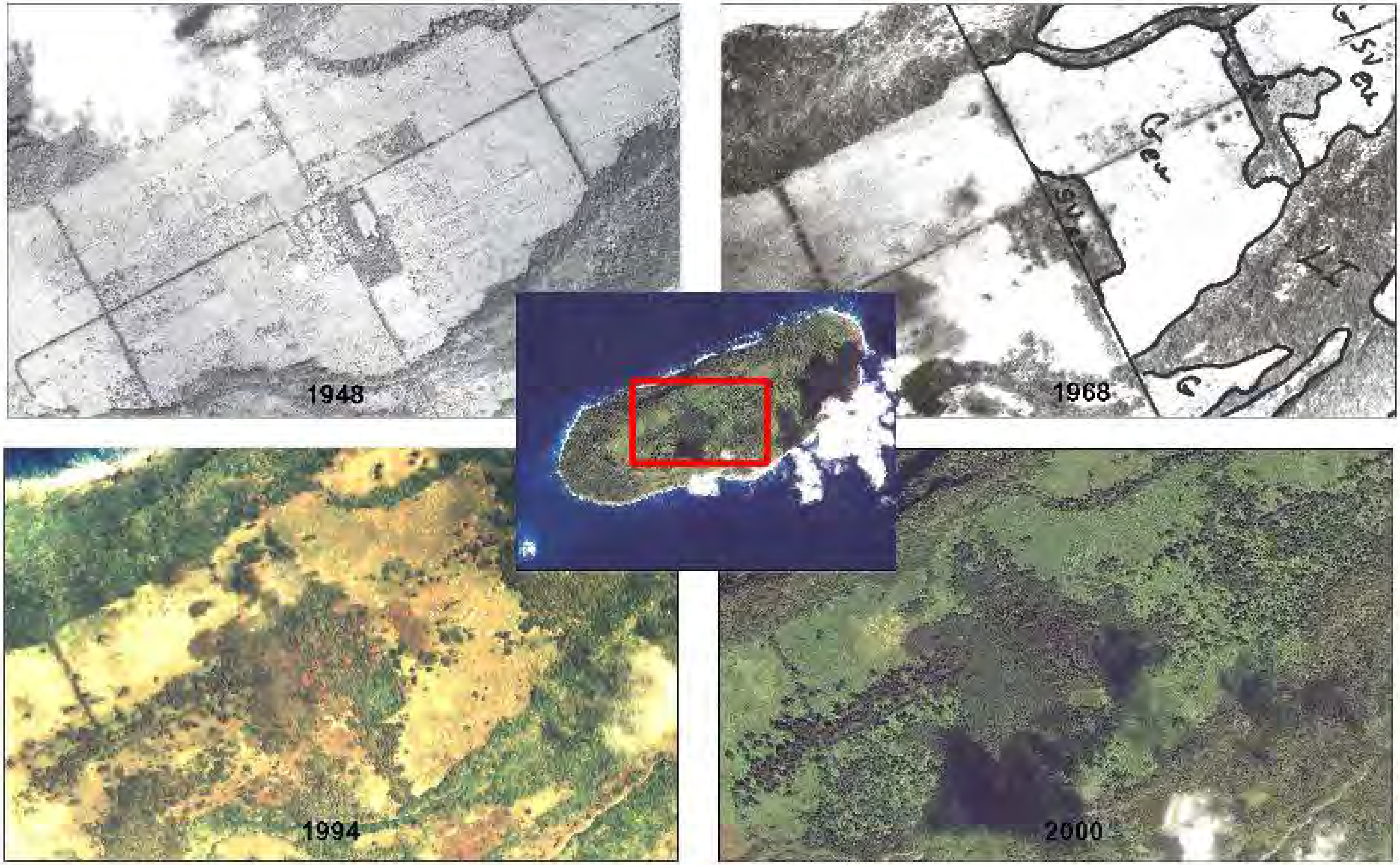


Figure 7. Vegetation changes in central Aguiguan, as shown by a series of aerial photos of center of the island. About half of the island was cleared for agriculture during the 1930s and 1940s (represented in the 1948 photo). Agriculture halted after WWII, and the fallow fields were dominated by grass (labeled G in the 1964 photo, and represented in yellow in the 1994 photo). Secondary forest expanded into the fallow fields and is represented in dark green in the bottom two photos. By 2000, the non-native shrub *Lantana camara* had replaced the grass in the fallow fields, and is represented in light green in the 2000 photo. One of the few remaining patches of grass is visible in the 2000 photo (just below the right corner of the central panel).

Table 1. List of birds detected from three different point-transect surveys on Tinian. In 1982 and 1996, 216 stations were sampled on 10 transects, and in 2008 254 stations were sampled on 14 transects. The number of stations occupied (# Stns Ocpd) and birds detected (# Dect), and indices of percent occurrence (% Occ) and birds per station (BPS), were calculated. Nomenclature generally follows the AOU checklist and Reichel and Glass (1991) with updates. Density estimates were produced for birds in bold.

		1982				1996				2008			
Species	Scientific Name	# Stns Ocpd	# Dect	% Occ	BPS	# Stns Ocpd	# Dect	% Occ	BPS	# Stns Ocpd	# Dect	% Occ	BPS
Red Junglefowl	*Gallus gallus*	45	105	20.8	0.49	0	0	0.0	0.00	45	77	17.7	0.30
White-tailed Tropicbird	*Phaethon lepturus*	0	0	0.0	0.00	0	0	0.0	0.00	3	5	1.2	0.02
Yellow Bittern	*Ixobrychus sinensis*	10	10	4.6	0.05	16	18	7.4	0.08	34	38	13.3	0.15
Pacific Reef-Egret	*Egretta sacra*	1	1	0.5	<0.01	1	1	0.5	<0.01	0	0	0.0	0.00
Pacific Golden-Plover	*Pluvialis fulva*	1	1	0.5	0.00	0	0	0.0	0.00	3	11	1.2	0.04
Ruddy Turnstone	*Arenaria interpres*	0	0	0.0	0.00	0	0	0.0	0.00	1	1	0.4	<0.01
Brown Noddy	*Anous stolidus*	0	0	0.0	0.00	0	0	0.0	0.00	1	1	0.4	<0.01
White Tern	*Gygis alba*	128	344	59.3	1.59	22	52	10.2	0.24	122	322	48.0	1.27
Island Collared-Dove	*Streptopelia bitorquata*	51	66	23.6	0.31	136	256	63.0	1.19	79	116	31.1	0.46
White-throated Ground-Dove	*Gallicolumba xanthonura*	13	16	6.0	0.07	23	23	10.6	0.11	64	82	25.2	0.32
Mariana Fruit-Dove	*Ptilinopus roseicapilla*	189	623	87.5	2.88	150	240	69.4	1.11	212	462	83.4	1.82
Collared Kingfisher	*Todiramphus chloris*	150	294	69.4	1.36	124	285	57.4	1.32	190	374	74.8	1.47
Micronesian Honeyeater	*Myzomela rubratra*	131	236	60.6	1.09	60	96	27.8	0.44	87	125	34.3	0.49
Tinian Monarch	*Monarcha takatsukasae*	187	539	86.6	2.50	173	500	80.1	2.31	178	361	70.1	1.42
Rufous Fantail	*Rhipidura rufifrons*	202	786	93.5	3.64	188	502	87.0	2.32	235	686	92.5	2.70
Bridled White-eye	*Zosterops saypani*	216	2,222	100.0	10.29	216	1,770	100.0	8.19	253	2,024	99.6	7.97

Micronesian Starling	*Aplonis opaca*	177	513	81.9	2.38	106	226	49.1	1.05	215	614	84.7	2.42
Eurasian Tree Sparrow	*Passer montanus*	1	1	0.5	<0.01	3	13	1.4	0.06	13	62	5.1	0.24

Table 2. Population density and abundance estimates for native and alien Tinian land birds from three point-transect surveys. Data from Engbring et al. (1986) transects only. First row: mean density (birds/km^2 ± SE, with 95% CI). Second row: bird abundance (sum of density by habitat type times the area of habitat types) with 95% CI. Agriculture, coastal and urban/residential habitat types were dropped for calculating bird abundance due to small sample size.

Species	1982	1996	2008
Yellow Bittern	1.5 ± 0.89 (0.5–4.4)	7.4 ± 2.49 (3.9–14.1)	18.2 ± 4.56 (11.2–29.6)
	127 (30–550)	764 (270–2,302)	1,695 (835–3,575)
White Tern	144.1 ± 17.24 (113.9–182.2)	25.3 ± 7.01 (14.8–43.2)	169.9 ± 19.66 (135.4–213.2)
	13,980 (9,349–21,512)	2,846 (1,121–7,300)	15,147 (10,067–23,041)
Island Collared-Dove	12.4 ± 2.04 (9.0–17.1)	34.3 ± 3.67 (27.8–42.3)	23.9 ± 3.24 (18.4–31.2)
	1,093 (642–2,024)	3,291 (2,296–4,777)	2,198 (1,374–3,648)
White-throated Ground-Dove	4.1 ± 1.45 (2.0–8.0)	4.6 ± 1.30 (2.7–8.0)	20.2 ± 3.91 (13.8–29.5)
	434 (136–1,421)	440 (174–1,147)	1,827 (1,045–3,226)
Mariana Fruit-Dove	42.6 ± 2.64 (37.7–48.1)	15.8 ± 1.23 (13.6–18.4)	33.1 ± 1.96 (29.4–37.1)
	3,909 (3,185–4,826)	1,539 (1,155–2,065)	3,029 (2,506–3,677)
Collared Kingfisher	7.0 ± 1.46 (4.7–10.5)	22.9 ± 3.28 (17.3–30.3)	61.3 ± 4.33 (53.3–70.4)
	570 (305–1,130)	2,268 (1,329–3,883)	5,439 (4,212–7,090)
Micronesian Honeyeater	77.2 ± 6.79 (64.9–91.7)	31.2 ± 4.26 (23.9–40.8)	41.3 ± 4.86 (32.8–52.0)
	7,859 (5,877–10,700)	2,847 (1,684–4,838)	3,716 (2,458–5,667)
Tinian Monarch	634.5 ± 37.88 (564.3–713.4)	705.7 ± 43.96 (624.3–797.6)	431.3 ± 30.75 (374.9–496.2)
	60,898 (49,484–75,398)	62,863 (50,476–78,758)	38,449 (29,992–49,849)
Rufous Fantail	641.2 ± 39.30 (568.4–723.3)	766.3 ± 40.85 (690.1–851.0)	975.0 ± 48.26 (884.6–1,074.6)
	58,336 (48,119–71,134)	67,191 (55,510–82,000)	86,112 (72,786–102,594)
Bridled White-eye	3,190.9 ± 101.79 (2,996.8–3,397.6)	2,731.9 ± 81.96 (2,575.5–2,897.8)	2,997.2 ± 105.80 (2,795.8–3,213.0)
	302,477 (270,218–338,821)	253,407 (225,258–286,044)	270,785 (239,579–306,772)
Micronesian Starling	133.9 ± 13.53 (109.8–163.3)	125.1 ± 13.34 (101.5–154.2)	349.5 ± 22.47 (308.0–396.6)
	11,543 (7,994–17,041)	10,841 (7,270–16,296)	30,088 (23,633–38,565)

Eurasian Tree Sparrow	2.1 ± 2.07 (0.4–10.7)	26.7 ± 16.42 (8.7–81.5)	110.2 ± 40.54 (54.7–222.2)
	155 (29–817)	1,244 (232–6,662)	2,111 (429–10,666)

Table 3. Repeated measures analysis of variance results for trends in Tinian land bird densities among years. Data from Engbring et al. (1986) transects only, excluding stations from agriculture, coastal and urban/residential habitat types. Trends are denoted as increasing (▲), decreasing (▼), or stable (▬). Significant changes are marked in bold. Degrees of freedom for the differences of least squares means (Diff LSM) are 398.

				Diff LSM								
		Fixed Effects		82-96			82-08			96-08		
Species	Trend	$F_{2,398}$	*p*	Est (SE)	*t*	Adj-*p*	Est (SE)	*t*	Adj-*p*	Est (SE)	*t*	Adj-*p*
Yellow Bittern	▲	13.57	<0.001	-0.04 (0.02)	-1.86	0.153	-0.10 (0.02)	-5.14	<0.001	-0.07 (0.02)	-3.29	0.003
White Tern	▬	43.18	<0.001	0.47 (0.06)	7.55	<0.001	-0.06 (0.06)	-0.91	0.634	-0.53 (0.06)	-8.46	<0.001
Island Collared-Dove	▲	16.22	<0.001	-0.14 (0.03)	-5.66	<0.001	-0.09 (0.03)	-3.38	0.002	0.06 (0.03)	2.28	0.060
White-throated Ground-Dove	▲	27.87	<0.001	<0.01 (0.02)	-0.42	0.906	-0.12 (0.02)	-6.67	<0.001	-0.11 (0.02)	-6.24	<0.001
Mariana Fruit-Dove	▼	64.54	<0.001	0.19 (0.02)	10.92	<0.001	0.05 (0.02)	2.73	0.018	-0.14 (0.02)	-8.19	<0.001
Collared Kingfisher	▲	87.05	<0.001	-0.11 (0.03)	-3.79	<0.001	-0.36 (0.03)	-12.84	<0.001	-0.26 (0.03)	-9.05	<0.001
Micronesian Honeyeater	▼	31.76	<0.001	0.27 (0.04)	7.59	<0.001	0.20 (0.04)	5.90	<0.001	-0.06 (0.04)	-1.69	0.209
Tinian Monarch	▼	10.65	<0.001	-0.09 (0.09)	-0.97	0.597	0.31 (0.09)	3.42	0.002	0.40 (0.09)	4.39	<0.001
Rufous Fantail	▲	19.55	<0.001	-0.24 (0.09)	-2.75	0.017	-0.54 (0.09)	-6.24	<0.001	-0.30 (0.09)	-3.49	0.002
Bridled White-eye	▬	5.26	0.006	0.16 (0.05)	3.24	0.004	0.07 (0.05)	1.42	0.330	-0.09 (0.05)	-1.81	0.166
Micronesian Starling	▲	67.87	<0.001	0.04 (0.07)	0.57	0.836	-0.64 (0.07)	-9.79	<0.001	-0.68 (0.07)	-10.36	<0.001
Eurasian Tree	▬	0.96	0.384	-0.02	-0.78	0.713	-0.03	-1.38	0.352	-0.01	-0.60	0.822

Sparrow	(0.02)	(0.02)	(0.02)

Table 4. Repeated measures analysis of variance results for year, region, and year-region interaction fixed effects in Tinian land bird densities. Data from Engbring et al. (1986) transects only. Dash indicates interaction test not conducted because one or both main effects results were non-significant. Differences of least squares means for the significant fixed effects (bold for interaction, italics for region) are presented in Appendix 2 and summarized in Figure 3.

	Fixed Effects					
	Year		Region		Interaction	
Species	$F_{2,392}$	P	$F_{3,196}$	P	$F_{6,392}$	P
Yellow Bittern	10.17	<0.001	0.20	0.899	—	—
White Tern	40.78	<0.001	4.15	0.007	1.71	0.116
Island Collared-Dove	19.67	<0.001	1.47	0.224	—	—
White-throated Ground-Dove	16.98	<0.001	5.19	0.002	6.60	<0.001
Mariana Fruit-Dove	66.10	<0.001	5.99	<0.001	3.76	0.001
Collared Kingfisher	81.67	<0.001	2.17	0.093	—	—
Micronesian Honeyeater	25.99	<0.001	10.89	<0.001	1.73	0.113
Tinian Monarch	8.94	<0.001	7.61	<0.001	3.10	0.006
Rufous Fantail	28.31	<0.001	5.23	0.002	6.63	<0.001
Bridled White-eye	9.29	<0.001	6.04	<0.001	11.58	<0.001
Micronesian Starling	62.05	<0.001	3.60	0.014	1.43	0.200
Eurasian Tree Sparrow	1.29	0.276	1.36	0.256	—	—

Table 5. One-way ANOVA and multiple comparisons results of Tinian Monarch densities by habitat types from the 2008 survey (data from all 14 transects). Agriculture habitat type was dropped from the analysis due to small sample size; only 2 survey stations were sampled. Significance was assessed at the alpha 0.05 level using Tukey's adjustment for multiple comparisons with 247 degrees of freedom (highlighted in bold). Habitat codes are LI – limestone forest; OF – open field; SF – secondary forest; TT – tangantangan thicket; and UR – urban/residential.

Fixed Effect	Num DF	Den DF	*F* Value	Pr > *F*
Habitat	4	247	6.24	<0.001

Habitat	Habitat	Estimate	Error	*t* Value	Adj *P*
LI	**OF**	0.76	0.203	3.75	**0.002**
LI	SF	0.01	0.173	0.04	1.000
LI	TT	0.31	0.165	1.85	0.348
LI	**UR**	1.11	0.382	2.91	**0.032**
OF	**SF**	-0.75	0.194	-3.89	**0.001**
OF	TT	-0.46	0.187	-2.43	0.111
OF	UR	0.35	0.392	0.89	0.900
SF	TT	0.30	0.154	1.94	0.298
SF	**UR**	1.10	0.377	2.93	**0.030**
TT	UR	0.80	0.374	2.15	0.201

Table 6. Measures of precision in Tinian Monarch 2008 densities for newly established transects, the original transects, the original transects in the same regions, and transects in limestone forest habitat[1].

Group	Density	SE	CV	Increased Precision
Original & New Transects	4.87	0.316	6.48	
Original Transects	4.51	0.32	7.09	8.6%
Limestone Forest Original & New Transects	6.41	0.735	11.48	
Limestone Forest Original Transects	4.97	1.152	23.20	50.5%
Carolinas & Diablo Regions Original & New Transects	5.03	0.392	7.80	
Carolinas & Diablo Regions Original Transects	4.46	0.409	9.18	15.0%
Carolinas Region Original & New Transects	3.73	0.544	14.56	
Carolinas Region Original Transects	3.62	0.661	18.23	20.1%
Diablo Region Original & New Transects	6.07	0.507	8.36	
Diablo Region Original Transects	5.07	0.488	9.62	13.1%

[1] New transects include 35 stations located in limestone forest and 2 stations in tangantangan thicket habitats, and were pooled for this analysis.

Table 7. List of birds detected from the 1982 and 2008 point-transect surveys on Aguiguan. In 1982, 66 stations were sampled on 4 transects (88 counts; several stations were counted more than once), and in 2008, 80 stations were sampled in 5 transects. The number of stations occupied (Stns Ocpd), birds detected (# Dect), indices of percent occurrence (% Occ), and birds per station (BPS) were calculated. Nomenclature generally follows the AOU checklist and Reichel and Glass (1991) with updates. Density estimates were produced for birds in bold. Scientific names are provided in superscript.

	1982				2008			
Species	# Stns Ocpd	# Dect	% Occ	BPS	# Stns Ocpd	# Dect	% Occ	BPS
Micronesian Megapode	8	14	9.1	0.16	11	15	13.8	0.19
White-tailed Tropicbird	1	1	1.1	0.01	—	—	—	—
Red-tailed Tropicbird[1]	8	13	9.1	0.15	—	—	—	—
Great Frigatebird[2]	1	2	1.1	0.02	—	—	—	—
Yellow Bittern	1	1	1.1	0.01	—	—	—	—
Brown Noddy	14	20	15.9	0.23	—	—	—	—
Black Noddy[3]	31	75	35.2	0.85	1	1	1.2	0.01
White Tern	54	218	61.4	2.48	34	84	42.5	1.05
Sooty Tern[4]	1	1	1.1	0.01	—	—	—	—
Island Collared-Dove	9	16	10.2	0.18	28	50	35	0.63
White-throated Ground-Dove	10	18	11.4	0.20	25	37	31.2	0.46
Mariana Fruit-Dove	87	757	98.9	8.60	75	240	93.8	3.00
Guam Swiftlet	26	157	29.6	1.78	9	27	11.2	0.34
Collared Kingfisher	56	154	63.6	1.75	53	101	66.2	1.26
Micronesian Honeyeater	87	745	98.9	8.47	74	174	92.5	2.18
Rufous Fantail	84	453	95.5	5.15	77	219	96.2	2.74
Golden White-eye	83	444	94.3	5.05	74	268	92.5	3.35
Bridled White-eye	88	823	100.0	9.35	77	758	96.2	9.48
Micronesian Starling	71	207	80.7	2.35	69	167	86.2	2.09

[1] = *Phaethon rubricauda*

[2] = *Fregata minor*

[3] = *Anous minutus*
[4] = *Onychoprion fuscatus*

Table 8. Population density and abundance estimates for native and alien Aguiguan land birds from two point-transect surveys. First row: mean density (birds/km^2 ± SE, with 95% CI). Second row: 2008 bird abundance (density by habitat times the habitat area) with 95% CI. Significance was assessed at the alpha 0.05 level using two-sample *z*-test (highlighted in bold). Change was defined as increasing (▲), decreasing (▼), or not significantly different (▬).

Species	1982	2008	*z* Value	*P*	Change
White Tern	169.6 ± 27.0 (124.2–231.6)	218.8 ± 44.2 (147.3–325.1)	-0.95	0.341	▬
		1,214 (604–3,651)			
Island Collared-Dove	4.4 ± 1.8 (2.0–9.7)	66.9 ± 16.7 (41.1–108.8)	-3.72	<0.001	▲
		307 (151–658)			
White-throated Ground-Dove	13.1 ± 4.8 (6.6–26.3)	100.2 ± 26.5 (59.9–167.6)	-3.23	0.001	▲
		484 (260–953)			
Mariana Fruit-Dove	107.5 ± 6.5 (95.4–121.1)	141.0 ± 10.8 (121.3–164.0)	-2.67	0.008	▲
		818 (604–1,170)			
Collared Kingfisher	13.1 ± 2.0 (9.7–17.8)	50.3 ± 6.6 (38.9–65.0)	-5.39	<0.001	▲
		347 (184–1,186)			
Micronesian Honeyeater	368.3 ± 19.6 (331.8–408.7)	336.2 ± 27.1 (286.7–394.1)	-0.96	0.337	▬
		2,128 (1,564–3,046)			
Rufous Fantail	568.8 ± 39.6 (496.0–652.2)	1.157.9 ± 89.3 (995.0–1,347.5)	-6.41	<0.001	▲
		6,429 (4,765–13,666)			
Golden White-eye	529.1 ± 40.6 (455.1–615.2)	1,292.6 ± 111.9 (1,089.7–1,533.4)	-6.41	<0.001	▲
		7,496 (4,983–17,387)			
Bridled White-eye	1,685.6 ± 102.3 (1,495.7–1,899.6)	6,771.2 ± 490.2 (5,867.6–7,814.1)	-10.15	<0.001	▲
		44,293 (32,246–63,031)			
Micronesian Starling	86.5 ± 10.9 (67.6–110.7)	505.2 ± 52.7 (411.5–620.3)	-7.78	<0.001	▲
		3,531 (1,902–12,374)			

Table 9. Comparison of density (birds/km2 and 95% confidence intervals) and change in the status of nine native land bird populations from the most recent point-transect surveys (Tinian and Aguiguan 2008, Saipan 2007) by island. A "—" denoted the species was not detected on the island. Changes are denoted as increasing (▲), decreasing (▼), or stable (▬). Results for Saipan are from Camp *et al*. (in press).

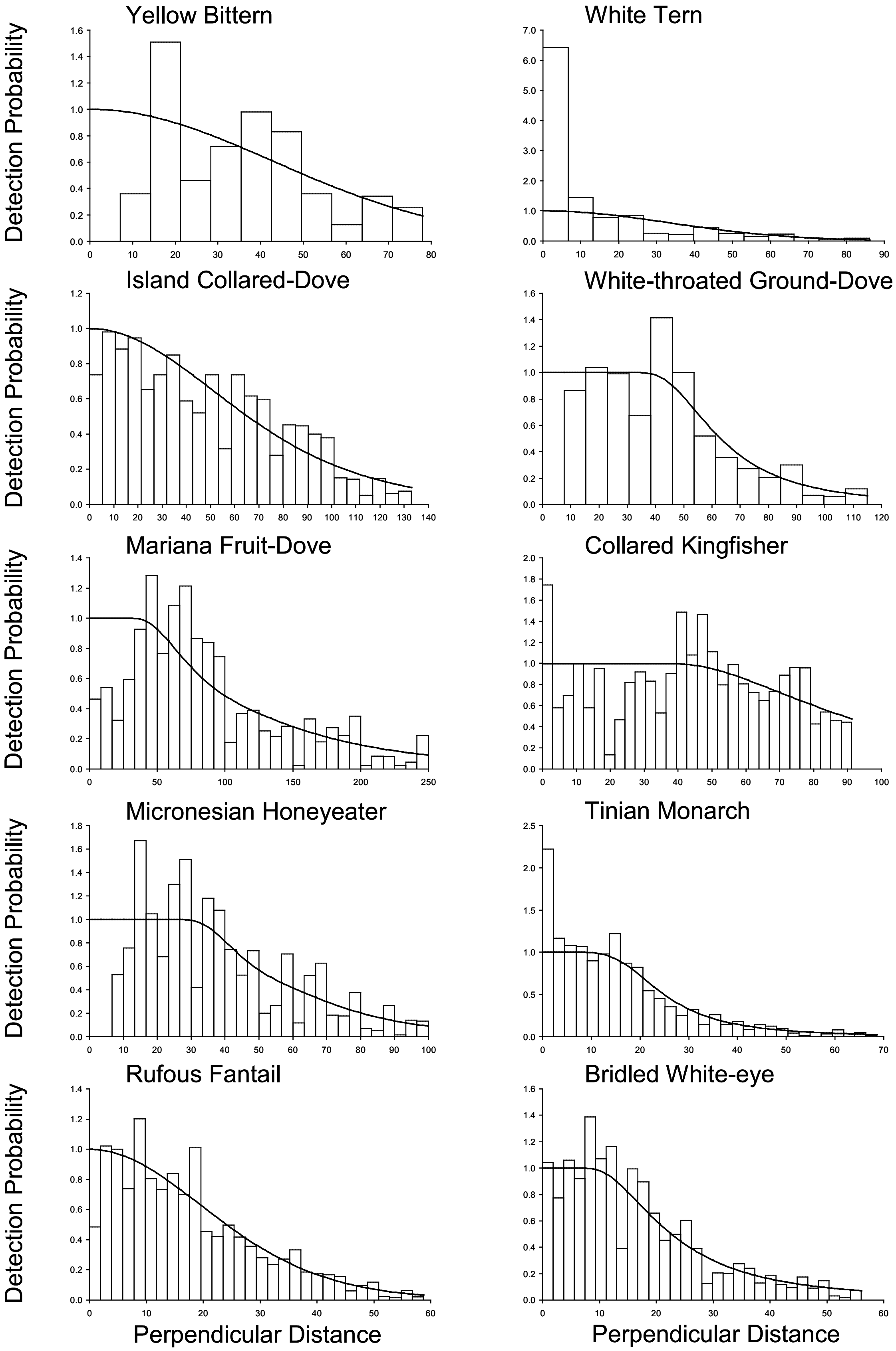
	Tinian		Aguiguan		Saipan	
Species	Density (95% CI)	Change	Density (95% CI)	Change	Density (95% CI)	Change
Yellow Bittern	18.2 (11.2–29.6)	▲	—		11.4 (4.8–21.2)	▲
White-Throated Ground-Dove	20.2 (13.8–29.5)	▲	100.2 (59.9–167.6)	▲	100.5 (77.1–127.9)	▲
Mariana Fruit-Dove	33.1 (29.4–37.1)	▼	141.0 (121.3–164.0)	▲	65.5 (53.0–79.8)	▬
Collared Kingfisher	61.3 (53.3–70.4)	▲	50.3 (38.9–65.0)	▲	25.8 (16.8–39.1)	▬
Micronesian Honeyeater	41.3 (32.8–52.0)	▼	336.2 (286.7–394.1)	▬	482.3 (383.5–651.5)	▲
Rufous Fantail	975.0 (884.6–1,074.6)	▲	1,157.9 (995.0–1,347.5)	▲	469.1 (394–1,601.5)	▼
Golden White-Eye	—		1,292.6 (1,089.7–1,533.4)	▲	711.8 (534.8–975.3)	▼
Bridled White-eye	2,997.2 (2,795.8–3,213.0)	▬	6,771.2 (5,867.6–7,814.1)	▲	4,713.3 (3,982.7–5,488.9)	▬
Micronesian Starling	349.5 (308.0–396.6)	▲	505.2 (411.5–620.3)	▲	161.9 (96.8–257.5)	▲

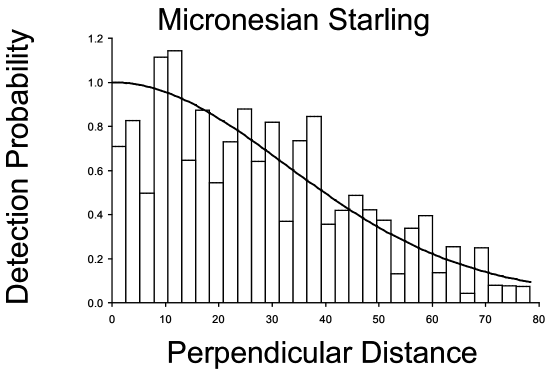
Appendix 1. Species data and models.

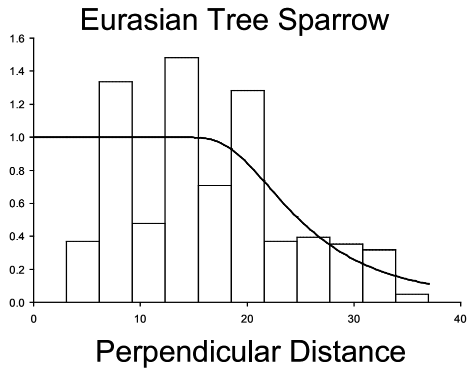
A. Detection function parameters used to derive population densities for each species on Tinian.

Species	Truncation	Key Model	Adjustment Terms	Covariates
Yellow Bittern	78.0	Half normal	None	None
White Tern	92.7	Half normal	None	None
Island Collared-Dove	133.0	Half normal	None	Observer
White-throated Ground-Dove	115.0	Hazard rate	None	None
Mariana Fruit-Dove	250.0	Hazard rate	None	Observer
Collared Kingfisher	91.2	Hazard rate	None	Observer
Micronesian Honeyeater	100.0	Hazard rate	None	Year
Tinian Monarch	68.6	Hazard rate	None	Observer
Rufous Fantail	58.7	Half normal	None	Observer
Bridled White-eye	56.0	Hazard rate	None	Observer
Micronesian Starling	78.3	Half normal	None	Observer
Eurasian Tree Sparrow	37.0	Hazard rate	None	None

C. Histograms of bird detections used to calculate population estimates on Tinian. The best fit lines for these data were modeled with program DISTANCE.



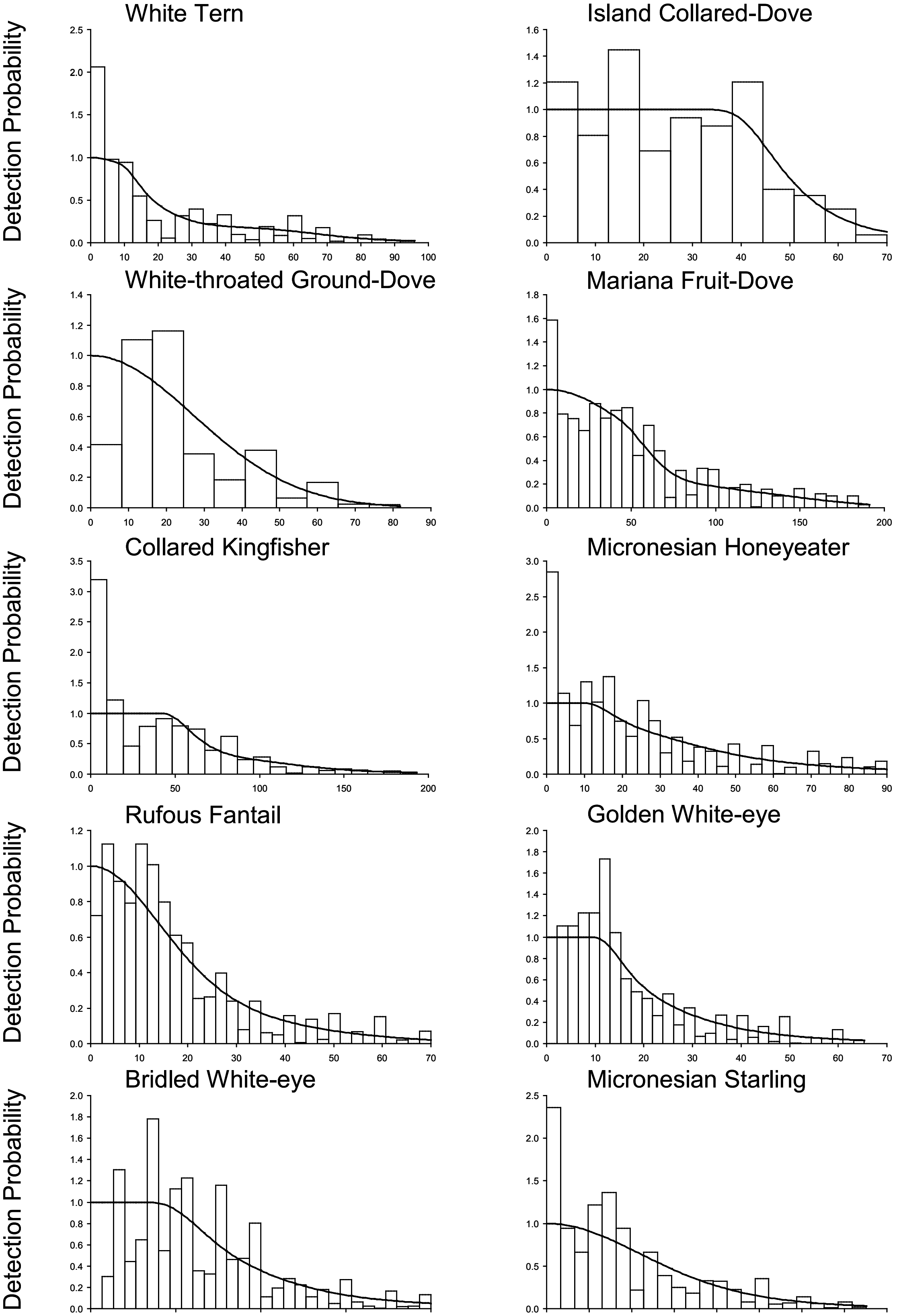

Micronesian Starling
Detection Probability
0.0
0.2
0.4
0.6
0.8
1.0
1.2
0
10
20
30
40
50
60
70
80
Perpendicular Distance


Eurasian Tree Sparrow
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
1.6
0
10
20
30
40
Perpendicular Distance

D. Detection function parameters used to derive population densities for each species on Aguiguan.

Species	Truncation	Key Model	Adjustment Terms	Covariates
White Tern	95.8	Half normal	Cosine (2,3)	Observer
Island Collared-Dove	70.0	Hazard rate	None	None
White-throated Ground-Dove	81.8	Half normal	None	None
Mariana Fruit-Dove	191.0	Hazard rate	Cosine (2)	Observer
Collared Kingfisher	193.0	Hazard rate	None	Year
Micronesian Honeyeater	90.0	Hazard rate	None	Observer
Rufous Fantail	70.0	Hazard rate	None	Observer
Golden White-eye	65.3	Hazard rate	None	Observer
Bridled White-eye	40.0	Hazard rate	None	Cloud
Micronesian Starling	75.1	Half normal	None	Observer

E. Histograms of bird detections used to calculate population estimates on Aguiguan. The best fit lines for these data were modeled with program DISTANCE.



Appendix 2. Results from region and year analyses for Tinian land birds.

A) Density estimates (birds/km^2), standard error (SE), and 95% confidence intervals (Lower and Upper 95% CI) by region and year.

Yellow Bittern

Region	Year	Estimate	SE	L 95%CI	U 95%CI
Carolinas	1982	0.0	0.00	0.0	0.0
	1996	4.0	2.92	1.1	14.8
	2008	21.8	7.10	11.6	40.9
Diablo	1982	0.0	0.00	0.0	0.0
	1996	8.8	4.81	3.2	24.3
	2008	22.0	7.69	11.2	43.1
Hagoi	1982	2.0	2.01	0.4	10.7
	1996	7.9	4.10	3.0	21.0
	2008	15.8	6.59	7.1	35.1
Masalog	1982	5.9	4.27	1.6	21.9
	1996	8.9	5.25	3.0	26.8
	2008	8.9	5.25	3.0	26.8

White Tern

Region	Year	Estimate	SE	L 95%CI	U 95%CI
Carolinas	1982	222.6	48.36	144.7	342.2
	1996	16.7	9.48	5.8	48.2
	2008	188.4	37.91	126.4	280.7
Diablo	1982	129.3	22.75	91.3	183.0
	1996	50.5	17.20	26.1	97.7
	2008	240.4	40.57	172.2	335.5
Hagoi	1982	112.0	24.79	72.2	173.5
	1996	5.5	3.83	1.5	19.4
	2008	95.6	24.72	57.4	159.1
Masalog	1982	106.5	30.64	60.1	188.6
	1996	16.4	16.40	3.0	88.9
	2008	110.6	29.96	64.5	189.7

Island Collared-Dove

Region	Year	Estimate	SE	L 95%CI	U 95%CI
Carolinas	1982	5.8	3.01	2.2	15.4
	1996	38.8	7.08	27.0	55.7
	2008	14.3	4.45	7.8	26.3
Diablo	1982	20.4	4.48	13.3	31.4
	1996	25.4	4.91	17.3	37.1
	2008	33.1	7.27	21.5	51.0
Hagoi	1982	5.7	2.24	2.7	12.2
	1996	32.4	6.65	21.6	48.6
	2008	21.0	4.34	13.9	31.6
Masalog	1982	15.7	4.58	8.8	28.0
	1996	48.6	9.06	33.5	70.6
	2008	24.3	5.75	15.1	39.0

White-throated Ground-Dove

Region	Year	Estimate	SE	L 95%CI	U 95%CI

Carolinas	1982	3.5	3.58	0.7	19.0
	1996	1.2	1.19	0.2	6.3
	2008	4.6	2.35	1.8	12.0
Diablo	1982	4.3	1.96	1.8	10.2
	1996	5.1	2.15	2.3	11.4
	2008	37.7	7.94	25.0	56.9
Hagoi	1982	1.2	1.17	0.2	6.2
	1996	7.0	2.88	3.1	15.4
	2008	20.9	5.79	12.1	35.8
Masalog	1982	8.7	4.60	3.2	23.7
	1996	5.2	3.02	1.8	15.5
	2008	7.0	4.27	2.2	21.8

Mariana Fruit-Dove

Region	Year	Estimate	SE	L 95%CI	U 95%CI
Carolinas	1982	53.7	4.72	45.1	64.0
	1996	12.4	1.98	9.0	17.0
	2008	35.4	3.44	29.2	43.0
Diablo	1982	37.8	2.94	32.4	44.1
	1996	21.7	2.32	17.6	26.9
	2008	38.0	2.85	32.8	44.1
Hagoi	1982	42.8	5.19	33.6	54.5
	1996	12.8	1.99	9.4	17.4
	2008	28.4	3.77	21.8	37.0
Masalog	1982	35.4	7.04	23.8	52.8
	1996	13.4	2.33	9.5	19.0
	2008	26.3	3.64	19.9	34.8

Collared Kingfisher

Region	Year	Estimate	SE	L 95%CI	U 95%CI
Carolinas	1982	5.5	2.52	2.3	13.2
	1996	15.7	4.72	8.7	28.3
	2008	51.6	7.47	38.7	68.9
Diablo	1982	8.7	2.76	4.7	16.1
	1996	34.8	6.13	24.6	49.3
	2008	68.3	7.33	55.2	84.5
Hagoi	1982	5.4	2.48	2.3	13.0
	1996	23.5	8.07	12.1	45.9
	2008	57.9	8.41	43.4	77.4
Masalog	1982	8.1	4.14	3.1	21.5
	1996	8.1	4.14	3.1	21.5
	2008	66.5	10.47	48.5	91.4

Micronesian Honeyeater

Region	Year	Estimate	SE	L 95%CI	U 95%CI
Carolinas	1982	91.3	14.03	67.3	123.9
	1996	52.4	10.06	35.8	76.7
	2008	67.6	11.77	47.8	95.5
Diablo	1982	97.8	10.21	79.6	120.3
	1996	34.8	7.26	23.1	52.5
	2008	43.5	7.52	30.9	61.2
Hagoi	1982	39.7	8.25	26.3	59.9

	1996	14.7	6.00	6.7	32.3
	2008	14.7	5.21	7.4	29.3
Masalog	1982	70.5	16.66	44.0	113.1
	1996	17.6	7.86	7.4	41.8
	2008	37.5	10.85	21.1	66.6

Tinian Monarch

Region	Year	Estimate	SE	L 95%CI	U 95%CI
Carolinas	1982	498.2	59.84	392.1	633.1
	1996	630.7	77.83	493.1	806.7
	2008	346.6	63.26	241.2	498.1
Diablo	1982	856.3	55.40	753.3	973.3
	1996	750.9	61.05	639.1	882.3
	2008	485.4	46.84	400.8	587.8
Hagoi	1982	637.6	69.30	513.3	791.9
	1996	742.8	92.48	579.6	952.0
	2008	451.9	58.83	348.6	585.7
Masalog	1982	380.7	86.11	242.0	598.9
	1996	668.5	107.43	483.8	923.8
	2008	417.8	66.85	302.8	576.5

Rufous Fantail

Region	Year	Estimate	SE	L 95%CI	U 95%CI
Carolinas	1982	661.9	85.71	511.2	857.0
	1996	910.1	78.08	766.9	1079.9
	2008	1042.1	104.31	853.5	1272.4
Diablo	1982	735.8	52.83	638.1	848.5
	1996	740.8	63.56	624.8	878.4
	2008	941.1	73.59	805.8	1099.0
Hagoi	1982	622.5	70.41	496.8	780.2
	1996	832.3	66.17	710.3	975.4
	2008	900.0	70.38	770.1	1051.7
Masalog	1982	446.6	98.39	287.2	694.6
	1996	507.5	93.23	350.8	734.3
	2008	1055.6	106.93	860.5	1295.0

Bridled White-eye

Region	Year	Estimate	SE	L 95%CI	U 95%CI
Carolinas	1982	3266.8	167.26	2949.0	3618.8
	1996	2575.7	129.82	2328.6	2849.1
	2008	3226.9	210.72	2831.7	3677.1
Diablo	1982	3638.8	174.30	3308.4	4002.1
	1996	3005.3	155.07	2712.0	3330.2
	2008	2452.9	153.80	2165.2	2778.8
Hagoi	1982	2637.7	162.75	2331.4	2984.2
	1996	2993.9	108.38	2785.5	3218.0
	2008	3452.9	216.50	3045.8	3914.5
Masalog	1982	3000.8	251.17	2533.1	3554.7
	1996	2014.2	165.16	1706.3	2377.6
	2008	3072.7	204.33	2686.2	3514.8

Micronesian Starling

Region	Year	Estimate	SE	L 95%CI	U 95%CI
Carolinas	1982	137.1	23.93	96.9	194.0
	1996	153.5	33.59	99.5	236.8
	2008	365.9	49.47	279.5	479.1
Diablo	1982	173.2	29.13	124.2	241.5
	1996	151.3	22.61	112.5	203.4
	2008	380.2	35.28	316.3	456.9
Hagoi	1982	134.5	20.97	98.6	183.5
	1996	80.7	19.21	50.4	129.2
	2008	363.2	42.03	288.4	457.5
Masalog	1982	48.4	17.42	23.9	98.3
	1996	96.9	24.71	58.2	161.2
	2008	242.2	31.92	185.7	315.8

Eurasian Tree Sparrow

Region	Year	Estimate	SE	L 95%CI	U 95%CI
Carolinas	1982	0.0	0.00	0.0	0.0
	1996	75.2	56.71	19.6	288.4
	2008	393.8	151.68	187.6	826.7
Diablo	1982	6.1	6.13	1.1	32.3
	1996	24.3	24.53	4.6	129.1
	2008	12.1	12.27	2.3	64.5
Hagoi	1982	0.0	0.00	0.0	0.0
	1996	0.0	0.00	0.0	0.0
	2008	0.0	0.00	0.0	0.0
Masalog	1982	0.0	0.00	0.0	0.0
	1996	0.0	0.00	0.0	0.0
	2008	49.2	39.24	11.9	203.8

B) Comparison of densities by region and year using repeated measures ANOVA for eight species with significant main effects (Table 4). Effect codes are Yr – year, Reg – region, and Y*R – interaction between year and region main effects.

White Tern

Effect	Region	Year	Region	Year	Estimate	SE	DF	*t* Value	Adj *P*
Yr		1982		1996	0.4920	0.0649	392	7.58	<.001
Yr		1982		2008	-0.0298	0.0649	392	-0.46	0.890
Yr		1996		2008	-0.5218	0.0649	392	-8.04	<.001
Reg	Carolina		Diablo		-0.0054	0.0795	196	-0.07	1.000
Reg	Carolina		Hagoi		0.2214	0.0855	196	2.59	0.050
Reg	Carolina		Masalog		0.1707	0.0947	196	1.80	0.275
Reg	Diablo		Hagoi		0.2268	0.0749	196	3.03	0.015
Reg	Diablo		Masalog		0.1761	0.0852	196	2.07	0.168
Reg	Hagoi		Masalog		-0.0507	0.0909	196	-0.56	0.944

White-throated Ground-Dove

Effect	Region	Year	Region	Year	Estimate	SE	DF	*t* Value	Adj *P*
Yr		1982		1996	-0.0042	0.0181	392	-0.23	0.971
Yr		1982		2008	-0.0934	0.0181	392	-5.16	<.001
Yr		1996		2008	-0.0891	0.0181	392	-4.93	<.001
Reg	Carolina		Diablo		-0.0845	0.0225	196	-3.75	0.001
Reg	Carolina		Hagoi		-0.0433	0.0242	196	-1.79	0.282
Reg	Carolina		Masalog		-0.0264	0.0268	196	-0.98	0.759
Reg	Diablo		Hagoi		0.0412	0.0212	196	1.94	0.214
Reg	Diablo		Masalog		0.0581	0.0241	196	2.41	0.079
Reg	Hagoi		Masalog		0.0169	0.0257	196	0.66	0.913
Yr*Reg	Carolina	1982	Diablo	1982	-0.0081	0.0355	576	-0.23	1.000
Yr*Reg	Carolina	1982	Hagoi	1982	0.0156	0.0382	576	0.41	1.000
Yr*Reg	Carolina	1982	Masalog	1982	-0.0416	0.0423	576	-0.98	0.998
Yr*Reg	Carolina	1982	Carolina	1996	0.0136	0.0380	392	0.36	1.000
Yr*Reg	Carolina	1982	Diablo	1996	-0.0148	0.0355	576	-0.42	1.000
Yr*Reg	Carolina	1982	Hagoi	1996	-0.0320	0.0382	576	-0.84	1.000
Yr*Reg	Carolina	1982	Masalog	1996	-0.0177	0.0423	576	-0.42	1.000
Yr*Reg	Carolina	1982	Carolina	2008	-0.0211	0.0380	392	-0.56	1.000
Yr*Reg	Carolina	1982	Diablo	2008	-0.2381	0.0355	576	-6.70	<.001
Yr*Reg	Carolina	1982	Hagoi	2008	-0.1210	0.0382	576	-3.16	0.072
Yr*Reg	Carolina	1982	Masalog	2008	-0.0273	0.0423	576	-0.64	1.000
Yr*Reg	Diablo	1982	Hagoi	1982	0.0238	0.0335	576	0.71	1.000
Yr*Reg	Diablo	1982	Masalog	1982	-0.0335	0.0381	576	-0.88	0.999
Yr*Reg	Diablo	1982	Carolina	1996	0.0217	0.0355	576	0.61	1.000
Yr*Reg	Diablo	1982	Diablo	1996	-0.0067	0.0288	392	-0.23	1.000
Yr*Reg	Diablo	1982	Hagoi	1996	-0.0239	0.0335	576	-0.71	1.000
Yr*Reg	Diablo	1982	Masalog	1996	-0.0096	0.0381	576	-0.25	1.000
Yr*Reg	Diablo	1982	Carolina	2008	-0.0130	0.0355	576	-0.37	1.000
Yr*Reg	Diablo	1982	Diablo	2008	-0.2299	0.0288	392	-7.99	<.001
Yr*Reg	Diablo	1982	Hagoi	2008	-0.1129	0.0335	576	-3.37	0.039
Yr*Reg	Diablo	1982	Masalog	2008	-0.0192	0.0381	576	-0.50	1.000

Yr*Reg	Hagoi	1982	Masalog	1982	-0.0572	0.0406	576	-1.41	0.962
Yr*Reg	Hagoi	1982	Carolina	1996	-0.0020	0.0382	576	-0.05	1.000
Yr*Reg	Hagoi	1982	Diablo	1996	-0.0304	0.0335	576	-0.91	0.999
Yr*Reg	Hagoi	1982	Hagoi	1996	-0.0476	0.0345	392	-1.38	0.966
Yr*Reg	Hagoi	1982	Masalog	1996	-0.0334	0.0406	576	-0.82	1.000
Yr*Reg	Hagoi	1982	Carolina	2008	-0.0368	0.0382	576	-0.96	0.998
Yr*Reg	Hagoi	1982	Diablo	2008	-0.2537	0.0335	576	-7.57	<.001
Yr*Reg	Hagoi	1982	Hagoi	2008	-0.1366	0.0345	392	-3.97	0.005
Yr*Reg	Hagoi	1982	Masalog	2008	-0.0429	0.0406	576	-1.06	0.996
Yr*Reg	Masalog	1982	Carolina	1996	0.0552	0.0423	576	1.30	0.978
Yr*Reg	Masalog	1982	Diablo	1996	0.0268	0.0381	576	0.70	1.000
Yr*Reg	Masalog	1982	Hagoi	1996	0.0096	0.0406	576	0.24	1.000
Yr*Reg	Masalog	1982	Masalog	1996	0.0239	0.0422	392	0.57	1.000
Yr*Reg	Masalog	1982	Carolina	2008	0.0205	0.0423	576	0.48	1.000
Yr*Reg	Masalog	1982	Diablo	2008	-0.1965	0.0381	576	-5.16	<.001
Yr*Reg	Masalog	1982	Hagoi	2008	-0.0794	0.0406	576	-1.95	0.724
Yr*Reg	Masalog	1982	Masalog	2008	0.0143	0.0422	392	0.34	1.000
Yr*Reg	Carolina	1996	Diablo	1996	-0.0284	0.0355	576	-0.80	1.000
Yr*Reg	Carolina	1996	Hagoi	1996	-0.0456	0.0382	576	-1.19	0.989
Yr*Reg	Carolina	1996	Masalog	1996	-0.0313	0.0423	576	-0.74	1.000
Yr*Reg	Carolina	1996	Carolina	2008	-0.0347	0.0380	392	-0.91	0.999
Yr*Reg	Carolina	1996	Diablo	2008	-0.2517	0.0355	576	-7.08	<.001
Yr*Reg	Carolina	1996	Hagoi	2008	-0.1346	0.0382	576	-3.52	0.024
Yr*Reg	Carolina	1996	Masalog	2008	-0.0409	0.0423	576	-0.97	0.998
Yr*Reg	Diablo	1996	Hagoi	1996	-0.0172	0.0335	576	-0.51	1.000
Yr*Reg	Diablo	1996	Masalog	1996	-0.0029	0.0381	576	-0.08	1.000
Yr*Reg	Diablo	1996	Carolina	2008	-0.0063	0.0355	576	-0.18	1.000
Yr*Reg	Diablo	1996	Diablo	2008	-0.2233	0.0288	392	-7.75	<.001
Yr*Reg	Diablo	1996	Hagoi	2008	-0.1062	0.0335	576	-3.17	0.070
Yr*Reg	Diablo	1996	Masalog	2008	-0.0125	0.0381	576	-0.33	1.000
Yr*Reg	Hagoi	1996	Masalog	1996	0.0143	0.0406	576	0.35	1.000
Yr*Reg	Hagoi	1996	Carolina	2008	0.0109	0.0382	576	0.28	1.000
Yr*Reg	Hagoi	1996	Diablo	2008	-0.2061	0.0335	576	-6.15	<.001
Yr*Reg	Hagoi	1996	Hagoi	2008	-0.0890	0.0345	392	-2.58	0.293
Yr*Reg	Hagoi	1996	Masalog	2008	0.0047	0.0406	576	0.12	1.000
Yr*Reg	Masalog	1996	Carolina	2008	-0.0034	0.0423	576	-0.08	1.000
Yr*Reg	Masalog	1996	Diablo	2008	-0.2204	0.0381	576	-5.78	<.001
Yr*Reg	Masalog	1996	Hagoi	2008	-0.1033	0.0406	576	-2.54	0.317
Yr*Reg	Masalog	1996	Masalog	2008	-0.0096	0.0422	392	-0.23	1.000
Yr*Reg	Carolina	2008	Diablo	2008	-0.2170	0.0355	576	-6.10	<.001
Yr*Reg	Carolina	2008	Hagoi	2008	-0.0999	0.0382	576	-2.61	0.277
Yr*Reg	Carolina	2008	Masalog	2008	-0.0062	0.0423	576	-0.15	1.000
Yr*Reg	Diablo	2008	Hagoi	2008	0.1171	0.0335	576	3.50	0.026
Yr*Reg	Diablo	2008	Masalog	2008	0.2108	0.0381	576	5.53	<.001
Yr*Reg	Hagoi	2008	Masalog	2008	0.0937	0.0406	576	2.31	0.474

Mariana Fruit-Dove

Effect	Region	Year	Region	Year	Estimate	SE	DF	*t* Value	Adj *P*
Yr		1982		1996	0.1941	0.0175	392	11.11	<.001
Yr		1982		2008	0.0522	0.0175	392	2.99	0.008
Yr		1996		2008	-0.1418	0.0175	392	-8.12	<.001
Reg	Carolina		Diablo		0.0185	0.0214	196	0.86	0.824
Reg	Carolina		Hagoi		0.0551	0.0230	196	2.39	0.082
Reg	Carolina		Masalog		0.0965	0.0255	196	3.78	0.001
Reg	Diablo		Hagoi		0.0366	0.0202	196	1.82	0.269
Reg	Diablo		Masalog		0.0780	0.0230	196	3.40	0.005
Reg	Hagoi		Masalog		0.0414	0.0245	196	1.69	0.332
Yr*Reg	Carolina	1982	Diablo	1982	0.1129	0.0341	578	3.31	0.047
Yr*Reg	Carolina	1982	Hagoi	1982	0.0836	0.0367	578	2.28	0.495
Yr*Reg	Carolina	1982	Masalog	1982	0.1822	0.0406	578	4.48	0.001
Yr*Reg	Carolina	1982	Carolina	1996	0.3105	0.0367	392	8.47	<.001
Yr*Reg	Carolina	1982	Diablo	1996	0.2347	0.0341	578	6.88	<.001
Yr*Reg	Carolina	1982	Hagoi	1996	0.3066	0.0367	578	8.35	<.001
Yr*Reg	Carolina	1982	Masalog	1996	0.3030	0.0406	578	7.46	<.001
Yr*Reg	Carolina	1982	Carolina	2008	0.0922	0.0367	392	2.52	0.333
Yr*Reg	Carolina	1982	Diablo	2008	0.1105	0.0341	578	3.24	0.058
Yr*Reg	Carolina	1982	Hagoi	2008	0.1779	0.0367	578	4.85	0.000
Yr*Reg	Carolina	1982	Masalog	2008	0.2070	0.0406	578	5.09	<.001
Yr*Reg	Diablo	1982	Hagoi	1982	-0.0293	0.0322	578	-0.91	0.999
Yr*Reg	Diablo	1982	Masalog	1982	0.0693	0.0366	578	1.89	0.763
Yr*Reg	Diablo	1982	Carolina	1996	0.1976	0.0341	578	5.79	<.001
Yr*Reg	Diablo	1982	Diablo	1996	0.1218	0.0278	392	4.38	0.001
Yr*Reg	Diablo	1982	Hagoi	1996	0.1937	0.0322	578	6.03	<.001
Yr*Reg	Diablo	1982	Masalog	1996	0.1901	0.0366	578	5.20	<.001
Yr*Reg	Diablo	1982	Carolina	2008	-0.0207	0.0341	578	-0.61	1.000
Yr*Reg	Diablo	1982	Diablo	2008	-0.0024	0.0278	392	-0.08	1.000
Yr*Reg	Diablo	1982	Hagoi	2008	0.0650	0.0322	578	2.02	0.679
Yr*Reg	Diablo	1982	Masalog	2008	0.0941	0.0366	578	2.57	0.298
Yr*Reg	Hagoi	1982	Masalog	1982	0.0986	0.0390	578	2.53	0.326
Yr*Reg	Hagoi	1982	Carolina	1996	0.2270	0.0367	578	6.18	<.001
Yr*Reg	Hagoi	1982	Diablo	1996	0.1511	0.0322	578	4.70	0.000
Yr*Reg	Hagoi	1982	Hagoi	1996	0.2231	0.0333	392	6.71	<.001
Yr*Reg	Hagoi	1982	Masalog	1996	0.2194	0.0390	578	5.63	<.001
Yr*Reg	Hagoi	1982	Carolina	2008	0.0086	0.0367	578	0.23	1.000
Yr*Reg	Hagoi	1982	Diablo	2008	0.0270	0.0322	578	0.84	1.000
Yr*Reg	Hagoi	1982	Hagoi	2008	0.0943	0.0333	392	2.84	0.170
Yr*Reg	Hagoi	1982	Masalog	2008	0.1234	0.0390	578	3.17	0.072
Yr*Reg	Masalog	1982	Carolina	1996	0.1284	0.0406	578	3.16	0.073
Yr*Reg	Masalog	1982	Diablo	1996	0.0525	0.0366	578	1.44	0.956
Yr*Reg	Masalog	1982	Hagoi	1996	0.1245	0.0390	578	3.19	0.066
Yr*Reg	Masalog	1982	Masalog	1996	0.1208	0.0407	392	2.97	0.123
Yr*Reg	Masalog	1982	Carolina	2008	-0.0900	0.0406	578	-2.21	0.540
Yr*Reg	Masalog	1982	Diablo	2008	-0.0716	0.0366	578	-1.96	0.721
Yr*Reg	Masalog	1982	Hagoi	2008	-0.0043	0.0390	578	-0.11	1.000

Yr*Reg	Masalog	1982	Masalog	2008	0.0249	0.0407	392	0.61	1.000
Yr*Reg	Carolina	1996	Diablo	1996	-0.0758	0.0341	578	-2.22	0.534
Yr*Reg	Carolina	1996	Hagoi	1996	-0.0039	0.0367	578	-0.11	1.000
Yr*Reg	Carolina	1996	Masalog	1996	-0.0076	0.0406	578	-0.19	1.000
Yr*Reg	Carolina	1996	Carolina	2008	-0.2184	0.0367	392	-5.96	<.001
Yr*Reg	Carolina	1996	Diablo	2008	-0.2000	0.0341	578	-5.86	<.001
Yr*Reg	Carolina	1996	Hagoi	2008	-0.1327	0.0367	578	-3.61	0.018
Yr*Reg	Carolina	1996	Masalog	2008	-0.1035	0.0406	578	-2.55	0.314
Yr*Reg	Diablo	1996	Hagoi	1996	0.0719	0.0322	578	2.24	0.523
Yr*Reg	Diablo	1996	Masalog	1996	0.0683	0.0366	578	1.87	0.779
Yr*Reg	Diablo	1996	Carolina	2008	-0.1425	0.0341	578	-4.18	0.002
Yr*Reg	Diablo	1996	Diablo	2008	-0.1242	0.0278	392	-4.47	0.001
Yr*Reg	Diablo	1996	Hagoi	2008	-0.0568	0.0322	578	-1.77	0.834
Yr*Reg	Diablo	1996	Masalog	2008	-0.0277	0.0366	578	-0.76	1.000
Yr*Reg	Hagoi	1996	Masalog	1996	-0.0037	0.0390	578	-0.09	1.000
Yr*Reg	Hagoi	1996	Carolina	2008	-0.2145	0.0367	578	-5.84	<.001
Yr*Reg	Hagoi	1996	Diablo	2008	-0.1961	0.0322	578	-6.10	<.001
Yr*Reg	Hagoi	1996	Hagoi	2008	-0.1288	0.0333	392	-3.87	0.007
Yr*Reg	Hagoi	1996	Masalog	2008	-0.0996	0.0390	578	-2.55	0.310
Yr*Reg	Masalog	1996	Carolina	2008	-0.2108	0.0406	578	-5.19	<.001
Yr*Reg	Masalog	1996	Diablo	2008	-0.1924	0.0366	578	-5.26	<.001
Yr*Reg	Masalog	1996	Hagoi	2008	-0.1251	0.0390	578	-3.21	0.063
Yr*Reg	Masalog	1996	Masalog	2008	-0.0960	0.0407	392	-2.36	0.439
Yr*Reg	Carolina	2008	Diablo	2008	0.0184	0.0341	578	0.54	1.000
Yr*Reg	Carolina	2008	Hagoi	2008	0.0857	0.0367	578	2.33	0.454
Yr*Reg	Carolina	2008	Masalog	2008	0.1148	0.0406	578	2.83	0.173
Yr*Reg	Diablo	2008	Hagoi	2008	0.0673	0.0322	578	2.09	0.627
Yr*Reg	Diablo	2008	Masalog	2008	0.0965	0.0366	578	2.64	0.262
Yr*Reg	Hagoi	2008	Masalog	2008	0.0292	0.0390	578	0.75	1.000

Micronesian Honeyeater

Effect	Region	Year	Region	Year	Estimate	SE	DF	*t* Value	Adj *P*
Yr		1982		1996	0.2518	0.0363	392	6.94	<.001
Yr		1982		2008	0.1876	0.0363	392	5.17	<.001
Yr		1996		2008	-0.0642	0.0363	392	-1.77	0.182
Reg	Carolina		Diablo		0.0323	0.0478	196	0.68	0.906
Reg	Carolina		Hagoi		0.2413	0.0514	196	4.70	<.001
Reg	Carolina		Masalog		0.1799	0.0569	196	3.16	0.010
Reg	Diablo		Hagoi		0.2090	0.0450	196	4.64	<.001
Reg	Diablo		Masalog		0.1476	0.0512	196	2.88	0.023
Reg	Hagoi		Masalog		-0.0615	0.0546	196	-1.13	0.674

Tinian Monarch

Effect	Region	Year	Region	Year	Estimate	SE	DF	*t* Value	Adj *P*
Yr		1982		1996	-0.1750	0.0925	392	-1.89	0.143
Yr		1982		2008	0.2156	0.0925	392	2.33	0.053
Yr		1996		2008	0.3905	0.0925	392	4.22	<.001

Reg	Carolina		Diablo		-0.4019	0.1180	196	-3.40	0.004
Reg	Carolina		Hagoi		-0.2164	0.1270	196	-1.70	0.324
Reg	Carolina		Masalog		0.1388	0.1406	196	0.99	0.757
Reg	Diablo		Hagoi		0.1854	0.1112	196	1.67	0.344
Reg	Diablo		Masalog		0.5406	0.1265	196	4.27	0.000
Reg	Hagoi		Masalog		0.3552	0.1349	196	2.63	0.045
Yr*Reg	Carolina	1982	Diablo	1982	-0.7112	0.1837	572	-3.87	0.007
Yr*Reg	Carolina	1982	Hagoi	1982	-0.3605	0.1976	572	-1.82	0.804
Yr*Reg	Carolina	1982	Masalog	1982	0.4406	0.2188	572	2.01	0.684
Yr*Reg	Carolina	1982	Carolina	1996	-0.3105	0.1942	392	-1.60	0.909
Yr*Reg	Carolina	1982	Diablo	1996	-0.4804	0.1837	572	-2.62	0.275
Yr*Reg	Carolina	1982	Hagoi	1996	-0.3738	0.1976	572	-1.89	0.764
Yr*Reg	Carolina	1982	Masalog	1996	-0.1663	0.2188	572	-0.76	1.000
Yr*Reg	Carolina	1982	Carolina	2008	0.2374	0.1942	392	1.22	0.987
Yr*Reg	Carolina	1982	Diablo	2008	-0.0871	0.1837	572	-0.47	1.000
Yr*Reg	Carolina	1982	Hagoi	2008	0.0120	0.1976	572	0.06	1.000
Yr*Reg	Carolina	1982	Masalog	2008	0.0689	0.2188	572	0.32	1.000
Yr*Reg	Diablo	1982	Hagoi	1982	0.3507	0.1731	572	2.03	0.675
Yr*Reg	Diablo	1982	Masalog	1982	1.1518	0.1969	572	5.85	<.001
Yr*Reg	Diablo	1982	Carolina	1996	0.4007	0.1837	572	2.18	0.564
Yr*Reg	Diablo	1982	Diablo	1996	0.2308	0.1473	392	1.57	0.920
Yr*Reg	Diablo	1982	Hagoi	1996	0.3374	0.1731	572	1.95	0.727
Yr*Reg	Diablo	1982	Masalog	1996	0.5449	0.1969	572	2.77	0.198
Yr*Reg	Diablo	1982	Carolina	2008	0.9486	0.1837	572	5.16	<.001
Yr*Reg	Diablo	1982	Diablo	2008	0.6241	0.1473	392	4.24	0.002
Yr*Reg	Diablo	1982	Hagoi	2008	0.7231	0.1731	572	4.18	0.002
Yr*Reg	Diablo	1982	Masalog	2008	0.7801	0.1969	572	3.96	0.005
Yr*Reg	Hagoi	1982	Masalog	1982	0.8011	0.2100	572	3.82	0.009
Yr*Reg	Hagoi	1982	Carolina	1996	0.0500	0.1976	572	0.25	1.000
Yr*Reg	Hagoi	1982	Diablo	1996	-0.1199	0.1731	572	-0.69	1.000
Yr*Reg	Hagoi	1982	Hagoi	1996	-0.0133	0.1762	392	-0.08	1.000
Yr*Reg	Hagoi	1982	Masalog	1996	0.1942	0.2100	572	0.92	0.999
Yr*Reg	Hagoi	1982	Carolina	2008	0.5979	0.1976	572	3.03	0.105
Yr*Reg	Hagoi	1982	Diablo	2008	0.2734	0.1731	572	1.58	0.916
Yr*Reg	Hagoi	1982	Hagoi	2008	0.3725	0.1762	392	2.11	0.613
Yr*Reg	Hagoi	1982	Masalog	2008	0.4294	0.2100	572	2.05	0.662
Yr*Reg	Masalog	1982	Carolina	1996	-0.7511	0.2188	572	-3.43	0.032
Yr*Reg	Masalog	1982	Diablo	1996	-0.9210	0.1969	572	-4.68	0.000
Yr*Reg	Masalog	1982	Hagoi	1996	-0.8144	0.2100	572	-3.88	0.007
Yr*Reg	Masalog	1982	Masalog	1996	-0.6069	0.2158	392	-2.81	0.179
Yr*Reg	Masalog	1982	Carolina	2008	-0.2033	0.2188	572	-0.93	0.999
Yr*Reg	Masalog	1982	Diablo	2008	-0.5278	0.1969	572	-2.68	0.240
Yr*Reg	Masalog	1982	Hagoi	2008	-0.4287	0.2100	572	-2.04	0.664
Yr*Reg	Masalog	1982	Masalog	2008	-0.3717	0.2158	392	-1.72	0.857
Yr*Reg	Carolina	1996	Diablo	1996	-0.1699	0.1837	572	-0.92	0.999
Yr*Reg	Carolina	1996	Hagoi	1996	-0.0633	0.1976	572	-0.32	1.000
Yr*Reg	Carolina	1996	Masalog	1996	0.1442	0.2188	572	0.66	1.000

Yr*Reg	Carolina	1996	Carolina	2008	0.5479	0.1942	392	2.82	0.175
Yr*Reg	Carolina	1996	Diablo	2008	0.2234	0.1837	572	1.22	0.988
Yr*Reg	Carolina	1996	Hagoi	2008	0.3225	0.1976	572	1.63	0.896
Yr*Reg	Carolina	1996	Masalog	2008	0.3794	0.2188	572	1.73	0.851
Yr*Reg	Diablo	1996	Hagoi	1996	0.1066	0.1731	572	0.62	1.000
Yr*Reg	Diablo	1996	Masalog	1996	0.3141	0.1969	572	1.60	0.910
Yr*Reg	Diablo	1996	Carolina	2008	0.7177	0.1837	572	3.91	0.006
Yr*Reg	Diablo	1996	Diablo	2008	0.3932	0.1473	392	2.67	0.245
Yr*Reg	Diablo	1996	Hagoi	2008	0.4923	0.1731	572	2.84	0.166
Yr*Reg	Diablo	1996	Masalog	2008	0.5493	0.1969	572	2.79	0.188
Yr*Reg	Hagoi	1996	Masalog	1996	0.2075	0.2100	572	0.99	0.998
Yr*Reg	Hagoi	1996	Carolina	2008	0.6112	0.1976	572	3.09	0.088
Yr*Reg	Hagoi	1996	Diablo	2008	0.2867	0.1731	572	1.66	0.887
Yr*Reg	Hagoi	1996	Hagoi	2008	0.3857	0.1762	392	2.19	0.559
Yr*Reg	Hagoi	1996	Masalog	2008	0.4427	0.2100	572	2.11	0.617
Yr*Reg	Masalog	1996	Carolina	2008	0.4037	0.2188	572	1.85	0.792
Yr*Reg	Masalog	1996	Diablo	2008	0.0792	0.1969	572	0.40	1.000
Yr*Reg	Masalog	1996	Hagoi	2008	0.1783	0.2100	572	0.85	1.000
Yr*Reg	Masalog	1996	Masalog	2008	0.2352	0.2158	392	1.09	0.995
Yr*Reg	Carolina	2008	Diablo	2008	-0.3245	0.1837	572	-1.77	0.835
Yr*Reg	Carolina	2008	Hagoi	2008	-0.2254	0.1976	572	-1.14	0.993
Yr*Reg	Carolina	2008	Masalog	2008	-0.1685	0.2188	572	-0.77	1.000
Yr*Reg	Diablo	2008	Hagoi	2008	0.0991	0.1731	572	0.57	1.000
Yr*Reg	Diablo	2008	Masalog	2008	0.1561	0.1969	572	0.79	1.000
Yr*Reg	Hagoi	2008	Masalog	2008	0.0570	0.2100	572	0.27	1.000

Rufous Fantail

Effect	Region	Year	Region	Year	Estimate	SE	DF	*t* Value	Adj *P*
Yr		1982		1996	-0.2980	0.0868	392	-3.43	0.002
Yr		1982		2008	-0.6521	0.0868	392	-7.52	<.001
Yr		1996		2008	-0.3542	0.0868	392	-4.08	0.000
Reg	Carolina		Diablo		0.0887	0.1147	196	0.77	0.866
Reg	Carolina		Hagoi		0.0847	0.1234	196	0.69	0.902
Reg	Carolina		Masalog		0.4970	0.1367	196	3.64	0.002
Reg	Diablo		Hagoi		-0.0040	0.1081	196	-0.04	1.000
Reg	Diablo		Masalog		0.4082	0.1230	196	3.32	0.006
Reg	Hagoi		Masalog		0.4122	0.1312	196	3.14	0.010
Yr*Reg	Carolina	1982	Diablo	1982	-0.4308	0.1748	564	-2.46	0.366
Yr*Reg	Carolina	1982	Hagoi	1982	-0.1465	0.1881	564	-0.78	1.000
Yr*Reg	Carolina	1982	Masalog	1982	0.4411	0.2083	564	2.12	0.610
Yr*Reg	Carolina	1982	Carolina	1996	-0.7967	0.1821	392	-4.38	0.001
Yr*Reg	Carolina	1982	Diablo	1996	-0.2829	0.1748	564	-1.62	0.902
Yr*Reg	Carolina	1982	Hagoi	1996	-0.5348	0.1881	564	-2.84	0.166
Yr*Reg	Carolina	1982	Masalog	1996	0.2863	0.2083	564	1.37	0.968
Yr*Reg	Carolina	1982	Carolina	2008	-0.7584	0.1821	392	-4.17	0.002
Yr*Reg	Carolina	1982	Diablo	2008	-0.5752	0.1748	564	-3.29	0.050
Yr*Reg	Carolina	1982	Hagoi	2008	-0.6196	0.1881	564	-3.29	0.049

Yr*Reg	Carolina	1982	Masalog	2008	-0.7916	0.2083	564	-3.80	0.009
Yr*Reg	Diablo	1982	Hagoi	1982	0.2843	0.1648	564	1.73	0.856
Yr*Reg	Diablo	1982	Masalog	1982	0.8719	0.1874	564	4.65	0.000
Yr*Reg	Diablo	1982	Carolina	1996	-0.3659	0.1748	564	-2.09	0.628
Yr*Reg	Diablo	1982	Diablo	1996	0.1479	0.1381	392	1.07	0.996
Yr*Reg	Diablo	1982	Hagoi	1996	-0.1040	0.1648	564	-0.63	1.000
Yr*Reg	Diablo	1982	Masalog	1996	0.7171	0.1874	564	3.83	0.008
Yr*Reg	Diablo	1982	Carolina	2008	-0.3276	0.1748	564	-1.87	0.775
Yr*Reg	Diablo	1982	Diablo	2008	-0.1444	0.1381	392	-1.05	0.997
Yr*Reg	Diablo	1982	Hagoi	2008	-0.1887	0.1648	564	-1.15	0.992
Yr*Reg	Diablo	1982	Masalog	2008	-0.3608	0.1874	564	-1.92	0.743
Yr*Reg	Hagoi	1982	Masalog	1982	0.5875	0.1999	564	2.94	0.131
Yr*Reg	Hagoi	1982	Carolina	1996	-0.6502	0.1881	564	-3.46	0.030
Yr*Reg	Hagoi	1982	Diablo	1996	-0.1364	0.1648	564	-0.83	1.000
Yr*Reg	Hagoi	1982	Hagoi	1996	-0.3883	0.1652	392	-2.35	0.443
Yr*Reg	Hagoi	1982	Masalog	1996	0.4328	0.1999	564	2.17	0.576
Yr*Reg	Hagoi	1982	Carolina	2008	-0.6120	0.1881	564	-3.25	0.056
Yr*Reg	Hagoi	1982	Diablo	2008	-0.4287	0.1648	564	-2.60	0.282
Yr*Reg	Hagoi	1982	Hagoi	2008	-0.4731	0.1652	392	-2.86	0.159
Yr*Reg	Hagoi	1982	Masalog	2008	-0.6451	0.1999	564	-3.23	0.060
Yr*Reg	Masalog	1982	Carolina	1996	-1.2377	0.2083	564	-5.94	<.001
Yr*Reg	Masalog	1982	Diablo	1996	-0.7240	0.1874	564	-3.86	0.007
Yr*Reg	Masalog	1982	Hagoi	1996	-0.9759	0.1999	564	-4.88	<.001
Yr*Reg	Masalog	1982	Masalog	1996	-0.1548	0.2024	392	-0.76	1.000
Yr*Reg	Masalog	1982	Carolina	2008	-1.1995	0.2083	564	-5.76	<.001
Yr*Reg	Masalog	1982	Diablo	2008	-1.0163	0.1874	564	-5.42	<.001
Yr*Reg	Masalog	1982	Hagoi	2008	-1.0606	0.1999	564	-5.31	<.001
Yr*Reg	Masalog	1982	Masalog	2008	-1.2326	0.2024	392	-6.09	<.001
Yr*Reg	Carolina	1996	Diablo	1996	0.5138	0.1748	564	2.94	0.132
Yr*Reg	Carolina	1996	Hagoi	1996	0.2618	0.1881	564	1.39	0.965
Yr*Reg	Carolina	1996	Masalog	1996	1.0830	0.2083	564	5.20	<.001
Yr*Reg	Carolina	1996	Carolina	2008	0.0382	0.1821	392	0.21	1.000
Yr*Reg	Carolina	1996	Diablo	2008	0.2215	0.1748	564	1.27	0.983
Yr*Reg	Carolina	1996	Hagoi	2008	0.1771	0.1881	564	0.94	0.999
Yr*Reg	Carolina	1996	Masalog	2008	0.0051	0.2083	564	0.02	1.000
Yr*Reg	Diablo	1996	Hagoi	1996	-0.2519	0.1648	564	-1.53	0.932
Yr*Reg	Diablo	1996	Masalog	1996	0.5692	0.1874	564	3.04	0.102
Yr*Reg	Diablo	1996	Carolina	2008	-0.4755	0.1748	564	-2.72	0.221
Yr*Reg	Diablo	1996	Diablo	2008	-0.2923	0.1381	392	-2.12	0.611
Yr*Reg	Diablo	1996	Hagoi	2008	-0.3366	0.1648	564	-2.04	0.663
Yr*Reg	Diablo	1996	Masalog	2008	-0.5087	0.1874	564	-2.71	0.223
Yr*Reg	Hagoi	1996	Masalog	1996	0.8211	0.1999	564	4.11	0.003
Yr*Reg	Hagoi	1996	Carolina	2008	-0.2236	0.1881	564	-1.19	0.990
Yr*Reg	Hagoi	1996	Diablo	2008	-0.0404	0.1648	564	-0.25	1.000
Yr*Reg	Hagoi	1996	Hagoi	2008	-0.0847	0.1652	392	-0.51	1.000
Yr*Reg	Hagoi	1996	Masalog	2008	-0.2568	0.1999	564	-1.28	0.981
Yr*Reg	Masalog	1996	Carolina	2008	-1.0447	0.2083	564	-5.02	<.001

Yr*Reg	Masalog	1996	Diablo	2008	-0.8615	0.1874	564	-4.60	0.000
Yr*Reg	Masalog	1996	Hagoi	2008	-0.9058	0.1999	564	-4.53	0.001
Yr*Reg	Masalog	1996	Masalog	2008	-1.0779	0.2024	392	-5.33	<.001
Yr*Reg	Carolina	2008	Diablo	2008	0.1832	0.1748	564	1.05	0.996
Yr*Reg	Carolina	2008	Hagoi	2008	0.1389	0.1881	564	0.74	1.000
Yr*Reg	Carolina	2008	Masalog	2008	-0.0332	0.2083	564	-0.16	1.000
Yr*Reg	Diablo	2008	Hagoi	2008	-0.0443	0.1648	564	-0.27	1.000
Yr*Reg	Diablo	2008	Masalog	2008	-0.2164	0.1874	564	-1.15	0.992
Yr*Reg	Hagoi	2008	Masalog	2008	-0.1720	0.1999	564	-0.86	0.999

Bridled White-eye

Effect	Region	Year	Region	Year	Estimate	SE	DF	*t* Value	Adj *P*
Yr		1982		1996	0.1766	0.0464	392	3.81	0.001
Yr		1982		2008	0.0071	0.0464	392	0.15	0.987
Yr		1996		2008	-0.1695	0.0464	392	-3.65	0.001
Reg	Carolina		Diablo		0.1128	0.0523	196	2.16	0.139
Reg	Carolina		Hagoi		0.0699	0.0563	196	1.24	0.601
Reg	Carolina		Masalog		0.2577	0.0623	196	4.14	0.000
Reg	Diablo		Hagoi		-0.0429	0.0493	196	-0.87	0.820
Reg	Diablo		Masalog		0.1449	0.0561	196	2.58	0.051
Reg	Hagoi		Masalog		0.1878	0.0598	196	3.14	0.010
Yr*Reg	Carolina	1982	Diablo	1982	-0.0421	0.0878	587	-0.48	1.000
Yr*Reg	Carolina	1982	Hagoi	1982	0.2615	0.0945	587	2.77	0.198
Yr*Reg	Carolina	1982	Masalog	1982	0.2141	0.1046	587	2.05	0.661
Yr*Reg	Carolina	1982	Carolina	1996	0.2121	0.0974	392	2.18	0.566
Yr*Reg	Carolina	1982	Diablo	1996	0.1620	0.0878	587	1.84	0.792
Yr*Reg	Carolina	1982	Hagoi	1996	0.1099	0.0945	587	1.16	0.991
Yr*Reg	Carolina	1982	Masalog	1996	0.6562	0.1046	587	6.27	<.001
Yr*Reg	Carolina	1982	Carolina	2008	-0.0335	0.0974	392	-0.34	1.000
Yr*Reg	Carolina	1982	Diablo	2008	0.3972	0.0878	587	4.52	0.001
Yr*Reg	Carolina	1982	Hagoi	2008	0.0169	0.0945	587	0.18	1.000
Yr*Reg	Carolina	1982	Masalog	2008	0.0813	0.1046	587	0.78	1.000
Yr*Reg	Diablo	1982	Hagoi	1982	0.3037	0.0828	587	3.67	0.015
Yr*Reg	Diablo	1982	Masalog	1982	0.2563	0.0942	587	2.72	0.220
Yr*Reg	Diablo	1982	Carolina	1996	0.2542	0.0878	587	2.89	0.147
Yr*Reg	Diablo	1982	Diablo	1996	0.2041	0.0739	392	2.76	0.200
Yr*Reg	Diablo	1982	Hagoi	1996	0.1520	0.0828	587	1.84	0.797
Yr*Reg	Diablo	1982	Masalog	1996	0.6983	0.0942	587	7.42	<.001
Yr*Reg	Diablo	1982	Carolina	2008	0.0086	0.0878	587	0.10	1.000
Yr*Reg	Diablo	1982	Diablo	2008	0.4393	0.0739	392	5.95	<.001
Yr*Reg	Diablo	1982	Hagoi	2008	0.0590	0.0828	587	0.71	1.000
Yr*Reg	Diablo	1982	Masalog	2008	0.1235	0.0942	587	1.31	0.977
Yr*Reg	Hagoi	1982	Masalog	1982	-0.0474	0.1004	587	-0.47	1.000
Yr*Reg	Hagoi	1982	Carolina	1996	-0.0495	0.0945	587	-0.52	1.000
Yr*Reg	Hagoi	1982	Diablo	1996	-0.0996	0.0828	587	-1.20	0.989
Yr*Reg	Hagoi	1982	Hagoi	1996	-0.1517	0.0884	392	-1.72	0.860
Yr*Reg	Hagoi	1982	Masalog	1996	0.3946	0.1004	587	3.93	0.006

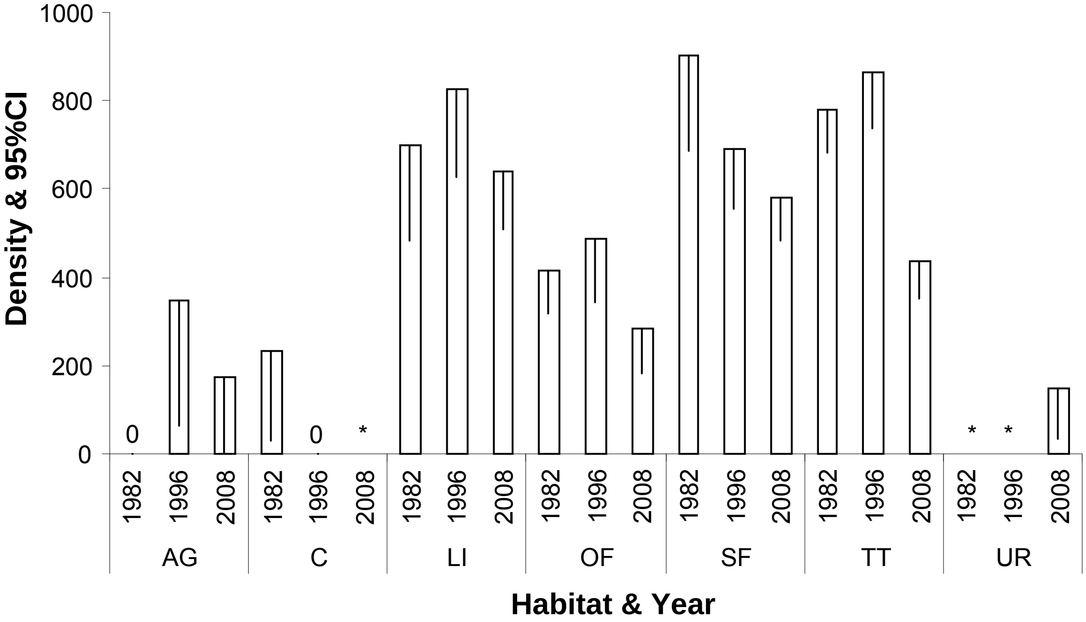
Yr*Reg	Hagoi	1982	Carolina	2008	-0.2950	0.0945	587	-3.12	0.081
Yr*Reg	Hagoi	1982	Diablo	2008	0.1357	0.0828	587	1.64	0.894
Yr*Reg	Hagoi	1982	Hagoi	2008	-0.2447	0.0884	392	-2.77	0.198
Yr*Reg	Hagoi	1982	Masalog	2008	-0.1802	0.1004	587	-1.79	0.820
Yr*Reg	Masalog	1982	Carolina	1996	-0.0021	0.1046	587	-0.02	1.000
Yr*Reg	Masalog	1982	Diablo	1996	-0.0522	0.0942	587	-0.55	1.000
Yr*Reg	Masalog	1982	Hagoi	1996	-0.1043	0.1004	587	-1.04	0.997
Yr*Reg	Masalog	1982	Masalog	1996	0.4420	0.1082	392	4.09	0.003
Yr*Reg	Masalog	1982	Carolina	2008	-0.2476	0.1046	587	-2.37	0.431
Yr*Reg	Masalog	1982	Diablo	2008	0.1831	0.0942	587	1.94	0.730
Yr*Reg	Masalog	1982	Hagoi	2008	-0.1972	0.1004	587	-1.96	0.717
Yr*Reg	Masalog	1982	Masalog	2008	-0.1328	0.1082	392	-1.23	0.987
Yr*Reg	Carolina	1996	Diablo	1996	-0.0501	0.0878	587	-0.57	1.000
Yr*Reg	Carolina	1996	Hagoi	1996	-0.1022	0.0945	587	-1.08	0.995
Yr*Reg	Carolina	1996	Masalog	1996	0.4441	0.1046	587	4.25	0.002
Yr*Reg	Carolina	1996	Carolina	2008	-0.2455	0.0974	392	-2.52	0.329
Yr*Reg	Carolina	1996	Diablo	2008	0.1851	0.0878	587	2.11	0.617
Yr*Reg	Carolina	1996	Hagoi	2008	-0.1952	0.0945	587	-2.07	0.648
Yr*Reg	Carolina	1996	Masalog	2008	-0.1307	0.1046	587	-1.25	0.985
Yr*Reg	Diablo	1996	Hagoi	1996	-0.0521	0.0828	587	-0.63	1.000
Yr*Reg	Diablo	1996	Masalog	1996	0.4942	0.0942	587	5.25	<.001
Yr*Reg	Diablo	1996	Carolina	2008	-0.1954	0.0878	587	-2.23	0.532
Yr*Reg	Diablo	1996	Diablo	2008	0.2352	0.0739	392	3.19	0.068
Yr*Reg	Diablo	1996	Hagoi	2008	-0.1451	0.0828	587	-1.75	0.842
Yr*Reg	Diablo	1996	Masalog	2008	-0.0806	0.0942	587	-0.86	0.999
Yr*Reg	Hagoi	1996	Masalog	1996	0.5463	0.1004	587	5.44	<.001
Yr*Reg	Hagoi	1996	Carolina	2008	-0.1433	0.0945	587	-1.52	0.935
Yr*Reg	Hagoi	1996	Diablo	2008	0.2874	0.0828	587	3.47	0.028
Yr*Reg	Hagoi	1996	Hagoi	2008	-0.0930	0.0884	392	-1.05	0.996
Yr*Reg	Hagoi	1996	Masalog	2008	-0.0285	0.1004	587	-0.28	1.000
Yr*Reg	Masalog	1996	Carolina	2008	-0.6897	0.1046	587	-6.59	<.001
Yr*Reg	Masalog	1996	Diablo	2008	-0.2590	0.0942	587	-2.75	0.206
Yr*Reg	Masalog	1996	Hagoi	2008	-0.6393	0.1004	587	-6.37	<.001
Yr*Reg	Masalog	1996	Masalog	2008	-0.5748	0.1082	392	-5.31	<.001
Yr*Reg	Carolina	2008	Diablo	2008	0.4307	0.0878	587	4.90	<.001
Yr*Reg	Carolina	2008	Hagoi	2008	0.0504	0.0945	587	0.53	1.000
Yr*Reg	Carolina	2008	Masalog	2008	0.1148	0.1046	587	1.10	0.995
Yr*Reg	Diablo	2008	Hagoi	2008	-0.3803	0.0828	587	-4.60	0.000
Yr*Reg	Diablo	2008	Masalog	2008	-0.3159	0.0942	587	-3.35	0.041
Yr*Reg	Hagoi	2008	Masalog	2008	0.0645	0.1004	587	0.64	1.000

Micronesian Starling

Effect	Region	Year	Region	Year	Estimate	SE	DF	*t* Value	Adj *P*
Yr		1982		1996	0.0097	0.0677	392	0.14	0.989
Yr		1982		2008	-0.6479	0.0677	392	-9.57	<.001
Yr		1996		2008	-0.6576	0.0677	392	-9.72	<.001
Reg	Carolina		Diablo		-0.0766	0.0877	196	-0.87	0.819

Reg	Carolina	Hagoi	0.0316	0.0944	196	0.34	0.987
Reg	Carolina	Masalog	0.2310	0.1045	196	2.21	0.124
Reg	Diablo	Hagoi	0.1082	0.0827	196	1.31	0.558
Reg	Diablo	Masalog	0.3076	0.0941	196	3.27	0.007
Reg	Hagoi	Masalog	0.1994	0.1003	196	1.99	0.196

Appendix 3. Break down of the Tinian Monarch population by habitat and year.



A) Plot of Tinian Monarch density estimates (birds/km^2) and lower 95% confidence interval by habitat and year from all transects (10 in 1982 and 1996, and 14 in 2008). Habitat types are AG – agriculture, C – coastal, LI – limestone forest, OF – open field, SF – secondary forest, TT – tangantangan thicket, and UR – urban/residential. No birds were detected in the agriculture habitat in 1982 or coastal habitat in 1996. No stations (indicated with *) were surveyed in the coastal habitat in 2008, and urban/residential habitat in 1996 and 2008.

B) Tinian Monarch density estimates (birds/km^2), standard error (SE), and 95% confidence intervals (Lower and Upper 95% CI) by habitat and year from all transects (10 in 1982 and 1996, and 14 in 2008). Habitat types are AG – agriculture, C – coastal, LI – limestone forest, OF – open field, SF – secondary forest, TT – tangantangan thicket, and UR – urban/residential. No birds were detected in the agriculture habitat in 1982 or coastal habitat in 1996. No stations (indicated with *) were surveyed in the coastal habitat in 2008, and urban/residential habitat in 1996 and 2008.

Habitat	Year	Estimate	SE	L 95%CI	U 95%CI
AG	1982	0.0			
	1996	349.4	201.96	63.561	1920.800
	2008	174.7	174.77	†	†
C	1982	232.9	116.66	30.770	1763.400
	1996	0.0			
	2008	*			
LI	1982	698.8	123.97	483.410	1010.200
	1996	825.9	111.49	625.330	1090.700
	2008	640.6	73.54	509.490	805.400
OF	1982	414.9	56.68	316.340	544.230
	1996	485.8	84.62	342.690	688.560
	2008	283.3	63.74	180.590	444.440
SF	1982	901.1	117.05	687.880	1180.400
	1996	691.2	76.45	553.930	862.540
	2008	582.4	54.28	483.960	700.740
TT	1982	778.2	51.52	682.940	886.810
	1996	863.2	68.80	737.510	1010.400
	2008	435.7	46.84	352.230	539.030
UR	1982	*			
	1996	*			
	2008	149.8	103.99	32.300	694.240

† Sample size was insufficient to estimate reliable confidence intervals.

C) Comparison of Tinian Monarch densities by habitat and year using repeated measures ANOVA from all transects (10 in 1982 and 1996, and 14 in 2008). Year and habitat fixed effects were significant but the year and habitat interaction was non-significant ($F_{8,623} = 0.62$, $p = 0.764$); therefore, only effects by habitat are presented here. Differences among years are presented in Table 4. Significant differences are highlighted in bold. Habitat types are LI – limestone forest, OF – open field, SF – secondary forest, and TT – tangantangan thicket; agriculture, coastal, and urban/residential (ACU) habitats were combined because insufficient numbers of stations were sampled in those habitats.

Effect	Num DF	Den DF	*F* Value	Pr > *F*
Habitat	4	645	15.04	<.0001

Effect	Effect	Estimate	Error	*t* Value	Adj *P*
ACU	**LI**	**-1.002**	**0.250**	**-4.00**	**<.001**
ACU	OF	-0.354	0.243	-1.45	0.592
ACU	**SF**	**-0.958**	**0.245**	**-3.91**	**<.001**
ACU	**TT**	**-0.999**	**0.236**	**-4.23**	**<.001**
LI	**OF**	**0.648**	**0.132**	**4.91**	**<.001**
LI	SF	0.044	0.134	0.33	0.998
LI	TT	0.003	0.118	0.03	1.000
OF	**SF**	**-0.604**	**0.119**	**-5.10**	**<.001**
OF	**TT**	**-0.645**	**0.100**	**-6.43**	**<.001**
SF	TT	-0.041	0.102	-0.40	0.995

SYSTEMATIC RODENT MONITORING

A Study of the Introduced Small Mammals of the Mariana Islands

Final Report to the USGS Brown Treesnake Project,
Fort Collins Science Center
Fort Collins, CO

submitted by

Andrew S. Wiewel, Amy A. Yackel Adams, and Gordon H. Rodda

United States Geological Survey
Fort Collins Science Center
2150 Centre Avenue, Building C
Fort Collins, CO 80526-8118

March 2008

ABSTRACT OF REPORT

Introduced small mammals frequently have detrimental impacts on island ecology, including competition with or predation on native flora and fauna. Introduced small mammals may also disrupt island trophic systems and alter large-scale ecosystem processes. However, our understanding of these effects is limited by incomplete knowledge of small mammal distribution, density, and biomass on many islands. Such information is especially critical in the Mariana Islands, where introduced small mammals are keystone prey for the introduced brown treesnake (*Boiga irregularis*) and small mammal density is inversely related to the effectiveness of brown treesnake control and management tools, such as mouse-attractant traps. Despite the importance of reliable small mammal population data for numerous conservation and management applications, researchers in the Mariana Islands (and elsewhere) often use sampling and analysis methods of questionable accuracy and precision, such as snaptrapping and count-based indices of abundance, perhaps because these methods are thought to be fast and inexpensive.

In an effort to address these concerns, we developed a robust and repeatable mark-recapture livetrapping methodology to determine introduced small mammal distribution, density, and biomass at 8 sites on Guam, 4 sites on Rota, 5 sites on Saipan, and 3 sites on Tinian. On each island, we sampled at least 1 grassland, *Leucaena* forest, and native limestone forest site. In addition, we conducted snaptrapping at these sites following livetrapping, which allowed direct comparison between these sampling methods as well as estimates in indices generated from them. Livetrapping and snaptrapping occurred between April 2005 and June 2007.

In chapter 1, we present density and biomass estimates generated from mark-recapture livetrapping sampling, and speculate on potential impacts on the ecology of Mariana Islands. Of the species captured, *Rattus rattus/R. tanezumi* (morphologically similar, genetic-based differentiation in progress; hereafter *R. rattus*) was most common across all habitats and islands. In contrast, *Suncus murinus* was not captured on Rota, *Mus musculus* was rarely captured at forested sites, and *R. exulans* and *R. norvegicus* were captured infrequently. Modeling of mark-recapture data indicated that neophobia, island, sex, reproductive status and rain amount influenced *R. rattus* capture probability, whereas time, island, and capture heterogeneity influenced *S. murinus* and *M. musculus* capture probability. Introduced small mammal density and biomass estimates generated from these models were much greater on Rota,

Saipan, and Tinian than on Guam, most likely a result of brown treesnake predation pressure on the latter island. *R. rattus* and *M. musculus* density and biomass were greatest in grassland, whereas *S. murinus* density and biomass were greatest in *Leucaena* forest. The high densities documented during this research suggest that introduced small mammals (especially *R. rattus*) may be impacting the abundance and diversity of native lizards, birds, and bats in the Mariana Islands. Ecological processes such as plant regeneration may also be affected. Further, brown treesnake control and management tools that rely on mouse attractants will be less effective on Rota, Saipan, and Tinian than on Guam. If the brown treesnake becomes established on these islands, high-density introduced small mammal populations may facilitate and support a high-density brown treesnake population, even as native species are reduced or extirpated.

In chapter 2, we investigate the precision of mark-recapture and removal abundance estimates generated from livetrapping and snaptrapping data and evaluate 2 count-based indices, number of individuals captured (M_{t+1}) and captures per unit effort (CPUE), as predictors of abundance. We also evaluate the cost and time associated with implementing livetrapping and snaptrapping and compare species-specific capture rates of selected live and snap traps. For all species, mark-recapture estimates were consistently more precise based on coefficients of variation and 95% confidence intervals. The predictive utility of both M_{t+1} and CPUE was relatively poor, but improved with increasing sampling duration over occasions 1–5. More importantly, modeling of sampling data revealed that underlying assumptions critical to the application of indices of abundance, such as spatially and temporally constant capture probability, were not met. Capture probability also varied as a function of covariates (sex, age, reproductive status, body size, and rain amount) for *R. rattus*. Snaptrapping was cheaper and faster than livetrapping, although the time difference was negligible when site preparation time was considered. We documented variable capture rates in different traps: *R. rattus* captures were greatest in Haguruma live and Victor snap traps, whereas *S. murinus* and *M. musculus* captures were greatest in Sherman live and Museum Special snap traps. While snaptrapping and count-based indices may have utility after validation against more rigorous sampling or estimation procedures, validation should occur across the full range of study conditions. Resources required for this level of validation would likely be better allocated towards implementing rigorous and robust methods.

ACKNOWLEDGMENTS

Funding for this research was provided by the U.S. Department of Interior, Office of Insular Affairs. We are indebted to numerous individuals who assisted with site preparation and small mammal sampling activities, including R. Bischof, V. Boyarski, I. Chellman, M. Christy, J. Farley, E. Fidler, G. George, A. Hambrick, T. Hinkle, P. Reynolds, S. Siers, J. Stanford, A. Tuggle, and J. Willey. Logistical support was provided by the CNMI DFW (A. Castro, N. Hawley, L. Williams, and R. Ulloa), Guam DAWR (P. Wenninger), U.S. Air Force (D. Lujan), USDA/APHIS Wildlife Services (C. Clark), U.S. Navy (A. Brooke, S. Vogt, and R. Wescom), and USFWS (C. Bandy, G. Deutscher, and C. Kessler). A. Ellingson assisted with sampling design and methodology, and S. Converse and T. R. Stanley provided helpful feedback during data analysis. We thank B. Lardner, W. Pitt, R. N. Reed, and K. Swift for comments on earlier drafts of this report.

TABLE OF CONTENTS

ABSTRACT OF REPORT ii
ACKNOWLEDGMENTS iv

CHAPTER 1: INTRODUCED SMALL MAMMAL DENSITY AND BIOMASS IN THE MARIANA ISLANDS: IMPLICATIONS FOR ISLAND ECOLOGY

INTRODUCTION 1

STUDY AREA 3

METHODS 4
  Study Site Selection 4
  Small Mammal Sampling 5
  Data Analysis 7

RESULTS 11
  Modeling Capture and Recapture Probability 12
  Density Estimates 14
  Biomass Estimates 15

DISCUSSION 16
  Modeling Capture and Recapture Probability 19
  Implications for Mariana Island Ecology and Brown Treesnake Control and Management 23

LITERATURE CITED 27

TABLES 37

FIGURES 46

APPENDIX 1A. Review of small mammal introductions and research in the Mariana Islands 59

APPENDIX 1B. Delta method procedures for calculating density and biomass variances 99

APPENDIX 1C. Comparison of density estimates from Programs MARK and DENSITY 114

CHAPTER 2: EVALUATING ABUNDANCE ESTIMATES AND THE ASSUMPTIONS OF A COUNT-BASED INDEX OF ABUNDANCE FOR SMALL MAMMALS

INTRODUCTION 122

METHODS 126
  Study Site Selection 127
  Small Mammal Sampling 127
  Data Analysis 129

RESULTS 134
Modeling Capture and Recapture Probability 134
Abundance Estimates...136
Evaluation of Count-Based Indices 137
Comparison of Livetrapping and Snaptrapping Capture Rates 138
Comparison of Livetrapping and Snaptrapping Cost and Effort..139

DISCUSSION...140
Comparison of Mark-Recapture and Removal Abundance Estimates 140
Evaluation of Count-Based Indices 141
Comparison of Livetrapping and Snaptrapping Capture Rates 144
Comparison of Livetrapping and Snaptrapping Cost and Effort..145
Management Implications 146

LITERATURE CITED...147

TABLES 154

FIGURES..169

APPENDIX 2A. Post-hoc investigation of possible geographic closure violations during small mammal sampling on Guam, Rota, Saipan, and Tinian, 2005–2006 181

APPENDIX 2B. Post-hoc evaluation of relationship between count-based indices and removal abundance estimates from snaptrapping data 186

CHAPTER 1: INTRODUCED SMALL MAMMAL DENSITY AND BIOMASS IN THE MARIANA ISLANDS: IMPLICATIONS FOR ISLAND ECOLOGY

INTRODUCTION

Introduced small mammals often have detrimental effects on island ecology (Atkinson 1985, Towns et al. 2006). Direct effects of introduced small mammals include competition with, or predation on, various amphibian (Worthy 1987, Towns and Daugherty 1994), avian (Fisher and Baldwin 1946, Wirtz 1972, Recher and Clark 1974, Atkinson 1977, Martin et al. 2000, Smith et al. 2006), invertebrate (Bremner et al. 1984; Kuschel and Worthy 1996; Brook 1999, 2000; Carlton and Hodder 2003; Priddel et al. 2003), mammalian (Daniel 1990, Goodman 1995, Pascal et al. 2005), and reptilian species (Whitaker 1973; Newman 1994; Towns 1994; Towns and Daugherty 1994; Cree et al. 1995; Hoare et al. 2007a,b), often resulting in population declines or even extirpation. Introduced small mammals may also suppress plant recruitment by consuming bark, flowers, foliage, fruits, seeds, or seedlings (Allen et al. 1994; Campbell and Atkinson 1999, 2002; McConkey et al. 2003; Wilson et al. 2003); in extreme cases this recruitment suppression can result in local extirpation (Campbell and Atkinson 1999, 2002). Less apparent but equally important indirect effects include disruption of island trophic systems (Fritts and Rodda 1998, Towns 1999) and nutrient cycling (Fukami et al. 2006), modification of vegetative community structure and successional patterns (Campbell and Atkinson 1999, 2002; Athens et al. 2002), and creation of novel vectors and reservoirs for diseases and parasites of both animals (Pickering and Norris 1996, Martina et al. 2006) and humans (Chanteau et al. 1998, Lindo et al. 2002, Bitam et al. 2006, Jiang et al. 2006).

Despite this growing body of evidence, our understanding of the effects of introduced small mammals on island ecology is far from complete. In an effort to provide a framework for future research, Parker et al. (1999) proposed that 3 factors determine the impact of introduced species on island ecosystems: their range, density or biomass, and effect per individual or per unit biomass. The authors suggest that range, density, and biomass are much easier to quantify than the per individual or unit biomass effect of introduced species (Parker et al. 1999). While this claim is valid from a procedural standpoint, in practice the range, density, and biomass of introduced small mammal species are poorly understood on many islands, severely limiting efforts to understand their effect on island ecology. This lack of knowledge is especially troubling for well-studied island systems, such as the Mariana Islands, infamous for the introduced brown treesnake (*Boiga irregularis*) and its negative impact on island ecology (Savidge 1987, Fritts and Rodda 1998, Rodda et al. 1999, Rodda and Savidge 2007). In spite of considerable research efforts associated with controlling the brown treesnake on Guam and preventing its spread to other vulnerable locations (Colvin et al. 2005), relatively little is known about the introduced small mammal community, including such basic information as the number and identity of introduced small mammal species currently present.

Based on a review of available data (Appendix 1A, Tables 1A.1 and 1A.2), the introduced small mammal community of the Mariana Islands consists of 5 or 6 species (with possible additional subspecies), ranging from the earliest introduction, *Rattus exulans*, which occurred no later than A.D. 1000–1200 (Steadman 1999) to the most recent introduction, *Suncus murinus*, first captured on Guam in 1953 (Peterson 1956). Later introductions include *Mus musculus*, first reported on Guam in 1819 (Freycinet 2003:88), and *R. norvegicus*, first reported on Saipan in the late 1800's (Kuroda 1938 cited by Wiles et al. 1990). Regarding the polytypic species *M. musculus*, it is not clear which, or how many, subspecies (*M. m. musculus*, *M. m. domesticus*, or *M. m. castaneus*; Musser and Carleton 2005:1400–1401) have been introduced. It is notable that Prager et al. (1998) found *M. m. castaneus* on Tinian, although this identification was based on genetic analysis of a single speciment. Two additional species, *R. rattus* and *R. tanezumi*, have been documented in the Mariana Islands (Baker 1946, Johnson 1962, Yosida et al. 1985), although their current status is unclear. The complex taxonomic history of these closely related species (Musser and Carleton 2005:1484–1487, 1489–1491), which were only recently separated based on karyotypic differences (*R. rattus*: $2n = 38$; *R. tanezumi*: $2n = 42$) as well as biochemical and morphological features (Schwabe 1979, Baverstock et al. 1983), complicates the investigation of historic introductions and current distribution. Additional confusion arises from the limited hybridization observed in both laboratory (Yosida et al. 1971) and wild (Baverstock et al. 1983) populations, which led Baverstock et al. (1983:978) to conclude that *R. rattus* and *R. tanezumi* "…are best considered as incipient species. Where they meet, they may introgress, become sympatric without interbreeding, or one may replace the other depending upon the prevailing biological conditions."

Attempts to use available historic data to investigate introduced small mammal density and distribution in the Mariana Islands are complicated by the common reliance on non-rigorous sampling techniques, such as low sampling effort and convenience sampling (Anderson 2001, 2003), and the frequent reporting of indices of density (Appendix 1A, Table 1A.2). As a result, retrospective comparisons across sites, habitats, or islands require unrealistic assumptions about the equality of detection probability across space and time (Anderson 2001, 2003), inhibiting our understanding of introduced small mammal populations as well as our ability to investigate possible effects of introduced small mammals on both the ecology of the Mariana Islands and brown treesnake control and management. Thus, our objectives were to: 1) determine the distribution of introduced small mammals across the major habitats of Guam and the nearby islands of Rota, Saipan, and Tinian and 2) generate robust estimates of introduced small mammal density and biomass in these locations using rigorous and repeatable mark-recapture livetrapping methods.

STUDY AREA

The Mariana Islands consist of 15 islands arrayed in a north-south arc between approximately 13° and 21° N and 144° and 146° E (Metteler 1986; Figure 1). The marine tropical climate of the Mariana Islands results in minimal seasonal temperature variation, with monthly averages ranging between 24° and 27° C (Mueller-Dombois and Fosberg 1998:241). Precipitation is seasonal, with a rainy season from July to October, and averages 2000–2500 mm per year (Mueller-Dombois and Fosberg 1998:241). Tropical storms and typhoons occur frequently in the Mariana Islands, especially during the rainy season (Mueller-Dombois and Fosberg 1998:241).

Introduced small mammal sampling occurred on the permanently inhabited islands of Guam, Rota, Saipan, and Tinian (Figure 1). Guam is the largest island in the Mariana chain (544 km^2) and also has the greatest human population (154,805; U.S. Census Bureau 2004). Rota (85 km^2), Saipan (115 km^2), and Tinian (101 km^2), are each much smaller and have low (Rota: 3,283; Tinian: 3,540) to moderate (Saipan: 62,392) human populations (U.S. Census Bureau 2003). Geologically, these islands consist of a mixture of upraised coral plateaus and weathered volcanic substrates, with exposed volcanic rock being least common on Tinian (Mueller-Dombois and Fosberg 1998:254). The dominant native vegetative community of the upraised coral plateaus is limestone forest, which is most common and least disturbed on Rota (Falanruw et al. 1989). Native limestone forests in the Mariana Islands have highly variable structure and species composition, primarily as a function of slope, aspect, and elevation as well as the frequency and extent of typhoon damage and human disturbance (Mueller-Dombois and Fosberg 1998:242, 270–271). Common native forest species include *Aglaia mariannensis*, *Artocarpus mariannensis*, *Cycas circinalis*, *Cynometra ramiflora*, *Elaeocarpus joga*, *Ficus prolixa*, *Guamia mariannae*, *Ochrosia mariannensis*, *Pandanus dubius* and *P. tectorius*, *Pisonia grandis*, and *Premna obtusifolia* (Falanruw et al. 1989, Mueller-Dombois and Fosberg 1998:271). Grasslands or sparsely-forested savannahs are typically found on areas of exposed volcanic substrate, especially in southern Guam, the Sabana region of Rota, and the central ridge of Saipan. These grassland/savannah habitats are generally dominated by *Dimeria chloridiformis*, *Miscanthus floridulus*, or *Pennisetum polystachyon*; other commonly encountered grassland species include *Casuarina equisetifolia*, *Dicranopteris linearis*, and *Lycopodium cernuum* (Falanruw et al. 1989, Mueller-Dombois and Fosberg 1998:259, 268, 272). On Tinian, which lacks extensive areas of exposed volcanic substrate, non-forested areas are generally covered by a mixture of invasive weeds, especially *Mimosa invisa* (Falanruw et al. 1989, Mueller-Dombois and Fosberg 1998:264). Human disturbance, including highly destructive activities during and after World War II, as well as frequent storm damage, have modified the vegetative community of large areas in the Mariana Islands. Many of these disturbed areas, especially on Guam, Saipan, and Tinian, have been recolonized by the introduced leguminous tree *Leucaena leucocephala*. This species often exists in nearly monotypic stands, but is also commonly found in association with *Flagellaria indica*, *Hibiscus tiliaceus*, *Nephrolepis biserrata* and *N. hirsutula*, and *Triphasia trifolia* (Falanruw et al. 1989, Mueller-Dombois and Fosberg 1998:264).

METHODS

For a complete description of the study site selection and small mammal sampling protocols used during this research (described below), please refer to Wiewel (2005).

Study Site Selection

We sampled 8 sites on Guam (one of which was sampled annually; Figure 2), 4 sites on Rota (Figure 3), 5 sites on Saipan (Figure 4), and 3 sites on Tinian (Figure 5) between April 2005 and June 2007 (Table 1). Study sites were identified using a combination of 1:24,000 and 1:25,000 scale topographical maps (U.S. Geological Survey 1999a,b,c; 2000) and 1:20,000 scale vegetation maps (Falanruw et al. 1989). Sites were evaluated based on habitat type, available area of relatively homogeneous habitat, and land ownership status. Selected sites represented the 3 major habitat types of the southern Mariana Islands: native limestone forest, grassland, and *L. leucocephala*-dominated secondary forest. Additional sites were selected near airports and seaports, independent of habitat type, based on a desire to better understand introduced small mammal populations in these areas. Both Johnson (1962) and Musser and Carleton (2005:1485) stated that on islands with *R. tanezumi*, *R. rattus* is restricted to ships in harbor and only rarely able to colonize onshore areas; thus, seaport (and presumably airport) areas were deemed important for understanding *R. rattus* and *R. tanezumi* distributions. Airports and seaports are also critical areas for control and management efforts aimed at preventing transport of brown treesnakes from Guam to other islands. Sites near airports and seaports generally included a mixture of habitat types (typically grassland and *L. leucocephala*-dominated secondary forest) and were classified as mixed habitat. With the exception of mixed habitat sites, potential sites contained ≥4 ha of relatively homogeneous habitat. Sites were located primarily on military and public lands because these areas generally offered larger tracts of homogeneous habitat and because information about private land ownership and permission for access were often difficult to obtain. On each island, at least 1 native limestone forest site, 1 grassland site, and 1 *L. leucocephala*-dominated secondary forest site were selected and sampled. Five sites were sampled near airports and seaports on Guam (*n* = 2), Rota (*n* = 1), and Saipan (*n* = 2; Table 1).

Small Mammal Sampling

Due to the uncertainty surrounding the status of *R. rattus* and *R. tanezumi* in the Mariana Islands, we collected genetic material from all captured *Rattus* to allow determination of species identification and distribution. Preliminary analysis of the cytochrome oxidase I mtDNA region of 8 specimens from northern and central Guam indicated that all were *R. diardii* (sensu Robins et al. 2007), rather than the expected *R. rattus* and *R. tanezumi*. Until samples from all islands are processed, however, we will use the more recognized term *R. rattus* to refer to the combined sample of unidentified *Rattus* species.

At each site, mark-recapture livetrapping was conducted for 5 consecutive nights on an 11 × 11 grid with 12.5 m intervals between each trap station (grid area = 1.56 ha). A single standard-length folding

Sherman live trap (229 × 89 × 76 mm; H.B. Sherman Traps, Inc., Tallahassee, FL) was placed at each trap station (*n* = 121) and a single Haguruma wire mesh live trap (approximately 285 × 210 × 140 mm; Standard Trading Co., Honolulu, HI) was placed at every other trap station (*n* = 36; Figure 6). This trapping design was based on the general home range requirements of the species most likely to be captured in each trap. For example, historic research on Guam suggests that *M. musculus* has an average home range diameter of approximately 50 m, with average female and male movements of 27 and 33 m, respectively, between captures (Baker 1946). Similarly, *S. murinus* home ranges on Guam typically range 14–60 m in diameter (Barbehenn 1974a). Thus, standard-length Sherman traps, which are more likely to capture *M. musculus* and *S. murinus* (Gragg 2004, Wiewel 2004b), were placed at 12.5 m intervals to increase the likelihood that individual *M. musculus* and *S. murinus* within the trapping grid were exposed to multiple traps. *Rattus* species typically have larger home ranges than *M. musculus* or *S. murinus*. For example, male *R. rattus* home ranges vary between approximately 0.94 ha (Dowding and Murphy 1994) and 4.2 ha (Lindsey et al. 1999), with females exhibiting slightly smaller home ranges. Spencer and Davis (1950) recorded movements between successive captures of <60 m for 66% of adult male, 77% of adult female, and 84% of juvenile *R. rattus*. *R. exulans* home ranges vary from 0.16 ha (Strecker 1962) to 2.8 ha (Lindsey et al. 1999). Thus, Haguruma traps, which are more likely to capture *Rattus* species (Gragg 2004, Wiewel 2004a) were spaced at 25 m intervals to better match the larger average home range of these species.

Closed traps were placed on the grid a minimum of 2 nights prior to the beginning of sampling to provide an opportunity for small mammals to acclimate to their presence. Traps were placed on the ground and, whenever possible, located next to or beneath clumps of grass, downed woody debris, or rocks to provide shelter from sun and rain. Traps were baited with a mixture of peanut butter, oats, and food-grade paraffin (Wiewel 2004b) and were checked beginning at 0730–0800 each day. Traps were closed during the day to minimize trap mortality. Traps were reopened at approximately 1600 and rebaited as necessary to ensure bait freshness.

Captured animals were examined and measured to determine species, sex, age, reproductive status, mass (g), head-body length (mm), tail length (mm), right hind foot length (mm), right ear length (mm), and testes length (mm; if applicable). Captured individuals were uniquely marked in each ear with numbered metal ear tags (*M. musculus* and *S. murinus*: small ear tags produced by S. Roestenburg, Riverton, UT; *Rattus* species: #1005-1, National Band and Tag Co., Newport, KY). Recaptured animals were examined to determine tag number. All capture, handling, and marking techniques followed guidelines approved by the American Society of Mammalogists (Gannon et al. 2007) and the U.S. Geological Survey Animal Care and Use Committee (Fort Collins Science Center).

Each site (except for CP05, CP06, and CP07; Table 1) was also sampled with 5 consecutive nights of snaptrapping during the week following livetrapping. Results of snaptrapping are described elsewhere (see Chapter 2); however, data collected during snaptrapping were included in the calculation of both body condition index (a covariate used in mark-recapture abundance estimation) and mean maximum distance moved (MMDM; used in density estimation).

Data Analysis

We estimated density and biomass separately for each species. First, we generated site- specific estimates of abundance using estimated capture and recapture probability modeled from livetrapping data. Because these estimates had no associated area component, our second step was to estimate the effective trapping area (ETA) for each site with reference to each species' mean maximum distance moved (MMDM) between captures. Third, we estimated density as abundance/ETA. Fourth, we determined mean body mass based on measurements of captured animals at each site. Fifth, for each site we estimated biomass as the product of site-specific density and site-specific mean body mass. Finally, we created variance-covariance matrices to separately calculate the variances of density and biomass estimates.

Data analysis generally followed an information-theoretic approach involving model selection and multi-model inference. Model selection was based on Akaike's Information Criterion (AIC; Akaike 1973) corrected for small sample size (AIC_c; Hurvich and Tsai 1989). Models were considered competitive with the top-ranked model when $\Delta AIC_c \leq 2.0$ (Burnham and Anderson 2002:131). Model-averaging was based on Akaike weights (Burnham and Anderson 2002:150) and included the entire model set except for models with nonsensical β or real parameter estimates, which were removed prior to model averaging. We defined nonsensical β estimates as those with standard error (SE) >> β (e.g., β =

16.8, SE(β) = 2084.6) and nonsensical real parameter estimates as those with SE = 0. Unless otherwise indicated, all estimates are presented as mean ± 1 SE.

Abundance Estimation.–Abundance estimates were generated from livetrapping data in Program MARK 4.3 (White and Burnham 1999) using the conditional likelihood closed capture-recapture model developed by Huggins (1989, 1991). The Huggins model uses estimates of capture probability and the number of individuals captured to estimate abundance. Encounter histories are used to estimate capture probability and can account for heterogeneity in capture probability from temporal, behavioral, and individual effects (both in the form of finite mixture distributions [Norris and Pollock 1996, Pledger 2000] and covariates [Huggins 1989, 1991]). In this context, mixture distributions are an attempt to deal with individual heterogeneity by grouping animals with similar capture probabilities into discrete classes for modeling purposes (Pledger 2000). For example, a 2-mixture distribution groups individuals into 2 classes of high and low capture probability. Similarly, covariates are variables thought to influence capture probability (and other demographic parameters) which, when added to capture probability models, may reduce unexplained heterogeneity and thereby improve parameter estimation (Pollock et al. 1984, Pollock 2002). Covariates may pertain to individual animals (e.g., age, sex, mass), in which case they are generally assumed constant over time for modeling purposes, or to the environment (e.g., temperature, precipitation), in which case they are generally assumed constant for all animals over a specified time span, such as 24 hours (Pollock et al. 1984, Pollock 2002).

In Program MARK, design matrices were coded to allow sites to be treated both individually and as groups, based on common attributes such as island or habitat. Capture and recapture probability were primarily modeled across these groups to increase statistical efficiency (i.e., reduce estimate variance) and allow abundance estimates to be generated from sites with few captures or recaptures (Bowden et al. 2003, White 2005, Conn et al. 2006, Converse et al. 2006). Models were specified using the logit link function to constrain parameter estimates to the range 0–1 and to allow the use of non-identity design matrices (Cooch and White 2005). Model building in Program MARK occurred in an iterative fashion, beginning with the traditional mark-recapture models (M_0, M_b, M_t, M_h, M_{tb}, M_{bh}, M_{th}, M_{tbh}) outlined by Otis et al. (1978), where subscripts indicate the type of capture probability variation dealt with by each model: *b* = behavioral variation, *t* = temporal variation, *h* = heterogeneity, and *0* = constant capture probability. Models incorporating heterogeneity effects were specified as 2-mixture models, based on concerns that our dataset would not support a more parameterized mixture model (Conn et al. 2006). Models were ranked based on AIC_c scores, with the top model being used for further model development. If the top ranked model included temporal variation, a set of neophobia models were fit to the dataset. Neophobia models allowed capture probability to vary during the first (neo1) or first and second (neo2) sampling occasions, while holding capture probability constant for the remaining sampling occasions. The motivation for neophobia models came both from literature accounts of neophobia for introduced small mammals (Inglis et al. 1996, Thorsen et al. 2000, Clapperton 2006), as well as observations of an increase in number of individuals captured after the first or second sampling occasion at many of our sites. As before, the top ranked model was used for further model development. The next subset of models added to the MARK analysis were parameterized to model capture probability, recapture probability, or both capture and recapture probability as a function of island, habitat, or site. This complexity was deemed necessary to investigate possible variation in capture and recapture probability across these groupings. We hypothesized that capture or recapture probability might differ between Guam (with brown treesnake predation pressure) and Rota, Saipan, and Tinian (without brown treesnake predation), so the island grouping was coded in 2 ways, with island[4] distinguishing between each island and island[2] distinguishing Guam from the combination of Rota, Saipan, and Tinian. Again, the top ranked model was used for further model development.

The final subset of models added to the MARK analysis contained combinations of 5 individual and 2 environmental covariates, beginning with the full model containing all covariates and proceeding through a series of more parsimonious models including only those covariates important for explaining capture probability. Covariate importance was assessed through examination of β values and 95% CIs, where covariates with non-zero overlapping 95% CIs were considered influential on capture probability. Model-averaged abundance estimates were then generated from this pool of models to account for model selection uncertainty, unless the top ranked model had a model weight > 0.90 (Burnham and Anderson 2002:150). Covariates under consideration included sex (male or female), age (adult or juvenile), reproductive status, body condition index, body size, rain previous night, and rain amount. Reproductive status (repstat) was a categorical variable that differentiated reproductively active adults from non-reproductive adults and juveniles; assignment of repstat class was based on mass and the presence of externally visible sexual characteristics such as descended testes for males and active lactation for females. Body condition index (bodycon) was calculated as the ratio between the observed and expected mass of an individual, where expected mass was determined from a linear regression of ln mass vs. ln head-body length. The expected mass regression was generated using mass and head-body measurements from all individuals (i.e., animals captured during both livetrapping and snaptrapping). For each species, variation in bodycon was modeled as a function of island[4], island[2], and habitat in an analysis of variance framework (Proc GLM, SAS Institute 2003; Table 2). A site-specific bodycon model was not considered because of sparse data for some sites, which might have biased bodycon estimates for individuals from those sites. Bodycon estimates from the top model (or the model-averaged bodycon estimate) for each species were included in MARK modeling. Body Size (size) was a species-specific composite variable created from a principle components analysis (Proc FACTOR, SAS Institute 2003) of mass, head-body length, tail length, hind foot length, and ear length measured for each captured individual. Rain previous night (rainprev) was a categorical measure of the presence or absence of rainfall during the night prior to each trap monitoring occasion. Finally, rain amount (rainamt) was a quantitative measure of the total rainfall (mm) at the center of the trapping grid during each 24-hour sampling occasion, with the exception of the first sampling occasion for which the rainfall measurement encompassed only a 12–16 hour period. Prior to including rainamt in MARK models, rainfall amounts for the 5 sampling occasions were examined for equality across sites. Based on overlapping 95% CIs, there was no effect of the abbreviated rainfall measurement period during the first sampling occasion (Table 3).

Density Estimation.–Species-specific density estimates were generated by dividing the model-averaged abundance estimates from Program MARK by estimates of the effective trapping area (ETA), where ETA was calculated as the total area encompassed by the trapping grid (1.56 ha) plus a boundary strip equal to ½ the mean maximum distance moved (MMDM) between captures for individuals captured ≥2 times (Wilson and Anderson 1985). For the purposes of MMDM calculation, livetrapping and snaptrapping data were combined to increase sample size, after first verifying that movements between captures were not significantly different between sampling methods. Snaptrapping movement observations occurred when animals captured and marked during livetrapping were recaptured during snaptrapping. The combination of livetrapping and snaptrapping datasets increased movement sample size by 41% for *M. musculus*, 45% for *S. murinus*, and 58% for *R. rattus*. For each species, variation in MMDM was modeled as a function of island[4], island[2], and habitat in an analysis of variance framework (Proc GLM, SAS Institute 2003; Table 4). A site-specific MMDM model was not considered because of sparse data for some sites. MMDM estimates from the top model (or model-averaged MMDM estimates) for each species were then used to calculated density. Variance-covariance matrices for density and ETA estimates were computed using the delta method (Seber 2002) and used to determine the variance of derived density estimates (Appendix 1B). We also evaluated an alternative density estimation technique implemented in Program DENSITY (Efford 2004), which avoids potential complications associated with the use of MMDM and ETA (Anderson et al. 1983, Efford 2004). Estimates from Program DENSITY were compared with our density estimates generated using Program MARK (Appendix 1C).

Biomass Estimation.—Biomass was calculated for each species as the product of site-specific density and site-specific mean body mass. For individuals captured multiple times, mean individual mass was used when estimating site-specific mean body mass. Variation in mass was modeled as a function of island[4], island[2], habitat, and site in an analysis of variance framework (Proc GLM, SAS Institute 2003; Table 5). Variance-covariance matrices for density and mass (using estimates from the top mass model or model-averaged mass estimates) were then computed using the delta method (Seber 2002) and used to determine the variance of the derived biomass estimates.

RESULTS

We captured 707 *R. rattus*, 298 *S. murinus*, 154 *M. musculus*, 16 *R. exulans*, and 5 *R. norvegicus* in 17,270 trap nights (Table 6). *R. rattus*, captured at 17 of 20 sites, was the only species captured in all sampled habitats and on all islands (Table 6). *S. murinus*, captured at 9 of 20 sites, was also captured in all sampled habitats but was not captured or observed on Rota (Table 6). In contrast, *M. musculus* was consistently captured at grassland sites only and at 8 of 20 sites overall (Table 6). *R. exulans* and *R. norvegicus* were rarely captured and were not included in density and biomass estimation. In general, captures of all species were greater on Rota, Saipan, and Tinian than on Guam.

Modeling Capture and Recapture Probability

R. rattus capture and recapture probability were best explained by an additive model (w_i = 0.871) allowing neophobic temporal variation (neo2) in capture probability for each island (island[4]; Figure 7), as well as capture probability variation by sex, repstat, and rainamt with recapture probability varying by island (island[4]), sex, repstat, and rainamt (Table 7). All plausible models contained the neo2 effect on capture probability; the best model without neo2 had no support (ΔAIC_c = 33.92). A post-hoc replacement of neo2 with neo1 in the top model reduced the parameter count by 4 but resulted in a less plausible model (ΔAIC_c = 9.81), whereas replacing neo2 with the fully parameterized time model resulted in a ΔAIC_c of 4.36. Attempts to model heterogeneity using mixture models generated nonsensical estimates for the mixture parameter (e.g., 0.52 ± 1.46, 95% CI = 0–1 for M_h or 0.98 ± 0.00, 95% CI = 0.98–0.98 for M_{tbh}). In contrast, covariates were useful for modeling heterogeneity; the addition of sex, repstat, and rainamt to capture and recapture probability greatly improved model fit compared to a model that allowed neophobic temporal variation (neo2) for each island (island[4]) without covariates (ΔAIC_c = 17.97). *R. rattus* capture and recapture probability were lower for males than for females (β_{sex} = -0.44 ± 0.15, 95% CI = -0.75– -0.14), higher for reproductively mature individuals ($\beta_{repstat}$ = 0.47 ± 0.15, 95% CI = 0.17–0.77), and positively correlated with rainfall ($\beta_{rainamt}$ = 0.02 ± 0.01, 95% CI = 0.01–0.04). Reproductively mature females were more than twice as likely to be captured and recaptured as non-mature males (Figures 8, 9). Model-averaged *R. rattus* abundance estimates generated from these models varied between sites, but were generally greatest on Tinian ($\hat{N}$ range = 86–194, n = 3) and Rota ($\hat{N}$ range = 18–186, n = 4), followed by Saipan ($\hat{N}$ range = 15–91, n = 5) and Guam ($\hat{N}$ range = 2–41, n = 9; Table 8).

S. murinus capture and recapture probability were best explained by an additive model allowing temporal variation and heterogeneity in capture probability and temporal variation for each island (island[4]) and heterogeneity in recapture probability (w_i = 0.994; Table 7). Estimated capture and recapture probability increased over time. The best model without a temporal effect had no support (ΔAIC_c = 32.43). Unexplained heterogeneity was approximated by 2 mixture classes which comprised 65% (low capture probability) and 35% (high capture probability) of the population. Thus, well over half of the population had an estimated maximum capture probability of <0.16 (Figure 10A) and a maximum recapture probability <0.19 (Figure 10B). Peak recapture probability for both mixture classes was observed on Guam (Figure 10B). In contrast to *R. rattus*, none of the covariates under consideration were useful for modeling heterogeneity in *S. murinus* capture or recapture probability; the best model containing covariates had essentially no weight ($w_i = 0.006$; Table 7). Due to the high level of support for the top model ($w_i = 0.994$), *S. murinus* abundance estimates were generated from this model alone. These estimates were varied between sites, but were generally greatest on Tinian ($\hat{N}$ range = 17–143, $n = 3$), followed by Saipan ($\hat{N}$ range = 14–71, $n = 5$) and Guam ($\hat{N}$ range = 0–20, $n = 9$; Table 8).

M. musculus capture and recapture probability were best explained by 3 additive models allowing both temporal variation and heterogeneity in these parameters (Table 7). In the top model ($w_i = 0.349$), temporal variation in capture probability varied between Guam and the combination of Rota, Saipan, and Tinian (island[2]). The second-ranked model ($w_i = 0.331$) differed only by the addition of temporal variation by island[2] on recapture probability. The third-ranked model ($w_i = 0.192$) differed from the top model by allowing the temporal variation in capture probability to differ for each island (island[4]). Model-averaged capture and recapture probability for Guam varied across time, whereas model-averaged capture and recapture probability for Rota, Saipan, and Tinian were relatively constant across sampling occasions (Figure 11). The best model without a temporal effect had no support ($\Delta AIC_c = 15.91$). Unexplained heterogeneity was approximated by 2 mixture classes which comprised 67.3% (low probability) and 32.7% (high probability) of the population. On Guam, capture and recapture probabilities differed by 0.35–0.54 between the low and high probability mixtures (Figure 11). For Rota, Saipan, and Tinian, over half of the population had estimated maximum capture and recapture probabilities <0.20 and <0.12, respectively, with the remainder of the population exhibiting high capture and recapture probabilities (Figure 11). As with *S. murinus*, none of the covariates under consideration were useful for modeling *M. musculus* abundance; the best model containing covariates had essentially no weight ($w_i = 0.003$; Table 7). Model-averaged *M. musculus* abundance estimates generated from these models were variable between sites, but was generally greatest on Saipan ($\hat{\bar{N}}$ range = 0–81, $n = 5$) and Rota ($\hat{\bar{N}}$ range = 2–53, $n = 4$), followed by Guam ($\hat{\bar{N}}$ range = 0–18, $n = 9$) and Tinian ($\hat{\bar{N}}$ range = 0–15, $n = 3$; Table 8).

Density Estimates

R. rattus MMDM varied primarily between islands (island[4]; w_i = 0.977; Table 4), and was greatest on Guam (35.6 ± 5.4 m, 95% CI = 24.6–46.7; *n* = 33), followed by Saipan (22.8 ± 2.5 m, 95% CI = 17.8–27.9; *n* = 100), Rota (14.5 ± 1.6 m, 95% CI = 11.4–17.6; *n* = 175), and Tinian (14.5 ± 1.3 m, 95% CI = 11.9–17.1; *n* = 180). When combined with the nominal grid area of 1.56 ha, these MMDM estimates resulted in ETAs of 2.58 ha for Guam, 2.19 ha for Saipan, and 1.95 ha for Rota and Tinian and mean *R. rattus* density estimates of 73.0/ha (*n* = 3) on Tinian, 53.5/ha (*n* = 4) on Rota, 25.6/ha (*n* = 5) on Saipan, and 5.1/ha (*n* = 9) on Guam (Table 9).

S. murinus MMDM varied primarily between habitats (w_i = 0.987; Table 4), and was greatest in grassland (29.2 ± 2.7 m, 95% CI = 23.7–34.7; *n* = 48), followed by mixed habitat (19.3 ± 3.2 m, 95% CI = 12.7–25.9; *n* = 25), *Leucaena* forest (16.3 ± 1.4 m, 95% CI = 13.6–19.0; *n* = 68), and native forest (14.2 ± 3.5 m, 95% CI = 6.4–22.0; *n* = 12). When combined with the nominal grid area of 1.56 ha, these MMDM estimates resulted in ETAs of 2.38 ha for grassland, 2.08 ha for mixed habitat, 2.00 ha for *Leucaena* forest, and 1.94 ha for native forest. Because many more *S. murinus* were captured on Saipan and Tinian than on Guam, we considered habitats separately for these areas. On Saipan and Tinian, mean estimated density was 52.8/ha (*n* = 2) in *Leucaena* forest, 24.2/ha (*n* = 2) in native forest, 20.2/ha (*n* = 2) in mixed habitat, and 9.7/ha (*n* = 2) in grassland (Table 9). On Guam, estimated density was 8.6/ha (*n* = 1) in grassland and 0/ha in the other habitats (*n* = 8; Table 9).

M. musculus MMDM varied primarily between islands (island[4]; w_i = 0.718), although there was also support for the simpler island model (island[2]) differentiating only between Guam and the other islands (w_i = 0.272; Table 4). Model-averaged MMDM was greatest on Guam (31.2 ± 3.6 m, 95% CI = 22.1–38.3; *n* = 25), followed by Saipan (22.8 ± 2.5 m, 95% CI = 17.9–27.7; *n* = 77), Rota (18.2 ± 3.2 m, 95% CI = 11.9–24.5; *n* = 59), and Tinian (11.7 ± 8.7 m, 95% CI = 0–28.0; *n* = 3). When combined with the nominal grid area of 1.56 ha, these MMDM estimates resulted in ETAs of 2.44 ha for Guam, 2.18 ha for Saipan, 2.03 ha for Rota, and 1.87 ha for Tinian. These model-averaged ETAs produced mean *M. musculus* density estimates of 15.8/ha (*n* = 4) on Rota, 7.7/ha (*n* = 5) on Saipan, 2.6/ha (*n* = 3) on Tinian, and 0.8/ha (*n* = 9) on Guam (Table 9).

Biomass Estimates

R. rattus, *S. murinus*, and *M. musculus* varied dramatically in morphology (Table 10), with mean *R. rattus* mass being much greater (121.9 ± 1.8 g, 95% CI = 118.3–125.5; *n* = 707) than mean *S. murinus* mass (25.7 ± 0.4 g, 95% CI = 25.0–26.5; *n* = 298) or mean *M. musculus* mass (12.5 ± 0.2 g, 95% CI = 12.1–12.9; *n* = 154). *R. rattus* mass varied by site (w_i = 1.000), whereas *S. murinus* and *M. musculus* mass varied primarily by habitat (w_i = 0.974) and island (w_i = 0.903), respectively (Table 5). Mean *S. murinus* mass was greatest in mixed habitat (28.6 ± 0.9 g, 95% CI = 26.8–30.3; *n* = 56), followed by *Leucaena* forest (26.0 ± 0.6 g, 95% CI = 24.9–27.1; *n* = 136), native forest (25.5 ± 0.6 g, 95% CI = 24.2–26.7; *n* = 62), and grassland (21.5 ± 1.0 g, 95% CI = 19.5–23.4; *n* = 44). Mean *M. musculus* mass was greatest on Tinian (14.4 ± 1.0 g, 95% CI = 12.2–16.7; *n* = 9), followed by Rota (12.8 ± 0.2 g, 95% CI = 12.4–13.3; *n* = 77), Saipan (12.1 ± 0.3 g, 95% CI = 11.5–12.6; *n* = 53), and Guam (11.0 ± 0.7 g, 95% CI = 9.4–12.6; *n* = 15).

R. rattus biomass was markedly greater than *S. murinus* or *M. musculus* biomass across sampled habitats and islands (Table 11). In fact, there was only 1 site (SAEN) where estimated *S. murinus* biomass was similar to *R. rattus* biomass, and 1 site (ACHU) where estimated *M. musculus* biomass was >5% of *R. rattus* biomass (Table 11). Mean estimated *R. rattus* biomass was greatest on Tinian and Rota, with maximum estimates of 11.6 and 9.8 kg/ha, respectively, and was roughly 3–8 times greater at sites on Rota, Saipan, and Tinian than on Guam (Figure 12, Table 11). Similarly, mean estimated *S. murinus* biomass was greatest on Tinian and Saipan, with maximum estimates of 1.9 and 0.9 kg/ha, respectively, and mean estimated *M. musculus* biomass was greatest on Rota and Saipan, with maximum estimates of 0.3 and 0.4 kg/ha, respectively (Figure 12, Table 11).

When evaluating biomass across habitats, we separated Rota, Saipan, and Tinian from Guam due to dramatically higher *R. rattus*, *S. murinus*, and *M. musculus* biomass on these islands. On Rota, Saipan, and Tinian, mean *R. rattus* biomass was greatest in grassland (Figure 13), with a maximum estimate of 11.6 kg/ha in this habitat (Table 11). In other habitats, mean estimated *R. rattus* biomass was roughly half that estimated for grassland (Figure 13), although maximum biomass estimates exceeded 8 kg/ha in both mixed habitat and native forest (Table 11). In contrast to *R. rattus*, mean estimated *S. murinus* biomass was lowest in grassland and highest in *Leucaena* forest on Saipan and Tinian (Figure 13), with a maximum estimate of 1.9 kg/ha in this habitat (Table 11). Mean estimated *M. musculus* biomass was greatest in grassland on Rota, Saipan, and Tinian (Figure 13), with a maximum estimate of 0.4 kg/ha in this habitat (Table 11). On Guam, mean estimated biomass was greatest in grassland for all species (Figure 13, Table 11). Biologically relevant levels of *R. rattus* biomass were also observed in *Leucaena* forest on Guam, although estimates were quite variable (0–2.9 kg/ha; Table 11) in this habitat. Introduced small mammal biomass was uniformly low (or non-existent) in mixed habitat and native forest on Guam (Figure 13, Table 11).

DISCUSSION

This study provides the first robust and reliable density and biomass estimates for introduced small mammals in grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Rota, Saipan, and Tinian. Density and biomass were greater on Rota, Saipan, and Tinian than on Guam. Overall, density and biomass were greatest in grassland and *Leucaena* forest habitats, and on Tinian and Rota.

Three species, *R. rattus*, *S. murinus*, and *M. musculus*, were commonly captured during this study. Two additional species, *R. exulans* and *R. norvegicus*, were captured infrequently and in very low numbers. Of these species, *R. rattus* attains the greatest density and biomass in the Mariana Islands. On Rota, Saipan, and Tinian *R. rattus* density and biomass estimates ranged from 6.9–99.9/ha and 1.0–11.6 kg/ha, respectively; maximum density and biomass were observed in grasslands and on Tinian. Maximum estimates of *R. rattus* density on Rota, Saipan, and Tinian are 2–3 times greater than the highest known historic values from Guam and also greater than estimates from other tropical Pacific islands, including Pohnpei (4.0–8.5/ha; Strecker 1962), Majuro (11.3/ha; Strecker 1962), Eniwetok (19.9/ha; Jackson 1967), and the Galapagos (0.2–18.9/ha; Clark 1980). Indeed, the peak densities observed during this study, especially on Rota and Tinian, are suggestive of population irruptions. Conversely, the fact that high density *R. rattus* populations were observed across habitats, islands, and time is not indicative of an irruptive event, and instead suggests that high density *R. rattus* populations may be fairly common on Rota, Saipan, and Tinian. Comparable (and even higher) densities have been recorded for *R. exulans* on small relatively competitor- and predator-free islands. On Kure Atoll, Wirtz (1972) documented a mean *R. exulans* density of 111.2/ha during monthly sampling from March 1964 to May 1965, with monthly estimates ranging from 49.4/ha to 185.3/ha. Similarly, on Tititiri Matangi

Island, New Zealand, Moller and Craig (1987) estimated peak *R. exulans* densities of 130 ± 20/ha in grassland and 101 ± 12/ha in forest during regular sampling from February 1975 to May 1977.

Estimated *R. rattus* density (0–15.9/ha) and biomass (0–2.9 kg/ha) were considerably lower on Guam. These density estimates are slightly lower than estimates from Guam in 1945 (10.9–30.0/ha; Baker 1946) and the early 1960's (18.8/ha; Barbehenn 1969, 1974b). It is notable that our estimates of Guam *R. rattus* density are generally lower in forest than in grassland, a pattern first observed in the mid-1980's by Savidge (forest: 0–2.5/ha, grassland: 36.4/ha; 1986). Gragg et al. (in prep) also found high *Rattus* species density (combined estimates for *R. exulans* and *R. rattus*: 14.7–69.8/ha) in southern Guam grasslands in 2002–2003. This pattern is at least partially attributable to variable brown treesnake predation pressure, as brown treesnake density is generally greater in forest than grassland habitats on Guam (Savidge 1987, 1991; Rodda and Dean-Bradley 2001).

S. murinus is generally less common than *R. rattus* in the Mariana Islands, although estimated *S. murinus* density exceeded *R. rattus* density on 2 sites where both were present. Nonetheless, the low mass of *S. murinus* (in relation to *R. rattus*) resulted in *S. murinus* biomass estimates that, with one exception, were only 1–37% of the estimated *R. rattus* biomass for the same site. Overall, *S. murinus* density exceeded 30/ha at 4 of the 9 sites where this species was captured. On Saipan and Tinian, *S. murinus* density and biomass were greater in forest than grassland, with the highest values occurring in *Leucaena* forest. In contrast, we did not capture *S. murinus* in 7 forest sites on Guam, again possibly an indication of brown treesnake predation pressure in Guam forests. Although *S. murinus* was reported from Rota in 1966 (Barbehenn 1974b), we neither captured nor observed this species during approximately 9 weeks spent on the island and believe it to be absent. In general, our estimates of *S. murinus* density are comparable to historic values from Guam (25.4/ha, Barbehenn 1969, 1974b; 19.1/ha, Savidge 1986) and more recent estimates from Saipan (16.7–27.3/ha, S. Vogt unpublished data). Our estimates are also similar to values obtained for the islands of Ile aux Aigrettes (29.2/ha) and Ile de la Passe (20/ha), located off the coast of Mauritius in the Indian Ocean (Varnham et al. 2002). However, our maximum estimated *S. murinus* density of 73.7/ha greatly exceeds known values, and could indicate an irruptive potential for this species in the Mariana Islands.

M. musculus is a relatively minor component of the introduced small mammal community in the Mariana Islands from a biomass standpoint, with estimates ranging from 0.01–0.45 kg/ha. However, *M. musculus* capture probability may have been negatively influenced by *R. rattus* activity (Brown et al. 1996, Weihong et al. 1999). To investigate this possibility, we added site-specific *R. rattus* density to the top *M. musculus* model in a post-hoc MARK analysis. As anticipated, *R. rattus* density had a negative effect on *M. musculus* capture probability (β = -0.008 ± 0.006, 95% CI = -0.019–0.003), although this effect was weak as demonstrated by the 95% CI that asymmetrically overlapped zero (Figure 14). Nonetheless, the trend of decreasing *M. musculus* capture probability with increasing *R. rattus* density suggests that this relationship warrants further investigation and should be considered during sampling design and data analysis. For example, the use of multiple trap types may decrease the likelihood of capture probability suppression of non-dominant species (Brown et al. 1996, Weihong et al. 1999, Gragg 2004). There was an indication of habitat specialization for *M. musculus*, as maximum density and biomass occurred at grassland and mixed habitat sites with patchy vegetative growth and exposed soil. Baker (1946:398) noted a similar preference for “open grass and brush land” and areas where “limestone soils are exposed” on Guam. Similar habitat preferences for this species have been noted for other tropical Pacific islands (Nicholson and Warner 1953, Berry and Jackson 1979). Overall, estimated *M. musculus* density ranged from 0.8–36.5/ha, exceeded 15/ha at 4 of the 8 sites where encountered, and was greater than *R. rattus* and *S. murinus* density at only 1 site where all 3 species were present. These estimates are comparable, though perhaps slightly lower than, historic (8.3–25.8/ha; Baker 1946) and more recent (18.5–104.0/ha; Gragg et al in prep) estimates from Guam.

When interpreting these (and other) density and biomass estimates, it is essential to recognize the potential for temporal variability in introduced small mammal populations. For example, annual sampling at a single site on Guam (CP05, CP06, CP07) demonstrated significant temporal variation in *R. rattus* density and biomass, which increased from 2.6/ha and 0.4 kg/ha in 2005 to 15.3/ha and 2.9 kg/ha in 2006. In 2007, 10 days of livetrapping (1570 trap nights) at this site yielded zero captures. Note that this sampling occurred at the same time each year (early May–early June) and therefore represents annual temporal variability. It is also possible that introduced small mammal density and biomass exhibit intra-annual temporal variability in the Mariana Islands. One slight complication is that this site is used for an ongoing, long-term brown treesnake population study (Rodda et al. 2007) and is surrounded by a snake- and ungulate-proof fence (i.e., brown treesnakes can not enter or exit and ungulates are excluded), suggesting that the site is not directly comparable with other forested areas on Guam. For example, the exclusion of introduced ungulates has resulted in rapid and dramatic shifts in vegetation structure and composition compared to the surrounding landscape (M. Christy, unpublished data). Nonetheless, the temporal variability in *R. rattus* density and biomass observed at this site suggests that introduced small mammal density and biomass may fluctuate greatly over relatively short time spans in the Mariana Islands. The potential for temporal variability should always be considered when interpreting density and biomass estimates, which are merely a snapshot of a dynamic population.

Modeling Capture and Recapture Probability

Our sampling design and data analysis approach allowed us to consider the importance of factors, including time, behavior, heterogeneity, sampling location (e.g., island, habitat), and various covariates, which can affect capture and recapture probability. By accounting for these factors during modeling, we were better able to generate robust and reliable estimates of density and biomass. Modeling identified several important sources of heterogeneity for *R. rattus* capture and recapture probability, including neophobia (capture probability only), island, sex, reproductive status, and rain amount.

We documented reduced capture probability for *R. rattus* on the first and second sampling occasion for traps placed on the grid 2 nights prior to the beginning of sampling. Neophobia has been previously documented in laboratory, commensal, and wild *Rattus* populations (Temme and Jackson 1979, Inglis et al. 1996, Thorsen et al. 2000, Priyambodo and Pelz 2003, Clapperton 2006), and should be an important consideration during sampling design. It is possible that an extended trap acclimation period (>2 nights) or trap pre-baiting could have reduced the neophobia effect, and these possibilities warrant further investigation.

R. rattus capture and recapture probability also varied between islands. Guam, Rota, Saipan, and Tinian differ in a number of biologically relevant ways, including land-use history, introduced ungulate density, and predator density, which might influence *R. rattus* populations. Each island has experienced significant but variable disturbance over the past century as a result of shifting land-use patterns and World War II. Notably, large areas of Rota, Saipan, and Tinian were converted to sugarcane production during Japanese occupation (1914–1944); these areas were largely abandoned following World War II (Bowers 2001). Wartime activities further damaged the native vegetation of these islands, such that post-war estimates of residual forest cover were only 23% for Rota, 5% for Saipan, and 2% for Tinian (Bowers 2001:206). Many disturbed areas, especially abandoned sugarcane fields and areas cleared by military activities, were recolonized by *L. leucocephala* which often persists in near-monotypic stands to the present day (Mueller-Dombois and Fosberg 1998:264). The vegetative community of the Mariana Islands has also been modified by introduced ungulates, although such effects differ between islands. Introduced feral pigs (*Sus scrofa*) and Phillipine deer (*Cervus mariannus*) are currently present on Guam, Rota, and Saipan (Stinson 1994, Vogt and Williams 2004, Wiles 2005), with possible detrimental effects on native forest species recruitment (Wiles et al. 1996, Ritter and Naugle 1999). Introduced pigs and deer once occurred on Tinian but have disappeared in recent years (Wiles et al. 1990). However, domestic cattle (*Bos taurus*) grazing over large areas of this island likely have detrimental impacts on native vegetation (Wiles et al. 1990). In addition, each island is home to a variable suite of predators capable of capturing *R. rattus*. The most obvious difference between islands, in terms of predators, is the high-density brown treesnake population on Guam (Rodda et al. 1999). Other potential predators, including feral cats (*Felis catus*; common on Rota and Tinian) and dogs (*Canis familiaris*; common on Guam), monitor lizards (*Varanus indicus*), collared kingfishers (*Halcyon chloris*; not present on Guam), Micronesian starlings (*Aplonis opaca*; uncommon on Guam), Mariana crows (*Corvus kubaryi*; Guam [rare] and Rota [uncommon] only), the introduced black drongo (*Dicrurus macrocercus*; Guam and Rota only), and the introduced cane toad (*Bufo marinus*) exist at variable densities in different habitats and on different islands (Stinson 1994, Vogt and Williams 2004, Wiles 2005). With the exception of the brown treesnake on Guam, the effect of predators on *R. rattus* in the Mariana Islands is presently unclear. It seems likely, however, that intra- and inter-island variability in predation pressure, habitat structure, and vegetative species composition influence *R. rattus* populations, suggesting that further investigation of these factors would be valuable.

Several covariates (sex, reproductive status, and rain amount) proved important for modeling *R. rattus* capture and recapture probability. Addition of these covariates improved the precision of site-specific abundance estimates by an average of 25.7% (range: 8.1–57.2%) relative to estimates produced by the same model without covariates. A secondary benefit of including covariates in models is the knowledge gained about individual or environmental factors that influence capture and recapture probability. In this case, the importance of sex and reproductive status for *R. rattus* capture and recapture probability is not surprising based on evidence of the importance of social structure and dominance hierarchies for *Rattus* species. For example, during long-term observations of a free-ranging *R. rattus* population, Ewer (1971) found that females were more aggressive than males and adults were generally dominant over juveniles. Similarly, for our data *R. rattus* capture and recapture probability were higher for females than for males, and for reproductively mature individuals than for non-reproductive individuals (both adults and juveniles). Alternatively, the high capture and recapture probability of reproductively mature females could simply be the result of increased energy requirements and foraging activity by these individuals, leading to increased encounters with traps and increased captures.

The importance of the environmental covariate rainfall amount, and the positive relationship between rainfall amount and *R. rattus* capture and recapture probability, is more difficult to interpret. Although speculative, one possible explanation is that *R. rattus* activity increased with rainfall as individuals searched for standing water. The limestone substrate of large areas of the Mariana Islands is highly permeable and available surface water is typically rare or nonexistent (Mueller-Dombois and Fosberg 1998:254). Outside of the moisture available in food items, water may be limited except immediately following a rainfall event. Unfortunately, it does not seem that *R. rattus* water requirements have been studied in detail and it is unknown if *R. rattus* in the Mariana Islands can meet daily water requirements through diet alone. Norman and Baudinette (1969) found that wild *R. rattus* collected on Green Island, Tasmania, had a mean minimum daily water requirement of only 5.95 ± 1.4 mL (representing 5.36 ± 0.8% body weight/day), although mean daily intake rose to 40.7 ± 15.0 mL (representing 21.5 ± 5.0% body weight/day) when water was provided *ad libitum*. Stomach content analysis of additional *R. rattus* collected from the same location revealed an average of 6.4 mL of free water in the diet, suggesting that individuals could meet minimum water requirements through diet alone (Norman and Baudinette 1969). The applicability of these data to *R. rattus* in the Mariana Islands is unknown, however, as conditions influencing daily water requirements (e.g., temperature, humidity, and diet) differ markedly between the Mariana Islands and Tasmania. Alternatively, rainfall amount might increase *R. rattus* activity by influencing food availability or palatability in some unknown fashion. Investigation of this speculative hypothesis would require detailed study of both the diet of *R. rattus* in the Mariana Islands and the impact of rainfall on the components of that diet. Regardless of the underlying relationship between capture and recapture probability and rainfall, the effect size was small in comparison to the effect size of sex and reproductive status. Rainfall in the Mariana Islands is highly variable, even over the spatial extent of our trapping grids, and may occur at any time. Improved rainfall measurement should incorporate more frequent recording of rainfall (minimally 12 hour intervals to differentiate daytime and nighttime rainfall) and perhaps multiple recording stations to better cover the area of the trapping grid.

In contrast to *R. rattus*, none of the covariates under consideration were important for modeling *S. murinus* and *M. musculus* capture and recapture probability. Instead, heterogeneity was accounted for through the use of 2-mixture models. This outcome was somewhat surprising, as mixture models are generally not well-supported for small datasets (Conn et al. 2006), such as our *S. murinus* (298 total individuals) and *M. musculus* (154 total individuals) datasets. Other researchers have documented apparent capture probability heterogeneity for both *S. murinus* (Seymour et al. 2005) and *M. musculus* (Drickamer et al. 1999, Conn et al. 2006). Notably, Seymour et al. (2005) found highly variable capture probabilities in a *S. murinus* population on Ile aux Aigrettes. Of the 759 *S. murinus* captured on this 25-ha island during a 7-month eradication attempt (96,613 trap nights), approximately 350 captures occurred during the first 3 nights of trapping and 89.3% of the total captures occurred by night 18. Infrequent captures (<25) occurred over the next 3.5 months, followed by a 3 month period of increasing captures across the island before the eradication attempt was abandoned. These results are suggestive of a population with at least 2 capture probability classes, with a large proportion of the population having high capture probability and a smaller proportion of the population having very low capture probability. In contrast, our results from the Mariana Islands indicated that roughly ⅓ of *S. murinus* had high capture probabilities, with the remainder having reduced capture probability.

Both *S. murinus* and *M. musculus* also exhibited temporal variation in capture and recapture probability. *S. murinus* capture and recapture probability increased over time. This pattern of reduced capture probability during early sampling occasions suggests neophobia, although this effect was not strongly supported in models. While some researchers have documented a similar response for *S. murinus* (e.g., Figure 6 in Seymour et al. 2005), the general consensus seems to be that *S. murinus* is neophilic and likely to investigate, rather than avoid, new objects (Churchfield 1990 cited by Seymour et al. 2005). Interpretation of high *S. murinus* recapture probability for Guam is difficult because of sparse data (14 animals captured at a single site) on this island. The pattern in temporal variation is less clear for *M. musculus*, although there does seem to be an indication of increasing capture and recapture probability over time for Rota, Saipan, and Tinian. Interpretation of the Guam capture and recapture probabilities is again complicated by sparse data (15 animals captured at a single site) on this island.

Modeling also indicated that *S. murinus* recapture probability varied between each island, and that *M. musculus* capture probability varied differed between Guam and the combination of Rota, Saipan, and Tinian. As with *R. rattus*, intra- and inter-island variation in habitat structure, vegetative species composition, and predator community could be biologically relevant for *S. murinus* and *M. musculus* populations and could explain these island-level differences in capture and recapture probability.

Implications for Mariana Island Ecology and Brown Treesnake Control and Management

Although little direct evidence currently exists for the Mariana Islands, it seems likely that the high-density introduced small mammal populations documented during this research have negative effects on native fauna and flora, and that introduced species (including small mammals) have modified Mariana Island ecosystems and ecosystem function (Fritts and Rodda 1998). In recent years, researchers have noted apparent declines of several avian species in the Mariana Islands, including the bridled white-eye (*Zosterops conspicillatus rotensis*; Amidon 2000, Fancy and Snetsinger 2001) and Mariana crow (*Corvus kubaryi*; Plentovich et al. 2005, U.S. Fish and Wildlife Service 2005) on Rota and the Micronesian megapode (*Megapodius laperouse*) and Mariana fruit dove (*Ptilinopus roseicapilla*) on Saipan (Craig 1999). Numerous hypotheses, including predation by introduced species (e.g., *Rattus*, black drongos, and feral cats), avian diseases or parasites, pesticides, and habitat degradation associated with land-use changes or typhoon damage, have been considered (Craig 1999, Amidon 2000, Fancy and Snetsinger 2001, Plentovich et al. 2005, U.S. Fish and Wildlife Service 2005, Ha et al. in prep). While predation by black drongos, diseases, and pesticides have largely been ruled out and habitat degradation is increasingly seen as an important factor in avian declines (e.g., Fancy and Snetsinger 2001, Ha et al. in prep), the role of introduced small mammals remains unclear. Predation by introduced *Rattus* species is often rejected as a cause of recent avian declines because ≥1 *Rattus* species have been present in the Mariana Islands for at least 1000 years. However, this rejection does not account for differential effects of various *Rattus* species on birds (Atkinson 1985, Thibault et al. 2002, Towns et al. 2006), as *R. exulans* (the earliest introduction to the Mariana Islands; Steadman 1999) is generally considered least detrimental to avian species. Perhaps more importantly, the potential impact of *R. diardii* or *R. tanezumi* on avian species is unknown, and the uncertainty surrounding the status and distribution of *R. diardii*, *R. rattus*, and *R. tanezumi* in the Mariana Islands further complicates matters. Further, temporal shifts in the presence or abundance of *Rattus* species may obscure their role in avian declines. High-density introduced small mammal populations on Rota, Saipan, and Tinian might also impact avian species through dietary competition, especially during the dry season when certain food items may become scarce. Food competition for invertebrate and reptile foods could be especially problematic for nesting birds, as these high protein prey items are required for nestlings.

Predation by introduced small mammals may also have direct negative effects on invertebrate or reptile populations in the Mariana Islands. Although *Rattus* species are often implicated in invertebrate and reptilian declines (Whitaker 1973; Bremner et al. 1984; Cree et al. 1995; Priddel et al. 2003; Hoare et al. 2007a,b), the insectivorous *S. murinus* may be more problematic for these taxa in the Mariana Islands. *S. murinus* has been implicated in the decline of native invertebrates and reptiles on Mauritius and nearby islands (Varnham et al. 2002). On Guam, Barbehenn (1974b) commented that no skinks were observed during hundreds of hours of small mammal trapping during the peak of the *S. murinus* irruption in the early 1960's, which contrasts with the current abundance and visibility of skinks on Guam. More recently, Fritts and Rodda (1998) noted large differences in mean skink density between Saipan, where *S. murinus* was common (2200 skinks/ha), areas on Guam with few *S. murinus* (8850 skinks/ha), and areas on Guam where both *S. murinus* and brown treesnakes were excluded (13,200 skinks/ha). Similarly, Rodda and Fritts (1992) implicated *S. murinus* in the decline of the pelagic gecko (*Nactus pelagicus*), when they found that this gecko was common on Rota, where *S. murinus* was absent, but highly localized (Guam) or rare or possibly extinct (Saipan and Tinian) on islands with high past or current *S. murinus* populations.

Recent research suggests that introduced small mammals have important impacts on the effectiveness of brown treesnake control efforts, which are highly dependent on traps using live, domestic mice (*M. musculus*) as attractants. These traps are placed around ports, airports, and other cargo-handling facilities on Guam, as well as in locations vulnerable to accidental brown treesnake introductions, such as Rota, Saipan, and Tinian. Mouse-attractant traps are also commonly deployed during the response to snake sightings in brown treesnake-free locations. However, research conducted on Guam suggests that brown treesnake trap capture rates are inversely related to introduced small mammal density. For example, Rodda et al. (2001) found a strong correlation ($r^2 = 0.90$) between brown treesnake trap capture rates and indices of small mammal density and documented a 7-fold increase in brown treesnake capture rates in areas of very low small mammal density on Guam. Similarly, Gragg et al. (2007) documented a 22–65% increase in brown treesnake trap capture probability after reducing rodent populations with localized rodenticide application. These findings suggest reduced effectiveness of mouse-attractant traps on Rota, Saipan, and Tinian. Further, the majority of brown treesnake control and eradication tools currently being developed and evaluated, such as various acetaminophen delivery devices (Savarie et al. 2001), also rely on mouse-based attractants and will likely be subject to the same reduction in effectiveness in areas of high introduced small mammal density.

A second, though perhaps less obvious, effect of introduced small mammals on brown treesnake control and management relates to their impact on island trophic systems and predator-prey relationships. On Guam, introduced prey species, including small mammals, skinks, and geckos, were abundant and widespread at the time of brown treesnake introduction following World War II (Baker 1946, Fritts and Rodda 1998). Because these introduced prey species evolved with various predators, they were better able to persist under brown treesnake predation than the predator-naïve native species of Guam. In so doing, introduced prey species supported a high-density brown treesnake population, even as native avian and reptilian species declined. By the time brown treesnake predation pressure began to reduce introduced prey densities and brown treesnake density also began to decline because of food limitations, many native species were already extinct. Unfortunately, the high introduced small mammal density and biomass documented on the islands of Rota, Saipan, and Tinian during this research suggests that a similar scenario could develop on these islands should a brown treesnake population become established.

LITERATURE CITED

Akaike, H. 1973. Information theory as an extension of the maximum likelihood principle. Pages 267–281 *in* B.N. Petrov and F. Csáki, editors. Second International symposium on information theory. Akademiai Kiado, Budapest, Hungary.

Allen, R.B., W.G. Lee, and B.D. Rance. 1994. Regeneration in indigenous forest after eradication of Norway rats, Breaksea Island, New Zealand. New Zealand Journal of Botany 32:429–439.

Amidon, F.A. 2000. Habitat relationships and life history of the Rota bridled white-eye (*Zosterops rotensis*). M.S. Thesis, Virginia Polytechnic Institute and State University, Blacksburg, Virginia, USA.

Anderson, D.R. 2001. The need to get the basics right in wildlife field studies. Wildlife Society Bulletin 29:1294–1297.

Anderson, D.R. 2003. Response to Engeman: index values rarely constitute reliable information. Wildlife Society Bulletin 31:288–291.

Anderson, D.R., K.P. Burnham, G.C. White, and D.L. Otis. 1983. Density estimation of small-mammal populations using a trapping web and distance sampling methods. Ecology 64:674–680.

Athens, J.S., H.D. Tuggle, J.V. Ward, and D.J. Welch. 2002. Avifaunal extinctions, vegetation change, and Polynesian impacts in prehistoric Hawai'i. Archaeology in Oceania 37:57–78.

Atkinson, I.A.E. 1977. A reassessment of factors, particularly *Rattus rattus* L., that influenced the decline of endemic forest birds in the Hawaiian Islands. Pacific Science 31:109–133.

Atkinson, I.A.E. 1985. The spread of commensal species of *Rattus* to oceanic islands and their effects on island avifaunas. Pages 35–81 *in* P.J. Moors, editor. Conservation of island birds. International Council for Bird Preservation, Technical Publication No. 3, Cambridge, England.

Baker, R.H. 1946. A study of rodent populations on Guam, Mariana Islands. Ecological Monographs 16:393–408.

Barbehenn, K.R. 1969. Responses of rodents and shrews to patterns of removal trapping. Pages 247–252 *in* D.W. Parrack, editor. Indian rodent symposium. Calcutta, India.

Barbehenn, K.R. 1974a. Estimating density and home range size with removal grids: the rodents and shrews of Guam. Acta Theriologica 19:191–234.

Barbehenn, K.R. 1974b. Recent invasions of Micronesia by small mammals. Micronesica 10:41–50.
Baverstock, P.R., M. Adams, L.R. Maxson, and T.H. Yosida. 1983. Genetic differentiation among karyotypic forms of the black rat, *Rattus rattus*. Genetics 105:969–983.

Berry, R.J. and W.B. Jackson. 1979. House mice on Enewetak Atoll. Journal of Mammalogy 60:222–225.

Bitam, I, B. Baziz, J.-M. Rolain, M. Belkaid, and D. Raoult. 2006. Zoonotic focus of plague, Algeria. Emerging Infectious Diseases 12:1975–1977.

Bowden, D.C., G.C. White, A.B. Franklin, and J.L. Ganey. 2003. Estimating population size with correlated sampling unit estimates. Journal of Wildlife Management 67:1–10.

Bowers, N.M. 2001. Problems of resettlement on Saipan, Tinian and Rota Mariana Islands. Occassional Historical Papers Series No. 7. CNMI Division of Historic Preservation, Saipan, Commonwealth of the Northern Mariana Islands.

Bremner, A.G., C.F. Butcher, and G.B. Patterson. 1984. The density of indigenous invertebrates on three islands in Breaksea Sound, Fiordland, in relation to the distribution of introduced mammals. Journal of the Royal Society of New Zealand 14:379–386.

Brook, F.J. 1999. Changes in the landsnail fauna of Lady Alice Island, northeastern New Zealand. Journal of the Royal Society of New Zealand 29:135–157.

Brook, F.J. 2000. Prehistoric predation of the landsnail *Placostylus ambagiosus* Suter (Stylommatophora: Bulimulidae), and evidence for the timing of establishment of rats in northernmost New Zealand. Journal of the Royal Society of New Zealand 30:227–241.

Brown, K.P., H. Moller, J. Innes, and N. Alterio. 1996. Calibration of tunnel tracking rates to estimate relative abundance of ship rats (*Rattus rattus*) and mice (*Mus musculus*) in a New Zealand forest. New Zealand Journal of Ecology 20:271–275.

Burnham, K.P., and D.R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Second Edition. Springer, New York, New York, USA.

Campbell, D.J., and I.A.E. Atkinson. 1999. Effects of kiore (*Rattus exulans* Peale) on recruitment of indigenous coastal trees on northern offshore islands of New Zealand. Journal of the Royal Society of New Zealand 29:265–290.

Campbell, D.J., and I.A.E. Atkinson. 2002. Depression of tree recruitment by the Pacific rat (*Rattus exulans* Peale) on New Zealand's northern offshore islands. Biological Conservation 107:19–35.

Carlton, J.T., and J. Hodder. 2003. Maritime mammals: terrestrial mammals as consumers in marine intertidal communities. Marine Ecology Progress Series 256:271–286.

Chanteau, S., L. Ratsifasoamanana, B. Rasoamanana, L. Rahalison, J. Randriambelosoa, J. Roux, and D. Rabeson. 1998. Plague, a reemerging disease in Madagascar. Emerging Infectious Diseases 4:101–104.

Churchfield, S. 1990. The natural history of shrews. Christopher Helm, London, U.K.

Clapperton, B.K. 2006. A review of the current knowledge of rodent behaviour in relation to control devices. Science for Conservation 263. New Zealand Department of Conservation, Wellington, New Zealand.

Clark, D.B. 1980. Population ecology of *Rattus rattus* across a desert-montane forest gradient in the Galapagos Islands. Ecology 61:1422–1433.

Cooch, E.G., and G.C. White. 2005. Program MARK: a gentle introduction. (http://www.phidot.org/software/mark/docs/book/).

Colvin, B.A., M.W. Fall, L.A. Fitzgerald and L.L. Loope. 2005. Review of Brown Treesnake problems and control programs: report of observations and recommendations. Report to Department of Interior, Office of Insular Affairs, Honolulu, Hawaii, USA.

Conn, P.B., A.D. Arthur, L.L. Bailey, and G.R. Singleton. 2006. Estimating the abundance of mouse populations of known size: promises and pitfalls of new methods. Ecological Applications 16:829–837.

Converse, S.J., G.C. White, and W.M. Block. 2006. Small mammal responses to thinning and wildfire in ponderosa pine-dominated forests of the southwestern United States. Journal of Wildlife Management 70:1711–1722.

Craig, R.J. 1999. Conservation of endangered white-eyes (Zosteropidae) in the tropical Pacific. Bird Conservation Research, Inc. Contribution No. 1:1–8.

Cree, A., C.H. Daugherty, and J.M. Hay. 1995. Reproduction of a rare New Zealand reptile, the tuatara, *Sphenodon punctatus*, on rat-free and rat-inhabited islands. Conservation Biology 9:373–383.

Daniel, M.J. 1990. Greater short-tailed bat. Pages 131–136 *in* C.M. King, editor. The handbook of New Zealand mammals. Oxford University Press, Auckland, New Zealand.

Davis, S.A., L.K. Akison, L.N. Farroway, G.R. Singleton, and K.E. Leslie. 2003. Abundance estimators and truth: accounting for individual heterogeneity in wild house mice. Journal of Wildlife Management 67:634–645.

Dowding, J.E. and E.C. Murphy. 1994. Ecology of ship rats (*Rattus rattus*) in a Kauri (*Agathis australis*) forest in Northland, New Zealand. New Zealand Journal of Ecology 18:19–28.

Drickamer, L.C., G.A. Feldhamer, D.G. Mikesic, and C.M. Holmes. 1999. Trap-response heterogeneity of house mice (*Mus musculus*) in outdoor enclosures. Journal of Mammalogy 80:410–420.

Efford, M.G. 2004. Density estimation in live-trapping studies. Oikos 106:598–610.

Ewer, R.F. 1971. The biology and behaviour of a free-living population of black rats (*Rattus rattus*). Animal Behaviour Monographs 4:127–174.

Falanruw, M.V.C., T.G. Cole and A.H. Ambacher. 1989. Vegetation survey of Rota, Tinian, and Saipan, Commonwealth of the Northern Mariana Islands. Resource Bulletin PSW-27. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, California, USA.

Fancy, S.G., and T.J. Snetsinger. 2001. What caused the population decline of the bridled white-eye on Rota, Mariana Islands? Pages 274–280 *in* J.M. Scott, S. Conant, and C. van Riper, III, editors. Evolution, ecology, conservation, and management of Hawaiian birds: a vanishing avifauna. Studies in Avian Biology No. 22, Cooper Ornithological Society.

Fisher, H.I., and P.H. Baldwin. 1946. War and the birds of Midway Atoll. Condor 48:3–15.

Freycinet, L.C.d. 2003. An account of the Corvette L' Uranie's sojourn at the Mariana Islands, 1819; supplemented with the Journal of Rose de Freycinet. Occasional Historic Papers No. 13, CNMI Division of Historic Preservation, Saipan, Commonwealth of the Northern Mariana Islands.

Fritts, T.H., and G.H. Rodda. 1998. The role of introduced species in the degradation of island ecosystems: a case history of Guam. Annual Review of Ecology and Systematics 29:113–140.

Fukami, T., D.A. Wardle, P.J. Bellingham, C.P.H. Mulder, D.R. Towns, G.W. Yeates, K.I. Bonner, M.S. Durrett, M.N. Grant-Hoffman, and W.M. Williamson. 2006. Above- and below-ground impacts of introduced predators in seabird-dominated island ecosystems. Ecology Letters 9:1299–1307.

Gannon, W.L., R.S. Sikes, and the Animal Care and Use Committee of the American Society of Mammalogists. 2007. Guidelines of the American Society of Mammalogists for the use of wild mammals in research. Journal of Mammalogy 88:809–823.

Goodman, S.M. 1995. *Rattus* on Madagascar and the dilemma of protecting the endemic rodent fauna. Conservation Biology 9:450–453.

Gragg, J.E. 2004. Rodent reduction for enhanced control of brown treesnakes (*Boiga irregularis*). M.S. Thesis, Colorado State University, Fort Collins, Colorado, USA.

Gragg, J.E., G.H. Rodda, J.A. Savidge, G.C. White, K. Dean-Bradley, and A.R. Ellingson. 2007. Response of brown treesnakes to reduction of their rodent prey. Journal of Wildlife Management 71:2311–2317.

Gragg, J.E., J.A. Savidge, G.H. Rodda, G.C. White, A.R. Ellingson, and K. Dean-Bradley. In Prep. Rodenticide effects on abundance and survival of rodents on Guam.

Ha, J.C., J. Buckley, and R.R. Ha. In Prep. Forest bird abundances and the potential for typhoon impact on bird populations on the island of Rota, Northern Mariana Islands.

Hoare, J.M., L.K. Adams, L.S. Bull, and D.R. Towns. 2007a. Attempting to manage complex predator-prey interactions fails to avert imminent extinction of a threatened New Zealand skink population. Journal of Wildlife Management 71:1576–1584.

Hoare, J.M., S. Pledger, N.J. Nelson, and C.H. Daugherty. 2007b. Avoiding aliens: behavioural plasticity in habitat use enables large, nocturnal geckos to survive Pacific rat invasions. Biological Conservation 136:510–519.

Huggins, R.M. 1989. On the statistical analysis of capture-recapture experiments. Biometrika 76:133–140.

Huggins, R.M. 1991. Some practical aspects of a conditional likelihood approach to capture experiments. Biometrics 47:725–732.

Hurvich, C.M., and C.-L. Tsai. 1989. Regression and time-series model selection in small samples. Biometrika 76:297–307.

Inglis, I.R., D.S. Shepherd, P. Smith, P.J. Haynes, D.S. Bull, D.P. Cowan, and D. Whitehead. 1996. Foraging behaviour of wild rats (*Rattus norvegicus*) towards new foods and bait containers. Applied Animal Behaviour Science 47:175–190.

Jackson, W.B. 1967. Productivity in high and low islands with special emphasis to rodent populations. Micronesica 3:5–15.

Jiang, J., D.W. Soeatmadji, K.M. Henry, S. Ratiwayanto, M.J. Bangs, and A.L. Richards. 2006. *Rickettsia felis* in *Xenopsylla cheopis*, Java, Indonesia. Emerging Infectious Diseases 12:1281–1283.

Johnson, D.H. 1962. Rodents and other Micronesian animals collected. Pages 21–38 *in* T.I. Storer, editor. Pacific island rat ecology: report of a study made on Ponape and adjacent islands 1955–1958. Bernice P. Bishop Museum Bulletin 225, Honolulu, Hawaii, USA.

Kuroda, N. 1938. A list of the Japanese mammals. Privately printed, Tokyo, Japan.

Kuschel, G., and T.H. Worthy. 1996. Past distribution of large weevils (Coleoptera: Curculionidae) in the South Island, New Zealand, based on Holocene fossil remains. New Zealand Entomologist 19:15–22.

Lindo, J.F., C. Waugh, J. Hall, C. Cunningham-Myrie, D. Ashley, M.L. Eberhard, J.J. Sullivan, H.S. Bishop, D.G. Robinson, T. Holtz, and R.D. Robinson. 2002. Enzootic *Angiostrongylus cantonensis* in rats and snails after an outbreak of human eosinophilic meningitis, Jamaica. Emerging Infectious Diseases 8:324–326.

Lindsey, G.D., S.M. Mosher, S.G. Fancy, and T.D. Smucker. 1999. Population structure and movements of introduced rats in an Hawaiian rainforest. Pacific Conservation Biology 5:94–102.

Martin, J.-L., J.-C. Thibault, and V. Bretagnolle. 2000. Black rats, island characteristics, and colonial nesting birds in the Mediterranean: consequences of an ancient introduction. Conservation Biology 14:1452–1466.

Martina, B.E.E., G. van Doornum, G.M. Dorrestein, H.G.M. Niesters, K.J. Stittelaar, M.A.B.I. Wolters, H.G.H. van Bolhuis, and A.D.M.E. Osterhaus. 2006. Cowpox virus transmission from rats to monkeys, the Netherlands. Emerging Infectious Diseases 12:1005–1007.

McConkey, K.R., D.R. Drake, H.J. Meehan, and N. Parsons. 2003. Husking stations provide evidence of seed predation by introduced rodents in Tongan rain forests. Biological Conservation 109:221–225.

Metteler, L.S. 1986. Pacific island names: a map and name guide to the new Pacific. Bernice P. Bishop Museum Miscellaneous Publication 34, Honolulu, Hawaii, USA.

Moller, H., and J.L. Craig. 1987. The population ecology of *Rattus exulans* on Tiritiri Matangi Island, and a model of comparative population dynamics in New Zealand. New Zealand Journal of Zoology 14:305–328.

Mueller-Dombois, D. and F.R. Fosberg. 1998. Vegetation of the tropical Pacific islands. Springer, New York, New York, USA.

Musser, G.G., and M.D. Carleton. 2005. Family Muridae. Pages 1189–1531 *in* D.E. Wilson and D.M. Reeder, editors. Mammal species of the world: a taxonomic and geographic reference. Third Edition. Smithsonian Institute Press, Washington, D.C., USA.

Newman, D.G. 1994. Effects of a mouse, *Mus musculus*, eradication programme and habitat change on lizard populations of Mana Island, New Zealand, with special reference to McGregor's skink, *Cyclodina macgregori*. New Zealand Journal of Zoology 21:443–456.

Nicholson, A.J., and D.W. Warner. 1953. The rodents of New Caledonia. Journal of Mammalogy 34:168–179.

Norris, J.L., III., and K.H. Pollock. 1996. Nonparametric MLE under two closed capture-recapture models with heterogeneity. Biometrics 52:639–649.

Norman, F.I., and R.V. Baudinette. 1969. Water economy and salt balance of an insular population of *Rattus rattus* Linnaeus. Journal of Mammalogy 50:487–493.

Otis, D.L., K.P. Burnham, G.C. White, and D.R. Anderson. 1978. Statistical inference from capture data on closed animal populations. Wildlife Monographs 62:1–135.

Parker, I.M., D. Simberloff, W.M. Londsdale, K. Goodell, M. Wonham, P.M. Kareiva, M.H. Williamson, B.V. Holle, P.B. Moyle, J.E. Byers, and L. Goldwasser. 1999. Impact: towards a framework for understanding the ecological effects of invaders. Biological Invasions 1:3–19.

Pascal, M., F. Siorat, O. Lorvelec, P. Yésou, and D. Simberloff. 2005. A pleasing consequence of Norway rat eradication: two shrew species recover. Diversity and Distributions 11:193–198.

Peterson, G.D., Jr. 1956. *Suncus murinus*, a recent introduction to Guam. Journal of Mammalogy 37:278–279.

Pickering, J., and C.A. Norris. 1996. New evidence concerning the extinction of the endemic murid *Rattus macleari* from Christmas Island, Indian Ocean. Australian Mammalogy 19:19–25.

Pledger, S. 2000. Unified maximum likelihood estimates for closed capture-recapture models using mixtures. Biometrics 56:434–442.

Plentovich, S., J.M. Morton, J. Bart, R.J. Camp, M.R. Lusk, N.C. Johnson, and E.A. VanderWerf. 2005. Population trends of Mariana crow (Corvus kubaryi) on Rota, Commonwealth of the Northern Mariana Islands. Bird Conservation International 15:211–224.

Pollock, K.H. 2002. The use of auxiliary variables in capture-recapture modeling: an overview. Journal of Applied Statistics 29:85–102.

Pollock, K.H., J.E. Hines, and J.D. Nichols. 1984. The use of auxiliary variables in capture-recapture and removal experiments. Biometrics 40:329–340.

Prager, E.M., C. Orrego, and R.D. Sage. 1998. Genetic variation and phylogeography of central Asian and other house mice, including a major new mitochondrial lineage in Yemen. Genetics 150:835-861.

Priddel, D., N. Carlile, M. Humphrey, S. Fellenberg, and D. Hiscox. 2003. Rediscovery of the 'extinct' Lord Howe Island stick-insect (*Dryococelus australis* (Montrouzier)) (Phasmatodea) and recommendations for its conservation. Biodiversity and Conservation 12:1391–1403.

Priyambodo, S. and H. Pelz. 2003. Studies on neophobic behaviour in Norway rats (*Rattus norvegicus* Berkenhout, 1769) from farms in Germany. Pages 155–158 *in* G.R. Singleton, L.A. Hinds, C.J. Krebs, and D.M. Spratt, editors. Rats, mice, and people: rodent biology and management. Australian Centre for International Agricultural Research, Canberra, Australia.

Recher, H.F., and S.S. Clark. 1974. A biological survey of Lord Howe Island with recommendations for the conservation of the island's wildlife. Biological Conservation 6:263–273.

Ritter, M.W., and C.M. Naugle. 1999. Population characteristics, germination and proposed management of *Elaeocarpus joga* Merr. on Guam: a regionally endemic tree. Micronesica 31:275–281.

Robins, J. H., M. Hingston, E. Matisoo-Smith, and H. A. Ross. 2007. Identifying Rattus species using mitochondrial DNA. Molecular Ecology Notes 7:717-729.

Rodda, G.H., and K. Dean-Bradley. 2001. Inventory of the reptiles of the War in the Pacific National Historic Park, Guam. Report to USGS, Fort Collins, Colorado, USA.

Rodda, G.H., K. Dean-Bradley, and C.P. Qualls. 2001. The effect of prey density on brown treesnake capture probability. Brown Treesnake 2001: Research & Management. Micronesica 37:181–182.

Rodda, G.H., and T.H. Fritts. 1992. The impact of the introduction of the brown tree snake, *Boiga irregularis*, on Guam's lizards. Journal of Herpetology 26:166–174.

Rodda, G.H., T.H. Fritts, M.J. McCoid, and E.W. Campbell III. 1999. An overview of the biology of the brown treesnake (*Boiga irregularis*), a costly introduced pest on Pacific Islands. Pages 44–80 *in* G.H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka. editors. Problem snake management: the habu and the brown treesnake. Cornell University Press, Ithaca, New York, USA.

Rodda, G.H., and J.A. Savidge. 2007. Biology and impacts of Pacific island invasive species. 2. *Boiga irregularis*, the brown tree snake (Reptilia: Colubridae). Pacific Science 61:307–324.

Rodda, G.H., J.A. Savidge, C.L. Tyrrell, M.T. Christy, and A.R. Ellingson. 2007. Size bias in visual searches and trapping of brown treesnakes on Guam. Journal of Wildlife Management 71:656–661.

SAS Institute. 2003. SAS/STAT software. Version 9.1. SAS Institute, Inc., Cary, North Carolina, USA.

Savarie, P.J., J.A. Shivik, G.C. White, J.C. Hurley, and L. Clark. 2001. Use of acetaminophen for large-scale control of brown treesnakes. Journal of Wildlife Management 65:356–365.

Savidge, J.A. 1986. The role of disease and predation in the decline of Guam's avifauna. Dissertation, University of Illinois, Urbana-Champaign, Illinois, USA.

Savidge, J.A. 1987. Extinction of an island forest avifauna by an introduced snake. Ecology 68:660–668.

Savidge, J.A. 1991. Population characteristics of the introduced brown tree snake (*Boiga irregularis*) on Guam. Biotropica 23:294–300.

Schwabe, H.W. 1979. Vergleichend-allometrische untersuchungen an den schädeln Europäischer und Asiatischer hausratten (*Rattus rattus* L.). Zeitschrift fur Säugetierkunde 44:354–360.

Seber, G.A.F. 2002. The estimation of animal abundance and related parameters. Second Edition. The Blackburn Press, New Jersey, USA.

Seymour, A., K. Varnham, S. Roy, S. Harris, L. Bhageerutty, S. Church, A. Harris, N.V. Jennings, C.O. Jones, A. Khadun, J. Mauremootoo, T. Newman, V. Tatayah, C. Webbon, and G. Wilson. 2005. Mechanisms underlying the failure of an attempt to eradicate the invasive Asian musk shrew *Suncus murinus* from an island nature reserve. Biological Conservation 125:23–35.

Smith, D.G., E.K. Shiinoki, and E.A. VanderWerf. 2006. Recovery of native species following rat eradication on Mokoli'i Island, O'ahu, Hawai'i. Pacific Science 60:299–303.

Spencer, H.J., and D.E. Davis. 1950. Movements and survival of rats in Hawaii. Journal of Mammalogy 31:154–157.

Steadman, D.W. 1999. The prehistory of vertebrates, especially birds, on Tinian, Aguiguan, and Rota, Northern Mariana Islands. Micronesica 31:319–345.

Stinson, D.W. 1994. Birds and mammals recorded from the Mariana Islands. Natural History Research Special Issue 1:333–344.

Strecker, R.L. 1962. Population levels. Pages 74–79 *in* T.I. Storer, editor. Pacific island rat ecology: report of a study made on Ponape and adjacent islands 1955–1958. Bernice P. Bishop Museum Bulletin 225, Honolulu, Hawaii, USA.

Temme, M., and W.B. Jackson. 1979. Criteria for trap evaluation. American Society of Testing Materials Special Technical Publication 680:58–67.

Thibault, J.-C., J.-L. Martin, A. Penloup, and J.-Y. Meyer. 2002. Understanding the decline and extinction of monarchs (Aves) in Polynesian islands. Biological Conservation 108:161–174.

Thorsen, M., R. Shorten, R. Lucking, and V. Lucking. 2000. Norway rats (*Rattus norvegicus*) on Frégate Island, Seychelles: the invasion; subsequent eradication attempts and implications for the island's fauna. Biological Conservation 96:133–138.

Towns, D.R. 1994. The role of ecological restoration in the conservation of Whitaker's skink (*Cyclodina whitakeri*), a rare New Zealand lizard (Lacertilia: Scincidae). New Zealand Journal of Zoology 21:457–471.

Towns, D.R. 1999. Interactions between geckos, honeydew scale insects and host plants revealed on islands in northern New Zealand, following eradication of introduced rats and rabbits. Pages 329–335 *in* C.R. Veitch and M.N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, U.K.

Towns, D., I. Atkinson, and C. Daugherty. 2006. Have the harmful effects of introduced rats on islands been exaggerated? Biological Invasions 8:863–891.

Towns, D.R., and C.H. Daugherty. 1994. Patterns of range contractions and extinctions in the New Zealand herpetofauna following human colonisation. New Zealand Journal of Zoology 21:315–339.

U.S. Census Bureau. 2003. Population and housing profile, 2000: The Commonwealth of the Northern Mariana Islands. Government Printing Office, Washington, D.C., USA.

U.S. Census Bureau. 2004. Population and housing profile, 2000: Guam. Government Printing Office, Washington, D.C., USA.

U.S. Fish and Wildlife Service. 2005. Draft revised recovery plan for the aga or Mariana crow (*Corvus kubaryi*). Portland, Oregon, USA.

U.S. Geological Survey. 1999a. Rota [map]. 1:25000. 7.5 Minute Series. Washington, D.C., USA.

U.S. Geological Survey. 1999b. Saipan [map]. 1:25000. 7.5 Minute Series. Washington, D.C., USA.

U.S. Geological Survey. 1999c. Tinian [map]. 1:25000. 7.5 Minute Series. Washington, D.C., USA.

U.S. Geological Survey. 2000. Guam [map]. 1:24000. 7.5 Minute Series. Washington, D.C., USA.

Varnham, K., S.S. Roy, A. Seymour, J. Mauremootoo, C.G. Jones, and S. Harris. 2002. Eradicating Indian musk shrews (*Suncus murinus*, Soricidae) from Mauritian offshore islands. Pages 342–349 *in* C.R. Veitch and M.N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, U.K.

Vogt, S.R., and L.L. Williams. 2004. Common flora and fauna of the Mariana Islands. Privately printed, Saipan, Northern Mariana Islands.

Weihong, J, C.R. Veitch, and J.L. Craig. 1999. An evaluation of the efficiency of rodent trapping methods: the effect of trap arrangement, cover type, and bait. New Zealand Journal of Ecology 23:45–51.

Whitaker, A.H. 1973. Lizard populations on islands with and without Polynesian Rats, *Rattus exulans* (Peale). Proceedings of the New Zealand Ecological Society 20:121–130.

White, G.C. 2005. Correcting wildlife counts using detection probabilities. Wildlife Research 32:211–216.

White, G.C., and K.P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46(Supplement):120–138.

Wiewel, A.S. 2004a. Systematic rodent monitoring project update: bait and trap preference. Report to USGS, Fort Collins, Colorado, USA.

Wiewel, A.S. 2004b. Systematic rodent monitoring project: WAPA Sherman trap comparison. Report to USGS, Fort Collins, Colorado, USA.

Wiewel, A.S. 2005. Systematic rodent monitoring protocol. Report to USGS, Fort Collins, Colorado, USA.

Wiles, G.J. 2005. A checklist of the birds and mammals of Micronesia. Micronesica 38:141–189.

Wiles, G.J., A.B. Amerson, Jr., and R.E. Beck, Jr. 1990. The mammals of Tinian, Mariana Islands. Micronesica 23:167–180.

Wiles, G.J., I.H. Schreiner, D. Nafus, L.K. Jurgensen, and J.C. Manglona. 1996. The status, biology, and conservation of *Serianthes nelsonii* (Fabaceae), an endangered Micronesian tree. Biological Conservation 76:229–239.

Wilson, K.R., and D.R. Anderson. 1985. Evaluation of two density estimators of small mammal population size. Journal of Mammalogy 66:13–21.

Wilson, D.J., W.G. Lee, R.A. Webster, and R.B. Allen. 2003. Effects of possums and rats on seedling establishment at two forest sites in New Zealand. New Zealand Journal of Ecology 27:147–155.

Wirtz, W.O., II. 1972. Population ecology of the Polynesian rat, *Rattus exulans*, on Kure Atoll, Hawaii. Pacific Science 26:433–464.

Worthy, T.H. 1987. Palaeoecological information concerning members of the frog genus *Leiopelma*: Leiopelmatidae in New Zealand. Journal of the Royal Society of New Zealand 17:409–420.

Yosida, T.H., H. Kato, K. Tsuchiya, and K. Moriwaki. 1971. Karyotypes and serum transferring patterns of hybrids between Asian and Oceanian black rats, *Rattus rattus*. Chromosoma 34:40–50.

Yosida, T.H., T. Udagawa, M. Ishibashi, K. Moriwaki, T. Yabe, and T. Hamada. 1985. Studies on the karyotypes of the black rats distributed in the Pacific and south Pacific islands, with special regard to the border line of the Asian and Oceanian type of black rats in the Pacific Ocean. Proceedings of the Japanese Academy, Series B 61:71–74.

TABLES

Table 1. Introduced small mammal sampling site coordinates and dates on Guam, Rota, Saipan, and Tinian, 2005–2007. Coordinates indicate the site centroid, and are presented in decimal degrees (WGS 84, UTM Zone 55). Note that CP05, CP06, and CP07 represent a single site sampled annually.

Site	Habitat	Dates Sampled	Latitude	Longitude
Guam				
MSRG	grassland	Jun 6–10, 2005	13.542	144.912
ASMF	*Leucaena* forest	May 30–Jun 3, 2005	13.512	144.870
CP05 CP06 CP07	*Leucaena* forest	May 2–6, 2005 May 15–19, 2006 Jun 6–10, 2007	13.640	144.865
GSYF	*Leucaena* forest	Nov 6–10, 2006	13.437	144.659
PAGO	*Leucaena* forest	Jun 20–24, 2005	13.417	144.783
GAHF	mixed	Oct 23–27, 2006	13.491	144.795
NMAR	native forest	May 16–20, 2005	13.378	144.672
RITL	native forest	Apr 18–22, 2005	13.648	144.863
Rota				
SABA	grassland	Jan 23–27, 2006	14.140	145.191
GAON	*Leucaena* forest	Jan 30–Feb 3, 2006	14.115	145.199
RAPF	mixed	Apr 10–14, 2006	14.170	145.240
ASAK	native forest	Apr 3–7, 2006	14.154	145.170
Saipan				
ACHU	grassland	Sep 19–23, 2005	15.238	145.773
OBYT	*Leucaena* forest	Sep 26–30, 2005	15.108	145.729
SAEN	mixed	Aug 22–26, 2006	15.127	145.727
SPOR	mixed	Aug 15–19, 2006	15.227	145.744
LATT	native forest	Sep 12–16, 2005	15.251	145.798
Tinian				
KAST	grassland	Oct 24–28, 2005	14.951	145.651
ABLE	*Leucaena* forest	Nov 7–11, 2005	15.076	145.640
LSUS	native forest	Oct 31–Nov 4, 2005	15.043	145.629

Table 2. Model selection results from analysis of variance of multiple models explaining variation in *Rattus rattus*, *Suncus murinus*, and *Mus musculus* body condition index (bodycon) on Guam, Rota, Saipan, and Tinian, 2005–2007, as a function of island[4] (each island modeled separately), island[2] (Guam vs. Rota, Saipan, and Tinian combined), and habitat. Results include the number of model parameters (*K*), relative Akaike's Information Criterion corrected for small sample size (ΔAIC_c), and Akaike weight (w_i).

	K	ΔAIC_c	w_i
R. rattus			
Bodycon(island[4])	6	0.00	1.000
Bodycon(island[2])	4	49.34	0.000
Bodycon(habitat)	6	70.70	0.000
S. murinus			
Bodycon(habitat)	6	0.00	0.999
Bodycon(island[4])	5	21.11	0.001
Bodycon(island[2])	4	23.90	0.000
M. musculus			
Bodycon(island[2])	6	0.00	0.656
Bodycon(island[4])	4	1.33	0.378
Bodycon(habitat)	6	9.38	0.006

Table 3. Mean ($\overline{X}$) rainfall (mm), standard error (SE), and 95% confidence intervals (95% CI) measured during livetrapping on Guam, Rota, Saipan, and Tinian, 2005–2007 ($n = 22$ sites). Average rainfall measurement period was 12–16 hours on occasion 1 and 24 hours on occasions 2–5.

	$\overline{X}$	SE	95% CI
Occasion 1	4.3	2.1	0.1–8.5
Occasion 2	4.8	1.6	1.6–7.9
Occasion 3	3.7	1.1	1.6–5.9
Occasion 4	6.1	1.8	2.5–9.7
Occasion 5	4.7	2.4	0–9.4

Table 4. Model selection results from analysis of variance of multiple models explaining variation in *Rattus rattus*, *Suncus murinus*, and *Mus musculus* mean maximum distance moved (MMDM) between captures during livetrapping on Guam, Rota, Saipan, and Tinian, 2005–2007, as a function of island[4], island[2], and habitat. Results include the number of model parameters (*K*), relative Akaike's Information Criterion corrected for small sample size (ΔAIC_c), and Akaike weight (w_i).

	K	ΔAIC_c	w_i
R. rattus			
MMDM(island[4])	6	0.00	0.977
MMDM(island[2])	4	7.48	0.023
MMDM(habitat)	6	22.66	0.000
S. murinus			
MMDM(habitat)	6	0.00	0.986
MMDM(island[4])	5	8.88	0.012
MMDM(island[2])	4	12.62	0.002
M. musculus			
MMDM(island[4])	6	0.00	0.718
MMDM(island[2])	4	1.94	0.272
MMDM(habitat)	6	8.54	0.010

Table 5. Model selection results from analysis of variance of multiple models explaining variation in *Rattus rattus*, *Suncus murinus*, and *Mus musculus* mass observed during livetrapping on Guam, Rota, Saipan, and Tinian, 2005–2007, as a function of island[4], island[2], habitat, and site. Results include the number of model parameters (*K*), relative Akaike's Information Criterion corrected for small sample size (ΔAIC_c), and Akaike weight (w_i).

	K	ΔAIC_c	w_i
R. rattus			
Mass(site)	20	0.00	1.000
Mass(island[2])	4	78.28	0.000
Mass(island[4])	6	79.86	0.000
Mass(habitat)	6	126.80	0.000
S. murinus			
Mass(habitat)	6	0.00	0.974
Mass(site)	11	7.24	0.026
Mass(island[2])	4	25.69	0.000
Mass(island[4])	5	26.71	0.000
M. musculus			
Mass(island[4])	6	0.00	0.903
Mass(island[2])	4	5.82	0.049
Mass(site)	10	5.88	0.048
Mass(habitat)	6	15.47	0.000

Table 6. *Mus musculus*, *Rattus exulans*, *R. norvegicus*, *R. rattus*, and *Suncus murinus* individuals captured (M_{t+1}) and total captures ($n_{.}$) during mark-recapture livetrapping in grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Rota, Saipan, and Tinian, 2005–2007.

		M. musculus		*R. exulans*		*R. norvegicus*		*R. rattus*		*S. murinus*	
Site	Habitat	M_{t+1}	$n_{.}$	M_{t+1}	$n_{.}$	M_{t+1}	$n_{.}$	M_{t+1}	$n_{.}$	M_{t+1}	$n_{.}$
Guam											
MSRG	grassland	15	29	1	2			22	24	14	22
ASMF	*Leucaena* forest							5	6		
CP05	*Leucaena* forest			1	1			4	4		
CP06								22	27		
CP07											
GSYF	*Leucaena* forest							13	14		
PAGO	*Leucaena* forest										
GAHF	mixed							1	1		
NMAR	native forest										
RITL	native forest										
Rota											
SABA	grassland	25	32					88	119		
GAON	*Leucaena* forest	19	27	13	16			42	63		
RAPF	mixed	32	51					106	146		
ASAK	native forest	1	2					11	11		
Saipan											
ACHU	grassland	51	96					41	63	19	41
OBYT	*Leucaena* forest	2	2	1	2	2	2	50	58	43	63
SAEN	mixed					1	1	8	8	47	59
SPOR	mixed					1	1	29	34	9	9
LATT	native forest							24	28	19	21
Tinian											
KAST	grassland	9	12			1	1	106	132	11	11
ABLE	*Leucaena* forest							55	81	93	113
LSUS	native forest							80	92	43	43

Table 7. Model selection results for mark-recapture modeling of capture (*p*) and recapture (*c*) probability for *Rattus rattus*, *Suncus murinus*, and *Mus musculus* livetrapping data collected on Guam, Rota, Saipan, and Tinian, 2005–2007. Parenthetical terms indicate the nesting structure of the previous variable (e.g., neo2(island[4]) specifies separate neophobia effects for each island). All heterogeneity models (h) used 2 finite mixtures to approximate individual heterogeneity. Results include the number of model parameters (*K*), relative Akaike's Information Criterion corrected for small sample size (ΔAIC_c), and Akaike weight (w_i).

	K	ΔAIC_c	w_i
Models for *R. rattus*			
p neo2(island[4]) + sex + repstat + rainamt *c* island[4] + sex + repstat + rainamt	16	0.00	0.871
p neo2(island[4]) + repstat + rainamt *c* island[4] + repstat + rainamt	15	5.72	0.050
p neo2(island[4]) + sex + age + repstat + bodycon + size + rainprev + rainamt *c* island[4] + sex + age + repstat + bodycon + size + rainprev + rainamt	20	6.97	0.027
p neo2(island[4]) + sex + rainamt *c* island[4] + sex + rainamt	15	7.12	0.025
p neo2(island[4]) + rainamt *c* island[4] + rainamt	14	7.56	0.020
p neo2(island[4]) + sex + repstat *c* island[4] + sex + repstat	15	9.61	0.007
Models for *S. murinus*			
p t + h *c* t(island[4]) + h	10	0.00	0.994
p t + h + sex + repstat + bodycon + size + rainprev + rainamt *c* t(island[4]) + h + sex + repstat + bodycon + size + rainprev + rainamt	16	10.33	0.006
Models for *M. musculus*			
p t(island[2]) + h *c* t + h	11	0.00	0.349
p t(island[2]) + h *c* t(island[2]) + h	13	0.10	0.331
p t(island[4]) + h *c* t + h	19	1.20	0.192
p t + h *c* t(island[2]) + h	9	4.36	0.039
p t(island[4]) + h *c* t(island[4]) + h	23	4.65	0.034
p t + h *c* t + h	7	5.01	0.029
p neo1 + h *c* h	4	6.79	0.012
p t + h *c* t(habitat) + h	11	7.94	0.007
p neo2 + h *c* h	5	8.82	0.004
p t(island[2]) + h + sex + repstat + bodycon + size + rainprev + rainamt *c* t + h + sex + repstat + bodycon + size + rainprev + rainamt	17	9.23	0.003

Table 8. Model-averaged closed population abundance estimates ($\hat{\bar{N}}$), standard errors (SE), and 95% confidence intervals (95% CI) for *Rattus rattus*, *S. murinus*, and *Mus musculus* captured during livetrapping in grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Rota, Saipan, and Tinian, 2005–2007.

		R. rattus			*S. murinus*			*M. musculus*		
Site	Habitat	$\hat{\bar{N}}$	SE	95% CI	$\hat{\bar{N}}$	SE	95% CI	$\hat{\bar{N}}$	SE	95% CI
Guam										
MSRG	grassland	41.1	9.4	22.8–59.5	20.3	5.8	15.4–43.2	17.5	4.0	9.6–25.3
ASMF	*Leucaena* forest	6.6	1.7	3.3–9.9						
CP05	*Leucaena* forest	6.8	2.6	1.7–11.9						
CP06		39.5	8.7	22.4–56.6						
CP07										
GSYF	*Leucaena* forest	22.9	5.8	11.5–34.3						
PAGO	*Leucaena* forest									
GAHF	mixed	1.8	1.2	0–4.1						
NMAR	native forest									
RITL	native forest									
Rota										
SABA	grassland	142.4	22.8	97.6–187.1				41.5	10.1	21.6–61.3
GAON	*Leucaena* forest	70.0	12.9	44.7–95.2				32.0	8.3	15.7–48.3
RAPF	mixed	186.4	31.0	125.7–247.2				53.2	12.7	28.4–78.1
ASAK	native forest	17.8	4.3	9.4–26.3				1.7	1.2	0–4.0
Saipan										
ACHU	grassland	72.2	13.9	44.9–99.5	28.8	8.5	21.3–61.4	80.5	17.4	46.5–114.6
OBYT	*Leucaena* forest	90.6	17.4	56.4–124.7	67.7	20.0	49.1–142.3	3.2	1.6	0.2–6.3
SAEN	mixed	15.0	4.7	5.9–24.1	70.6	19.1	52.8–141.9			
SPOR	mixed	54.8	11.7	31.9–77.7	13.6	4.4	9.9–31.6			
LATT	native forest	47.1	11.1	25.4–68.8	29.9	9.4	21.5–65.8			
Tinian										
KAST	grassland	194.4	34.5	126.8–262.1	17.3	5.8	12.4–40.2	14.6	4.4	5.9–23.3
ABLE	*Leucaena* forest	85.6	14.1	58.0–113.2	143.0	39.5	105.8–288.5			
LSUS	native forest	146.1	26.3	94.6–197.6	63.7	16.9	48.1–127.0			

Table 9. *Rattus rattus*, *Suncus murinus*, and *Mus musculus* density estimates ($\hat{D}$; animals/ha), standard errors (SE), and 95% confidence intervals (95% CI) in grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Rota, Saipan, and Tinian, 2005–2007. Blank entries indicate zero captures, and therefore zero estimated density.

		R. rattus			*S. murinus*			*M. musculus*		
Site	Habitat	$\hat{D}$	SE	95% CI	$\hat{D}$	SE	95% CI	$\hat{D}$	SE	95% CI
Guam										
MSRG	grassland	15.9	3.7	8.6–23.2	8.6	2.5	3.7–13.5	7.2	1.7	3.9–10.5
ASMF	*Leucaena* forest	2.6	0.7	1.2–4.0						
CP05	*Leucaena* forest	2.6	1.0	0.6–4.6						
CP06		15.3	3.5	8.4–22.2						
CP07										
GSYF	*Leucaena* forest	8.9	2.3	4.4–13.4						
PAGO	*Leucaena* forest									
GAHF	mixed	0.7	0.5	0–1.7						
NMAR	native forest									
RITL	native forest									
Rota										
SABA	grassland	73.2	11.9	49.9–96.5				20.7	5.0	10.9–30.5
GAON	*Leucaena* forest	36.0	6.7	22.9–49.1				16.0	4.1	8.0–24.0
RAPF	mixed	95.8	16.1	64.2–127.4				26.5	6.3	14.2–38.8
ASAK	native forest	9.2	2.2	4.9–13.5				0.8	0.6	0–2.0
Saipan										
ACHU	grassland	33.0	6.4	20.5–45.5	13.4	3.7	6.1–20.7	36.5	8.1	20.6–52.4
OBYT	*Leucaena* forest	41.4	8.1	25.5–57.3	31.6	10.2	11.6–51.6	1.5	0.7	0.1–2.9
SAEN	mixed	6.9	2.1	2.8–11.0	32.9	9.6	14.1–51.7			
SPOR	mixed	25.1	5.4	14.5–35.7	6.3	2.2	2.0–10.6			
LATT	native forest	21.6	5.1	11.6–31.6	14.0	5.2	3.8–24.2			
Tinian										
KAST	grassland	99.9	17.9	64.8–135.0	8.9	2.5	4.0–13.8	8.2	2.7	2.9–13.5
ABLE	*Leucaena* forest	44.0	7.3	29.7–58.3	73.7	20.1	34.3–113.1			
LSUS	native forest	75.1	13.6	48.4–101.8	32.8	9.6	14.0–51.6			

Table 10. Mean ($\overline{X}$) and standard error (SE) mass (g), head + body length (mm), tail length (mm), hind foot length (mm), and ear length (mm) of adult *Rattus rattus*, *Suncus murinus*, and *Mus musculus* captured during livetrapping on Guam, Rota, Saipan, and Tinian, 2005–2007.

	Guam		Rota		Saipan		Tinian	
	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE	$\overline{X}$	SE
R. rattus								
n	65		263		218		277	
Mass	183.69	5.70	136.04	1.97	137.40	1.48	130.63	1.76
Head + Body Length	194.53	2.63	184.73	0.98	177.96	0.77	179.46	0.99
Tail Length[a]	205.11	2.76	195.63	1.11	189.04	1.11	194.50	1.19
Hind Foot Length	32.18	0.23	31.17	0.07	30.68	0.08	30.87	0.08
Ear Length	20.21	0.16	20.17	0.06	19.38	0.07	20.22	0.08
S. murinus								
N	30				236		208	
Mass	22.72	1.29			25.68	0.44	24.43	0.44
Head + Body Length	97.61	2.16			104.96	0.56	101.17	0.61
Tail Length[b]	63.98	1.18			67.19	0.34	66.55	0.34
Hind Foot Length	15.06	0.22			15.04	0.06	15.14	0.05
M. musculus								
N	19		98		73		10	
Mass	11.75	0.51	12.56	0.22	11.97	0.26	14.59	0.89
Head + Body Length	66.38	1.08	71.86	0.46	71.03	0.62	76.81	1.76
Tail Length[c]	77.43	0.87	75.04	0.49	76.72	0.69	79.76	1.75
Hind Foot Length	13.65	0.14	14.51	0.07	13.60	0.08	14.46	0.22
Ear Length	10.46	0.19	10.68	0.06	10.27	0.08	10.90	0.19

[a] Excludes *R. rattus* with damaged tails (corrected *n* = 62, 239, 202, and 251, respectively).

[b] Excludes *S. murinus* with damaged tails (corrected *n* = 30, 232, and 206, respectively).

[c] Excludes *M. musculus* with damaged tails (corrected *n* = 17, 85, 67, and 10, respectively).

Table 11. *Rattus rattus*, *Suncus murinus*, and *Mus musculus* biomass estimates ($\hat{Biom}$; kg/ha), standard errors (SE), and 95% confidence intervals (95% CI) in grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Rota, Saipan, and Tinian, 2005–2007. Blank entries indicate zero captures, and therefore zero estimated biomass.

		R. rattus			*S. murinus*			*M. musculus*		
Site	Habitat	$\hat{Biom}$	SE	95% CI	$\hat{Biom}$	SE	95% CI	$\hat{Biom}$	SE	95% CI
Guam										
MSRG	grassland	2.42	0.58	1.28–3.56	0.20	0.06	0.08–0.32	0.08	0.02	0.04–0.12
ASMF	*Leucaena* forest	0.70	0.19	0.33–1.07						
CP05		0.39	0.16	0.08–0.70						
CP06	*Leucaena* forest	2.88	0.66	1.59–4.17						
CP07										
GSYF	*Leucaena* forest	1.36	0.37	0.63–2.09						
PAGO	*Leucaena* forest									
GAHF	mixed	0.06	0.05	0–0.16						
NMAR	native forest									
RITL	native forest									
Rota										
SABA	grassland	9.80	1.62	6.62–12.98				0.26	0.07	0.12–0.40
GAON	*Leucaena* forest	4.63	0.89	2.89–6.37				0.20	0.05	0.10–0.30
RAPF	mixed	8.85	1.54	5.83–11.87				0.34	0.08	0.18–0.50
ASAK	native forest	1.03	0.28	0.48–1.58				0.01	0.01	0–0.03
Saipan										
ACHU	grassland	4.13	0.83	2.50–5.76	0.24	0.08	0.08–0.40	0.45	0.10	0.25–0.65
OBYT	*Leucaena* forest	4.31	0.87	2.60–6.02	0.88	0.27	0.35–1.41	0.01	0.01	0–0.03
SAEN	mixed	0.96	0.32	0.33–1.59	0.98	0.28	0.43–1.53			
SPOR	mixed	3.03	0.68	1.70–4.36	0.18	0.06	0.06–0.30			
LATT	native forest	3.18	0.76	1.69–4.67	0.40	0.13	0.15–0.66			
Tinian										
KAST	grassland	11.57	2.11	7.43–15.71	0.16	0.05	0.06–0.26	0.11	0.04	0.03–0.19
ABLE	*Leucaena* forest	5.09	0.88	3.37–6.81	1.87	0.52	0.85–2.89			
LSUS	native forest	8.78	1.63	5.59–11.97	0.83	0.25	0.34–1.32			

FIGURES

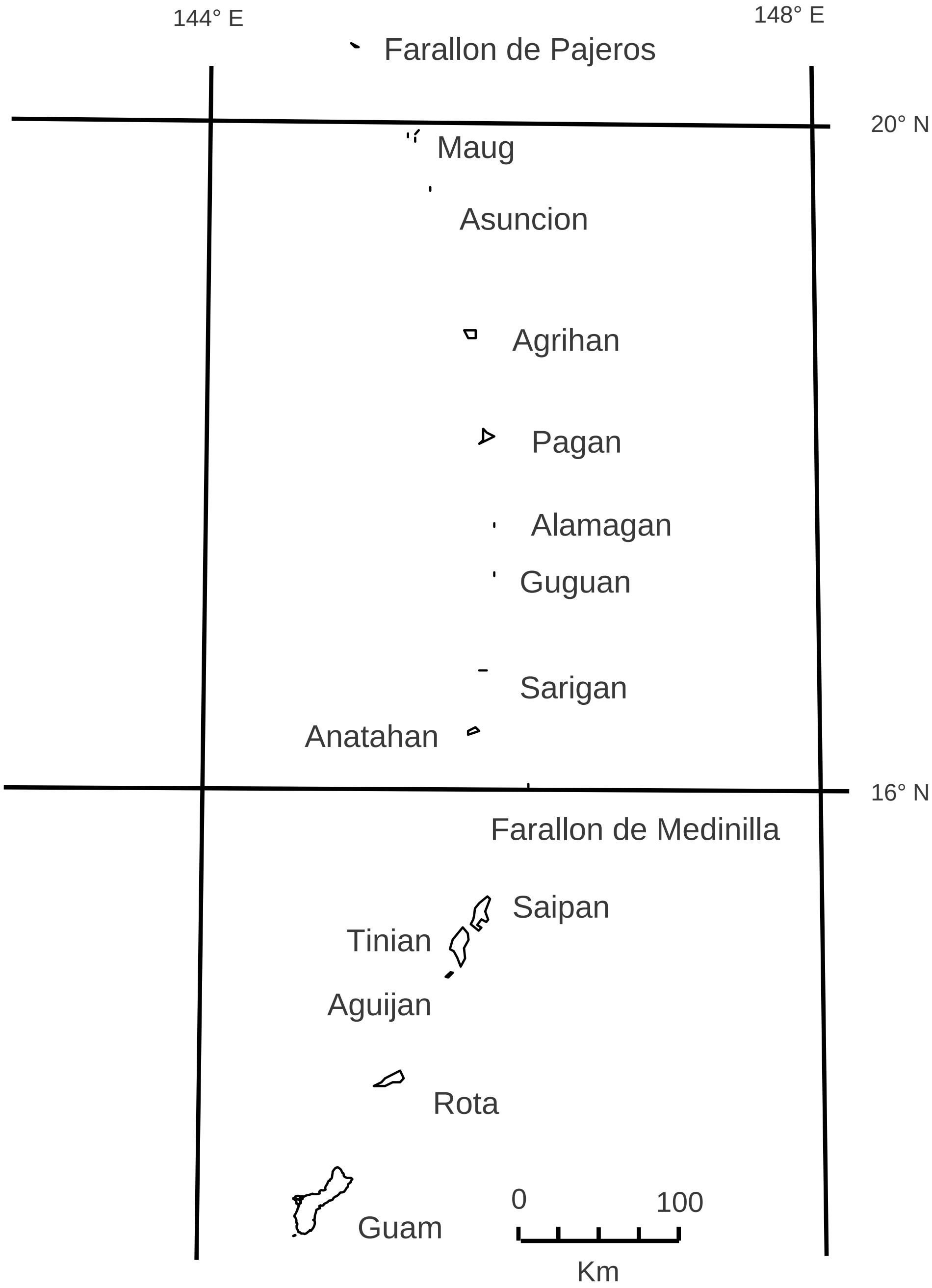


Figure 1. Map of the principal Mariana Islands.

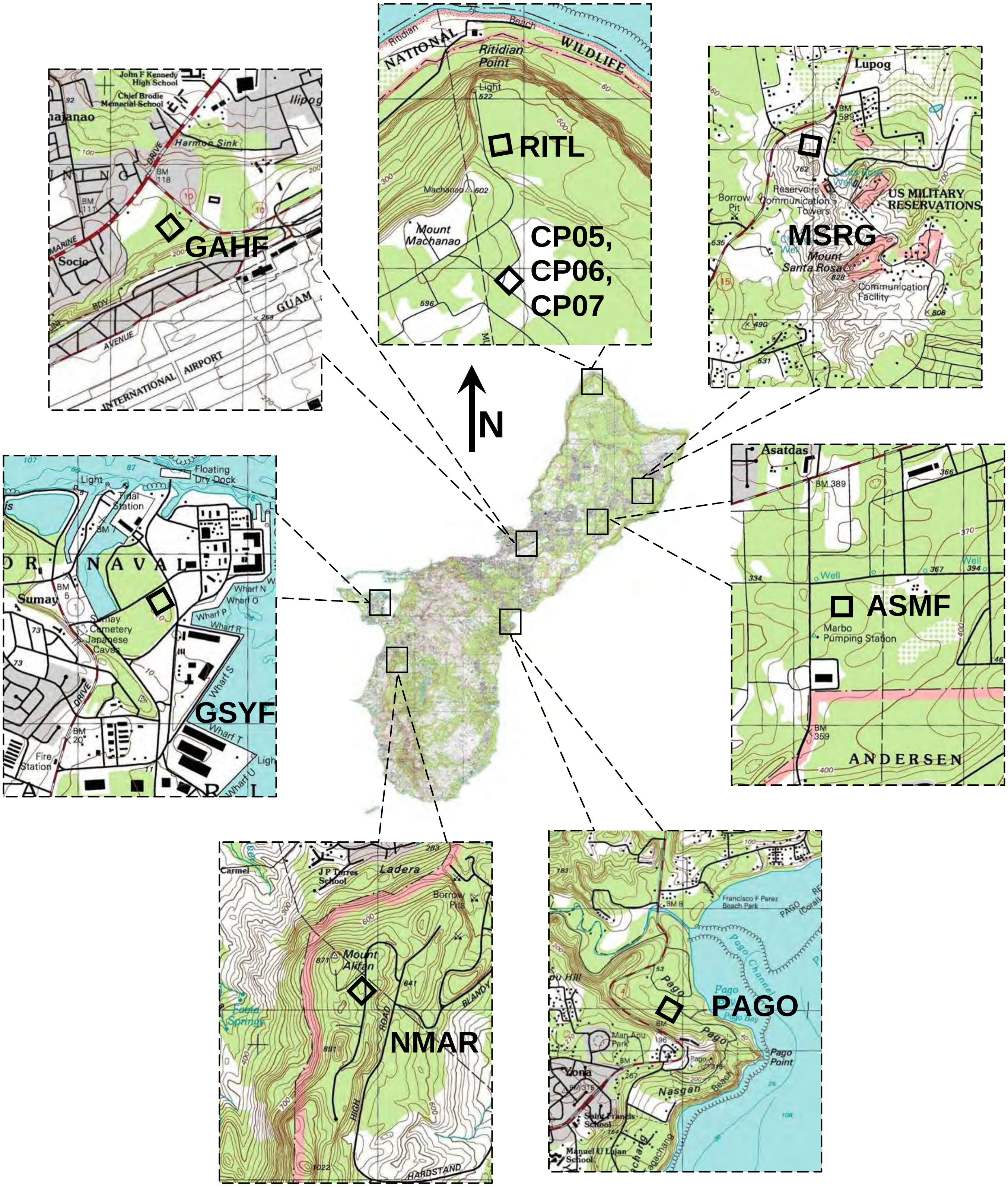


Figure 2. Introduced small mammal sampling locations on Guam, Mariana Islands, 2005–2007. Sampling grids are delineated with bold squares, which represent an area of 125 m^2 (1.56 ha). See Table 1 for site coordinates, sampling dates, and habitat classifications. Note that CP05, CP06, and CP07 represent a single site sampled annually.

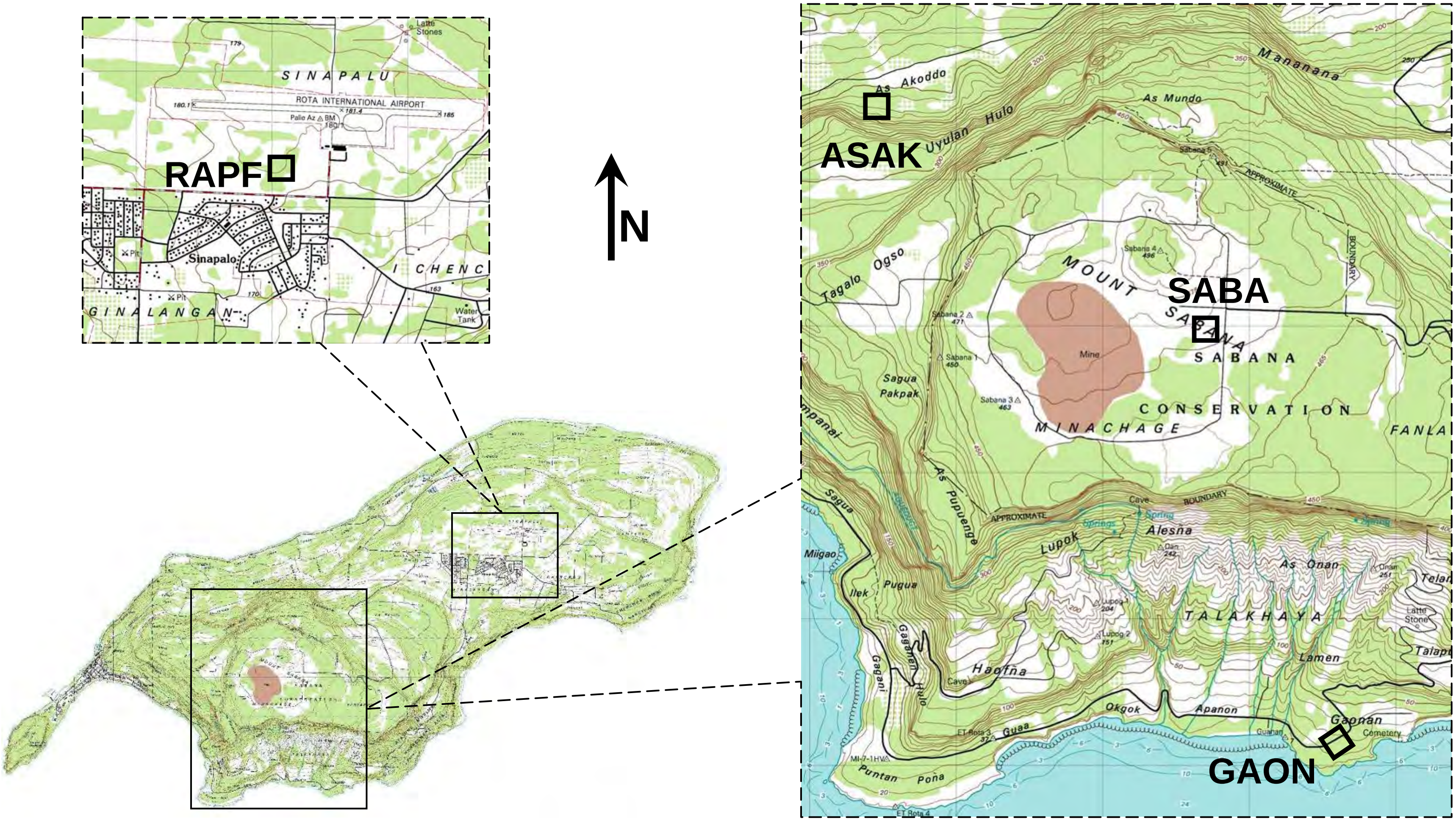


Figure 3. Introduced small mammal sampling locations on Rota, Mariana Islands, 2005–2007. Sampling grids are delineated with bold squares, which represent an area of 125 m^2 (1.56 ha). See Table 1 for site coordinates, sampling dates, and habitat classifications.

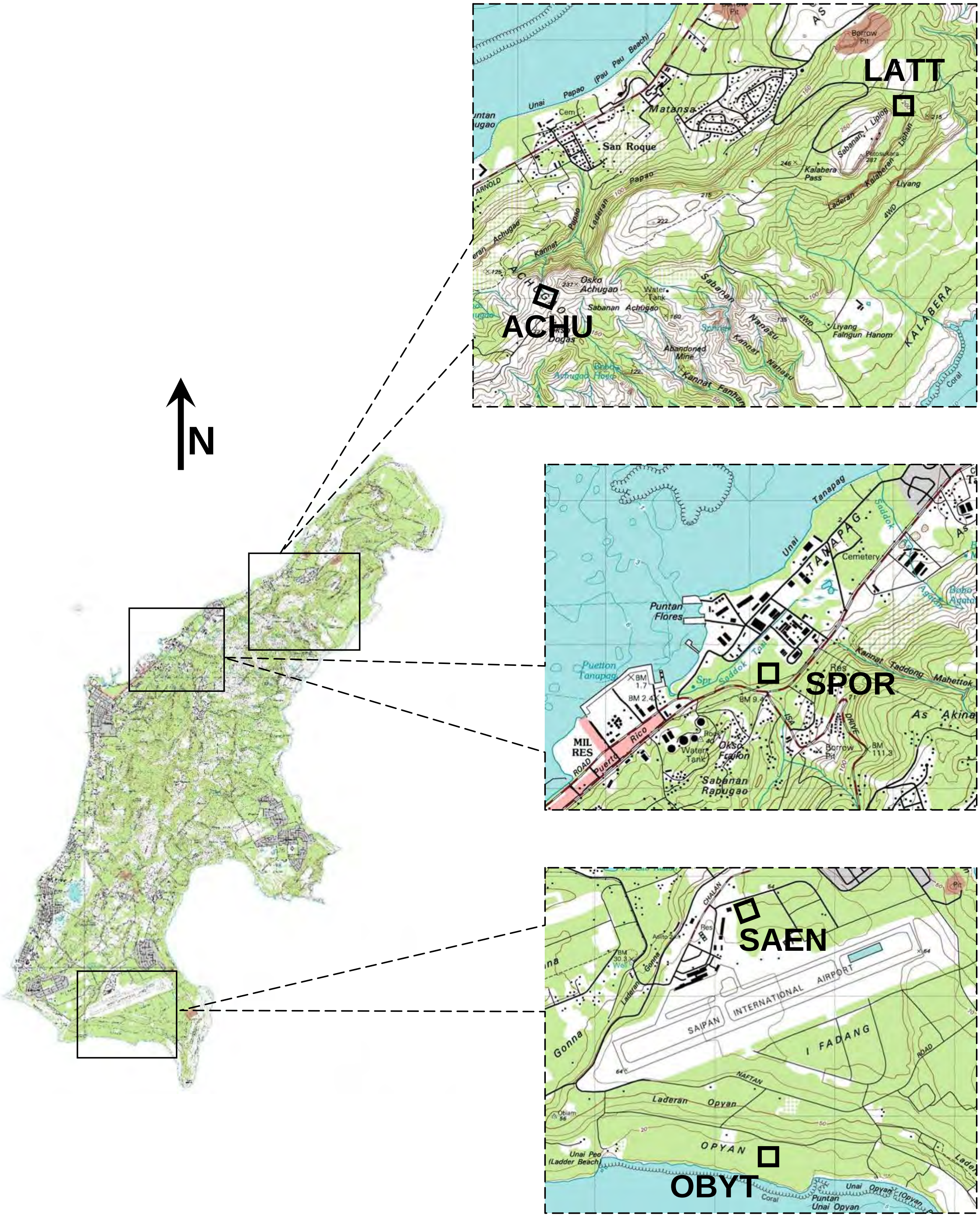


Figure 4. Introduced small mammal sampling locations on Saipan, Mariana Islands, 2005–2007. Sampling grids are delineated with bold squares, which represent an area of 125 m^2 (1.56 ha). See Table 1 for site coordinates, sampling dates, and habitat classifications.

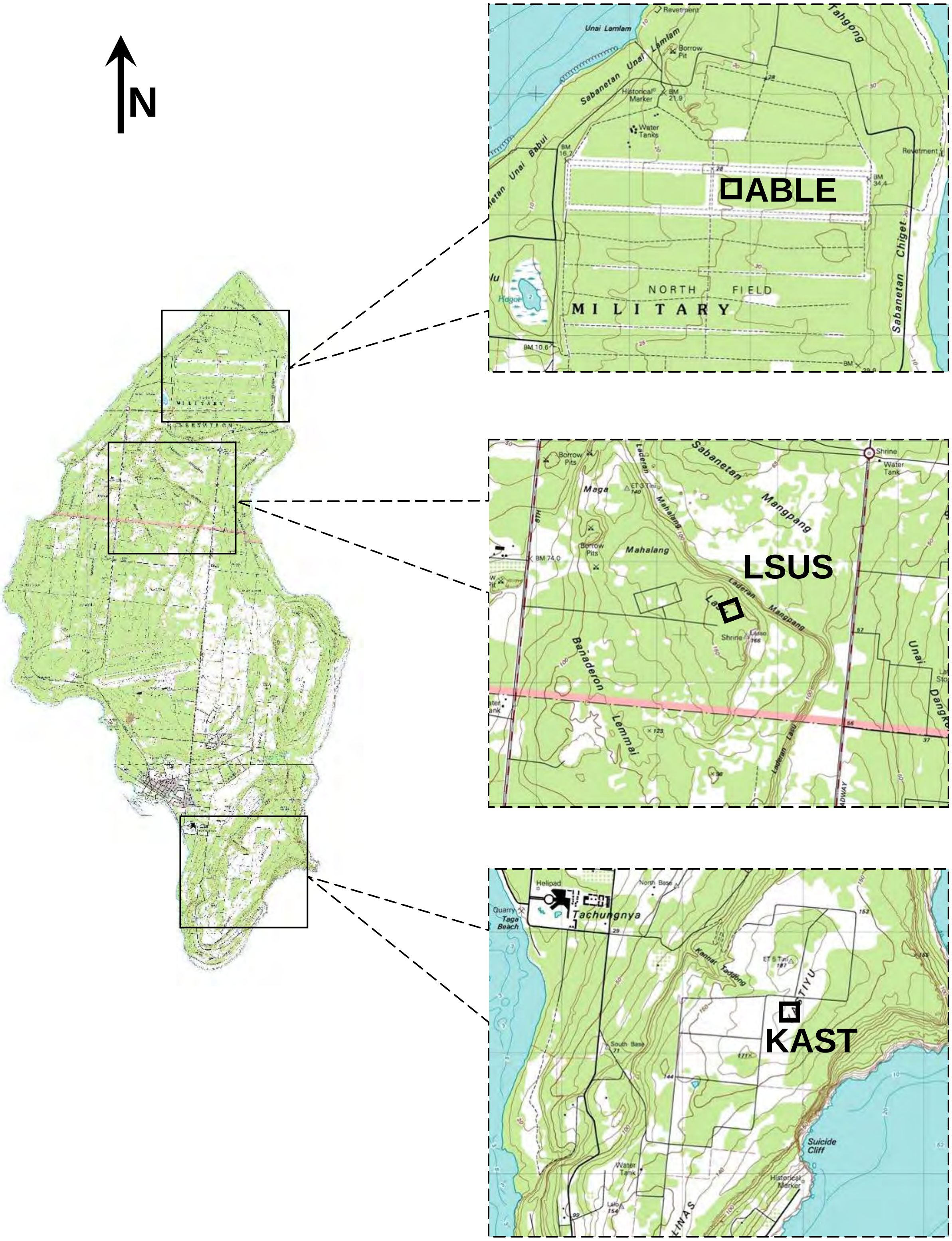


Figure 5. Introduced small mammal sampling locations on Tinian, Mariana Islands, 2005–2007. Sampling grids are delineated with bold squares, which represent an area of 125 m^2 (1.56 ha). See Table 1 for site coordinates, sampling dates, and habitat classifications.

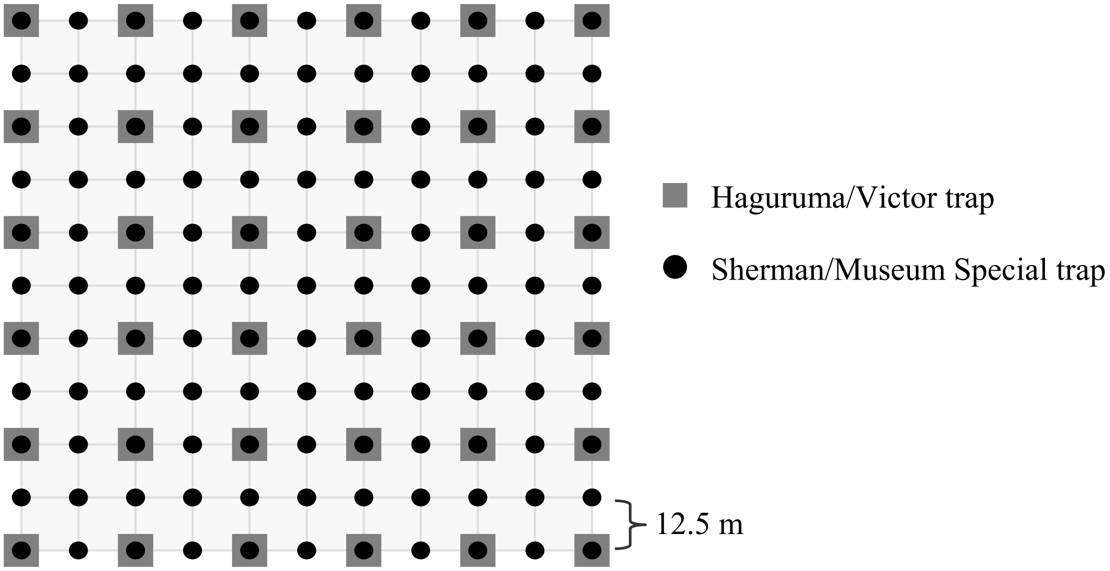


Figure 6. Schematic diagram of 11 × 11 grid (nominal area = 1.56 ha) used during mark-recapture livetrapping on Guam, Rota, Saipan, and Tinian, 2005–2007.

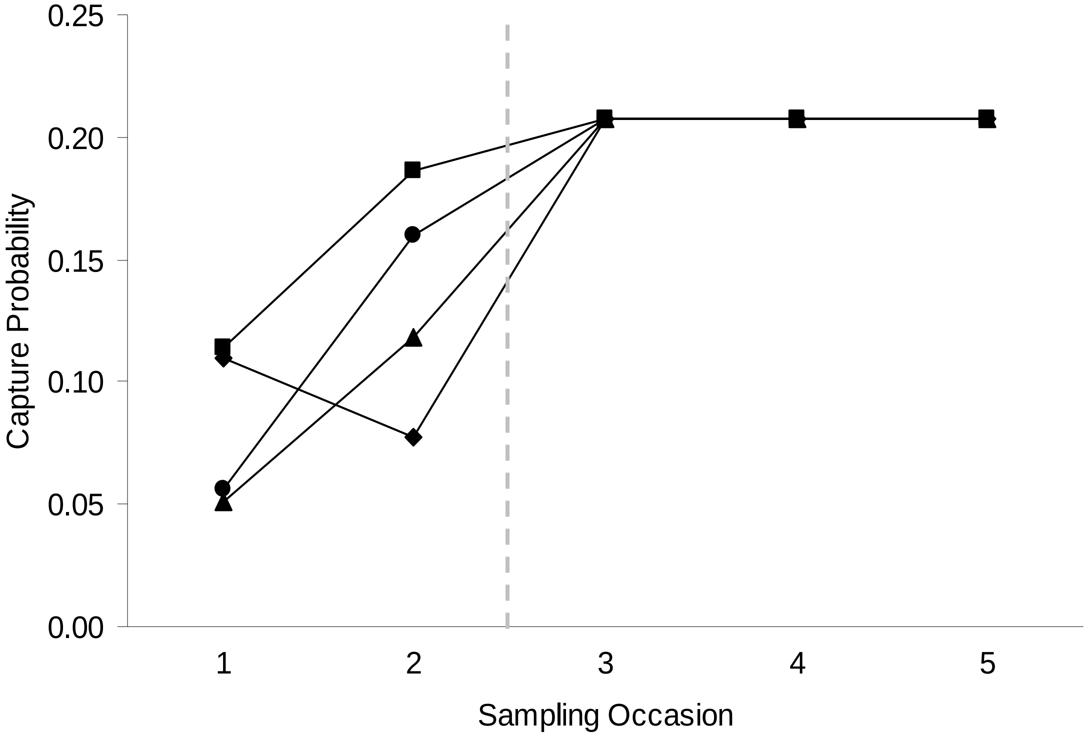


Figure 7. Island-specific neophobia effect on sampling occasion 1 and 2 *Rattus rattus* capture probabilities from mark-recapture livetrapping conducted on Guam (♦), Rota (■), Saipan (▲), Tinian (●), 2005–2007. The dashed line delineates the neophobia effect.

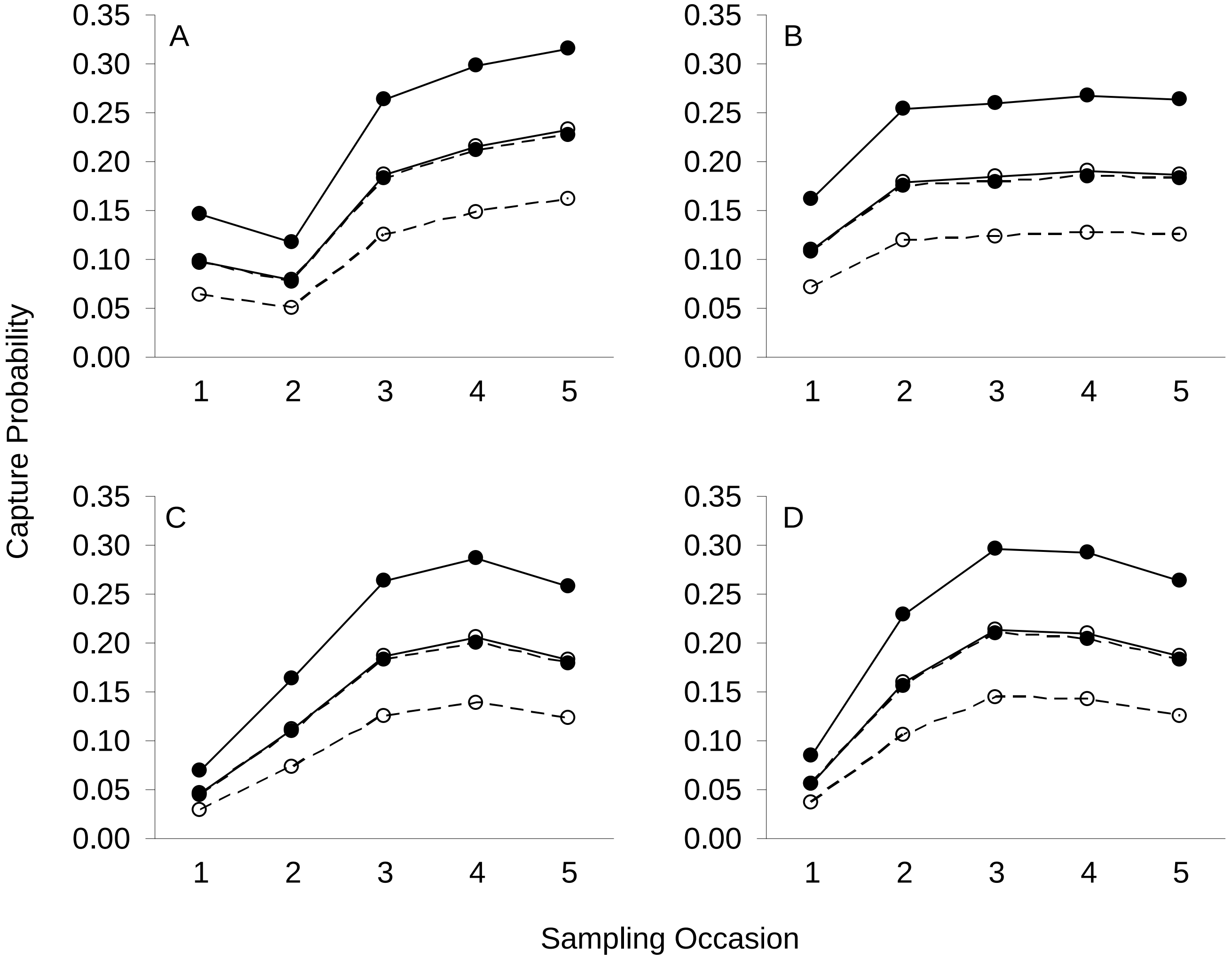


Figure 8. Effect of sex (female = •, male = ○) and reproductive status (reproductively active = solid line, non-reproductive = dashed line) on *Rattus rattus* livetrapping capture probability on Guam (A), Rota (B), Saipan (C), and Tinian (D), 2005–2007.

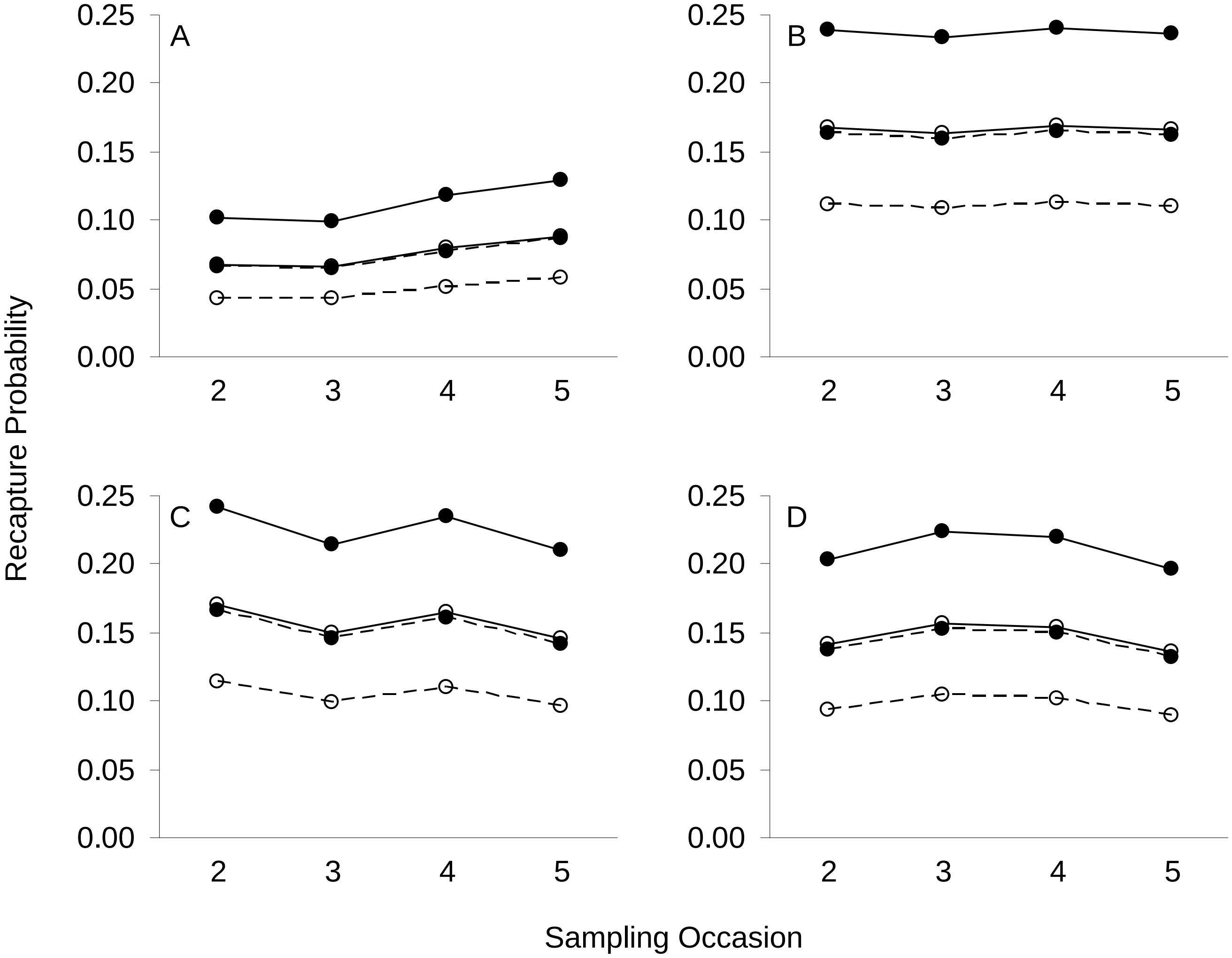


Figure 9. Effect of sex (female = •, male = ○) and reproductive condition (reproductively active = solid line, non-reproductive = dashed line) on *Rattus rattus* livetrapping recapture probability on Guam (A), Rota (B), Saipan (C), and Tinian (D), 2005–2007.

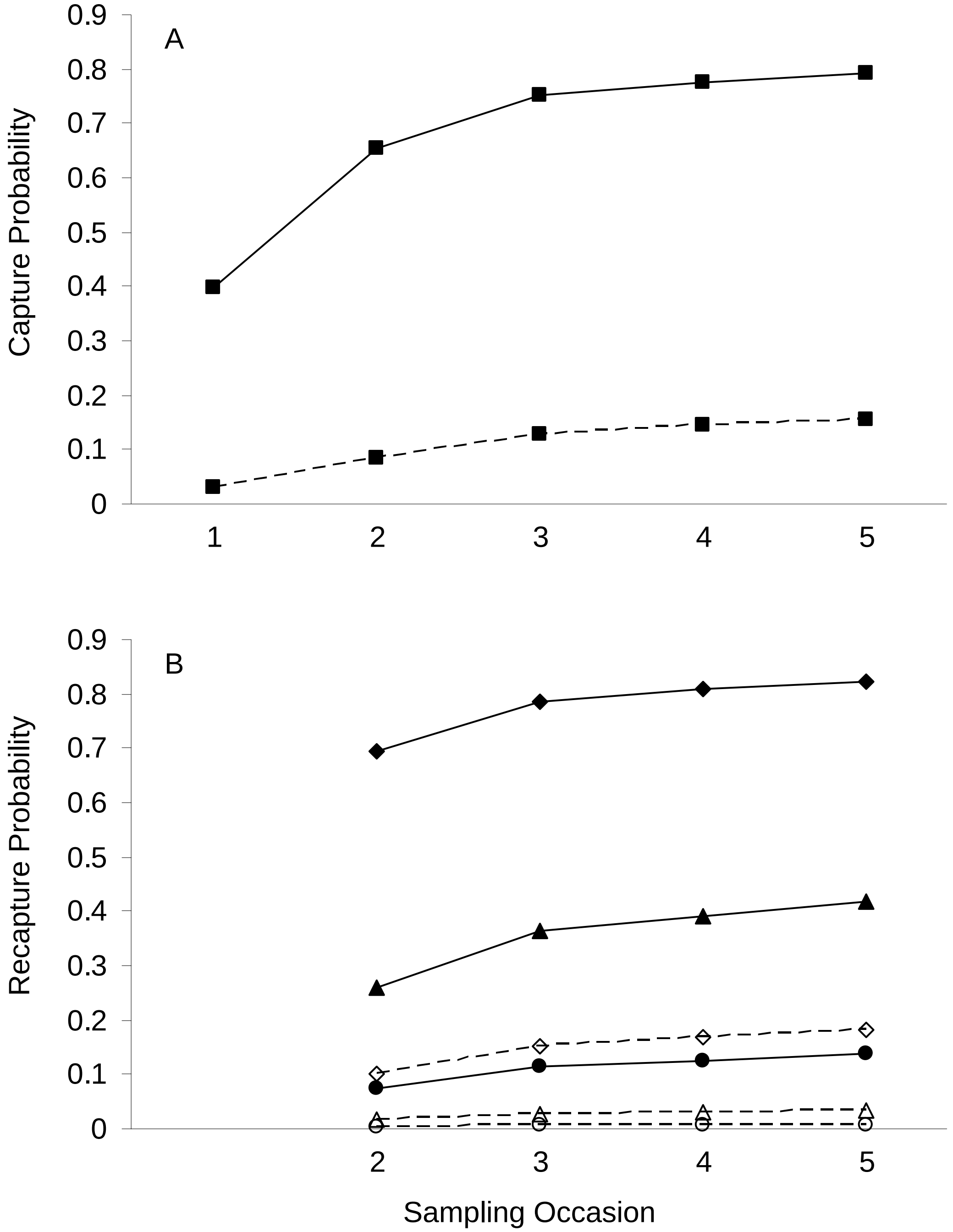


Figure 10. *Suncus murinus* livetrapping capture (A) and recapture (B) probability for high (35% of population; solid line) and low (65% of population; dashed line) mixture classes. Mixture-specific capture probabilities for all islands combined are indicated by ■ in panel A. Island-specific recapture probabilities are presented for Guam (♦), Saipan (▲), and Tinian (•) in panel B. *S. murinus* was not captured on Rota.

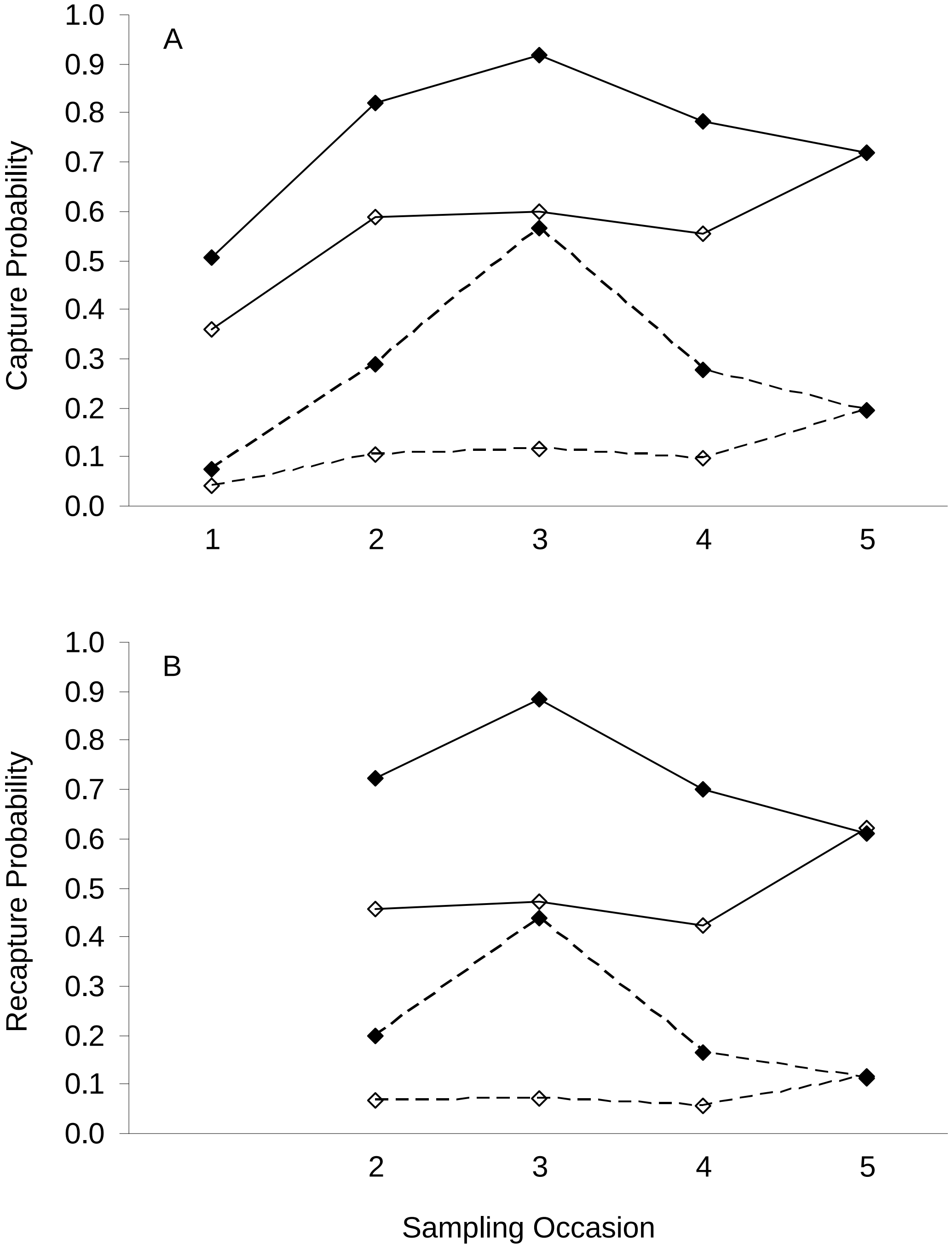


Figure 11. *Mus musculus* livetrapping capture (A) and recapture (B) probability for high (32.7% of population; solid line) and low (67.3% of population; dashed line) mixture classes. Mixture-specific capture and recapture probabilities are presented for Guam (♦) and the combination of Rota, Saipan, and Tinian (◊).

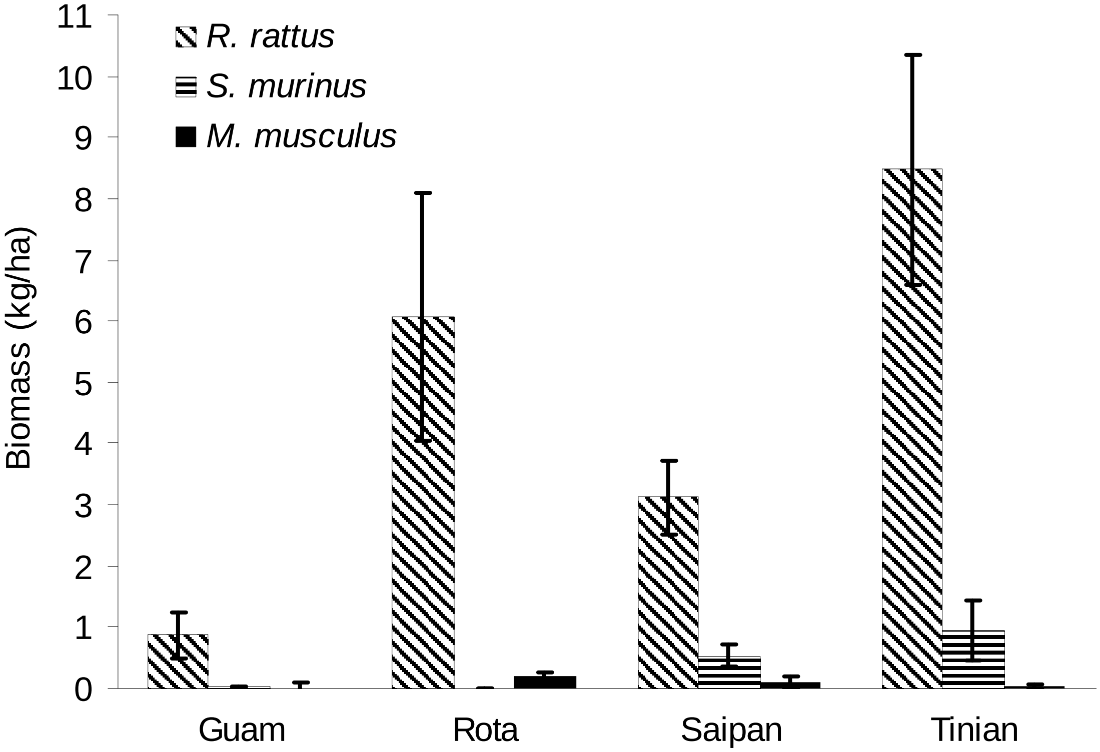


Figure 12. Mean *Rattus rattus*, *Suncus murinus*, and *Mus musculus* biomass estimates derived from mark-recapture livetrapping on Guam, Rota, Saipan, and Tinian, 2005–2007. Bars indicate ± 1 SE.

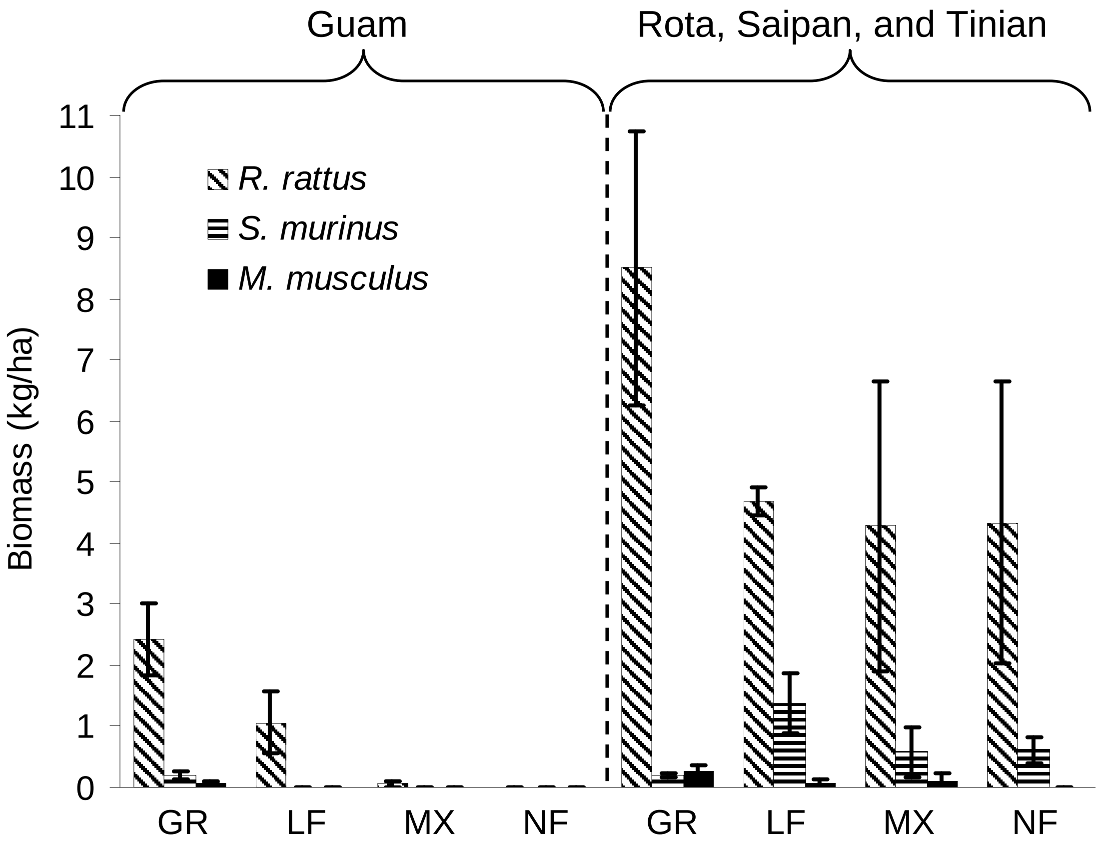


Figure 13. Mean *Rattus rattus*, *Suncus murinus*, and *Mus musculus* biomass estimates derived from mark-recapture livetrapping of grassland (GR), *Leucaena* forest (LF), mixed (MX), and native forest (NF) habitats on Guam, Rota, Saipan, and Tinian, 2005–2007. Bars indicate ± 1 SE.

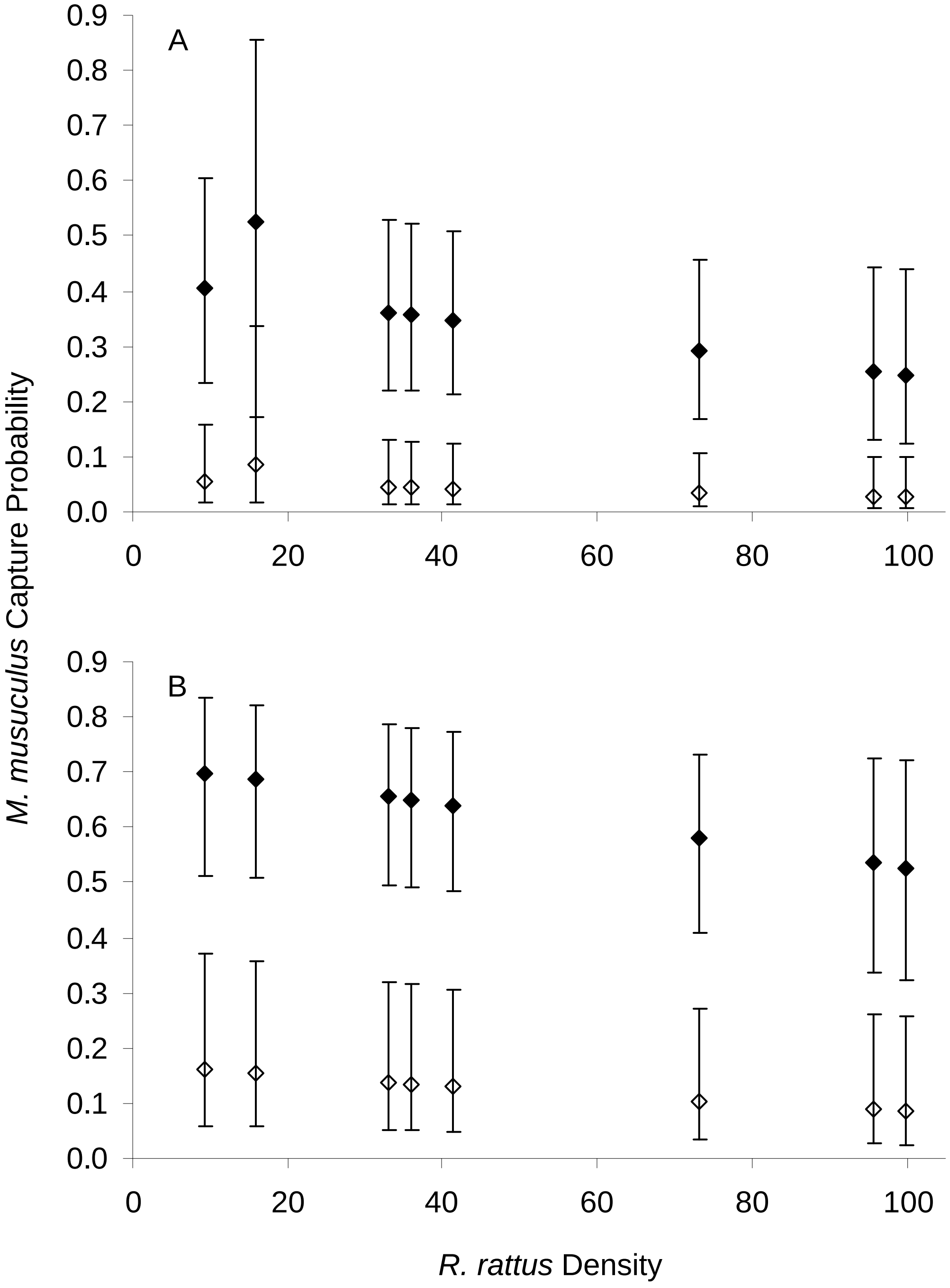


Figure 14. Effect of *R. rattus* density on *Mus musculus* livetrapping capture probability (β = -0.008 ± 0.006, 95% CI = -0.019–0.003) on Guam, Rota, Saipan, and Tinian, 2005–2007. Site-specific capture probability estimates are presented for high (32.7% of population; ♦) and low (67.3% of population; ◊) mixture classes on sampling occasion 1 (A) and sampling occasion 5 (B). Bounded bars indicate 95% CIs.

APPENDIX 1A. Review of small mammal introductions and research in the Mariana Islands

The native terrestrial mammalian fauna of the Mariana Islands is limited to the Marianas fruit bat (*Pteropus mariannus*), the little Mariana fruit bat (*Pteropus tokudae*), and the Pacific sheath-tailed bat (*Emballonura semicaudata*); all other terrestrial mammalian species historically or currently present were almost certainly introduced by humans (Stinson 1994, Wiles 2005). The Mariana Islands are somewhat unique in that they were one of the first non-continental Pacific island chains reached by both prehistoric humans (ca. 2300–1500 B.C.; Rainbird 1994, Athens and Ward 2004) and Europeans (A.D. 1521; Rogers 1995, Barratt 2003), resulting in a long history of introductions. However, attempts to determine when these introductions occurred and to identify them to the species level are complicated by limited archaeological evidence of prehistoric fauna, uncertain or unspecific identification of species (and islands) by early naturalists, and high levels of both past and present taxonomic uncertainty and revision. More importantly, the combination of extensive habitat modification, beginning with the arrival of prehistoric humans (Athens and Ward 2004), and massive movements of people and goods, especially following Spanish colonization (Russell 1998, Barratt 2003) and continuing through World War II (Bowers 2001), have likely influenced the introduced small mammal community in unknown ways. Due to these disturbances, it is likely that the composition of the introduced small mammal community in the Mariana Islands has shifted over time as newly introduced species supplanted earlier introductions or species were reintroduced to islands where they had not previously established successful population, such that a complete and accurate delineation of the history of introduced small mammals in the Mariana Islands may not be feasible. Nonetheless, there is considerable value in reviewing the available evidence of small mammal introductions and historic research, especially as this information may help researchers understand the impacts these species have had, and continue to have, on the ecology of the Mariana Islands.

History of Small Mammal Introductions

Archaeological, linguistic, and palaeoenvironmental evidence suggests that the Mariana Islands were colonized by people from the Philippines, beginning as early as ca. 2300 BC (Athens and Ward 2004) and no later than ca. 1500 BC (Rainbird 1994). These early Pacific explorers frequently transported *Rattus exulans*, either inadvertently (Tate 1935:147) or perhaps deliberately as a food resource (Roberts

1991, Matisoo-Smith and Robins 2004), suggesting that this species, native to Southeast Asia (Musser and Carleton 2005:1469–1470), was the first small mammal introduced in the Mariana Islands. While the precise date of this introduction is unknown, the earliest known *R. exulans* bone evidence in the Mariana Islands does not occur until ca. AD 1000–1200 (Steadman 1999), well after the accepted date of human colonization.

All available evidence suggests that the second small mammal introduced to the Mariana Islands was also a member of the *Rattus* genus, although there is considerable disagreement regarding both the specific identification and date of this introduction. Two of the species in question, *R. rattus* and *R. tanezumi*, are closely related species that have only recently been separated by taxonomists based on karyotype (*R. rattus*: $2n = 38$; *R. tanezumi*: $2n = 42$) as well as biochemical and morphological features (Schwabe 1979, Baverstock et al. 1983). The complex and evolving taxonomy of these closely related species (Musser and Carleton 2005:1484–1487, 1489–1491), and of the *Rattus* genus in general (Robins et al. 2007), complicates investigation of both historic and current distributions. Regarding *R. rattus* and *R. tanezumi*, additional confusion arises from limited hybridization observed in both laboratory (Yosida et al. 1971) and wild (Baverstock et al. 1983) populations, which led Baverstock et al. (1983:978) to conclude that they "…are best considered as incipient species. Where they meet, they may introgress, become sympatric without interbreeding, or one may replace the other depending upon the prevailing biological conditions."

Based on known historic ranges, viable introduction pathways to the Mariana Islands exist for both *R. rattus* and *R. tanezumi*. *R. rattus*, native to the Indian subcontinent (Musser and Carleton 2005:1484), was introduced to Europe as a human commensal and, beginning ca. AD 1500, transported across the globe on European ships (Atkinson 1985, Nowak 1999:1521). Thus, this species could have reached the Mariana Islands no earlier than European discovery of the islands by Ferdinand Magellan in 1521 (Atkinson 1985). In contrast, the native range of *R. tanezumi* extended into Southeast Asia, from which early introductions into island Southeast Asia, including the Malaysia and the Philippines (Musser and Carleton 2005:1489), put this species in position for potential transport by prehistoric Pacific explorers originating in this region. While a prehistoric introduction of *R. tanezumi* to the Mariana Islands is therefore possible, it seems that all scientific literature suggesting or referencing the prehistoric introduction of *R. tanezumi* can be linked back to an unpublished manuscript cited in the influential reference "Mammals of the South-West Pacific & Moluccan Islands" (Flannery 1995). This manuscript (White and Flannery unpublished manuscript) described bone evidence, dated ca. AD 1000, of a large *Rattus* species on Fais Atoll, Yap. Based on comparisons with museum specimens as well as the fact that the bones were thought older than the earliest European voyages to Micronesia the authors concluded that the bones belonged to *R. tanezumi*. White and Flannery (unpublished manuscript) also reviewed paleontological bone evidence from Guam, Rota, and Pagan in the Mariana Islands (in addition to other Micronesian islands) and speculated that these specimens were also *R. tanezumi* based on size and the estimated date of deposition. The authors concluded that *R. tanezumi* was a prehistoric introduction throughout Micronesia and, further, that the introduction of this species preceded the introduction of *R. exulans* throughout the region (White and Flannery unpublished manuscript). More recent research in the Mariana Islands, however, offers no support for this hypothesis (Steadman 1999). Further, recent advances in molecular identification of rodent skeletal remains suggests that traditional measures of bone morphology may not be adequate for differentiating between *R. exulans* and other introduced *Rattus* species, all of which have variable and overlapping size distributions on different Pacific islands (Matisoo-Smith and Allen 2001). Thus, the likelihood of a prehistoric introduction of *R. tanezumi* to the Mariana Islands remains uncertain.

If not a prehistoric introduction, it is possible that *R. tanezumi*, like *R. rattus*, was introduced following European discovery of the Mariana Islands. Attempting to date these introductions is difficult, although there is some evidence to suggest that introductions were unlikely immediately following European discovery. Most importantly, relatively few European ships visited the Mariana Islands between Magellan's visit and 1565 when the islands were formally claimed by Spain (Driver 1988, Barratt 2003, Driver 2005). However, beginning around that time ship traffic associated with the Acapulco to Manilla galleon route began to regularly pass through the Mariana Islands (Driver 1988, Barratt 2003, Driver 2005), creating potential pathways for the introduction of both *R. rattus* (likely introduced by Europeans in Acapulco [Musser and Carleton 2005:1486] and possibly in Manilla as well) and *R. tanezumi* (likely present in Manilla; Musser and Carleton 2005:1489). Possible evidence for *R. tanezumi* transport by Spanish galleons traveling between Manila and Acapulco is provided by the recent discovery of 2n = 42 karyotype rats in western Mexico (Alonso et al. 1982). Even with increased ship traffic associated with growing trade, the introduction risk in the Mariana Islands was probably low because of the common practice of anchoring ships offshore and approaching land in small, open boats (Atkinson 1985, Barratt 2003, Driver 2005), due to the rarity of suitable inshore anchorages and apprehension of native islanders. Introduction opportunities surely increased following the establishment of a permanent Spanish settlement on Guam in 1668, which resulted in increased movements of people and goods to and from the Mariana Islands (Driver 1988, Barratt 2003, Driver 2005). Opportunities for *R. rattus* and *R. tanezumi* introduction (or reintroduction) to the Mariana Islands could have only increased further over time, as administration by Spain, Germany, Japan, and the United States brought people, goods, and ships from multiple areas to the islands (Driver 1988, Rogers 1995, Bowers 2001, Barratt 2003, Driver 2005).

An additional *Rattus* species, *R. norvegicus*, was introduced to the Mariana Islands at some point following Spanish colonization. The native range of *R. norvegicus* includes the Hondo region of Japan, southeastern Siberia, and northern China, from which the species reached Europe by ca. 1700 (Musser and Carleton 2005:1478–1480). As noted by Atkinson (1985), *R. norvegicus* replaced *R. rattus* as the common *Rattus* species in American and European ports, and consequently on American and European ships, between ca. 1700 and 1830. Indeed, most documented *Rattus* introductions during this time period were *R. norvegicus* (Atkinson 1985). Alternatively, a direct introduction from Japan to the Mariana Islands seems possible, given the native range of *R. norvegicus*. The earliest known reference for *R. norvegicus* in the Mariana Islands comes from Saipan, where the species was found by the late 1800's (Kuroda 1938 cited by Wiles et al. 1990). In contrast, Enders (1949) suggested that *R. norvegicus* was first introduced to Saipan during Japanese occupation between 1914 and 1944. Marshall (1962b) found *R. norvegicus* on Saipan in 1944–1945, and the first documented occurrence of *R. norvegicus* on Guam was not until 1962 (Barbehenn 1974). On both Guam and Saipan, *R. norvegicus* had a limited distribution and low overall abundance during extensive sampling conducted in the late 1950's and early 1960's (Barbehenn 1974), lending some support to a relatively recent introduction of this species.

Two additional species, *Mus musculus* and *Suncus murinus*, have been introduced to the Mariana Islands. As with the *Rattus* species, there is considerable uncertainty regarding both the date and identity of *M. musculus* introductions in the Mariana Islands. The taxonomy of the polytypic species *M. musculus* has been shaped by both natural and human-mediated radiation away from a presumed origin in either the northern Indian subcontinent (Boursot et al. 1996, Din et al. 1996) or west-central Asia (Prager et al. 1998). Recent research distinguishes 5 subspecific groups (*M. m. musculus*, *M. m. domesticus*, *M. m. castaneus*, *M. m. bactrianus*, and *M. m. gentilulus*) based on genetic and morphological traits, although these subspecies freely hybridize when sympatric (Boursot et al. 1993, Prager et al. 1998, Musser and Carleton 2005:1400–1402). The first 3 subspecies are the most widespread, with *M. m. musculus* ranging from Eastern Europe through Northern Asia including Japan, *M. m. domesticus* occurring throughout Western Europe, the Mediterranean including Northern Africa, and Southwest Asia, and *M. m. castaneus* extending from Central Asia through Southeast Asia and Japan (Musser and Carleton 2005:1400–1401). The remaining subspecies have more restricted ranges, possibly due to geographic constraints, with *M. m. bactrianus* occurring in mountain valleys in Afghanistan and *M. m. gentilulus* occurring in Yemen in the Southern Arabian Peninsula, although there is some speculation that this subspecies may be found throughout the Persian Gulf and Eastern Africa (Musser and Carleton 2005:1401).

Of these, *M. m. domesticus* is generally recognized as the subspecies most commonly transported (and introduced) during European colonization of North and South America, Australia, and numerous islands (Musser and Carleton 2005:1401). Nonetheless, the location of the Mariana Islands in relation to the ranges of various *M. musculus* subspecies suggests multiple potential avenues for introduction of *M. m. musculus*, *M. m. domesticus*, or *M. m. castaneus* to the Mariana Islands from Europe (*M. m. musculus* or *M. m. domesticus*), Southeast Asia (*M. m. castaneus*), or Japan (*M. m. musculus* or *M. m. castaneus*). Further, the presence of *M. m. castaneus* in Southeast Asia presents the possibility of a prehistoric introduction to the Mariana Islands, although to date no evidence of such an introduction is available. Instead, the earliest known reference for *M. musculus* in the Mariana Islands is 1819, when a French expedition to Guam noted “prodigious” rat and mice populations (Freycinet 2003:88). Although interesting, this information is not highly informative as it provides no means for discriminating between subspecies. Further, there is a possibility that the French expedition might have incorrectly identified the small (in comparison to *R. rattus* or *R. norvegicus*) *R. exulans* as mice, especially if they were unfamiliar with this species.

Additional clues about the subspecific identity of *M. musculus* in the Mariana Islands may be provided by morphological traits, such as tail length, and degree of commensalism. In general, both *M. m. domesticus* and *M. m. castaneus* have tails longer than their head and body, whereas the tail of *M. m. musculus* is shorter than its head and body (Boursot et al. 1993), suggesting that the long-tailed *M. musculus* present in the Mariana Islands (see Table 10 in main body of Chapter 1) is *M. m. domesticus* or *M. m. castaneus* (or both). Of these subspecies, *M. m. domesticus* establishes both commensal and permanent outdoor populations in warm regions, whereas *M. m. castaneus* is strictly a human commensal in tropical climates (Boursot et al. 1993). It therefore seems likely that feral populations in the Mariana Islands are *M. m. domesticus*, but the subspecific identity of commensal populations remains unclear. It is notable that Prager et al. (1998) found *M. m. castaneus* on Tinian, although this identification was based on genetic analysis of a single specimen. Clearly, additional research is necessary to clarify the subspecific identity of *M. musculus* in the Mariana Islands.

S. murinus, the most recent and best documented small mammal introduction to the Mariana Islands, was first observed on Guam in 1953 (Peterson 1956), on Saipan in 1962 and Rota in 1966 (Barbehenn 1974), and on Tinian in 1974 (Owen 1974). A single *S. murinus* was reportedly observed on Guguan in 1984 (Eldredge 1988), although no additional observations have been made since that time. Peterson (1956) suggested that the *S. murinus* on Guam were introduced from the Philippines. It is likely that other introductions in the Mariana Islands originated from the Guam population, although direct introductions from the Philippines may have occurred.

Current Distribution of Introduced Small Mammals

A review of recent accounts of faunal distribution in the Mariana Islands (Stinson 1994, Vogt and Williams 2004, Wiles 2005) and research pertaining to ≥1 islands (e.g., Pratt and Lemke 1984, Wiles et al. 1990, Rice and Stinson 1992), in addition to sampling conducted by ASW during 2005–2007 (described in the main text of Chapter 1 and 2), were summarized to determine the current distribution of introduced small mammals in the Mariana Islands (Table A.1). Not surprisingly, introduced small mammal diversity seems to be greatest in the southern, human-inhabited islands of Guam, Rota, Saipan, and Tinian. It is notable that no recent evidence exists for the presence of *R. norvegicus* or *S. murinus* on Rota or *R. exulans* on Tinian (Table A.1). *R. norvegicus* has apparently never been documented on Rota, although it is unclear if the most suitable areas for this species, such as the seaport, have been sampled. *S. murinus* was observed on Rota in 1966 (observation by R.P. Owens reported in Barbehenn 1974), but has not been documented since. Notably, this highly conspicuous species was not observed during approximately 9 weeks spent on Rota by ASW in 2005–2006. *R. exulans* was captured on Tinian following World War II (Marshall 1962a), but has not been documented since. Additional targeted sampling for *R. norvegicus* on Rota and *R. exulans* on Tinian is recommended to clarify this uncertainty.

The small, isolated, sparsely populated northern islands seem to have low introduced small mammal diversity, with *R. exulans* apparently the only species on many islands (Table A.1). *R. rattus* has been reported from 2 northern islands, Agrihan and Pagan, and unidentified *Rattus* species have been reported from Farallon de Pajaros, Maug Islands, and Asuncion Island (Table A.1). The observations from Farallon de Pajaros and Asuncion Island (Pratt and Lemke 1984) mention the presence of tunnels, which implies *R. norvegicus*. Note, however, that *R. rattus* will excavate burrows when above ground cover is scarce, a situation that may be prevalent on Farallon de Pajaros (Eldredge 1983). Stinson's (1994) suggestion that *R. norvegicus* is present (noted as uncommon) on Farallon de Pajaros is probably referencing Pratt and Lemke (1984), although the ambiguous "DFW files" (interpreted as indicating a record on file with the CNMI Division of Fish and Wildlife) is the only reference provided. Observations from Maug Islands (Eldredge et al. 1977, Eldredge 1983, Pratt and Lemke 1984, Rice and Stinson 1992) consistently mention small rats, implying *R. exulans*, although Eldredge et al. (1977) suggested that the observations may have been *R. rattus*. One additional observation, made at Naftan Rock offshore from Aguijan, requires further clarification. Here, Lemke et al. (1985) observed burrows and chewed bird bones which the authors attributed to the presence of an unidentified *Rattus* species. As noted previously, these burrows suggest, but do not confirm, the presence of *R. norvegicus*.

It should be noted that the northern islands are rarely visited and have been subject to limited terrestrial scientific investigation, such that undiscovered introduced small mammal species could be present. Although the northern islands were largely abandoned by humans during much of the Spanish administration of the Mariana Islands, activities associated with copra production, including construction of several villages and accompanying infrastructure, during German (1899–1914) and Japanese (1914–1944) administration of the northern islands (Russell 1998, C. Kessler, personal communication) certainly provided opportunities for small mammal introductions. Additional scientific visits to the northern islands, with a focus on documenting the density and distribution of introduced species, would be extremely valuable for understanding the ecology of the Mariana Islands.

Little information pertaining to habitat-specific distributions is currently available. This is not surprising, as successful introduced species are often habitat generalists. On Guam, Baker (1946) noted that *M. musculus* and *R. exulans* were rarely found in undisturbed limestone forest, and that *R. exulans* (in contrast to *M. musculus* and *R. tanezumi* [called *R. mindanensis* by Baker]) was rarely found near human habitation. Both Johnson (1962) and Musser and Carleton (2005:1485) suggested that when both species were present, *R. tanezumi* (called *R. r. mansorius* by Johnson) largely excluded *R. rattus* such that this species was found on ships in harbors but only rarely on shore. More recently, Yosida et al. (1985) collected *R. tanezumi* from houses on Guam, suggesting that this species may move freely between commensal and wild habitats. On Guam, *S. murinus* was able to colonize the entire island by 1958 (only 5 years after the first documentation; Barbehenn 1962), and was found in all available habitats during widespread sampling conducted in the early 1960's (Barbehenn 1969, 1974). Similarly, *S. murinus* was found throughout Saipan within 18 months of first documentation on the island (Barbehenn 1974). In contrast, *R. norvegicus* was much slower to colonize new areas on Guam and Saipan (Barbehenn 1974), and may be more strictly commensal than other introduced small mammals in the Mariana Islands (Marshall 1962b, Wiles et al. 1990).

Introduced Small Mammal Density in the Mariana Islands

Early, qualitative accounts of introduced small mammals in the Mariana Islands are indicative of high density populations (Table A.2). The earliest known record of introduced small mammals in the Mariana Islands is from Rota in 1602 AD, when a Spanish priest, Fray Juan Pobre de Zamora, noted that rats were so numerous they destroyed half of the planted corn crop (Russell 2002). Rats were also quite numerous on Tinian in 1742, when a British expedition led by Lord Anson stopped at the island for provisions (Thomas 1971 cited by Wiles et al. 1990). As noted previously, Freycinet (2003:88) commented on "prodigious" rat and mice populations on Guam in 1819, "whose noxious tribes here constitute a veritable scourge for the husbandman and storekeeper alike." Beginning during Spanish administration of the Mariana Islands (1668–1899), significant effort was expended in reducing introduced small mammal populations (Rogers 1995). During German administration of the Mariana Islands (1899–1914), a 5 pfennig per rat bounty was offered to encourage residents to actively reduce populations (Bowers 2001). Japanese administrators of the Mariana Islands (1914–1944) also initiated programs aimed at reducing introduced small mammal populations (Bowers 2001). A slightly different approach was taken on Guam in 1919 when the American governor passed a law requiring all male residents to deliver 5 dead rats per month or be fined $0.25 (Rogers 1995). It seems, however, that neither rewards nor penalties led to a significant population reductions in the Mariana Islands. In 1947, rats continued to "overrun the islands" of Rota, Saipan, and Tinian, causing "excessive destruction of small chickens and crops," and forcing "farmers to abandon the planting of crops for which the rats have a preference" (Bowers 2001).

Following World War II, quantitative studies of introduced small mammals began to occur in the Mariana Islands (Table A.2). Taken as a whole, however, post-World War II introduced small mammal research is relatively limited in the Mariana Islands, especially outside of the populated islands of Guam, Rota, Saipan, and Tinian. Much of the research that has been conducted is unpublished and exists only in internal agency reports (e.g., U.S. Fish and Wildlife Service, CNMI Department of Fish and Wildlife), and many of these reports contain only observational data. While useful for documenting inter-island and possibly inter-habitat distributions, these data have little utility for investigating introduced small mammal density. Interpretation of much of the available non-observational data is complicated by inconsistent documentation of sampling methodology and results, the common reliance on non-rigorous sampling techniques, such as low sampling effort and convenience sampling (Anderson 2001, 2003), and the frequent reporting of indices of density (Table A.2). These issues, while understandable given the logistical constraints imposed on research activities by the isolation and rugged nature of the Northern Mariana Islands, severely limit the utility of available data. For example, variable snaptrapping capture rates (e.g., compare sampling events on Rota; Table A.2) might reflect variable density between sites or habitats, but could also result from fluctuating capture probability associated with any number of factors, including season, habitat, weather, or sampling methodology (grid vs. transect). It is essential to consider these confounding factors when evaluating and interpreting any historic research, including the research reviewed below.

On Guam, the first known quantitative study of introduced small mammals occurred in 1945, when Baker (1946) documented relatively high, but variable, densities of *M. musculus*, *R. exulans*, and *R. tanezumi* (called *R. mindanensis* by Baker) at Mount Santa Rosa in northeastern Guam (Table A.2). Island-wide sampling conducted during the late 1950's and early 1960's on Guam suggested that *S. murinus* density equaled or exceed the combined density of other introduced small mammals, and also suggested that *R. exulans* and *R. tanezumi* (called *R. r. mansorius* by Barbehenn) density remained relatively constant in comparison to Baker's (1946) data (Barbehenn 1962, 1969, 1974). Note that this sampling occurred only 5–10 years after the proposed introduction of S. murinus to Guam in 1953 (Peterson 1956), suggesting rapid colonization and population growth. In contrast, *M. musculus* density seemed to have declined dramatically between 1945 (Baker 1946) and the early 1960's (Barbehenn 1969, 1974; Table A.2). This decline may have resulted from predation by *S. murinus* (Barbehenn 1974) as well as other factors including predation by the introduced brown treesnake (*Boiga irregularis*; Savidge 1987).

More recent sampling on Guam suggests declining introduced small mammal populations, especially in forested areas. For example, sampling conducted by King et al. (unpublished manuscript) and Savidge (1986) during the 1980's and early 1990's documented low *M. musculus*, *R. exulans*, *R. rattus*/*R. tanezumi*, and *S. murinus* density in various forest habitats (Table A.2). In contrast, introduced small mammal populations remained relatively high in sampled grasslands (King et al., unpublished manuscript; Savidge 1986). Similarly, Gragg (2004) documented high *M. musculus* and *Rattus* species (*R. exulans* and *R. rattus* were not differentiated in this study) density at 4 grassland plots in southern Guam in 2002 and 2003 (Table A.2). In contrast to previous grassland sampling, however, *S. murinus* was captured infrequently (Gragg 2004; Table A.2). It is unclear whether this result indicates a recent decline in *S. murinus* density in grasslands or if Gragg's (2004) study site encompassed an area of low *S. murinus* density not representative of the more general situation in grasslands on Guam. It is possible that the low *S. murinus* density observed by Gragg (2004) was an artifact of an unknown trap bias, as Gragg (2004) employed mark-recapture livetrapping, whereas most earlier research on Guam involved snaptrapping. The apparent long-term decline in introduced small mammal populations on Guam, especially in forest areas, is generally attributed to brown treesnake predation (Savidge 1987, Fritts and Rodda 1998). The relative persistence of introduced small mammal populations in grassland areas may result from lower brown treesnake density (and presumably lower predation pressure) in this habitat (Savidge 1987, 1991; Rodda and Dean-Bradley 2001).

Sampling data from Rota, Saipan, and Tinian suggests high-density *R. rattus*/*R. tanezumi* populations in most sampled habitats, as well as high-density *S. murinus* populations on Saipan and Tinian. *S. murinus* may no longer be present on Rota (Table A.2; J. Esselstyn and R. Ulloa, personal communication). Available data also suggest that *M. musculus*, *R. exulans*, and *R. norvegicus* occur at low densities on these islands (Table A.2), or perhaps they are sparsely distributed and have not been adequately sampled to date. It is difficult to make strong inferences about introduced small mammal populations on the remaining islands, with the possible exception of Aguijan, which is the best studied of the non-inhabited islands. On Aguijan, the available data suggests that current *R. exulans* densities may be higher than they were immediately following World War II (Table A.2), although it is not possible to make comparisons between *R. exulans* density on Aguijan and other islands based on available data.

Literature Cited

Alonso, R.A., D.A.F. Villagomez, A. Sanchez, and J.M. Cantu. 1982. Asian karyotype (2n=42) in the black rat (*Rattus rattus*) in Mexico. Annales de Genetique 25:133–135.

Amidon, F.A. 2000. Habitat relationships and life history of the Rota bridled white-eye (*Zosterops rotensis*). M.S. Thesis, Virginia Polytechnic Institute and State University, Blacksburg, Virginia, USA.

Anderson, D.R. 2001. The need to get the basics right in wildlife field studies. Wildlife Society Bulletin 29:1294–1297.

Anderson, D.R. 2003. Response to Engeman: index values rarely constitute reliable information. Wildlife Society Bulletin 31:288–291.

Arriola, L.A., S. Vogt, and J.B. Cruz. 1999. Federal aid in wildlife restoration program: annual report fiscal year 1999. Report to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Athens, J.S. and J.V. Ward. 2004. Holocene vegetation, savanna origins and human settlement of Guam. Pages 15–30 *in* V. Attenbrow and R. Fullagar, eds. A Pacific odyssey: archaeology and anthropology in the Western Pacific. Papers in honour of Jim Specht. Records of the Australian Museum, Supplement 29. Sydney, Australia.

Atkinson, I.A.E. 1985. The spread of commensal species of *Rattus* to oceanic islands and their effects on island avifaunas. Pages 35–81 *in* P.J. Moors, editor. Conservation of island birds. International Council for Bird Preservation, Technical Publication No. 3, Cambridge, England.

Baker, R.H. 1946. A study of rodent populations on Guam, Mariana Islands. Ecological Monographs 16:393–408.

Barbehenn, K.R. 1962. The house shrew on Guam. Pages 247–256 *in* T.I. Storer, editor. Pacific island rat ecology: report of a study made on Ponape and adjacent islands 1955–1958. Bernice P. Bishop Museum Bulletin 225, Honolulu, Hawaii, USA.

Barbehenn, K.R. 1969. Responses of rodents and shrews to patterns of removal trapping. Pages 247–252 *in* D.W. Parrack, editor. Indian rodent symposium. Calcutta, India.

Barbehenn, K.R. 1974. Recent invasions of Micronesia by small mammals. Micronesica 10:41–50.

Barratt, G. 2003. The Chamorros of the Mariana Islands: early European records, 1521–1721. Occasional Historic Papers No. 10, CNMI Division of Historic Preservation, Saipan, Commonwealth of the Northern Mariana Islands.

Baverstock, P.R., M. Adams, L.R. Maxson, and T.H. Yosida. 1983. Genetic differentiation among karyotypic forms of the black rat, *Rattus rattus*. Genetics 105:969–983.

Bowers, N.M. 2001. Problems of resettlement on Saipan, Tinian and Rota Mariana Islands. Occassional Historical Papers Series No. 7. CNMI Division of Historic Preservation, Saipan, Commonwealth of the Northern Mariana Islands.

Boursot, P., J.-C. Auffray, J. Britton-Davidian, and F. Bonhomme. 1993. The evolution of house mice. Annual Review of Ecology and Systematics 24:119–152.

Boursot, P., W. Din, R. Anand, D. Darviche, B. Dod, F. Von Deimling, G.P. Talwar, and F. Bonhomme. 1996. Origin and radiation of the house mouse: mitochondrial DNA phylogeny. Journal of Evolutionary Biology 9:391–415.

Campbell, E.W. 1995. Report on Aguiguan Island reptile and rodent surveys conducted between 21–27 March 1995 and 30 May–4 June 1995. Report to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Cruz, J. 2002. Reduction of feral ungulates on Anatahan, Commonwealth of the Northern Mariana Islands. Report to U.S. Navy, Pacific Division, Pearl Harbor, Hawaii, USA.

Cruz, J., L. Arriola, N. Johnson, and G.Beauprez. 2000a. Wildlife and vegetation surveys: Agrihan 2000. Technical Report #8 to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Cruz, J., L. Arriola, N. Johnson, and G.Beauprez. 2000b. Wildlife and vegetation surveys: Aguiguan 2000. Technical Report #2 to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Cruz, J., L. Arriola, N. Johnson, and G.Beauprez. 2000c. Wildlife and vegetation surveys: Alamagan 2000. Technical Report #4 to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Cruz, J., L. Arriola, N. Johnson, and G.Beauprez. 2000d. Wildlife and vegetation surveys: Anatahan 2000. Technical Report #6 to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Cruz, J., L. Arriola, N. Johnson, and G.Beauprez. 2000e. Wildlife and vegetation surveys: Guguan 2000. Technical Report #3 to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Cruz, J., L. Arriola, N. Johnson, and G.Beauprez. 2000f. Wildlife and vegetation surveys: Pagan 2000. Technical Report #7 to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Cruz, J., L. Arriola, N. Johnson, and G.Beauprez. 2000g. Wildlife and vegetation surveys: Sarigan 2000. Technical Report #5 to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Cruz, J., L. Williams, N. Hawley, J. Esselstyn, C. Kessler, and O. Bourquin. 2003. Wildlife and vegetation surveys prior to feral animal control Anatahan 2002. Technical Report #10 to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Davis, C.J. 1954. Report of the Davis expedition to Agiguan July–August, 1954. Hawaii Board of Commissioners of Agriculture and Forestry, Hilo, Hawaii, USA.

Din, W, R. Anand, P. Boursot, D. Darviche, B. Dod, E. Jouvin-Marche, A. Orth, G.P. Talwar, P.-A. Cazenave, and F. Bonhomme. 1996. Origin and radiation of the house mouse: clues from nuclear genes. Journal of Evolutionary Biology 9:519–539.

Downs, T. 1948. Amphibians and reptiles of Tinian Island. Transactions of the Kansas Academy of Science 51:112–116.

Driver, M.G. 1988. Cross, sword, and silver: the nascent Spanish colony in the Mariana Islands. Pacific Studies 11:21–51.

Driver, M.G. 2005. The Spanish governors of the Mariana Islands; notes on their activities and the saga of the Palacio, their residence and the seat of colonial government in Agańa, Guam. Micronesian Area Research Center Educational Series No. 27, Mangilao, Guam.

Eldredge, L.G. 1984. Aguijan revisited. Report to the University of Guam Marine Laboratory, Mangilao, Guam.

Eldredge, L.G. 1988. Case studies of the impacts of introduced animal species on renewable resources in the U.S.-affiliated Pacific islands. Imamura, Carolyn K. and Towle, E. OTA commissioned papers, integrated renewable resource management for U.S. insular areas ("Islands Study" 1987), Vol. 1. A.26–A.46. Washington, D.C., US Congress Office of Technology Assessment.

Eldredge, L.G. 1983. Summary of environmental and fishing information on Guam and the Commonwealth of the Northern Mariana Islands: historical background, description of the islands, and review of the climate, oceanography, and submarine topography. U.S. Deptartment of Commerce, National Oceanic and Atmospheric Administration, National Marine Fish Service, Southwest Fisheries Center.

Eldredge, L.G., R.T. Tsuda, P.H. Moore, M.I. Chernin, and S. Neudecker. 1977. A natural history of Maug, Northern Mariana Islands. University of Guam Marine Laboratory Technical Report 43:1–87.

Enders, R.K. 1949. A field study of the rats in the Marianas and Palaus. Scientific Investigations in Micronesia–Pacific Science Board, Report No. 3.

Esselstyn, J., J.B. Cruz, L.L. Williams, and N. Hawley. 2003. Wildlife and vegetation surveys: Aguiguan 2002. Technical Report #9 to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Flannery, T.F. 1995. Mammals of the South-West Pacific & Moluccan Islands. Comstock/Cornell University Press, Ithaca, New York, USA.

Freycinet, L.C.d. 2003. An account of the Corvette L' Uranie's sojourn at the Mariana Islands, 1819; supplemented with the Journal of Rose de Freycinet. Occasional Historic Papers No. 13, CNMI Division of Historic Preservation, Saipan, Commonwealth of the Northern Mariana Islands.

Fritts, T.H., and G.H. Rodda. 1988. Brown tree snakes, *Boiga irregularis*, at the PCON-M223 (ROTHR) construction site on Guam. Report to U.S. Fish and Wildlife Service, Honolulu, Hawaii, USA.

Fritts, T.H., and G.H. Rodda. 1998. The role of introduced species in the degradation of island ecosystems: a case history of Guam. Annual Review of Ecology and Systematics 29:113–140.

Glass, P.O., and D.T. Aldan. 1987. Guguan field trip report, 27 May–4 June 1987. Report to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Gragg, J.E. 2004. Rodent reduction for enhanced control of brown treesnakes (*Boiga irregularis*). M.S. Thesis, Colorado State University, Fort Collins, Colorado, USA.

Gragg, J.E., J.A. Savidge, G.H. Rodda, G.C. White, A.R. Ellingson, and K. Dean-Bradley. In Prep. Rodenticide effects on abundance and survival of rodents on Guam.

Johnson, D.H. 1962. Rodents and other Micronesian animals collected. Pages 21–38 *in* T.I. Storer, editor. Pacific island rat ecology: report of a study made on Ponape and adjacent islands 1955–1958. Bernice P. Bishop Museum Bulletin 225, Honolulu, Hawaii, USA.

Kessler, C.C. 2002. Eradication of feral goats and pigs and consequences for other biota on Sarigan Island, Commonwealth of the Northern Mariana Islands. Pages 132–140 *in* C.R. Veitch and M.N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, U.K.

King, K.A., C.E. Grue, and C.M. Fecko. Recent changes in the distribution and abundance of the musk shrew (*Suncus murinus*) and other small mammals on Guam. Unpublished manuscript.

Kosaka, E., T.O. Lemke, T.K. Pratt, D. Aldan, and C. Falig. 1983. Agiguan field trip report, 11–14 July 1983. Report to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Kuroda, N. 1938. A list of the Japanese mammals. Privately printed, Tokyo, Japan.

Kuroda, N. 1939. Distribution of mammals in the Japanese Empire. Journal of Mammalogy 20:37–50.

Lemke, T.O., T.K. Pratt, C.M. Falig, and D. Aldan. 1985. Aguijan (Goat Island) field trip, January 30–February 3 1985. Report to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Lusk, M.R., P. Bruner, and C. Kessler. 2000. The avifauna of Farallon de Medinilla, Mariana Islands. Journal of Field Ornithology 71:22–33.

Marshall, J.T., Jr. 1962a. Geographic distribution and color phases of Micronesian rodents. Pages 39–44 *in* T.I. Storer, editor. Pacific island rat ecology: report of a study made on Ponape and adjacent islands 1955–1958. Bernice P. Bishop Museum Bulletin 225, Honolulu, Hawaii, USA.

Marshall, J.T., Jr. 1962b. Norway rat. Pages 239–240 *in* T.I. Storer, editor. Pacific island rat ecology: report of a study made on Ponape and adjacent islands 1955–1958. Bernice P. Bishop Museum Bulletin 225, Honolulu, Hawaii, USA.

Matisoo-Smith, E., and J.S. Allen. 2001. Name that rat: molecular and morphological identification of Pacific rodent remains. International Journal of Osteoarchaeology 11:34–42.

Matisoo-Smith, E., and J.H. Robins. 2004. Origins and dispersals of Pacific peoples: evidence from mtDNA phylogenies of the Pacific rat. Proceedings of the National Academy of Sciences 101:9167–9172.

Morton, J.M, S. Plentovich, and T. Sharp. 1999. Reproduction and juvenile dispersal of Mariana crows (*Corvus kubaryi*) on Rota 1996-1999. Draft report to U.S. Fish and Wildlife Service, Honolulu, HI, USA.

Morton, J.M., and T. Sharp. 1997. Reproduction and juvenile dispersal of Mariana crows (*Corvus kubaryi*) on Rota. Progress report to U.S. Fish and Wildlife Service, Honolulu, HI, USA.

Musser, G.G., and M.D. Carleton. 2005. Family Muridae. Pages 1189–1531 *in* D.E. Wilson, and D.M. Reeder, editors. Mammal species of the world: a taxonomic and geographic reference. Third Edition. Smithsonian Institute Press, Washington, D.C., USA.

Nelson, L., Jr., and F.W. Clark. 1973. Correction for sprung traps in catch/effort calculations of trapping results. Journal of Mammalogy 54:295–298.

Nowak, R.M. 1999. Walker's mammals of the world. Volume 2. Sixth Edition. Johns Hopkins University Press, Baltimore, Maryland, USA.

Owen, R.P. 1974. Environmental impact study on the terrestrial fauna and flora of Tinian with respect to the proposed establishment of a U.S. military base on that island. Report to Trust Territory of the Pacific Islands Biological Laboratory, Koror, Palau.

Peterson, G.D., Jr. 1954. Report on progress of carnivorous snail experiment on Agiguan, Marianas Islands. Invertebrate Consultants Commission for the Pacific, Pacific Science Board, Natural Resources Council.

Peterson, G.D., Jr. 1956. *Suncus murinus*, a recent introduction to Guam. Journal of Mammalogy 37:278–279.

Prager, E.M., C. Orrego, and R.D. Sage. 1998. Genetic variation and phylogeography of Central Asian and other house mice, including a major new mitochondrial lineage in Yemen. Genetics 150:835–861.

Pratt, T.K., and T.O. Lemke. 1984. Wildlife field trip report from the Townsend Cromwell Raioma cruise, 18 February–5 March, 1984. Report to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Rainbird, P. 1994. Prehistory in the northwest tropical Pacific: the Caroline, Mariana, and Marshall Islands. Journal of World Prehistory 8:293–349.

Reichel, J.D., C. Falig, J. Barcinas, and H. Cabrera. 1988a. Field trip report, 6–9 June 1988. Report to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Reichel, J.D., P.O. Glass, J.S. Cepeda, and J.F. Reubenog. 1988b. Field trip report, Anatahan, 27–29 Sep 1988. Report to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Rice, C.G. and D.W. Stinson. 1992. Field trip report: Chiba Institute Northern islands trip, 11–20 May, 24 May–9 June. Report to CNMI Division of Fish and Wildlife, Saipan, Commonwealth of the Northern Mariana Islands.

Roberts, M. 1991. Origin, dispersal routes, and geographic distribution of *Rattus exulans*, with special reference to New Zealand. Pacific Science 45:123–130.

Robins, J.H., M. Hingston, E. Matisoo-Smith, and H.A. Ross. 2007. Identifying *Rattus* species using mitochondrial DNA. Molecular Ecology Notes 7:717–729.

Rodda, G.H., and K. Dean-Bradley. 2001. Inventory of the reptiles of the War in the Pacific National Historic Park, Guam. Report to USGS, Fort Collins, Colorado, USA.

Rogers, R.F. 1995. Destiny's landfall: a history of Guam. University of Hawaii Press, Honolulu, Hawaii, USA.

Russell, S. 1998. Gani revisited: a historical overview of the Mariana Archipelago’s northern islands. Pacific Studies 21:83–105.

Russell, S. 2002. The island of Rota: an archaeological and historical overview. Occasional Historic Papers No. 11, CNMI Division of Historic Preservation, Saipan, Commonwealth of the Northern Mariana Islands.

Savidge, J.A. 1986. The role of disease and predation in the decline of Guam's avifauna. Dissertation, University of Illinois, Urbana-Champaign, Illinois, USA.

Savidge, J.A. 1987. Extinction of an island forest avifauna by an introduced snake. Ecology 68:660–668.

Savidge, J.A. 1991. Population characteristics of the introduced brown tree snake (*Boiga irregularis*) on Guam. Biotropica 23:294–300.

Schwabe, H.W. 1979. Vergleichend-allometrische untersuchungen an den schädeln Europäischer und Asiatischer hausratten (*Rattus rattus* L.). Zeitschrift fur Säugetierkunde 44:354–360.

Steadman, D.W. 1999. The prehistory of vertebrates, especially birds, on Tinian, Aguiguan, and Rota, Northern Mariana Islands. Micronesica 31:319–345.

Stinson, D.W. 1994. Birds and mammals recorded from the Mariana Islands. Natural History Research Special Issue 1:333–344.

Tate, G.H.H. 1935. Rodents of the genera *Rattus* and *Mus* from the Pacific islands. Bulletin of the American Museum of Natural History 68:145–178.

Thomas, P. 1971. Journal of a voyage to the south-seas. De Capo Press, New York, New York, USA.

U.S. Fish and Wildlife Service. 1986. Fish and wildlife resources of the Haputo Ecological Reserve Area. Report to U.S. Fish and Wildlife Service, Honolulu, Hawaii, USA.

Vogt, S.R., and L.L. Williams. 2004. Common flora and fauna of the Mariana Islands. Privately printed, Saipan, Northern Mariana Islands.

White, J.P., and T. Flannery. Murids from Fais Island. Unpublished manuscript.

Wiles, G.J. 2005. A checklist of the birds and mammals of Micronesia. Micronesica 38:141–189.

Wiles, G.J., A.B. Amerson, Jr., and R.E. Beck, Jr. 1990. The mammals of Tinian, Mariana Islands. Micronesica 23:167–180.

Yosida, T.H., H. Kato, K. Tsuchiya, and K. Moriwaki. 1971. Karyotypes and serum transferring patterns of hybrids between Asian and Oceanian black rats, *Rattus rattus*. Chromosoma 34:40–50.

Yosida, T.H., T. Udagawa, M. Ishibashi, K. Moriwaki, T. Yabe, and T. Hamada. 1985. Studies on the karyotypes of the black rats distributed in the Pacific and south Pacific islands, with special regard to the border line of the Asian and Oceanian type of black rats in the Pacific Ocean. Proceedings of the Japanese Academy, Series B 61:71–74.

Table A.1. Distribution of introduced small mammal species in the Mariana Islands, where species observations are indicated with an "O," anecdotal references (without accompanying evidence) are indicated with a "H," and captures are indicated with a "C." Brackets indicate uncertain species identification. Islands are listed in order from north to south.

<table>
<tr><th>Island</th><th><i>Mus musculus</i></th><th><i>Rattus exulans</i></th><th><i>Rattus norvegicus</i></th><th><i>Rattus rattus/ R. tanezumi</i></th><th><i>Suncus murinus</i></th></tr>
<tr><td rowspan="2">Farallon de Pajaros[a]</td><td rowspan="2"></td><td colspan="3">|----- --- O[d] --- -----|</td><td rowspan="2"></td></tr>
<tr><td></td><td>H[d]</td><td></td></tr>
<tr><td>Maug Islands[a]</td><td></td><td colspan="3">|----- --- O[d] --- -----|</td><td></td></tr>
<tr><td>Asuncion Island[a]</td><td></td><td colspan="3">|----- --- O[d] --- -----|</td><td></td></tr>
<tr><td>Agrihan[b]</td><td></td><td></td><td></td><td>C[d]</td><td></td></tr>
<tr><td>Pagan[b]</td><td></td><td></td><td></td><td>C[d]</td><td></td></tr>
<tr><td>Alamagan[b]</td><td></td><td>C[d]</td><td></td><td></td><td></td></tr>
<tr><td>Guguan[a]</td><td></td><td>C[d]</td><td></td><td></td><td>H[d,f]</td></tr>
<tr><td>Sarigan[b]</td><td></td><td>C[d]</td><td></td><td></td><td></td></tr>
<tr><td>Anatahan[b]</td><td></td><td>C[d]</td><td></td><td></td><td></td></tr>
<tr><td>Farallon de Medinilla[a]</td><td></td><td>H[d]</td><td></td><td></td><td></td></tr>
<tr><td>Saipan[c]</td><td>C[d,e]</td><td>C[d,e]</td><td>C[d,e]</td><td>C[d,e]</td><td>C[d,e]</td></tr>
<tr><td>Tinian[c]</td><td>C[d,e]</td><td>C[d, f]</td><td>C[e]</td><td>C[d,e]</td><td>C[d,e]</td></tr>
<tr><td rowspan="2">Aguijan[a]</td><td rowspan="2"></td><td>C[d]</td><td></td><td></td><td rowspan="2"></td></tr>
<tr><td colspan="3">|----- ---H[d,f,g]--- -----|</td></tr>
<tr><td>Rota[c]</td><td>C[d,e]</td><td>C[d,e]</td><td></td><td>C[d,e]</td><td>H[d,f]</td></tr>
<tr><td>Guam[c]</td><td>C[d,e]</td><td>C[d,e]</td><td>C[d,e]</td><td>C[d,e]</td><td>C[d,e]</td></tr>
</table>

[a] Uninhabitated in modern (post-WWII) era: Farallon de Pajaros, Maug Islands, Asuncion Island, and Guguan designated as nature preserves by the constitution of the Commonwealth of the Northern Mariana Islands. Farallon de Medinilla leased as a bombing range by U.S. military.

[b] Intermittent human settlement in modern (post-WWII) era: Agrihan and Alamagan currently have 5–10 residents each, Pagan recently recolonized by 2 families (C. Kessler, personal communication).

[c] Permanent human populations in modern (post-WWII) era.

[d] See island-specific references in Table A.2.

[e] Captured by ASW during 2005–2007.

[f] No recent observations, status unknown

[g] Evidence for *R. norvegicus* pertains to Naftan Rock, located south of Aguijan.

Table A.2. Summary of known introduced small mammal records in the Mariana Islands. Islands are listed in alphabetical order. Unless otherwise indicated, all information taken directly from cited references; information modified or calculated by ASW is indicated in bold. Blanks in table indicate that information was either not present or not interpretable in the cited reference. Brackets indicate uncertain species identification or unclear reference between observations/captures and a specific sampling unit. In the Sampling Date column, "Occasions" indicate the specified duration of sampling. In the Trap Type column, "Placement" indicates the count and placement of traps for a single sampling occasion, whereas "Effort" indicates the total sampling effort in trap nights (TN). Sampling results indicate the number of individuals captured, the number of captures/100 TN or 100 corrected TN (CTN; where the correction is for traps closed without a capture, following the method described by Nelson and Clark 1973), or the number of individuals/ha. In all cases, these density estimates represent nominal densities, where the number of captured or estimated individuals is divided by the area of the sample unit (i.e., no attempt to estimate the effective trapping area).

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Agrihan										
Pratt and Lemke 1984	Feb 22–23, 1984	mid-elevation		incidental observations				|– – – – –	observed large rat in tree fern	
Cruz et al. 2000a	Aug 11–14, 2000 (4)	introduced forest (southwest coast, near anchorage)	peanut butter	225 m transect (25 m spacing)	Victor rat snap (10 ground, 10 tree; 80 TN)				5 captures **6.3/100 TN** 8.3/100 CTN	
Aguijan										
Enders 1949	1949						|– – – – –	comments that rats are hard to find	– – – – –|	
Owen 1952 (cited in Davis 1954 and Eldredge 1984)	1950 or possibly 1952						|– – – – –	reported rats extremely scarce or absent	– – – – –|	
Peterson 1954				visual searches			|– – – – –	no evidence of rats, despite extensive searching	– – – – –|	
Davis 1954	Jul 21–Aug 11, 1954	"various habitats"	coconut, bacon, and bread	trapping and visual searches	"Japanese-type"		|– – – – –	observed 2 rats	– – – – –|	
Kosaka et al. 1983	Jul 11–14, 1983			incidental observations			frequent observations			
Lemke et al. 1985	Jan 30–Feb 3, 1985	Naftan Rock		incidental observations			|– – – – –	observed sign (burrows, chewed bird bones) of unknown rat	– – – – –|	

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Aguijan continued										
Reichel et al. 1988a	Jun 6–9, 1988	Guano Cave, other areas		incidental observations and specimen collection			observed single *R. exulans*; collected other rats for ID			
Stinson 1994				review of published and unpublished reports			common			
Campbell 1995 [a]	May 31–Jun 3, 1995 (4, 3)	native limestone forest	peanut butter + rolled oats	375 m transect (25 m spacing)	snap (16 ground, 16 tree; 112 TN)		10.7/100 CTN			
	Jun 2–Jun 4, 1995 (3)	savanna	peanut butter + rolled oats	325 m transect (25 m spacing)	snap (14 ground; 42 TN)		5.3/100 CTN			
Cruz et al. 2000b	Apr 2–5, 2000 (3)	introduced forest (upper plateau near camp)	peanut butter	600 m transect (25 m spacing)	snap (25 ground, 25 tree; 150 TN)		16 captures **10.7/100 TN** 12.5/100 CTN			
		native limestone forest (near or on transect 4)	peanut butter	600 m transect (25 m spacing)	snap (25 ground, 25 tree; 150 TN)		18 captures **12.0/100 TN** 16.5/100 CTN			
		savannah (upper plateau near camp)	peanut butter	600 m transect (25 m spacing)	snap (25 ground; 75 TN)		5 captures **6.7/100 TN** 9.8/100 CTN			
Esselstyn et al. 2003	Mar 14–21, 2002 (3)	*Leucaena* forest (upper plateau, south of camp)	peanut butter	275 m transect (25 m spacing)	Victor snap (12 ground, 12 tree; 72 TN)		6 captures **8.3/100 TN**			
		native limestone forest (between Transects 2 and 4)	peanut butter	275 m transect (25 m spacing)	Victor snap (12 ground, 12 tree; 72 TN)		1 capture **1.4/100 TN**			
		Cocos stand (northeast of camp)					|– – – – –	observed several unknown rats	– – – – –|	
Alamagan										
Cruz et al. 2000c [b]	Jun 11–15, 2000 (3)	mixed secondary forest (northwest slope)	peanut butter	600 m transect (25 m spacing)	Victor rat snap (25 tree; 75 TN)					
		mixed *Cocos* forest (near camp)					3 captures			

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Anatahan										
Reichel et al. 1988b	Sep 27–29, 1988			incidental observations			observed small rats			
Vogt (unpublished data)	Jul 1999 (3)	degraded native forest		700 m transect (25 m spacing)	Victor snap (29 ground; 87 TN)		2 captures **2.3/100 TN** **3.3/100 CTN**			
Cruz et al. 2000d	Jul 10–12, 2000 (3)	native forest (northwestern and southern coast)	peanut butter	675 m transect + 1000 m transect (25 m spacing)	Victor rat snap (28 ground/tree mix; 84 TN + 41 ground/tree mix; 123 TN)		12 captures **5.8/100 TN** 6.8/100 CTN			
Cruz 2002, Cruz et al. 2003 [c]	Apr 25–May 2, 2002	coastal forest dominated by *Barringtonia*	peanut butter	275 m transect (25 m spacing)	Victor snap (12 ground, 12 tree)		— | |			
		coastal scrub dominated by *Cocos*	peanut butter	275 m transect (25 m spacing)	Victor snap (12 ground, 12 tree)		2 captures (tentative identification)			
		upland forest dominated by *Hibiscus*	peanut butter	275 m transect (25 m spacing)	Victor snap (12 ground, 12 tree)		| | —			
Asuncion Island										
Pratt and Lemke 1984	Feb 28–29, 1984			incidental observations			| – – – – –	observed single rat and rat sign (tunnels)	– – – – – |	
Farallon de Medinilla										
Lusk et al. 2000 [d]	Nov 4, 1996	grass, shrubs, and isolated short trees					present (no reference)			
Farallon de Pajaros										
Pratt and Lemke 1984	Feb 27, 1984			incidental observations				unidentified large rat collected; observed extensive rat sign (scat and tunnels)	– – – – – |	
Stinson 1994				review of published and unpublished reports				uncommon		

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Guam										
Freycinet 2003	1819			incidental observations		|– – – – –	observed “prodigious” rats and mice	– – – – – – – –	– – – – –|	
Baker 1946 [e]	May 8–21, 1945 (14)	grassland, modified forest (1 mile west of Mt. Santa Rosa)	rolled oats + ground coconut	~90 × 115 m site (~9 m spacing; ~1.2 ha)	Sherman live (90 ground; 1260 TN)	20 captures 8.3/ha	7 captures 6.5/ha		37 captures 30.0/ha	
	Jun 21–30, 1945 (10)	grassland, modified forest (1 mile west of Mt. Santa Rosa)	rolled oats + ground coconut	~90 × 115 m site (~9 m spacing; ~1.2 ha)	Sherman live (90 ground; 900 TN)	17 captures 10.6/ha	7 captures 5.7/ha		22 captures 18.3/ha	
	Jul 31–Aug 9, 1945 (10)	grassland, modified forest (1 mile west of Mt. Santa Rosa)	rolled oats + ground coconut	~90 × 115 m site (~9 m spacing; ~1.2 ha)	Sherman live (90 ground; 900 TN)	19 captures 13.4/ha	4 captures 3.1/ha		13 captures 12.7/ha	
	Sep 28–Oct 6, 1945 (9)	grassland, modified forest (1 mile west of Mt. Santa Rosa)	rolled oats + ground coconut	~90 × 115 m site (~9 m spacing; ~1.25 ha)	Sherman live (90 ground; 810 TN)	34 captures 25.8/ha	18 captures 15.0/ha		12 captures 10.9/ha	
	Oct 19–24, 1945 (6)	grassland, modified forest (1 mile west of Mt. Santa Rosa)	rolled oats + ground coconut	~90 × 115 m site (~9 m spacing; ~1.2 ha)	Sherman live (90 ground; 540 TN) + snap (90 ground; 540 TN)	25 captures 20.7/ha	26 captures 21.7/ha		30 captures 24.0/ha	
Marshall 1962a	~1945–1960			review of specimens deposited at U.S. National Museum		collected	collected		collected	collected
Barbehenn 1962 [f]	Jan 1–May 20, 1958 (4 nights/transect)	grassy/brushy areas near human-use areas (island-wide)		46 transects (3–3.7 m spacing)	wooden-base snap (~ 100 ground; 21876 TN)	|– – – – –	– – – – – – – –	1613 captures **7.4/100 TN**	– – – – –|	1609 captures **7.4/100 TN**
Barbehenn 1969, 1974 [g]	May 1962–May 1964 (≥ 4 nights/grid)	grassland, shrubland, *Leucaena* forest (island-wide)	fresh coconut	twenty-three 8 × 8 grids (~15 m spacing; ~1.1 ha each)	mouse snap (64–128 ground), Museum Special snap (64 ground), and rat snap (128 ground; 5888–7360 TN)	115 captures **~4.5/ha (average across grids)**	340 captures **~13.4/ha (average across grids)**	— | | |	513 captures **~20.3/ha (average across grids)**	704 captures **~27.8/ha (average across grids)**
		grassland, shrubland, *Leucaena* forest (island-wide)	fresh coconut	six 8 × 8 grids (~30 m spacing; ~4.4 ha each)	mouse snap (64–128 ground), Museum Special snap (64 ground), and rat snap (128 ground; 1536–1920 TN)	118 captures **~4.5/ha (average across grids)**	387 captures **~14.7/ha (average across grids)**	| 6 captures | | |	337 captures **~12.8/ha (average across grids)**	422 captures **~16.0/ha (average across grids)**
	(usually 4 nights/transect)	grass, scrub, and forest edge near human-use areas (island-wide)	fresh coconut	42 transects (~3.7 m spacing)	wooden-base rat snap (~100 ground; ~16800 TN)	121 captures **~0.7/100 TN**	287 captures **~1.7/100 TN**	| | | —	1324 captures **~7.9/100 TN**	2805 captures **~16.7/100 TN**

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Guam continued										
King et al. (unpublished manuscript)[h]	Jun 4–Jul 4, 1981 (1 night/transect)	limestone forest (Ritidian, Tarague)	canned dog food or peanut butter + oatmeal (alternate traps)	three 1500 m transects (~8 m spacing)	Victor mouse snap (200 ground; 600 TN)	2 captures 0.3/100 TN	2 captures 0.3/100 TN			19 captures 3.2/100 TN
		second-growth forest (Andersen NE, Andersen NW)	canned dog food or peanut butter + oatmeal (alternate traps)	three 1500 m transects (~8 m spacing)	Victor mouse snap (200 ground; 600 TN)	6 captures 1.0/100 TN	7 captures 1.2/100 TN			16 captures 2.7/100 TN
		mixed forest (Dededo, Ipapao, Andersen Marbo Annex)	canned dog food or peanut butter + oatmeal (alternate traps)	five 1500 m transects (~8 m spacing)	Victor mouse snap (200 ground; 1000 TN)		2 captures 0.2/100 TN			7 captures 0.7/100 TN
		ravine forest (Chaot River, High Road, Almagosa Springs)	canned dog food or peanut butter + oatmeal (alternate traps)	three 1500 m transects (~8 m spacing)	Victor mouse snap (200 ground; 600 TN)		1 capture 0.2/100 TN			1 capture 0.2/100 TN
		savannah (Mt. Tenjo, Sigua Falls, Roberto's, NASA Tracking Station)	canned dog food or peanut butter + oatmeal (alternate traps)	four 1500 m transects (~8 m spacing)	Victor mouse snap (200 ground; 800 TN)	5 captures 0.6/100 TN				15 captures 1.9/100 TN
		swamp (Agana Swamp)	canned dog food or peanut butter + oatmeal (alternate traps)	1500 m transect (~8 m spacing)	Victor mouse snap (200 ground; 200 TN)					1 capture 0.5/100 TN
		urban (Barrigada)	canned dog food or peanut butter + oatmeal (alternate traps)	1500 m transect (~8 m spacing)	Victor mouse snap (200 ground; 200 TN)		1 capture 0.5/100 TN			8 captures 4.0/100 TN

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Guam continued										
Savidge 1986 [i]	May 1984 (5)	grassland, modified forest (Baker (1946) site)	coconut	10 × 10 grid (10 m spacing; 0.81 ha)	mouse snap (150 ground; 750 TN) + rat snap (100 ground; 500 TN)				2 captures **0.2/100 TN** **2.5/ha**	
	Feb 1985 (5)	grassland, shrubland, *Leucaena* forest (Two Lover's Point)	coconut + peanut butter (mouse snap); fresh coconut (rat snap)	8 × 8 grid (15 m spacing; 1.1 ha)	mouse snap (80 ground; 400 TN) + rat snap (128 ground; 640 TN)				2 captures **0.2/100 TN** **1.8/ha**	
	Feb–Mar 1985 (5)	grassland, shrubland, *Leucaena* forest (Northwest Field)	coconut + peanut butter (mouse snap); fresh coconut (rat snap)	8 × 8 grid (15 m spacing; 1.1 ha)	mouse snap (80 ground; 400 TN) + rat snap (128 ground; 640 TN)					
	Mar 1985 (5)	grassland, shrubland, *Leucaena* forest (Anderson South)	coconut + peanut butter (mouse snap); fresh coconut (rat snap)	8 × 8 grid (15 m spacing; 1.1 ha)	mouse snap (80 ground; 400 TN) + rat snap (128 ground; 640 TN)	11 captures **1.1/100 TN** **10.0/ha**				
	Apr 1985 (5)	savannah (NASA Tracking Station)	coconut + peanut butter (mouse snap); fresh coconut (rat snap)	8 × 8 grid (15 m spacing; 1.1 ha)	mouse snap (64 ground; 320 TN) + rat snap (128 ground; 640 TN)	96 captures **10.0/100 TN** **87.3/ha**			40 captures **4.2/100 TN** **36.4/ha**	21 captures **2.2/100 TN** **19.1/ha**
U.S. Fish and Wildlife Service 1986 [j]	Apr–May 1986	coastal strand (Haputo Beach)	fresh coconut	1 transect	mouse snap (21 ground) + rat snap (21 ground)					
	Apr–May 1986	secondary forest (Haputo Road)	fresh coconut	1 transect (~ 10 m spacing)	mouse snap (21 ground) + rat snap (21 ground)					
Fritts and Rodda 1988 [k]	Apr 29–May 8 1988 (3)	mixed forest (Northwest Field)	coconut or peanut butter + oats	490 m transect (10 m spacing)	Victor snap (50 ground; 150 TN)		3 captures **1.3/100 TN**			
		mixed forest (NCTAMS, near Haputo Beach trailhead)	coconut or peanut butter + oats	480 m transect (10 m spacing)	Victor snap (49 ground; 147 TN)					

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Guam continued										
King et al. (unpublished manuscript)[h]	Jun 3–27, 1994 (1 night/transect)	coastal strand (Ritidian, Tarague)	canned dog food or peanut butter + oatmeal (alternate traps)	two 1500 m transects (~8 m spacing)	Victor mouse snap (200 ground; 400 TN)					
			fresh coconut	two 750 m transects (~8 m spacing)	Victor rat snap (100 ground; 200 TN)				2 captures 1.0/100 TN	
		limestone forest (Ritidian, Tarague)	canned dog food or peanut butter + oatmeal (alternate traps)	five 1500 m transects (~8 m spacing)	Victor mouse snap (200 ground; 1000 TN)					
			fresh coconut	two 750 m transects (~8 m spacing)	Victor rat snap (100 ground; 200 TN)				1 capture 0.5/100 TN	
		second-growth forest (Andersen NE, Andersen NW)	canned dog food or peanut butter + oatmeal (alternate traps)	three 1500 m transects (~8 m spacing)	Victor mouse snap (200 ground; 600 TN)	2 captures 0.3/100 TN				
			fresh coconut	two 750 m transects (~8 m spacing)	Victor rat snap (100 ground; 200 TN)				3 captures 1.5/100 TN	
		mixed forest (NCTAMS, Andersen Marbo Annex)	canned dog food or peanut butter + oatmeal (alternate traps)	two 1500 m transects (~8 m spacing)	Victor mouse snap (200 ground; 400 TN)					
			fresh coconut	two 750 m transects (~8 m spacing)	Victor rat snap (100 ground; 200 TN)					
		ravine forest (Chaot River, High Road, Almagosa Springs)	canned dog food or peanut butter + oatmeal (alternate traps)	three 1500 m transects (~8 m spacing)	Victor mouse snap (200 ground; 600 TN)					
			fresh coconut	three 750 m transects (~8 m spacing)	Victor rat snap (100 ground; 300 TN)				1 capture 0.3/100 TN	

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Guam continued										
King et al. (unpublished manuscript) [h] continued	Jun 3–27, 1994 (1 night/transect)	savannah (Mt. Tenjo, Sigua Falls, Roberto's, NASA Tracking Station)	canned dog food or peanut butter + oatmeal (alternate traps)	four 1500 m transects (~8 m spacing)	Victor mouse snap (200 ground; 800 TN)	9 captures 1.1/100 TN	1 capture 0.1/100 TN			10 captures 1.3/100 TN
			fresh coconut	four 750 m transects (~8 m spacing)	Victor rat snap (100 ground; 400 TN)		4 captures 1.0/100 TN		18 captures 4.5/100 TN	14 captures 3.5/100 TN
		urban (Naval Air Station Barracks, Naval Station Barracks)	canned dog food or peanut butter + oatmeal (alternate traps)	two 1500 m transects (~8 m spacing)	Victor mouse snap (200 ground; 400 TN)					15 captures 3.8/100 TN
			fresh coconut	two 750 m transects (~8 m spacing)	Victor rat snap (100 ground; 200 TN)				1 capture 0.5/100 TN	4 captures 2.0/100 TN
Stinson 1994				review of published and unpublished reports		common	uncommon	uncommon	common	uncommon
Gragg 2004 [l], Gragg et al. in prep	Jul 20–25, 2002 (6)	grassland with scattered patches of trees (Ija, Inarajan, Plot 1)	fresh coconut	9 × 9 grid (12.5 m spacing; 1 ha)	Haguruma live (28 ground; 168 TN), long Sherman live (25 ground; 150 TN), and standard Sherman live (28 ground; 168 TN)	17 captures 18.5/ha			15 captures 19.5/ha	21 captures
		(Ija, Inarajan, Plot 2)	fresh coconut	9 × 9 grid (12.5 m spacing; 1 ha)	Haguruma live (28 ground; 168 TN), long Sherman live (25 ground; 150 TN), and standard Sherman live (28 ground; 168 TN)	32 captures 50.4/ha			25 captures 36.6/ha	
		(Ija, Inarajan, Plot 3)	fresh coconut	9 × 9 grid (12.5 m spacing; 1 ha)	Haguruma live (28 ground; 168 TN), long Sherman live (25 ground; 150 TN), and standard Sherman live (28 ground; 168 TN)	42 captures 53.6/ha			29 captures 41.3/ha	
		(Ija, Inarajan, Plot 4)	fresh coconut	9 × 9 grid (12.5 m spacing; 1 ha)	Haguruma live (28 ground; 168 TN), long Sherman live (25 ground; 150 TN), and standard Sherman live (28 ground; 168 TN)	52 captures 68.0/ha			13 captures 18.2/ha	

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Guam continued										
Gragg 2004 [l], Gragg et al. in prep continued	Sep 21–26, 2002 (6)	(Ija, Inarajan, Plot 1)	fresh coconut	9 × 9 grid (12.5 m spacing; 1 ha)	Haguruma live (28 ground; 168 TN), long Sherman live (25 ground; 150 TN), and standard Sherman live (28 ground; 168 TN)	15 captures 15.2/ha			7 captures 15.7/ha	
		(Ija, Inarajan, Plot 4)	fresh coconut	9 × 9 grid (12.5 m spacing; 1 ha)	Haguruma live (28 ground; 168 TN), long Sherman live (25 ground; 150 TN), and standard Sherman live (28 ground; 168 TN)	22 captures 31.8/ha			8 captures 14.7/ha	
	Jun 24–29, 2003 (5)	grassland with scattered patches of trees (Ija, Inarajan, Plot 1)	fresh coconut	9 × 9 grid (12.5 m spacing; 1 ha)	Haguruma live (28 ground; 140 TN), long Sherman live (25 ground; 125 TN), and standard Sherman live (28 ground; 140 TN)	40 captures 40.2/ha			15 captures 24.9/ha	12 captures
		(Ija, Inarajan, Plot 2)	fresh coconut	9 × 9 grid (12.5 m spacing; 1 ha)	Haguruma live (28 ground; 140 TN), long Sherman live (25 ground; 125 TN), and standard Sherman live (28 ground; 140 TN)	48 captures 67.0/ha			29 captures 34.2/ha	
		(Ija, Inarajan, Plot 3)	fresh coconut	9 × 9 grid (12.5 m spacing; 1 ha)	Haguruma live (28 ground; 140 TN), long Sherman live (25 ground; 125 TN), and standard Sherman live (28 ground; 140 TN)	13 captures 24.0/ha			20 captures 20.1/ha	
		(Ija, Inarajan, Plot 4)	fresh coconut	9 × 9 grid (12.5 m spacing; 1 ha)	Haguruma live (28 ground; 140 TN), long Sherman live (25 ground; 125 TN), and standard Sherman live (28 ground; 140 TN)	47 captures 61.0/ha			19 captures 23.8/ha	
	Aug 20–25, 2003 (6)	(Ija, Inarajan, Plot 2)	fresh coconut	9 × 9 grid (12.5 m spacing; 1 ha)	Haguruma live (28 ground; 168 TN), long Sherman live (25 ground; 150 TN), and standard Sherman live (28 ground; 168 TN)	63 captures 104.0/ha			24 captures 69.8/ha	
		(Ija, Inarajan, Plot 3)	fresh coconut	9 × 9 grid (12.5 m spacing; 1 ha)	Haguruma live (28 ground; 168 TN), long Sherman live (25 ground; 150 TN), and standard Sherman live (28 ground; 168 TN)	13 captures 12.6/ha			10 captures 27.8/ha	

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Guguan										
Eldredge 1983		southern interior of island		incidental observations			small rats common			
Glass and Aldan 1987	May 28–Jun 4, 1987			incidental observations			observed small rats			
Eldredge 1988 [m]				incidental observations						observed single individual
Rice and Stinson 1992	May 17–18, 1992			incidental observations			found 2 small rodent skulls, probably *R. exulans*			
Stinson 1994				review of published and unpublished reports			common			
Cruz et al. 2000e	Jun 7–9, 2000 (3)	native forest (western slope)	peanut butter	550 m transect (25 m spacing)	Victor rat snap (23 tree; 69 TN)					
	(1)	savannah (near camp and large lava flow)	peanut butter	225 m transect (25 m spacing)	Victor rat snap (10 ground; 10 TN)		6 captures **60.0/100 TN** 70.6/100 CTN			
Kessler 2002 [n]				incidental observations			comment that *R. exulans* is very common			
Maug Islands										
Eldredge et al. 1977, Eldredge 1983	Jan and Jul 1975, Nov 1977, and Jul 1981	East Island (north end, abandoned cistern, Japanese weather station ruins)		incidental observations			|– – – – –	observed *R. exulans* or *R. rattus*; noted activity near *Terminalia* trees	– – – – –|	
Pratt and Lemke 1984	Feb 24–26, 1984	North Island		incidental observations			|– – – – –	observed small rat during day	– – – – –|	
		West Island		incidental observations			|– – – – –	observed 5 small rats	– – – – –|	
Rice and Stinson 1992	Jun 2–5, 1992	North and East Island		incidental observations			|– – – – –	observed small rats	– – – – –|	

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Pagan										
Pratt and Lemke 1984	Feb 19–21, 1984	near abandoned buildings where unknown rats were observed during previous visit		trapping and incidental observations	“rat”		| – – – – –	none captured or observed	– – – – – |	
Cruz et al. 2000f [o]	Aug 4–6, 2000 (3)	near barracks	peanut butter	opportunistic placement of traps	Victor rat snap					
		introduced forest (near landing area)	peanut butter	≤ 450 m transect (25 m spacing)	Victor rat snap (≤ 19 ground; ≤ 57 TN)					
		native forest (near landing area)	peanut butter	≤ 450 m transect (25 m spacing)	Victor rat snap (≤ 19 ground; ≤ 57 TN)				8 captures **≥ 14.0/100 TN**	
Rota										
Russell 2002	1602			observations by Fray Juan Pobre de Zamora			| – – – – –	observed abundant rats	– – – – – |	
Marshall 1962a	~1945–1960			review of specimens deposited at U.S. National Museum			collected		collected	
Bowers 2001	1947						| – – – – –	unidentified rats “overrun” island	– – – – – |	
Barbehenn 1974	Sep 1966									established; cites R.P. Owen, pers. comm..
Stinson 1994 [p]				review of published and unpublished reports		uncommon	uncommon?		uncommon?	common
Amidon 1999 [q]	Apr 1999 (5)	mature limestone forest (sites: 2HA, 1HB, 1LB, 1HC, 1LC, 1HD, 2HD, 1LD)	fresh coconut + peanut butter	eight 100 m transects (25 m spacing)	“snap traps” (5 alternating tree and ground; 200 TN)				31 captures 15.5/100 TN **23.1/100 CTN**	

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Rota continued										
Morton and Sharp 1997, Morton et al. 1999 [r]	Mar 25–30, 1997 (6)	coastal limestone forest with *Eugenia* understory (Mochong)	fresh coconut + peanut butter	two 500 m transects (25 m spacing)	“snap traps” (20 alternating tree and ground; 240 TN)				22 captures **9.2/100 TN** **14.9/100 CTN**	
		coastal limestone forest with *Eugenia* understory (Rail-release)	fresh coconut + peanut butter	two 500 m transects (25 m spacing)	“snap traps” (20 alternating tree and ground; 240 TN)				26 captures **10.8/100 TN** **17.7/100 CTN**	
		primary limestone forest (Golf Course)	fresh coconut + peanut butter	two 500 m transects (25 m spacing)	“snap traps” (20 alternating tree and ground; 240 TN)				25 captures **10.4/100 TN** **15.2/100 CTN**	
		primary limestone forest (Palii)	fresh coconut + peanut butter	two 500 m transects (25 m spacing)	“snap traps” (20 alternating tree and ground; 240 TN)				50 captures **20.8/100 TN** **33.8/100 CTN**	
	Aug 26–31, 1997 (6)	coastal limestone forest with *Eugenia* understory (Mochong)	fresh coconut + peanut butter	two 500 m transects (25 m spacing)	“snap traps” (20 alternating tree and ground; 240 TN)				9 captures **3.8/100 TN** **6.0/100 CTN**	
		coastal limestone forest with *Eugenia* understory (Rail-release)	fresh coconut + peanut butter	two 500 m transects (25 m spacing)	“snap traps” (20 alternating tree and ground; 240 TN)				26 captures **10.8/100 TN** **17.8/100 CTN**	
		primary limestone forest (Golf Course)	fresh coconut + peanut butter	two 500 m transects (25 m spacing)	“snap traps” (20 alternating tree and ground; 240 TN)				14 captures **5.8/100 TN** **7.1/100 CTN**	
		primary limestone forest (Palii)	fresh coconut + peanut butter	two 500 m transects (25 m spacing)	“snap traps” (20 alternating tree and ground; 240 TN)				11 captures **4.6/100 TN** **7.5/100 CTN**	
	Apr 1999 (5)	coastal limestone forest with *Eugenia* understory (Mochong)	fresh coconut + peanut butter	two 500 m transects (25 m spacing)	“snap traps” (20 alternating tree and ground; 200 TN)				32 captures 16.0/100 TN **23.5/100 CTN**	
		coastal limestone forest with *Eugenia* understory (Rail-release)	fresh coconut + peanut butter	two 500 m transects (25 m spacing)	“snap traps” (20 alternating tree and ground; 200 TN)				49 captures 24.5/100 TN **32.8/100 CTN**	
		primary limestone forest (Golf Course)	fresh coconut + peanut butter	two 500 m transects (25 m spacing)	“snap traps” (20 alternating tree and ground; 200 TN)				23 captures 11.5/100 TN **16.9/100 CTN**	
		primary limestone forest (Palii)	fresh coconut + peanut butter	two 500 m transects (25 m spacing)	“snap traps” (20 alternating tree and ground; 200 TN)				13 captures 6.5/100 TN **11.4/100 CTN**	

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Rota continued										
Esselstyn and Ulloa (unpublished data) [s]	Jul 28–30, 2002 (3)	degraded native forest (Mochong A)	coconut + peanut butter	190 m transect (10 m spacing)	Victor rat snap (20 tree; 60 TN, but traps were checked twice/day)				61 captures **50.8/100 trap checks**	
		degraded native forest (Mochong B)	coconut + peanut butter	190 m transect (10 m spacing)	Victor rat snap (20 tree; 60 TN, but traps were checked twice/day)				63 captures **52.5/100 trap checks**	
		degraded native forest (Mochong C)	coconut + peanut butter	190 m transect (10 m spacing)	Victor rat snap (20 tree; 60 TN, but traps were checked twice/day)				73 captures **60.8/100 trap checks**	
		degraded native forest (Lalayak A)	coconut + peanut butter	190 m transect (10 m spacing)	Victor rat snap (20 tree; 60 TN, but traps were checked twice/day)				67 captures **55.8/100 trap checks**	
		degraded native forest (Lalayak B)	coconut + peanut butter	190 m transect (10 m spacing)	Victor rat snap (20 tree; 60 TN, but traps were checked twice/day)				71 captures **59.2/100 trap checks**	
		degraded native forest (Lalayak C)	coconut + peanut butter	190 m transect (10 m spacing)	Victor rat snap (20 tree; 60 TN, but traps were checked twice/day)				58 captures **48.3/100 trap checks**	
	Sep 25–27, 2002 (3)	immature native forest (Pekngasu)	coconut + peanut butter	8 × 8 grid (10 m spacing; 0.49 ha)	Victor rat snap (64 tree; 192 TN)				22 captures **11.5/100 TN** **13.2/100 CTN**	
	Sep 29–Oct 4, 2002 (6)	immature native forest (Pekngasu)	coconut + peanut butter	8 × 8 grid (10 m spacing; 0.49 ha)	Victor rat snap (64 tree; 384 TN)				21 captures **5.5/100 TN** **6.1/100 CTN**	
	Oct 8–11, 2002 (4)	immature native forest (Pekngasu)	coconut + peanut butter	8 × 8 grid (10 m spacing; 0.49 ha)	Victor rat snap (64 tree; 256 TN)				21 captures **8.2/100 TN** **9.5/100 CTN**	
	Oct 15–18, 2002 (4)	immature native forest (Pekngasu)	coconut + peanut butter	8 × 8 grid (10 m spacing; 0.49 ha)	Victor rat snap (64 tree; 256 TN)				9 captures **3.5/100 TN** **3.8/100 CTN**	
	Oct 22–25, 2002 (4)	immature native forest (Pekngasu)	coconut + peanut butter	8 × 8 grid (10 m spacing; 0.49 ha)	Victor rat snap (64 tree; 256 TN)				2 captures **0.8/100 TN** **0.8/100 CTN**	

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Rota continued										
Esselstyn and Ulloa (unpublished data) continued	May 2003 (3)	native forest (< 50 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				17 captures **18.9/100 TN** 23.8/100 CTN	
		native forest (< 50 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				17 captures **18.9/100 TN** 24.6/100 CTN	
		native forest (300–350 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				36 captures **40.0/100 TN** 50.7/100 CTN	
		native forest (300–350 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				31 captures **34.4/100 TN** 39.2/100 CTN	
	Aug 2003 (3)	native forest (< 50 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				24 captures **26.7/100 TN** 33.1/100 CTN	
		native forest (< 50 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				14 captures **15.6/100 TN** 19.2/100 CTN	
		native forest (300–350 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				42 captures **46.7/100 TN** 56.0/100 CTN	
		native forest (300–350 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				35 captures **35.6/100 TN** 44.8/100 CTN	
	Oct 2003 (3)	native forest (< 50 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				7 captures **7.8/100 TN** 14.3/100 CTN	
		native forest (< 50 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				18 captures **20.0/100 TN** 25.0/100 CTN	
		native forest (300–350 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				30 captures **33.3/100 TN** 41.4/100 CTN	
		native forest (300–350 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				49 captures **54.4/100 TN** 60.1/100 CTN	

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Rota continued										
Esselstyn and Ulloa (unpublished data) continued	Feb 2004 (3)	native forest (< 50 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				14 captures **15.6/100 TN** 19.7/100 CTN	
		native forest (< 50 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				20 captures **22.2/100 TN** 27.2/100 CTN	
		native forest (300–350 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				42 captures **46.7/100 TN** 58.3/100 CTN	
		native forest (300–350 m elevation)	coconut + peanut butter	three 225 m transects (25 m spacing)	Victor rat snap (10 tree; 90 TN)				42 captures **46.7/100 TN** 57.5/100 CTN	
Amar and Ulloa (unpublished data)	Dec 17–19, 2003 (3)	introduced forest (Aga Tasi)	coconut + peanut butter	5 × 5 grid with empty center (25 m spacing; 1 ha)	Victor rat snap (24 tree; 72 TN)				15 captures **20.8/100 TN** 33.3/100 CTN	
	Jan 21–23, 2004 (3)	native forest (Gayaugon)	coconut + peanut butter	5 × 5 grid with empty center (25 m spacing; 1 ha)	Victor rat snap (24 tree; 72 TN)				9 captures **12.5/100 TN** 13.8/100 CTN	
	Feb 19–21, 2004 (3)	introduced forest (Fruit farm 2)	coconut + peanut butter	5 × 5 grid with empty center (25 m spacing; 1 ha)	Victor rat snap (24 tree; 72 TN)				15 captures **20.8/100 TN** 38.5/100 CTN	
	Feb 26–28, 2004 (3)	native forest (As Bake)	coconut + peanut butter	5 × 5 grid with empty center (25 m spacing; 1 ha)	Victor rat snap (24 tree; 72 TN)				18 captures **25.0/100 TN** 46.2/100 CTN	
	Mar 3–5, 2004 (3)	native forest (Tetohge)	coconut + peanut butter	5 × 5 grid with empty center (25 m spacing; 1 ha)	Victor rat snap (24 tree; 72 TN)				9 captures **12.5/100 TN** 14.1/100 CTN	
	Mar 17–19, 2004 (3)	introduced forest (Fruit farm 2b)	coconut + peanut butter	5 × 5 grid with empty center (25 m spacing; 1 ha)	Victor rat snap (24 tree; 72 TN)				15 captures **20.8/100 TN** 28.3/100 CTN	
	May 5–7, 2004 (3)	native forest (Pictograph Cave)	coconut + peanut butter	5 × 5 grid with empty center (25 m spacing; 1 ha)	Victor rat snap (24 tree; 72 TN)				6 captures **8.3/100 TN** 9.5/100 CTN	

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Rota continued										
Amar and Ulloa (unpublished data) continued	May 5–7, 2004 (3)	native forest (Open field 1)	coconut + peanut butter	5 × 5 grid with empty center (25 m spacing; 1 ha)	Victor rat snap (24 tree; 72 TN)				14 captures **19.4/100 TN** 23.7/100 CTN	
	May 12–14, 2004 (3)	introduced forest (Fruit farm 4)	coconut + peanut butter	5 × 5 grid with empty center (25 m spacing; 1 ha)	Victor rat snap (24 tree; 72 TN)				24 captures **33.3/100 TN** 44.4/100 CTN	
	May 19–21, 2004 (3)	native forest (Bird Sanctuary)	coconut + peanut butter	5 × 5 grid with empty center (25 m spacing; 1 ha)	Victor rat snap (24 tree; 72 TN)				18 captures **25.0/100 TN** 38.3/100 CTN	
	May 19–21, 2004 (3)	native forest (Open field 2)	coconut + peanut butter	5 × 5 grid with empty center (25 m spacing; 1 ha)	Victor rat snap (24 tree; 72 TN)				8 captures **11.1/100 TN** 13.6/100 CTN	
	May 26–28, 2004 (3)	native forest (Gayaugon B)	coconut + peanut butter	5 × 5 grid with empty center (25 m spacing; 1 ha)	Victor rat snap (24 tree; 72 TN)				13 captures **18.1/100 TN** 20.3/100 CTN	
	Jun 9–11, 2004 (3)	native forest (Quarry)	coconut + peanut butter	5 × 5 grid with empty center (25 m spacing; 1 ha)	Victor rat snap (24 tree; 72 TN)				29 captures **40.3/100 TN** 52.7/100 CTN	
Saipan										
Kuroda 1939[t]						present		present	present	
Marshall 1962a	~1945–1960			review of specimens deposited at U.S. National Museum		collected	collected	collected	collected	
Bowers 2001	1947						|–––––	unidentified rats “overrun” island	–––––|	
Enders 1949	1949	variety of habitats and human-use areas		trapping, visual searches, and incidental observations	“standard rat traps” (both ground and tree placement)	present	captured	captured; speculated that population was in decline	captured; most abundant rat	

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Saipan continued										
Barbehenn 1974 [u]	Aug 29–Sep 3, 1962	variety of habitats across island		8 sites		|– – – – –	138 captures	— | |	– – – – –|	20 captures
	Feb 11–18, 1963	variety of habitats across island		5 sites		|– – – – –	212 captures	includes 19 *R. norvegicus*	– – – – –|	149 captures
	Aug 26–Sep 5, 1963	variety of habitats across island		14 sites		|– – – – –	151 captures	captures | |	– – – – –|	195 captures
	Feb 25–Mar 5, 1964	variety of habitats across island		13 sites		|– – – – –	240 captures	| —	– – – – –|	149 captures
Stinson 1994 [v]				review of published and unpublished reports		uncommon	uncommon	common	uncommon?	common
Vogt (unpublished data) [w]	Apr 1–3, 1997 (6 sampling occasions)	mixed secondary forest bordering wetland (American Memorial Park)	canned cat food	7 × 9 grid (15 × 10 m spacing; 0.72 ha)	minnow funnel traps with one-way flaps (63 ground; 378 trap occasions)					33 captures 27.3/ha
	Nov 17–21, 1997 (10 sampling occasions)	*Leucaena* forest (near Saipan Airport)	canned cat food	11 × 11 grid (10 m spacing; 1 ha)	minnow funnel traps with one-way flaps (121 ground; 1210 trap occasions)					50 captures 26.4/ha
	Apr 13–15, 1998 (6 sampling occasions)	native limestone forest (near Bird Island)	canned cat food	11 × 11 grid (10 m spacing; 1 ha)	minnow funnel traps with one-way flaps (121 ground; 726 trap occasions)					70 captures 16.7/ha
CNMI-DFW (unpublished data)	Apr 25–28, 2000 (4)	Airport		trapping	snap (57–64; 235 TN)		|– – – – –	27 captures **11.5/100 TN** **16.0/100 CTN**	– – – – –|	12 captures **5.1/100 TN** **7.1/100 CTN**
	May 1–2, 2000 (2)	Airport		trapping	snap (64–66; 130 TN)		|– – – – –	6 captures **4.6/100 TN** **5.6/100 CTN**	– – – – –|	14 captures **10.8/100 TN** **13.2/100 CTN**
	May 4–5, 2000 (2)	Airport		trapping	snap (61–64; 125 TN)		|– – – – –	26 captures **20.8/100 TN** **36.4/100 CTN**	– – – – –|	17 captures **13.6/100 TN** **23.8/100 CTN**
	May 11–12, 2000 (2)	Airport		trapping	snap (31–33; 64 TN)		|– – – – –	7 captures **10.9/100 TN** **16.7/100 CTN**	– – – – –|	8 captures **12.5/100 TN** **19.1/100 CTN**
	May 16–17, 2000 (2)	Airport		trapping	snap (33; 66 TN)		|– – – – –	7 captures **10.6/100 TN** **12.7/100 CTN**	– – – – –|	9 captures **13.6/100 TN** **16.4/100 CTN**

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Saipan continued										
CNMI-DFW (unpublished data) continued	May 22–26, 2000 (5)	Airport		trapping	snap (29–33; 161 TN)		|– – – – –	8 captures **5.0/100 TN** **5.9/100 CTN**	– – – – –|	9 captures **5.6/100 TN** **6.6/100 CTN**
	May 30–Jun 2, 2000 (4)	Airport		trapping	snap (33; 132 TN)		|– – – – –	7 captures **5.3/100 TN** **6.1/100 CTN**	– – – – –|	1 capture **0.8/100 TN** **0.9/100 CTN**
	Feb 29–Mar 3, 2000 (4)	Marpi		trapping	snap (97–99; 392 TN)		|– – – – –	4 captures **1.0/100 TN** **1.1/100 CTN**	– – – – –|	2 captures **0.5/100 TN** **0.6/100 CTN**
	Mar 7–10, 2000 (4)	Marpi		trapping	snap (99; 396 TN)					2 captures **0.5/100 TN** **0.6/100 CTN**
	Mar 14–16, 2000 (3)	Marpi		trapping	snap (99–102; 301 TN)		|– – – – –	2 captures **0.7/100 TN** **0.8/100 CTN**	– – – – –|	
Sachtleben (unpublished data) [x]	Jul 3–8, 2003 (6)	*Leucaena* forest (Obyan)	fresh coconut	100 m long trapping line transect	Sherman live (100 ground; 600 TN)	4 captures	4 captures		10 captures, 4 recaptures	142 captures
	Jul 14–20, 2003 (6)	*Leucaena* forest (Bird Island)	fresh coconut	100 m long trapping line transect	Sherman live (100 ground/tree mix; 600 TN)		1 capture	1 capture		25 captures, 8 recaptures
	Jul 3–8, 2003 (6)	native forest (Laolao Bay)	fresh coconut	100 m long trapping line transect	Sherman live (100 ground; 600 TN)					117 captures
	Jul 14–20, 2003 (6)	native forest (Marpi)	fresh coconut	100 m long trapping line transect	Sherman live (100 ground/tree mix; 600 TN)					31 captures, 18 recaptures
Sarigan										
Arriola et al. 1999	Jul 4–8, 1999 (4)	mixed *Cocos* forest (USFWS transect 3)	baked coconut + peanut butter	850 m transect (25 m spacing)	Victor snap (35 ground; 140 TN)		35 captures **25.0/100 TN** **33.0/100 CTN**			
	Jul 4–8, 1999 (3)	native forest (USFWS transect 5)	baked coconut + peanut butter	1075–1200 m transect (25 m spacing)	Victor snap (44–49 ground; 141 TN)		6 captures **4.3/100 TN** **4.8/100 CTN**			
Vogt (unpublished data)	Jul 1999 (3)	native forest			Victor rat snap (43 ground; 129 TN)		6 captures **4.7/100 TN** **5.6/100 CTN**			

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Sarigan continued										
Kessler 2002 [y]	Jul 1999 Jul 2000		peanut butter	two 725 m transects (25 m spacing)	"large" snap (30 ground)		< 6 captures each year			
Cruz et al. 2000g	Jul 4–6, 2000 (3)	mixed *Cocos* forest (transect 3)	peanut butter	600 m transect (25 m spacing)	Victor rat snap (25 ground; 75 TN)		3 captures **4.0/100 TN** 5.7/100 CTN			
Tinian										
Thomas 1971 (cited in Wiles et al. 1990)	1742			incidental observations by Lord Anson			|–––––	unidentified rats numerous	–––––|	
Downs 1948	May 31–Oct 17, 1945			incidental observations		|–––––	observed rats and mice	–––––––	–––––|	
Marshall 1962a	~1945–1960			review of specimens deposited at U.S. National Museum		collected	collected		collected	
Bowers 2001	1947						|–––––	unidentified rats "overrun" island	–––––|	
Owen 1974	Jan 18–25, 1974			incidental observations					3 unknown rats observed, tentative *R. rattus* identification	single observation at hotel
Wiles et al. 1990 [z]	Jan 5–12 and May 10–14, 1985 (1)	*Leucaena* forest	peanut butter, toasted coconut, or dampened oatmeal	90–590 m transect(s) (10 m spacing)	8 x 14 or 9 x 18 cm snap (10–60 ground; 198 TN)				8 captures **4.0/100 TN**	5 captures **2.5/100 TN**
		open fields	peanut butter, toasted coconut, or dampened oatmeal	90–590 m transect(s) (10 m spacing)	8 x 14 or 9 x 18 cm snap (10–60 ground; 123 TN)	1 capture **0.81/100 TN**			3 captures **2.4/100 TN**	1 captures **0.8/100 TN**
		secondary vegetation	peanut butter, toasted coconut, or dampened oatmeal	90–590 m transect(s) (10 m spacing)	8 x 14 or 9 x 18 cm snap (10–60 ground; 67 TN)				2 captures **3.0/100 TN**	2 captures **3.0/100 TN**

Reference	Sampling Date (Occasions)	Sampling Habitat (Location)	Bait	Sampling Methodology	Trap Type (Placement; Effort)	*Mus musculus*	*Rattus exulans*	*Rattus norvegicus*	*Rattus rattus/ R. tanezumi*	*Suncus murinus*
Tinian continued										
Wiles et al. 1990 [z] continued	Jan 5–12 and May 10–14, 1985 (1)	strand vegetation	peanut butter, toasted coconut, or dampened oatmeal	90–590 m transect(s) (10 m spacing)	8 x 14 or 9 x 18 cm snap (10–60 ground; 47 TN)				8 captures **17.0/100 TN**	
		native forest	peanut butter, toasted coconut, or dampened oatmeal	90–590 m transect(s) (10 m spacing)	8 x 14 or 9 x 18 cm snap (10–60 ground; 25 TN)				Observed but not captured	Observed but not captured
		municipal dump	peanut butter, toasted coconut, or dampened oatmeal	90–590 m transect(s) (10 m spacing)	8 x 14 or 9 x 18 cm snap (10–60 ground; 25 TN)				1 captures **4.0/100 TN**	Observed but not captured
Stinson 1994				review of published and unpublished reports		uncommon	rare	uncommon	common	common
Vogt (unpublished data)	Mar 2007 (3)	*Leucaena* forest (road to Puntan Tahgong)		980 m transect (20 m spacing)	Victor rat snap (50 ground; 150 TN)				67 captures **44.7/100 TN** **78.8/100 CTN**	15 captures **10.0/100 TN** **17.6/100 CTN**
		native forest (road to Lasu)		980 m transect (20 m spacing)	Victor rat snap (50 ground; 150 TN)				42 captures **28.0/100 TN** **53.8/100 CTN**	22 captures **14.7/100 TN** **28.2/100 CTN**

---

[a] At native limestone forest site, tree traps were active for 4 nights, ground traps were active for 3 nights. An odd trap check schedule (traps checked hourly for 3 hours in evening, then again following morning) complicates calculation of TNs for comparison to other studies. In reference, the reported captures/100 CTN is < the reported captures/100 TN, which is not possible.

[b] Results reported in reference do not agree with Methods. For example, reported TN (168 CTN) is not possible based on number of traps and sampling duration. Also, sampling conducted in mixed *Cocos* forest, where 3 *R. exulans* were captured, is not mentioned in Methods and there is no indication of sampling effort.

[c] References do not indicate sampling duration; without this information, one can not calculate sampling effort or capture rates. Reference does not indicate which transect(s) captures occurred on.

[d] Reference reports the results of an avian visual survey, but also mentioned that *R. exulans* is present, although no supporting evidence was provided.

[e] Study employed mark-recapture livetrapping, except for final sampling event (Oct 19–24) which included both live and snap traps. Traps were places somewhat systematically at an average spacing of < 10 m. Note that Baker excluded animals that died during livetrapping from his density calculations, which could affect comparison with other studies. Including these animals in density calculations can result in significantly different results. For example, 10 *M. musculus* died during the first livetrapping session; leaving these animals in the density calculation results in an estimate of 16.7/ha vs. Baker’s value of 8.3/ha.

[f] Sampling conducted by Guam Sanitation Section. Reference reports 5469 traps active for 4 nights, for a total of 21876 TN. Reference reports combined capture totals for *M. musculus* and *Rattus* species.

[g] Trapping grids had 15.2 or 30.5 m spacing. Each trap station on grid contained 2 rat snap traps, 1 or 2 mouse snap traps, and 1 Museum Special snap trap. Transect sampling conducted by Guam Sanitation Section. Reference does not provide site-specific capture counts, limiting results to presentation of overall average captures/ha. Six *R. norvegicus* captured over duration of sampling, but reference does not provide indication of capture location or circumstances. Reference reports that many additional animals were captured during extended sampling (many grids were sampled > 4 nights) but offers no indication of additional sampling effort or number of additional captures.

[h] Transects located exclusively in human-made openings in vegetation (e.g., jeep trails, roadsides, utility right-of-ways). It is possible, especially for *M. musculus* and *S. murinus*

which have relatively small home ranges, that this approach sampled only "edge" habitats bordering these openings rather than the targeted habitats (e.g., native forest, ravine forest, etc.). Transects were sampled for 1 night only, except for Tarague in 1981 (alternate halves of limestone forest transect sampled on consecutive nights), Ipapao in 1981 (mixed forest transect sampled twice over consecutive nights), and Tarague in 1994 (2 limestone forest mouse trap transects each sampled twice over consecutive nights). Note that 3 *R. norvegicus* captured in 1981 and 2 *R. tanezumi* captured in 1994 are not presented in summary tables in reference. Also, reference text indicates 25 *R. tanezumi* captures in 1994, whereas reference table 2 indicates 26 *R. tanezumi* captures.

[i] Baker's (1946) site resampled with a 10 × 10 grid with trap stations spaced 10 m apart. Each trap station had 1 rat snap trap and 1–2 mouse snap traps. Other sampling events utilized 8 x 8 grids with trap stations spaced 15 m apart. Each trap station had 2 rat snap traps and 1–2 mouse snap traps. Note that nominal grid areas calculated in reference are incorrect (too large).

[j] Reference does not provide sampling duration information, so it is not possible to verify the sampling effort information (125 rat snap and 90 mouse snap TN).

[k] Note that 1 of 3 *R. exulans* was actually captured on an adhesive trap deployed for concurrent lizard surveys.

[l] Study employed mark-recapture livetrapping. Note that each trap station contained a single Haguruma, long Sherman, or standard Sherman trap. Traps allocation occurred (roughly) on an alternating row pattern. Summary data are only included here for pre-treatment and control (no rodenticide) sampling grids. Note that during Jun 24–29, 2003, sampling period data from Jun 25 had to be discarded because of a marking issue, resulting in 5 occasions for this sampling event. Reference does not separate *R. exulans* and *R. rattus* captures, but does comment that unequivocal *R. exulans* specimens made up ~3% of sample, whereas unequivocal *R. rattus* specimens made up >90% of sample. Reference does not present *S. murinus* capture information for each grid, only total captures per year.

[m] Only known record of *S. murinus* outside of Guam, Rota, Saipan, and Tinian. Status of sighting is unclear, although recent research trips to Guguan have not commented on presence of *S. murinus*.

[n] Reference reports research conducted on Sarigan, but mentions that *R. exulans* is very common and commonly observed during daylight on Guguan.

[o] Reference does not provide necessary information (number of traps) to calculate sampling effort or capture rates. Note that the value for TN presented in reference is not possible based on the maximum number of traps (≤ 19) and sampling duration described.

[p] Justification for common status of *S. murinus* is unclear, as only known reference of this species for Rota (and the reference cited by Stinson) is Barbehenn 1974, which does not comment on abundance.

[q] Reference indicates uncertainty about species identification and suggest either *R. exulans* or *R. tanezumi*. Although reference provides captures/100 CTN, they were not calculated in a comparable fashion to other data in this summary and are not included here.

[r] Additional sampling (6 consecutive nights) was conducted on these study sites in March and August, 1998, although neither reference provides adequate information for summary of these data. Both references indicate uncertainty about species identification and suggest either *R. exulans* or "some variant of *R. rattus*" (possibly *R. tanezumi*?).

[s] Traps checked twice per day at Mochong A, B, and C and Lalayak A, B, and C, which complicates calculation of TN for comparison with other studies.

[t] Reference comments on presence of 2 *Mus* species on Saipan: *M. musculus momiyamai* and *M. caroli boninensis.* Validity of this claim is unclear; taxonomy of *Mus* has undergone extensive revision in recent years and the current identification of the *Mus* species in the Mariana Islands is *M. musculus castaneus* (Musser and Carleton 2005:1401).

[u] Note that captures were divided into *S. murinus* and other introduced small mammals. The other introduced small mammal category included 19 *R. norvegicus*, although no information is provided for capture location or date.

[v] Justification for common status of *R. norvegicus* is unclear. Other references for *R. norvegicus* on Saipan offer little information related to abundance; available information seems to suggest that this species is actually rare on Saipan.

[w] Study employed mark-recapture livetrapping, and was targeted specifically for *S. murinus*. Traps checked twice per day, effectively doubling sampling occasions.

[x] Study employed limited mark-recapture livetrapping. *S. murinus* was not marked during Jul 3–8 sampling events, so recaptures make up an unknown portion of the indicated captures. Traps placed in trees (Jul 14–20 sampling events) recorded zero captures.

[y] Reference provides limited methodology and results. Reference states that S. Vogt conducted sampling, so Jul 1999 sampling may be same as in the previous record (S. Vogt unpublished data), where more complete methods and results are available.

[z] Reference does not provide specific information about number of sampled transects, the number of traps per transect, or the number of each of the 2 trap types used. Note that the 8 x 14 and 9 x 18 cm snap traps are similar in dimensions to Museum Special and Victor rat snap traps, respectively.

APPENDIX 1B. Delta method procedures for calculating density and biomass variances

Density Variance Estimates

Introduced small mammal density estimates were generated using species-specific abundance estimates (Program MARK 4.3; White and Burnham 1999) and mean maximum distance moved (MMDM) estimates from mark-recapture sampling conducted on Guam, Rota, Saipan, and Tinian, 2005–2007. Variance estimates were derived using the following steps. Note that matrices created in Steps 1–5 can most easily be created in MS Excel, using values generated from SAS code provided below. Matrices and matrix algebra described in Step 6 occur in Proc IML (SAS Institute 2003).

1. Output a model-averaged variance-covariance matrix for abundance estimates ($\hat{\bar{N}}$). In Program MARK, select the "Output" tab, select "Model Averaging," and then select "Derived." In the "Model Averaging Parameter Selection" box that opens, select the parameters of interest and check the box for "Export Variance-Covariance Matrix to a dBase file." This *.dbf file includes both the model-averaged variance-covariance matrix and the model-averaged parameter estimates, of which we are interested only in the former.

2. Produce a variance-covariance matrix for MMDM model(s). Generate the variance-covariance matrix in SAS, with the "COV" option in an LSMEANS statement (see SAS Code A). Use the sum of squares error (SSE) and the sample size (corrected total *df* + 1) to estimate $\hat{\sigma}^2$ to use in calculating AIC$_c$ values (($\log(\ell(\hat{\theta})) = -n/2 * \log(\hat{\sigma}^2)$) and weights to aid in model selection. Use the variance covariance matrix from the top model unless model selection uncertainty exists (top model AIC$_c$ $w_i < 0.90$), in which case the MMDM variance-covariance matrices should be model-averaged (see SAS Code 2)).

   The MMDM variance-covariance matrix is created by taking the values and applying them to the appropriate site-specific location in an identity matrix. For example, the *Rattus rattus* identity matrix includes all sites with captures ($n = 18$). The top MMDM model (MMDM as a function of Island; see Table 4 in main body of Chapter 1) provides COV output for each island; these values are applied to each site based on its island location, such that Guam sites receive the Guam COV value, Rota sites receive the Rota COV values, etc.

3. Calculate the effective trapping area (ETA), based on the MMDMs determined from modeling. This can be done using the following equation (see Williams et al. 2002:314–315 for further explanation):

$$ETA = Area_{grid} + \left(\left(\frac{MMDM}{2}\right) * 2L\right) + \left(\left(\frac{MMDM}{2}\right) * 2W\right) + \left(4 * \left(\frac{\pi * \left(\frac{MMDM}{2}\right)^2}{4}\right)\right),$$

   which simplifies to

$$ETA = Area_{grid} + ((MMDM) * L) + ((MMDM) * W) + \left(\pi * \left(\frac{MMDM}{2}\right)^2\right).$$

As an example, for *R. rattus* on Guam:

$$ETA = 15625\ m + ((35.617\ m)*125\ m) + ((36.617\ m)*125\ m) + \left(\pi * \left(\frac{35.617\ m}{2}\right)^2\right)$$

$$= 25525.58\ m^2 = 2.55\ ha$$

4. Create a matrix of the partial derivative of ETA with respect to MMDM using the following equation:

$$\frac{\partial\, ETA}{MMDM} = 0 + L + W + 2(MMDM)$$

As an example, for *R. rattus* on Guam $\frac{\partial\, ETA}{MMDM} = 0 + 125\ m + 125\ m + 2(35.617\ m) = 312.39\ m$.

This value is calculated for each site, and placed in the appropriate site-specific location in an identity matrix.

5. Create a variance covariance matrix of the partial derivative of density with respect to $\hat{\hat{N}}$, and density with respect to ETA. In effect, this matrix contains 2, side-by-side identity matrices.

Because of the form of the density formula ($Density = \frac{\hat{\hat{N}}}{ETA}$), these partial derivatives are calculated using the following equations:

$$\frac{\partial\, Density}{\hat{\hat{N}}} = \frac{1}{ETA} \qquad \text{and} \qquad \frac{\partial\, Density}{ETA} = \frac{-\left(\hat{\hat{N}}\right)}{(ETA)^2}$$

As an example, for *R. rattus* on Guam $\frac{\partial\, Density}{\hat{\hat{N}}} = \frac{1}{25525.58\ m^2}$ = 0.0000392,

and for *R. rattus* at site MSRG on Guam $\frac{\partial\, Density}{ETA} = \frac{-(41.134)}{\left(25525.58\ m^2\right)^2}$ = -0.0000000631/m^2

These values are calculated for each site, and placed in the appropriate site-specific location in an identity matrix (as noted above, this is essentially 2, side-by-side identity matrices) with $\frac{\partial\, Density}{\hat{\hat{N}}}$ on the left side of the matrix and $\frac{\partial\, Density}{ETA}$ on the right side of the matrix.

6. Perform the necessary matrix algebra to create new variance-covariance matrices using SAS Proc IML (see SAS Code 3). In the SAS code, there are a number of Proc IMPORT statements that import worksheets from an Excel spreadsheet. Imported worksheets contain the variance-covariance

matrices created in Steps 1–5 above: Program MARK $\hat{N}$ variance-covariance, MMDM variance-covariance, $\frac{\partial ETA}{MMDM}$ variance-covariance, and $\frac{\partial Density}{\hat{N}}$ *and* $\frac{\partial Density}{ETA}$ variance-covariance.

Imported datasets are used by Proc IML to generate new variance-covariance matrices, beginning with a variance-covariance matrix for ETA. Symbolically, this involves multiplying 3 matrices:

$$\left(Var\text{–}Cov\ of\ \frac{\partial ETA}{MMDM} \right) * \left(Var\text{–}Cov\ \ of\ \ MMDM \right) * \left(Var\text{–}Cov\ of\ \frac{\partial ETA}{MMDM} \right)^T ,$$

where the T indicates that this matrix is transposed. For *R. rattus*, each of these matrices is 18 × 18; as a result of matrix algebra rules this multiplication process produces an 18 × 18 matrix.

Next, Proc IML is used to generate a variance-covariance matrix combining the Program MARK $\hat{N}$ variance-covariance matrix and the ETA variance-covariance matrix created in the proceeding step. With the *R. rattus* data, these 18 × 18 matrices are combined to produce a 36 × 36 identity matrix, with Program MARK $\hat{N}$ variance-covariance in the upper left quadrant and the ETA variance-covariance in the lower right, and zeros filling in the upper right and lower left quadrants of the matrix. Note that no matrix algebra or other manipulation is involved in this step; instead, existing variance-covariance matrices are combined into a new, larger variance-covariance matrix.

Finally, Proc IML is used to generate a variance-covariance matrix for the density estimates generated from Program MARK $\hat{N}$ and MMDM. Symbolically, this involves multiplying 3 matrices:

$$\left(Var\text{–}Cov\ of\ \frac{\partial Density}{\hat{N}}\ and\ \frac{\partial Density}{ETA} \right) * \left(Var\text{–}Cov\ \ of\ \hat{N}\ and\ ETA \right) * \left(Var\text{–}Cov\ of\ \frac{\partial Density}{\hat{N}}\ and\ \frac{\partial Density}{ETA} \right)^T$$

For *R. rattus*, these matrices are [18 × 36]*[36 × 36]*$[36 \times 18]^T$ which produces an 18 × 18 matrix of density estimate variances.

Values of interest (variance estimates for density by site) lie along the diagonal of the matrix. Variances are converted to standard errors by taking their square root.

Biomass Variance Estimates

Because biomass estimates were derived from density estimates, biomass variance determination builds from the density variance determination described above. Introduced small mammal biomass estimates were generated using species-specific density estimates and mass estimates from sampling conducted on Guam, Rota, Saipan, and Tinian, 2005–2007. Variance estimates were derived using the following steps:

1. Steps 1–6 above would be repeated for biomass variance determination (if not already completed for density variance determination). Many of the matrices created in these steps are carried over for biomass variance determination. Note that matrices created in Steps 2 and 3 below can most easily

be created in Excel, using data generated from SAS code provided below. Matrices and matrix algebra described in Step 4 below occur in Proc IML (SAS Institute 2003).

2. Produce a variance-covariance matrix for mass model(s). Generate the variance-covariance matrix in SAS, with the "COV" option in an LSMEANS statement (see SAS Code 4). Use the sum of squares error (SSE) and the sample size (corrected total *df* + 1) to estimate $\hat{\sigma}^2$ to use in calculating AIC$_c$ values ($(\log(\ell(\hat{\theta})) = -n/2 * \log(\hat{\sigma}^2)$) and weights for each model to aid in model selection. Use the variance covariance matrix from the top model unless model selection uncertainity exists (top model AIC$_c$ $w_i < 0.90$), in which case the mass variance-covariance matrices should be model-averaged (follows identical procedure as in SAS Code 2)).

   The mass variance-covariance matrix is created by taking the values and applying them to the appropriate site-specific location in an identity matrix. For example, the *R. rattus* identity matrix includes all sites with captures ($n = 18$). The top mass model (mass as a function of site; see Table 5 in main body of Chapter 1) provides COV output for each site, which is entered along the main diagonal of the identity matrix.

3. Create a variance covariance matrix of the partial derivative of biomass with respect to density, and biomass with respect to mass. In effect, this matrix contains 2, side-by-side identity matrices. Because of the form of the biomass formula ($Biomass = Density\ x\ Mass$), these partial derivatives are calculated using the following equations:

   $$\frac{\partial\, Biomass}{Density} = Mass \qquad \text{and} \qquad \frac{\partial\, Biomass}{Mass} = Density$$

   As an example, for *R. rattus* at site KAST on Tinian $\frac{\partial\, Biomass}{Density} = Mass = 115.828$ g,

   and for *R. rattus* at site RAPF on Rota $\frac{\partial\, Biomass}{Mass} = Density = 95.8$/ha

   These values are calculated for each site, and placed in the appropriate site-specific location in an identity matrix (as noted above, this is essentially 2, side-by-side identity matrices) with $\frac{\partial\, Biomass}{Density}$ on the left side of the matrix and $\frac{\partial\, Biomass}{Mass}$ on the right side of the matrix.

4. Perform the necessary matrix algebra to create new variance-covariance matrices using SAS Proc IML (see SAS Code 3). In the SAS code, there are a number of Proc IMPORT statements that import worksheets from an Excel spreadsheet. Imported worksheets contain the variance-covariance matrix created in Step 6 from the Density variance determination procedure above, as well as Steps 2 and 3 from the Biomass determination procedure: density estimate variance-covariance, mass variance-covariance, and $\frac{\partial\, Biomass}{Density}$ *and* $\frac{\partial\, Biomass}{Mass}$ variance-covariance.

   Next, Proc IML is used to generate a variance-covariance matrix combining the density variance-covariance matrix and the mass variance-covariance matrix created in the proceeding steps. With the

R. rattus data, these 18 × 18 matrices are combined to produce a 36 × 36 identity matrix, with the density variance-covariance in the upper left quadrant and the mass variance-covariance in the lower right, and zeros filling in the upper right and lower left quadrants of the matrix. Note that no matrix algebra or other manipulation is involved in this step; instead, existing variance-covariance matrices are combined into a new, larger variance-covariance matrix.

Finally, Proc IML is used to generate a variance-covariance matrix for the biomass estimates generated from density estimates and mass estimates. Symbolically, this involves multiplying 3 matrices:

$$\left(Var\text{–}Cov\ of\ \frac{\partial\, Biomass}{Density}\ and\ \frac{\partial\, Biomass}{Mass}\right)*\left(Var\text{–}Cov\ of\ Density\ and\ Mass\right)*\left(Var\text{–}Cov\ of\ \frac{\partial\, Biomass}{Density}\ and\ \frac{\partial\, Biomass}{Mass}\right)^T$$

For *R. rattus*, these matrices are [18 × 36]*[36 × 36]*[36 × 18]^{T} which produces an 18 × 18 matrix of density estimate variances.

Values of interest (variance estimates for biomass by site) lie along the diagonal of the matrix. Variances are converted to standard errors by taking their square root.

Literature Cited

SAS Institute. 2003. SAS/STAT software. Version 9.1. SAS Institute, Inc., Cary, North Carolina, USA.

White, G.C., and K.P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46(Supplement):120–138.

Williams, B.K., J.D. Nichols, and M.J. Conroy. 2002. Estimating abundance for closed populations with mark-recapture methods. Pages 289–332 *in* Analysis and management of animal populations. Academic Press, San Diego, California, USA.

SAS Code 1. Code for outputting MMDM variance-covariance values – Density: Step 2 (filename: GLM – Spp MMDM.sas).

```
options formdlim = '-' ps = 80 ls = 95;
data MMDM;
input Spp $ Session $ Island $ Island2 $ Site $ Habitat $ Survey $ AnimalID MoveDis;
/*      Island2: Guam vs. CNMI
        Survey: M-R = Mark-Recapture; REM = Removal
        AnimalID: Unique ID number (assigned by site) for each animal
        MoveDis: Distance in meters between subsequent recaptures           */

cards;
MM  1   Guam    Guam    MSRG    GR  M-R     002  17.68
MM  1   Guam    Guam    MSRG    GR  REM     002  27.95
…       (many more rows of data)
SM  9   Tinian  CNMI    LSUS    NF  REM     233  0
run;

proc sort;
 by Spp;
 where Spp='RR';                /* Select Spp */
 run;

proc sort;
 by Session;
 run;

proc glm data=MMDM;
 title '** MoveDis=Island GLM **';
 class Island;
 model MoveDis=Island /solution;
 lsmeans Island / noprint out=out1 cov;
 run;
proc print data=out1;
 run;

proc glm data=MMDM;
 title '** MoveDis=Guam vs. CNMI GLM **';
 class Island2;
 model MoveDis=Island2 /solution;
 lsmeans Island2 / noprint out=out2 cov;
 run;
proc print data=out2;
 run;

proc glm data=MMDM;
 title '** MoveDis=Habitat GLM **';
 class Habitat;
 model MoveDis=Habitat /solution;
 lsmeans Habitat / noprint out=out3 cov;
 run;
proc print data=out3;
 run;

proc glm data=MMDM;
 title '** MoveDis=Site GLM **';
 class Site;
 model MoveDis=Site /solution;
 lsmeans Site / noprint out=out4 cov;
 run;
proc print data=out4;
 run;
quit;
```

SAS Code 2. Code for obtaining variance-covariance matrix and model-averaging multiple variance-covariance matrices – Density: Step 2 (filename: IML – Mass and MMDM Var-Cov ModAvg.sas).

```
options formdlim='-' ps=80 ls=100;

/***  Model Averaging of Parameter Estimates and Variance-Covariance Matrices  ***/

Title '***  Model Averaging of MM Mass Var-Cov Matrix  ***';

Proc Import out= work.Mass_Est
  DATAFILE= "C:\Documents and Settings\wiewela\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML MM ModAvg Data.xls"
  DBMS=EXCEL2000 REPLACE;
  SHEET="'Mass_Est$'";
  GETNAMES=No;

Proc Import out=work.Mass_Isl
  DATAFILE= "C:\Documents and Settings\wiewela\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML MM ModAvg Data.xls"
  DBMS=EXCEL2000 REPLACE;
  SHEET="'Mass_Isl$'";
  GETNAMES=No;

Proc Import out=work.Mass_GCNMI
  DATAFILE= "C:\Documents and Settings\wiewela\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML MM ModAvg Data.xls"
  DBMS=EXCEL2000 REPLACE;
  SHEET="'Mass_GCNMI'";
  GETNAMES=No;

Proc Import out=work.Mass_Hab
  DATAFILE= "C:\Documents and Settings\wiewela\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML MM ModAvg Data.xls"
  DBMS=EXCEL2000 REPLACE;
  SHEET="'Mass_Hab'";
  GETNAMES=No;

Proc Import out=work.Mass_Site
  DATAFILE= "C:\Documents and Settings\wiewela\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML MM ModAvg Data.xls"
  DBMS=EXCEL2000 REPLACE;
  SHEET="'Mass_Site'";
  GETNAMES=No;

/***  Create vectors containing parameter estimates for each Model  ***/;

Proc IML;
Use Mass_Est;
 read all var {F1} into estModel1;
 read all var {F2} into estModel2;
 read all var {F3} into estModel3;
 read all var {F4} into estModel4;

/***  Create Var-Cov matrices for each Model  ***/
/***  This corresponds to 8 columns in the Var-Cov matrix, i.e., 8 estimates of interest  ***/

Use Mass_Isl; read all var {F1 F2 F3 F4 F5 F6 F7 F8} into varModel1;
Use Mass_GCNMI; read all var {F1 F2 F3 F4 F5 F6 F7 F8} into varModel2;
Use Mass_Hab; read all var {F1 F2 F3 F4 F5 F6 F7 F8} into varModel3;
Use Mass_Site; read all var {F1 F2 F3 F4 F5 F6 F7 F8} into varModel4;

/***  Enter model weights for each model  ***/

weights={0.90278 0.04914 0.00040 0.04768};

/*** Model Average the Estimates  ***/

Mean=weights[1]*estModel1+weights[2]*estModel2+weights[3]*estModel3+weights[4]*estModel4;

/*** Model Average the Var-Cov Matrices  ***/

VarAve=weights[1]*(varModel1+(estModel1-Mean)*(estModel1-Mean)`)
+weights[2]*(varModel2+(estModel2-Mean)*(estModel2-Mean)`)
+weights[3]*(varModel3+(estModel3-Mean)*(estModel3-Mean)`)
+weights[4]*(varModel4+(estModel4-Mean)*(estModel4-Mean)`);

/***  Print the model averaged betas and Var-Cov Matrix of the betas  ***/

print Mean VarAve;
quit;

/*************************************************************************************************************************************/

Title '***  Model Averaging of MM MMDM Var-Cov Matrix  ***';
```

```
Proc Import out= work.MMDM_Est
  DATAFILE= "C:\Documents and Settings\wiewela\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML MM ModAvg Data.xls"
  DBMS=EXCEL2000 REPLACE;
  SHEET="'MMDM_Est$'";
  GETNAMES=No;

Proc Import out=work.MMDM_Isl
  DATAFILE= "C:\Documents and Settings\wiewela\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML MM ModAvg Data.xls"
  DBMS=EXCEL2000 REPLACE;
  SHEET="'MMDM_Isl$'";
  GETNAMES=No;

Proc Import out=work.MMDM_GCNMI
  DATAFILE= "C:\Documents and Settings\wiewela\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML MM ModAvg Data.xls"
  DBMS=EXCEL2000 REPLACE;
  SHEET="'MMDM_GCNMI'";
  GETNAMES=No;

Proc Import out=work.MMDM_Hab
  DATAFILE= "C:\Documents and Settings\wiewela\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML MM ModAvg Data.xls"
  DBMS=EXCEL2000 REPLACE;
  SHEET="'MMDM_Hab'";
  GETNAMES=No;

/***  Create vectors containing parameter estimates for each Model  ***/;

Proc IML;
Use MMDM_Est;
 read all var {F1} into estModel1;
 read all var {F2} into estModel2;
 read all var {F3} into estModel3;

/***  Create Var-Cov matrices for each Model  ***/
/***  This corresponds to 8 columns in the Var-Cov matrix, i.e., 8 estimates of interest  ***/

Use MMDM_Isl; read all var {F1 F2 F3 F4 F5 F6 F7 F8} into varModel1;
Use MMDM_GCNMI; read all var {F1 F2 F3 F4 F5 F6 F7 F8} into varModel2;
Use MMDM_Hab; read all var {F1 F2 F3 F4 F5 F6 F7 F8} into varModel3;

/***  Enter model weights for each model  ***/

weights={0.71811 0.27183 0.01006};

/*** Model Average the Estimates  ***/

Mean=weights[1]*estModel1+weights[2]*estModel2+weights[3]*estModel3;

/*** Model Average the Var-Cov Matrices  ***/

VarAve=weights[1]*(varModel1+(estModel1-Mean)*(estModel1-Mean)`)
+weights[2]*(varModel2+(estModel2-Mean)*(estModel2-Mean)`)
+weights[3]*(varModel3+(estModel3-Mean)*(estModel3-Mean)`);

/***  Print the model averaged betas and Var-Cov Matrix of the betas  ***/

print Mean VarAve;
quit;
```

SAS Code 3. Code for creating new variance-covariance matrices – Density: Step 6; Biomass: Step 4 (filename: IML – Delta Method Var-Cov Matrices.sas).

```
options formdlim = '-' ps=80 ls=200;

/***  SAS Code for Delta Method Calculation of Density Estimate Variances  ***/

Title "RR Density Estimate Variance Determination by delta method";

/***  Import appropriate files  ***/

/***  Model-averaged N estimates from Program MARK  ***/

Proc Import out=work.RR_Nhat
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'RR_Nhat'";
 GETNAMES=no;

/***  Island-specific MMDM variance-covariance estimates from SAS Analysis  ***/

Proc Import out=work.RR_MMDM
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'RR_MMDM'";
 GETNAMES=no;

/***  Partial derivatives of ETA|MMDM  ***/

Proc Import out=work.RR_ETA
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'RR_Partial_A'";
 GETNAMES=no;

/***  Partial derivatives of Density|N-hat and Density|ETA  ***/

Proc Import out=work.RR_Density
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'RR_Partial_D'";
 GETNAMES=no;

/***  Site-specific Mass variance-covariance estimates from SAS Analysis  ***/

Proc Import out=work.RR_Mass_Site
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'RR_Mass_Site'";
 GETNAMES=no;

/***  Partial derivatives of Biomass|D and Biomass|Mass  ***/

Proc Import out=work.RR_Biomass
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'RR_Partial_B'";
 GETNAMES=no;

Proc IML;
 Use RR_Nhat;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9 F10 F11 F12 F13 F14 F15 F16 F17 F18} into RR_Nhat_var;
 Use RR_MMDM;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9 F10 F11 F12 F13 F14 F15 F16 F17 F18} into RR_MMDM_var;
 Use RR_ETA;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9 F10 F11 F12 F13 F14 F15 F16 F17 F18} into RR_ETA_var;
 Use RR_Density;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9 F10 F11 F12 F13 F14 F15 F16 F17 F18
 F19 F20 F21 F22 F23 F24 F25 F26 F27 F28 F29 F30 F31 F32 F33 F34 F35 F36} into RR_Density_var;
 Use RR_Mass_Site;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9 F10 F11 F12 F13 F14 F15 F16 F17 F18} into RR_Mass_var;
 Use RR_Biomass;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9 F10 F11 F12 F13 F14 F15 F16 F17 F18
 F19 F20 F21 F22 F23 F24 F25 F26 F27 F28 F29 F30 F31 F32 F33 F34 F35 F36} into RR_Biomass_var;

/***  Density  ***/

/***  Create Var-Cov Matrix for Effective Trapping Area (ETA)  ***/

 RR_VC_ETA=RR_ETA_var*RR_MMDM_var*RR_ETA_var`;
 print RR_VC_ETA;
```

```
/***  Create Matrix with RR_Nhat_var in the upper left of the matrix, and RR_VC_ETA in the lower right  ***/

 RR_VC_Nhat_ETA=block(RR_Nhat_var,RR_VC_ETA);
 print RR_VC_Nhat_ETA;

/***  Create Var-Cov Matrix for Density Estimates  ***/

 RR_VC_Density = RR_Density_var*RR_VC_Nhat_ETA*RR_Density_var`;
 print RR_VC_Density;

/***  Note that this is in animals/m2  Conversion to animals/ha  ***/

 C=I(18);
 Convert = C*10000;
 RR_VC_Density_ha = Convert*RR_VC_Density*Convert`;
 print RR_VC_Density_ha;

/***  Biomass  ***/

/***  Create Matrix with RR_VC_Density_ha in the upper left of the matrix, and RR_Mass_Site in the lower right  ***/

 RR_VC_Density_Mass=block(RR_VC_Density_ha,RR_Mass_var);
 print RR_VC_Density_Mass;

/***  Create Var-Cov Matrix for Biomass Estimates  ***/

 RR_VC_Biomass = RR_Biomass_var*RR_VC_Density_Mass*RR_Biomass_var`;
 print RR_VC_Biomass;

/***  Note that this is in g/ha  Conversion to kg/ha  ***/

 Convert2 = C*0.001;
 RR_VC_Biomass_kg_ha = Convert2*RR_VC_Biomass*Convert2`;
 print RR_VC_Biomass_kg_ha;
quit;

/******************************************************************************************************************************************************/

Title "MM Density Estimate Variance Determination by delta method";

/***  Import appropriate files  ***/

/***  Model-averaged N estimates from Program MARK  ***/

Proc Import out=work.MM_Nhat
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'MM_Nhat'";
 GETNAMES=no;

/***  Model-averaged MMDM estimates from SAS Analysis  ***/

Proc Import out=work.MM_MMDM_ModAvg
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'MM_MMDM_ModAvg'";
 GETNAMES=no;

/***  Partial derivatives of ETA|MMDM  ***/

Proc Import out=work.MM_ETA
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'MM_Partial_A'";
 GETNAMES=no;

/***  Partial derivatives of Density|N-hat and Density|ETA  ***/

Proc Import out=work.MM_Density
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'MM_Partial_D'";
 GETNAMES=no;

/***  Model-averaged Mass variance-covariance estimates from SAS Analysis  ***/

Proc Import out=work.MM_Mass_ModAvg
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'MM_Mass_ModAvg'";
 GETNAMES=no;

/***  Partial derivatives of Biomass|D and Biomass|Mass  ***/
```

```
Proc Import out=work.MM_Biomass
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'MM_Partial_B'";
 GETNAMES=no;

Proc IML;
 Use MM_Nhat;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8} into MM_Nhat_var;
 Use MM_MMDM_ModAvg;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8} into MM_MMDM_var;
 Use MM_ETA;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8} into MM_ETA_var;
 Use MM_Density;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9 F10 F11 F12 F13 F14 F15 F16} into MM_Density_var;
 Use MM_Mass_ModAvg;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8} into MM_Mass_var;
 Use MM_Biomass;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9 F10 F11 F12 F13 F14 F15 F16} into MM_Biomass_var;

/***  Density  ***/

/***  Create Var-Cov Matrix for ETA  ***/

 MM_VC_ETA=MM_ETA_var*MM_MMDM_var*MM_ETA_var`;
 print MM_VC_ETA;

/***  Create Matrix with MM_Nhat_var in the upper left of the matrix, and MM_VC_ETA in the lower right  ***/

 MM_VC_Nhat_ETA=block(MM_Nhat_var,MM_VC_ETA);
 print MM_VC_Nhat_ETA;

/***  Create Var-Cov Matrix for Density Estimates  ***/

 MM_VC_Density = MM_Density_var*MM_VC_Nhat_ETA*MM_Density_var`;
 print MM_VC_Density;

/***  Note that this is in animals/m2  Conversion to animals/ha  ***/

 C=I(8);
 Convert = C*10000;
 MM_VC_Density_ha = Convert*MM_VC_Density*Convert`;
 print MM_VC_Density_ha;

/***  Biomass  ***/

/***  Create Matrix with MM_VC_Density_ha in the upper left of the matrix, and MM_Mass_Site in the lower right  ***/

 MM_VC_Density_Mass=block(MM_VC_Density_ha,MM_Mass_var);
 print MM_VC_Density_Mass;

/***  Create Var-Cov Matrix for Biomass Estimates  ***/

 MM_VC_Biomass = MM_Biomass_var*MM_VC_Density_Mass*MM_Biomass_var`;
 print MM_VC_Biomass;

/***  Note that this is in g/ha  Conversion to kg/ha  ***/

 Convert2 = C*0.001;
 MM_VC_Biomass_kg_ha = Convert2*MM_VC_Biomass*Convert2`;
 print MM_VC_Biomass_kg_ha;
quit;

/****************************************************************************************************************************************************************/

Title "SM Density Estimate Variance Determination by delta method";

/***  Import appropriate files  ***/

/***  Model-averaged N estimates from Program MARK  ***/

Proc Import out=work.SM_Nhat
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'SM_Nhat'";
 GETNAMES=no;

/***  Island-specific MMDM estimates from SAS Analysis  ***/

Proc Import out=work.SM_MMDM
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'SM_MMDM'";
```

```
 GETNAMES=no;

/***  Partial derivatives of ETA|MMDM  ***/

Proc Import out=work.SM_ETA
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'SM_Partial_A'";
 GETNAMES=no;

/***  Partial derivatives of Density|N-hat and Density|ETA  ***/

Proc Import out=work.SM_Density
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'SM_Partial_D'";
 GETNAMES=no;

/***  Habitat-specific Mass variance-covariance estimates from SAS Analysis  ***/

Proc Import out=work.SM_Mass_Hab
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'SM_Mass_Hab'";
 GETNAMES=no;

/***  Partial derivatives of Biomass|D and Biomass|Mass  ***/

Proc Import out=work.SM_Biomass
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'SM_Partial_B'";
 GETNAMES=no;

Proc IML;
 Use SM_Nhat;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9} into SM_Nhat_var;
 Use SM_MMDM;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9} into SM_MMDM_var;
 Use SM_ETA;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9} into SM_ETA_var;
 Use SM_Density;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9 F10 F11 F12 F13 F14 F15 F16 F17 F18} into SM_Density_var;
 Use SM_Mass_Hab;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9} into SM_Mass_var;
 Use SM_Biomass;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9 F10 F11 F12 F13 F14 F15 F16 F17 F18} into SM_Biomass_var;

/***  Density  ***/

/***  Create Var-Cov Matrix for ETA  ***/

 SM_VC_ETA=SM_ETA_var*SM_MMDM_var*SM_ETA_var`;
 print SM_VC_ETA;

/***  Create Matrix with SM_Nhat_var in the upper left of the matrix, and SM_VC_ETA in the lower right  ***/

 SM_VC_Nhat_ETA=block(SM_Nhat_var,SM_VC_ETA);
 print SM_VC_Nhat_ETA;

/***  Create Var-Cov Matrix for Density Estimates  ***/

 SM_VC_Density = SM_Density_var*SM_VC_Nhat_ETA*SM_Density_var`;
 print SM_VC_Density;

/***  Note that this is in animals/m2  Conversion to animals/ha  ***/

 C=I(9);
 Convert = C*10000;
 SM_VC_Density_ha = Convert*SM_VC_Density*Convert`;
 print SM_VC_Density_ha;

/***  Biomass  ***/

/***  Create Matrix with SM_VC_Density_ha in the upper left of the matrix, and SM_Mass_Site in the lower right  ***/

 SM_VC_Density_Mass=block(SM_VC_Density_ha,SM_Mass_var);
 print SM_VC_Density_Mass;

/***  Create Var-Cov Matrix for Biomass Estimates  ***/

 SM_VC_Biomass = SM_Biomass_var*SM_VC_Density_Mass*SM_Biomass_var`;
 print SM_VC_Biomass;
```

```
/***  Note that this is in g/ha  Conversion to kg/ha  ***/

 Convert2 = C*0.001;
 SM_VC_Biomass_kg_ha = Convert2*SM_VC_Biomass*Convert2`;
 print SM_VC_Biomass_kg_ha;
quit;

/*******************************************************************************************************************************************************************************************/

/***  The top model for SM has 99% of the weight. It may be more appropriate
      to use var-cov from this model rather than the model-averaged var-cov. ***/

Title "SM (Top Model) Density Estimate Variance Determination by delta method";

/***  Import appropriate files  ***/

/***  Model-averaged N estimates from Program MARK  ***/

Proc Import out=work.SM_Nhat_TopMod
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'SM_Nhat_TopMod'";
 GETNAMES=no;

/***  Island-specific MMDM estimates from SAS Analysis  ***/

Proc Import out=work.SM_MMDM
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'SM_MMDM'";
 GETNAMES=no;

/***  Partial derivatives of ETA|MMDM  ***/

Proc Import out=work.SM_ETA
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'SM_Partial_A'";
 GETNAMES=no;

/***  Partial derivatives of Density|N-hat and Density|ETA  ***/

Proc Import out=work.SM_Density
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'SM_Partial_D'";
 GETNAMES=no;

/***  Habitat-specific Mass variance-covariance estimates from SAS Analysis  ***/

Proc Import out=work.SM_Mass_Hab
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'SM_Mass_Hab'";
 GETNAMES=no;

/***  Partial derivatives of Biomass|D and Biomass|Mass  ***/

Proc Import out=work.SM_Biomass
 DATAFILE= "C:\Documents and Settings\wiewela.FORT\My Documents\USGS BTS Project\Sys Rod Mon\Data Analyses\SAS\IML VarCov Data.xls"
 DBMS=EXCEL2000 REPLACE;
 SHEET="'SM_Partial_B'";
 GETNAMES=no;

Proc IML;
 Use SM_Nhat_TopMod;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9} into SM_Nhat_TopMod_var;
 Use SM_MMDM;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9} into SM_MMDM_var;
 Use SM_ETA;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9} into SM_ETA_var;
 Use SM_Density;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9 F10 F11 F12 F13 F14 F15 F16 F17 F18} into SM_Density_var;
 Use SM_Mass_Hab;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9} into SM_Mass_var;
 Use SM_Biomass;
 read all var {F1 F2 F3 F4 F5 F6 F7 F8 F9 F10 F11 F12 F13 F14 F15 F16 F17 F18} into SM_Biomass_var;

/***  Density  ***/

/***  Create Var-Cov Matrix for ETA  ***/

 SM_VC_ETA=SM_ETA_var*SM_MMDM_var*SM_ETA_var`;
 print SM_VC_ETA;
```

```
/***  Create Matrix with SM_Nhat_TopMod_var in the upper left of the matrix, and SM_VC_ETA in the lower right  ***/

 SM_VC_Nhat_TopMod_ETA=block(SM_Nhat_TopMod_var,SM_VC_ETA);
 print SM_VC_Nhat_TopMod_ETA;

/***  Create Var-Cov Matrix for Density Estimates  ***/

 SM_VC_Density = SM_Density_var*SM_VC_Nhat_TopMod_ETA*SM_Density_var`;
 print SM_VC_Density;

/***  Note that this is in animals/m2  Conversion to animals/ha  ***/

 C=I(9);
 Convert = C*10000;
 SM_VC_Density_ha = Convert*SM_VC_Density*Convert`;
 print SM_VC_Density_ha;

/***  Biomass  ***/

/***  Create Matrix with SM_VC_Density_ha in the upper left of the matrix, and SM_Mass_Site in the lower right  ***/

 SM_VC_Density_Mass=block(SM_VC_Density_ha,SM_Mass_var);
 print SM_VC_Density_Mass;

/***  Create Var-Cov Matrix for Biomass Estimates  ***/

 SM_VC_Biomass = SM_Biomass_var*SM_VC_Density_Mass*SM_Biomass_var`;
 print SM_VC_Biomass;

/***  Note that this is in g/ha  Conversion to kg/ha  ***/

 Convert2 = C*0.001;
 SM_VC_Biomass_kg_ha = Convert2*SM_VC_Biomass*Convert2`;
 print SM_VC_Biomass_kg_ha;
quit;
```

SAS Code 4. Code for outputting Mass variance-covariance values – Biomass: Step 2 (filename: GLM – Spp Mass for Biomass Calculations.sas).

```
options formdlim = '-' ps = 80 ls = 95;
data MASS;
input Spp $ Exclude $ Survey $ Island $ Island2 $ Habitat $ Site $ AnimalID $ Sex $ Age $ RepCond $ Mass;
/*    Exclude:   1 = all captured individuals (M-R and REM)
                 2 = marked animals recaptured during REM
      Survey: M-R = Mark-Recapture; REM = Removal
      Island2: Guam vs. CNMI
      Sex: Male = 1, Female = 0
      Age: Adult = 1, Juv = 0
      RepCond: Mature = 1, Immature = 0
      AnimalID: Unique ID number (assigned by site) for each animal      */

cards;
MM   1     M-R  Saipan    CNMI     GR   ACHU     ACHU_MM_001 0    1    0    13
MM   1     M-R  Saipan    CNMI     GR   ACHU     ACHU_MM_053 0    1    0    9.75
…          (many more rows of data)
SM   2     REM Saipan     CNMI     HET  SPOR     SPOR_SM_406  1    1    1    32
run;

proc sort;
 by Spp;
 where Spp='RR';                    /* Select Spp */
 run;

proc sort;
 by Survey Site;
 run;

proc glm data=MASS;
 title '** M-R Mass=Island GLM **';
 where Survey='M-R';
 class Island;
 model Mass=Island /solution;
 lsmeans Island / noprint out=out1 cov;
 run;
proc print data=out1;
 run;

proc glm data=MASS;
 title '** M-R Mass=Guam vs. CNMI GLM **';
 where Survey='M-R';
 class Island2;
 model Mass=Island2 /solution;
 lsmeans Island2 / noprint out=out2 cov;
 run;
proc print data=out2;
 run;

proc glm data=MASS;
 title '** M-R Mass=Habitat GLM **';
 where Survey='M-R';
 class Habitat;
 model Mass=Habitat /solution;
 lsmeans Habitat / noprint out=out3 cov;
 run;
proc print data=out3;
 run;

proc glm data=MASS;
 title '** M-R Mass=Site GLM **';
 where Survey='M-R';
 class Site;
 model Mass=Site /solution;
 lsmeans Site / noprint out=out4 cov;
 run;
proc print data=out4;
 run;
quit;
```

APPENDIX 1C. Comparison of density estimates from Programs MARK and DENSITY

Density estimation from grid-based, mark-recapture sampling is complicated by difficulties in quantifying the true area sampled. Using the area of the grid itself (often referred to as naïve or nominal density) does not account for the unknown boundary area used by animals living along the edges of the grid, thereby underestimating the area sampled and producing a positively biased density estimate (Anderson et al. 1983, Wilson and Anderson 1985, Efford 2004). Alternative methods for determining the true area sampled by the grid, often referred to as the effective trapping area (ETA), include adding a boundary strip equal to ½ the average home range to the trapping grid (Dice 1938), using captures on nested subgrids within the grid to estimate the size of the boundary strip (Otis et al. 1978), or using a measure of animal movement, such as the mean maximum distance moved (MMDM), to determine the ETA (Wilson and Anderson 1985). A number of conceptual and procedural arguments against these ETA estimation methods have been presented in recent years (Anderson et al. 1983, Wilson and Anderson 1985). The primary objections relate to the influence of trap spacing and the number of recaptures on estimates of animal movements, and therefore estimates of ETA (Wilson and Anderson 1985). An alternative approach for determining density, using the recently developed software package Program DENSITY (Efford 2004), attempts to avoid the issue of determining ETA altogether. Instead, DENSITY uses an inverse prediction procedure to find a hypothetical density of animals, given the sampling methodology employed by the researcher, which could produce the capture and recapture results obtained during sampling.

We compared density estimates generated using the inverse prediction procedure in DENSITY to more traditional density estimates generated by dividing mark-recapture livetrapping abundance estimates generated using Program MARK by estimated ETA. Because we did not know true small mammal density on our sites, we could not directly evaluate the accuracy or precision of density estimates derived from DENSITY and MARK. Instead, our evaluation of these methods was based on species-specific comparisons of:

1. density estimates and variances, under the assumption that estimates with lower variance provide more useful information than estimates with higher variance, and

2. the ability of each method to produce density estimates from field data, including datasets with limited captures and recaptures.

Density Estimation Methods

Program DENSITY: Site- and species-specific density estimates were generated in DENSITY 3.3. DENSITY avoids issues associated with estimating the ETA by using an inverse prediction procedure to find a hypothetical density of animals, given the sampling methodology employed by the researcher, which would produce the mark-recapture trapping results obtained by the researcher. DENSITY allows the researcher to select among several estimators, including both traditional estimators such as M_0, M_t, M_b, and M_h (Otis et al. 1978, Burnham and Overton 1978) as well more recently developed estimators, including $M_{h\ \text{Chao}}$ and $M_{h\ \text{Chao modified}}$ (Chao 1987), $M_{h\ \text{2-point mixture}}$ (Pledger 2000), $M_{h\ \text{Beta-binomial}}$ (Dorazio and Royle 2003), and $M_{th\ \text{Chao coverage 1 and 2}}$ (Lee and Chao 1994). Efford (2004) states the choice of estimator may be relatively unimportant for density estimation, but goes on to recommend $M_{th\ \text{Chao coverage 1}}$ or $M_{th\ \text{Chao coverage 2}}$ (Lee and Chao 1994) as estimators that have proven especially robust to heterogeneity in capture probability in many field situations. However, to facilitate comparison between density estimates, we chose to select the estimator in DENSITY which most closely approximated the top model identified for each species during MARK modeling.

Program MARK: Site-and species-specific abundance estimates were generated using the Huggins (1989, 1991) conditional likelihood closed population model available in MARK 4.3 (White and Burnham 1999; see full description of modeling approach in Methods of Chapter 1). These estimates were combined with estimates of ETA to generate density estimates. ETA was calculated as the total area encompassed by the nominal trapping area (1.56 ha) plus a boundary strip equal to ½ the MMDM between captures (Wilson and Anderson 1985). MMDM was estimated separately for each species using multiple analysis of variance models (Proc GLM, SAS Institute 2003), where the candidate models allowed MMDM to vary by island, Guam and RST (Rota, Saipan, and Tinian combined), and habitat. A site-specific MMDM model was not considered because of concerns that limited movement data for several sites. Candidate MMDM models were evaluated based on AIC_c scores and MMDM estimates and variances were model-averaged as necessary to account for model selection uncertainty (Burnham and Anderson 2002:150). For each species, mark-recapture and removal sampling data were combined to increase MMDM sample size, after first verifying that movements were not significantly different between sampling methods. Removal sampling movement observations occurred when animals originally captured during mark-recapture were recaptured during removal sampling. The combination of mark-recapture and removal sampling datasets increased movement sample size by 58% for *Rattus rattus*, 45% for *Suncus murinus*, and 41% for *Mus musculus*. Variance-covariance matrices of the density and ETA estimates were computed using the delta method (Seber 2002), from which we were able to determine the variance of derived density estimates (Appendix 1B).

Evaluation of Density Estimates

Density estimates for *R. rattus*, *S. murinus*, and *M. musculus* were generated in Program DENSITY using the M_t estimator (*R. rattus*) and the $M_{\text{th Chao coverage 1}}$ estimator (*S. murinus* and *M. musculus*), as these estimators most closely approximated the structure of the top model for each species in our Program MARK analyses (see Table 7 in main body of Chapter 1). In contrast to modeling in MARK, which allowed us to generate density estimates for all sites where *R. rattus* (Table C.1), *S. murinus* (Table C.2), and *M. musculus* (Table C.3) were captured, DENSITY produced estimates for only 44.4% of sites with *R. rattus* captures, 55.6% of sites with *S. murinus* captures, and 37.5% of sites with *M. musculus* captures under the default inverse prediction parameters. While it was not surprising that DENSITY failed to produce estimates for sites with low captures and recaptures (e.g., most of the Guam sites), it also failed to produce estimates for several sites with large numbers of captures and recaptures such as RAPF (*R. rattus* $M_{t+1} = 106$, $n_{.} = 146$) and KAST (*R. rattus* $M_{t+1} = 106$, $n_{.} = 132$). Modification of inverse prediction parameters, including increasing the number of simulation replicates, decreasing the required precision, and excluding “extreme” movement observations (as recommended in the DENSITY Help file) improved the success rate of the inverse prediction procedure, such that density estimates were generated for 2 additional sites for both *R. rattus* (Table C.1) and *M. musculus* (Table C.3), including all sites with seemingly adequate numbers of captures and recaptures. Modification of inverse prediction procedures did not result in additional successful *S. murinus* density estimates (Table C.2).

Site-specific comparisons between DENSITY (for sites with estimates) and MARK revealed essentially equivalent density estimates (Tables C.1, C.2, C.3). Discrepancies between density estimates were generally small, with no apparent pattern to which method produced the greater estimate or variance. More importantly, on a site-specific basis, the 95% CIs overlapped for all estimates, and in all but 4 occasions (1 *R. rattus*, 2 *M. musculus*, and 1 *S. murinus*) the density estimate from DENSITY was included in the 95% CI from MARK, and vice versa (Tables C.1, C.2, C.3). However, DENSITY could not generate estimates for sites with sparse capture or recapture data, limiting the utility of this software. This limitation is thought to be addressed in the updated version of the software (Version 4.0) to be released in Fall 2007.

Literature Cited

Anderson, D.R., K.P. Burnham, G.C. White, and D.L. Otis. 1983. Density estimation of small-mammal populations using a trapping web and distance sampling methods. Ecology 64:674–680.

Burnham, K.P., and D.R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Second Edition. Springer, New York, New York, USA.

Burnham, K.P., and W.S. Overton. 1978. Estimation of the size of a closed population when capture probabilities vary among animals. Biometrika 65:625–633.

Chao, A. 1987. Estimating the population size for capture-recapture data with unequal catchability. Biometrics 43:783–791.

Dice, L.R. 1938. Some census methods for mammals. Journal of Wildlife Management 2:119–130.

Dorazio, R.M., and J.A. Royle. 2003. Mixture models for estimating the size of a closed population when capture rates vary among individuals. Biometrics 59:351–364.

Efford, M.G. 2004. Density estimation in live-trapping studies. Oikos 106:598–610.

Huggins, R.M. 1989. On the statistical analysis of capture-recapture experiments. Biometrika 76:133–140.

Huggins, R.M. 1991. Some practical aspects of a conditional likelihood approach to capture experiments. Biometrics 47:725–732.

Lee, S.-M., and A. Chao. 1994. Estimating population size via sample coverage for closed capture-recapture models. Biometrics 50:88–97.

Otis, D.L., K.P. Burnham, G.C. White, and D.R. Anderson. 1978. Statistical inference from capture data on closed animal populations. Wildlife Monographs 62:1–135.

Pledger, S. 2000. Unified maximum likelihood estimates for closed capture-recapture models using mixtures. Biometrics 56:434–442.

Seber, G.A.F. 2002. The estimation of animal abundance and related parameters. Second Edition. The Blackburn Press, New Jersey, USA.

White, G.C., and K.P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46(Supplement):120–138.

Wilson, K.R., and D.R. Anderson. 1985. Evaluation of two density estimators of small mammal population size. Journal of Mammalogy 66:13–21.

Table C.1. *Rattus rattus* density estimates ($\hat{D}$), standard errors (SE), and 95% confidence intervals (95% CI) generated from mark-recapture livetrapping of grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Rota, Saipan, and Tinian, 2005–2007, using Program MARK 4.3 and Program DENSITY 3.3. For Program DENSITY, “Failed” indicates that the program could not generate a density estimate.

		Program MARK			Program DENSITY		
Site	Habitat	$\hat{D}$	SE	95% CI	$\hat{D}$	SE	95% CI
Guam[a]							
MSRG	grassland	15.9	3.7	8.6–23.2	Failed		
ASMF	*Leucaena* forest	2.6	0.7	1.2–4.0	Failed		
CP05	*Leucaena* forest	2.6	1.0	0.6–4.6	Failed		
CP06		15.3	3.5	8.4–22.2	12.7	5.7	5.5–29.6
GSYF	*Leucaena* forest	8.9	2.3	4.4–13.4	Failed		
GAHF	mixed	0.7	0.5	0–1.7	Failed		
Rota							
SABA	grassland	73.2	11.9	49.9–96.5	71.1	15.1	47.1–107.3
GAON	*Leucaena* forest	36.0	6.7	22.9–49.1	23.2	5.7	14.5–37.2
RAPF	mixed	95.8	16.1	64.2–127.4	80.2	11.2	61.1–105.3
ASAK	native forest	9.2	2.2	4.9–13.5	Failed		
Saipan							
ACHU	grassland	33.0	6.4	20.5–45.5	18.3	4.8	11.1–30.3
OBYT	*Leucaena* forest	41.4	8.1	25.5–57.3	60.0	28.6	24.7–145.8
SAEN	mixed	6.9	2.1	2.8–11.0	Failed		
SPOR	mixed	25.1	5.4	14.5–35.7	31.0	15.1	12.5–76.6
LATT	native forest	21.6	5.1	11.6–31.6	Failed		
Tinian							
KAST	grassland	99.9	17.9	64.8–135.0	92.7	17.7	64.0–134.1
ABLE	*Leucaena* forest	44.0	7.3	29.7–58.3	30.6	5.8	21.3–44.1
LSUS	native forest	75.1	13.6	48.4–101.8	103.3	32.0	57.0–187.0

[a] Zero *R. rattus* captured at 4 sites (2 *Leucaena* forest, 2 native forest).

Table C.2. *Suncus murinus* density estimates ($\hat{D}$), standard errors (SE), and 95% confidence intervals (95% CI) generated from mark-recapture livetrapping of grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Saipan, and Tinian, 2005–2007, using Program MARK 4.3 and Program DENSITY 3.3. For Program DENSITY, "Failed" indicates that the program could not generate a density estimate. *S. murinus* was not captured on Rota.

		Program MARK			Program DENSITY		
Site	Habitat	$\hat{D}$	SE	95% CI	$\hat{D}$	SE	95% CI
Guam[a]							
MSRG	grassland	8.5	2.5	3.6–13.4	6.0	4.4	1.7–21.6
Saipan							
ACHU	grassland	13.4	3.7	6.1–20.7	5.8	2.3	2.8–12.2
OBYT	*Leucaena* forest	31.6	10.2	11.6–51.6	29.4	10.3	15.1–57.2
SAEN	mixed	32.9	9.6	14.1–51.7	29.5	9.3	16.2–57.7
SPOR	mixed	6.3	2.2	2.0–10.6	Failed		
LATT	native forest	14.0	5.2	3.8–24.2	Failed		
Tinian							
KAST	grassland	8.9	2.5	4.0–13.8	Failed		
ABLE	*Leucaena* forest	73.7	20.1	34.3–113.1	99.5	26.4	59.7–165.9
LSUS	native forest	32.8	9.6	14.0–51.6	Failed		

[a] Zero *S. murinus* captured at 9 sites (6 *Leucaena* forest, 1 mixed habitat, and 2 native forest).

Table C.3. *Mus musculus* density estimates ($\hat{D}$), standard errors (SE), and 95% confidence intervals (95% CI) generated from mark-recapture livetrapping of grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Rota, Saipan, and Tinian, 2005–2007, using Program MARK 4.3 and Program DENSITY 3.3. For Program DENSITY, "Failed" indicates that the program could not generate a density estimate.

		Program MARK			Program DENSITY		
Site	Habitat	$\hat{D}$	SE	95% CI	$\hat{D}$	SE	95% CI
Guam[a]							
MSRG	grassland	7.2	1.7	3.9–10.5	3.8	1.9	1.5–9.4
Rota							
SABA	grassland	20.7	5.0	10.9–30.5	21.8	17.5	5.5–86.9
GAON	*Leucaena* forest	16.0	4.1	8.0–24.0	19.9	16.1	5.0–80.1
RAPF	mixed	26.5	6.3	14.2–38.8	24.6	8.4	12.8–47.1
ASAK	native forest	0.8	0.6	0–2.0	Failed		
Saipan[b]							
ACHU	grassland	36.5	8.1	20.6–52.4	25.4	4.1	18.5–34.9
OBYT	*Leucaena* forest	1.5	0.7	0.1–2.9	Failed		
Tinian[c]							
KAST	grassland	8.2	2.7	2.9–13.5	Failed		

[a] Zero *M. musculus* captured at 9 sites (6 *Leucaena* forest, 1 mixed habitat, and 2 native forest).
[b] Zero *M. musculus* captured at 3 sites (2 mixed habitat and 1 native forest).
[c] Zero *M. musculus* captured at 2 sites (1 *Leucaena* forest and 1 native forest).

CHAPTER 2: EVALUATING ABUNDANCE ESTIMATES AND THE ASSUMPTIONS OF A COUNT-BASED INDEX OF ABUNDANCE FOR SMALL MAMMALS

INTRODUCTION

One of the most common questions in ecological research or management is: "How many are there?" Ideally, this question would be answered with a complete count, or census, of the population of interest. In practice, many species have life history traits which complicate population censuses. For example, most small mammal species are cryptic, nocturnal, and have spatially and temporally variable densities which generally preclude the use of a population census. Thus, small mammal researchers are often able to sample only a portion of the population, from which they make inference about the entire population of interest (Lancia et al. 2005). This approach requires that researchers select the most appropriate and reliable sampling and data analysis method(s) available, based on research objectives and the ecology of target species (Pollock et al. 2002, Witmer 2005).

Two of the most commonly used small mammal sampling methods are livetrapping and snaptrapping (Lancia et al. 2005). The primary difference between these methods is that livetrapping yields live animal captures, whereas captured animals are killed during snaptrapping. The usefulness of snaptrapping for long-term monitoring of small mammal abundance, distribution, or diversity is questionable, as the sampling method is highly disruptive to the study population (Sullivan et al. 2003) and could confound effects of interest. Further, direct comparisons between livetrapping and snaptrapping (or analogous methods such as seining and electrofishing) have produced equivocal or conflicting results, leading to uncertainty about the utility of methods which removal animals from the study population (Stickel 1946a, Yang et al. 1970, Bohlin and Sundstrom 1977, Rodgers et al. 1992, Sullivan et al. 2003, Rosenberger and Dunham 2005). Despite these potential issues, snaptrapping is often used alone or in conjunction with other methods such as pitfall sampling to study small mammal demography, habitat preferences, or response to management activities (e.g., Roberts and Craig 1990, Mitchell et al. 1995, Christian et al. 1997, Bellows et al. 2001, Ecke et al. 2002), perhaps because the method is thought to be faster and cheaper than livetrapping.

Both livetrapping and snaptrapping can provide data suitable for abundance estimation, although the removal of animals from the population during snaptrapping limits applicable estimation methods (Otis et al. 1978, White et al. 1982). During livetrapping, captured animals can be uniquely marked and released back into the study population, where they are available for recapture during subsequent sampling occasions, a process commonly referred to as mark-recapture (also capture-recapture or capture-mark-recapture) sampling. If continued over multiple sampling occasions, the capture history of marked and unmarked individuals on each sampling occasion can be used to generate a mark-recapture abundance estimate ($\hat{N}_{\mathrm{M-R}}$). For example, over 2 sampling occasions

$$\hat{N}_{\mathrm{M-R}} = \frac{n_1 n_2}{m_2},$$

where n_1 is the number of individuals captured and marked on occasion 1, n_2 is the number of individuals captured on occasion 2, and m_2 is the number of marked individuals recaptured on occasion 2 (Pollock et al. 1990). This most basic mark-recapture abundance estimator (the Lincoln-Petersen estimator) is subject to several assumptions: population closure (no births, deaths, emigration or immigration during sampling), no loss or misidentification of unique animal identifiers, and equal probability of capture for all animals on each sampling occasion (Williams et al. 2002). While the first assumption, population closure, can be relaxed if open population models are considered (Jolly 1965, Seber 1965), the scope of this paper is restricted to closed populations. Extending sampling over additional occasions allows the use of more sophisticated mark-recapture abundance estimators which relax the assumption of equal capture probability, which is unlikely to hold for wild populations and can cause bias in Lincoln-Peterson abundance estimates (Carothers 1973, Otis et al. 1978). These estimators have the general form

$$\hat{N}_{\mathrm{M-R}} = \frac{M_{\mathrm{t+1}}}{\hat{p}},$$

where $M_{\mathrm{t+1}}$ is the count of unique animals captured during sampling and $\hat{p}$ is the estimated cumulative capture probability, defined as the proportion of the total population captured and estimated from the relationship between new captures and recaptures over multiple sampling occasions (Nichols 1992). Like the Lincoln-Petersen estimator, these estimation methods are subject to the assumption of population closure and no tag loss or misidentification, but also add the assumption that capture probability is appropriately modeled (Williams et al. 2002). Following the nomenclature of Otis et al. (1978), capture probability can be modeled as being constant (M_0) or allowed to vary over time (M_t), between marked and unmarked individuals (i.e., behavioral response to capture; M_b), between all individuals (i.e., individual heterogeneity; M_h), or combinations of these factors (M_{tb}, M_{bh}, M_{th}, M_{tbh}). Recent advances in estimator development permit even more complex models, such as those incorporating mixture models (Norris and Pollock 1996, Pledger 2000) or covariates (Pollock et al. 1984; Huggins 1989, 1991). For a complete discussion of the history and development of mark-recapture abundance estimation methods, refer to Otis et al. 1978, White et al. 1982, Nichols 1992, Williams et al. 2002, and Lancia et al. 2005.

In contrast to livetrapping, the removal of captured animals from the population during snaptrapping eliminates the possibility of recaptures. This constraint limits suitable abundance estimation methods, as fewer data are available to model capture probability relative to mark-recapture sampling (Otis et al. 1978, White et al. 1982). Over 2 sampling occasions, snaptrapping can be used to generate removal abundance estimates ($\hat{N}_{\text{REM}}$) of the general form

$$\hat{N}_{\text{REM}} = \frac{n_1^2}{n_1 - n_2},$$

where n_1 is the number of individuals removed on occasion 1 and n_2 is the number of individuals removed on occasion 2 (Pollock 1991). Note that the form of this estimator requires that $n_1 > n_2$ (i.e., captures must decline from the first to the second sampling occasion to produce a viable estimate). Sampling over additional occasions allows $\hat{N}_{\text{REM}}$ to be generalized as

$$\hat{N}_{\text{REM}} = \frac{M_{t+1}}{\hat{p}},$$

where p is now estimated from the change in number of animals captured over successive sampling occasions (Pollock 1991). The generalized estimator does not relax the requirement that captures decline over successive sampling occasions; in fact, without this decline one cannot accurately estimate p or generate robust and unbiased $\hat{N}_{\text{REM}}$ from snaptrapping data (Otis et al. 1978, White et al. 1982, Pollock 1991). Early removal estimation methods assumed that capture probability was constant for all animals and across all sampling occasions (Zippin 1956, 1958). More recent estimation methods relax this assumption somewhat, but all removal abundance estimation methods assume population closure (outside of known removals associated with sampling) and that capture probability is modeled correctly. Recommended removal abundance estimators include M_b and M_{bh} (Otis et al. 1978), where the removal of animals is considered an extreme behavioral response to initial capture (i.e., recapture probability = 0). As with mark-recapture abundance estimation methods, recent advances allow more sophisticated models incorporating mixture models (Norris and Pollock 1996, Pledger 2000) or covariates (Pollock et al. 1984; Huggins 1989, 1991). The applicability of various estimators (M_b and M_{bh}) to both livetrapping and snaptrapping data illustrates an important and sometimes unrecognized point: livetrapping data can also be used to generate $\hat{N}_{\text{REM}}$ if recaptures are not considered during the estimation process (Otis et al. 1978). For a more thorough discussion of the history and development of removal abundance estimation methods, refer to Otis et al. 1978, White et al. 1982, and Williams et al. 2002.

Many researchers using either livetrapping or snaptrapping choose to forgo abundance estimation altogether and instead report only count-based indices of abundance, such as the number of individuals captured (M_{t+1}; Otis et al. 1978) or the captures per unit effort (CPUE; White et al. 1982). Index proponents suggest that indices require less analytical expertise and are subject to fewer or less restrictive assumptions than abundance estimation methods (Engeman 2005), while providing data suitable for relative comparisons between populations across space and time (Engeman 2003, 2005). Careful consideration of conditions surrounding the application of indices, however, suggests that they are not without potentially restrictive assumptions. For example, the assumption of population closure is critical for any comparison between populations, regardless of the metric the comparison is based on. More importantly, the inherent assumption of any index is that the relationship between the index and true abundance is monotonic, proportional, and constant across space and time (Nichols 1992; Anderson 2001, 2003). Unfortunately, few researchers test this assumption, either by evaluating indices against known populations (Conn et al. 2006), double sampling using both an index and a more rigorous sampling technique (Eberhardt and Simmons 1987, Slade and Blair 2000), or through simulation (McKelvey and Pearson 2001). Failure to validate indices limits their utility for making inference about animal populations (Nichols 1992; Anderson 2001, 2003).

As part of a larger study of introduced small mammal populations in the Mariana Islands, we evaluated livetrapping and snaptrapping in terms of sampling efficiency (the cost and time associated with implementing each method) and numerical estimation. Based on experience with these sampling methods, our *a priori* expectation was that snaptrapping would be both cheaper and faster than livetrapping. To evaluate numerical estimates, we compared site- and species-specific abundance estimates and count-based indices of abundance (M_{t+1} and CPUE) generated from each sampling method. *A priori*, we expected mark-recapture abundance estimates generated from livetrapping data to be more precise (i.e., have a smaller coefficient of variation and narrower 95% confidence intervals) than removal abundance estimates or count-based indices generated from either livetrapping or snaptrapping data. We based this hypothesis on the amount of information used to generate each estimate or index ($\hat{N}_{\mathrm{M-R}} > \hat{N}_{\mathrm{REM}} > \mathrm{CPUE} > M_{t+1}$) as discussed above, as well as the anticipated robustness of each metric to violations of critical assumptions, such as population closure, and sources of variation in capture probability. For example, temporal variation in capture probability would invalidate both M_{t+1} and CPUE, as the relationship between the indices and true abundance would no longer be constant across time. More importantly, these indices provide no means of recognizing temporal variation in capture probability, and therefore no criteria for determining the validity of the index. Removal abundance estimate methods are more robust to temporal variation in capture probability than indices, although such variation is likely to reduce the accuracy and precision of these estimates (Otis et al. 1978); if captures do not decline over time, however, removal abundance estimation methods will produce inaccurate and imprecise estimates. In contrast, mark-recapture abundance estimation methods are well suited for identifying and accounting for temporal variation in capture probability if sufficient data are available for modeling (Otis et al. 1978). Further discussion of the robustness of each metric to various sources of capture probability variation is provided in Table 1. Finally, because indices are frequently generated from short duration sampling events, we investigated the effects of sampling duration on index performance by evaluating M_{t+1} and CPUE generated from 1, 3, and 5 days of livetrapping and snaptrapping data, with the *a priori* expectation that the precision of these indices would increase with increased sampling duration.

METHODS

For a complete description of the study site selection and small mammal sampling protocols used during this research (described below), please refer to Wiewel (2005).

Study Site Selection

Sampling was conducted during 2005–2006 on Guam, Rota, Saipan, and Tinian in the Mariana Islands, an archipelago of 15 islands arrayed in a north-south arc between approximately 13° and 21° N and 144° and 146° E (Metteler 1986; Figure 1). Potential study sites were identified using a combination of 1:24,000 and 1:25,000 scale topographical maps (U.S. Geological Survey 1999a,b,c; 2000) and 1:20,000 scale vegetation maps (Falanruw et al. 1989). These sites were then evaluated based on habitat type, available area of relatively homogeneous habitat, and land ownership status. Selected sites represented the 3 major habitat types of the southern Mariana Islands: grassland, native limestone forest, and secondary forest dominated by *Leucaena leucocephala*, an introduced leguminous tree (Mueller-Dombois and Fosberg 1998). Additional sites were selected near airports and seaports, independent of habitat type, to better understand introduced small mammal populations in these areas which are important for brown treesnake (*Boiga irregularis*) control and management in the Mariana Islands. These sites generally included several habitat types (typically grassland and *L. leucocephala*-dominated secondary forest) and are classified as mixed habitat. With the exception of mixed habitat sites, potential sites contained ≥4 ha of relatively homogeneous habitat. Selected sites were located primarily on military and public lands because these areas frequently offered larger tracts of homogeneous habitat and because accurate information about private land ownership was often difficult to obtain. Sampling occurred at 7 sites on Guam, 4 on Rota, 5 on Saipan, and 3 on Tinian (Table 2). On each island, at least 1 grassland site, 1 native limestone forest site, and 1 *L. leucocephala*-dominated secondary forest site were selected and sampled. Five sites were sampled near airports and seaports on Guam ($n = 2$), Rota ($n = 1$), and Saipan ($n = 2$; Table 2).

Small Mammal Sampling

At each selected site, sampling activities occurred over 2 weeks and consisted of (in chronological order) a 2-day live trap acclimation period, a 5-day livetrapping period, a 2-day snap trap acclimation period, and a 5-day snaptrapping period. During acclimation periods traps were placed on the trapping grid but not baited. Species targeted during sampling include *Mus musculus*, *Rattus exulans*, *R. norvegicus*, *R. rattus*, and *Suncus murinus*, all of which are introduced in the Mariana Islands. There is uncertainty regarding the status of *R. rattus* and a closely related and morphologically similar species, *R.*

tanezumi (Musser and Carleton 2005:1484–1487, 1489–1491), in the Mariana Islands. Due to this uncertainty, we collected genetic material from all captured *Rattus* and are in the process of confirming species identification and distribution. Analysis of the cytochrome oxidase I mtDNA region of 8 specimens from northern and central Guam indicated that all were *R. diardii* (sensu Robins et al. 2007), rather than the expected *R. rattus* and *R. tanezumi*. Until samples from all islands are processed, however, we will use the more recognized term *R. rattus* to refer to the combined sample of unidentified *Rattus* species.

Sampling was conducted on an 11 × 11 grid with 12.5 m intervals between each trap station (nominal area = 1.56 ha). During livetrapping, a single standard-length folding Sherman live trap (229 × 89 × 76 mm; H.B. Sherman Traps, Inc., Tallahassee, FL) was placed at each trap station (*n* = 121) and a single Haguruma wire mesh live trap (approximately 285 × 210 × 140 mm; Standard Trading Co., Honolulu, HI) was placed at every other trap station (*n* = 36; Figure 2). Immediately following the final check of live traps, each Sherman trap was replaced with a single Museum Special snap trap (141 × 70 × 15 mm; Woodstream Corporation, Lititz, PA) and each Haguruma trap was replaced with a single Victor rat snap trap (175 × 84 × 28 mm; Model M201, Woodstream Corporation, Lititz, PA).

Trap selection and spacing were determined based on a combination of literature review and preliminary testing of these sampling parameters. Two types of live and snap traps were used to maximize captures of target species, based on preliminary trap evaluations in the Mariana Islands (Gragg 2004, Wiewel 2004a,b). Trap spacing was selected based on review of target species' home range and movement patterns. Thus, Sherman and Museum Special traps, which we believed would best capture *M. musculus* and *S. murinus* (Gragg 2004, Wiewel 2004b), were spaced at 12.5 m intervals to match the relatively small average home ranges of these species (Baker 1946; Barbehenn 1969, 1974). Similarly, Haguruma and Victor traps, which we believed would be more appropriate for capturing *Rattus* species (Gragg 2004, Wiewel 2004a), were spaced at 25 m intervals to better match the larger average home ranges of these species (Baker 1946, Strecker 1962, Barbehenn 1974, Dowding and Murphy 1994, Lindsey et al. 1999).

Traps were placed on the ground and, whenever possible, located next to or beneath clumps of grass, downed woody debris, or rocks to provide shelter from sun and rain. Traps were baited with a mixture of peanut butter, oats, and food-grade paraffin (Wiewel 2004a) and were checked beginning around 0730–0800 each day. Traps were closed throughout the day to minimize trap mortality, reopened at approximately 1600, and rebaited as necessary to ensure bait freshness. We recorded the time required to complete daily activities associated with each sampling method, including trap baiting, trap monitoring, and captured animal processing, for comparative purposes.

Animals captured during livetrapping were uniquely marked in each ear with appropriately sized numbered metal ear tags (*M. musculus* and *S. murinus*: small ear tags produced by S. Roestenburg, Riverton, UT; *Rattus* species: #1005-1, National Band and Tag Co., Newport, KY), allowing us to identify recaptured individuals. During both livetrapping and snaptrapping, captured animals were examined and measured to determine species, sex, age, and reproductive status, mass (g), head-body length (mm), tail length (mm), right hind foot length (mm), right ear length (mm), and testes length (mm; if applicable). All capture, handling, and marking techniques followed guidelines approved by the American Society of Mammalogists (Gannon et al. 2007) and the U.S. Geological Survey Animal Care and Use Committee (Fort Collins Science Center). Animals captured during snaptrapping were disposed of away from study sites and human-use areas.

Data Analysis

Data analysis generally followed an information-theoretic approach involving model selection and multi-model inference. Model selection was based on Akaike's Information Criterion (AIC; Akaike 1973) corrected for small sample size (AIC_c; Hurvich and Tsai 1989). Models were considered competitive with the top-ranked model when $\Delta AIC_c \leq 2.0$ (Burnham and Anderson 2002:131). Model-averaging was based on Akaike weights (Burnham and Anderson 2002:150) and included the entire model set except for models with nonsensical β or real parameter estimates, which were removed prior to model averaging. We defined nonsensical β estimates as those with standard error (SE) >> β (e.g., β = -18.6, SE(β) = 475.6) and nonsensical real parameter estimates as those with SE = 0. Unless otherwise indicated, all estimates are presented as mean ± 1 SE.

Data collected during livetrapping and snaptrapping were used to generate 3 distinct site- and species-specific abundance estimates. Mark-recapture abundance estimates were generated from livetrapping data and removal abundance estimates were generated from both livetrapping data (after excluding recapture information from the dataset) and snaptrapping data. Livetrapping and snaptrapping data were treated similarly during abundance estimation, with the exception that the modeling of removal abundance estimates required that recapture probability be constrained to 0 (i.e., no possibility of recapture), which precluded the creation of models allowing a behavioral response to initial capture or any variation in recapture probability. All abundance estimates were generated in Program MARK 4.3 (White and Burnham 1999) using the conditional likelihood closed capture-recapture model developed by Huggins (1989, 1991). The Huggins model uses estimates of capture probability and the number of individuals captured to estimate abundance. Encounter histories are used to estimate capture probability and can account for heterogeneity in capture probability from temporal, behavioral, and individual effects (both in the form of mixture distributions [Norris and Pollock 1996, Pledger 2000] and individual covariates [Huggins 1989, 1991]). In this context, mixture distributions are an attempt to deal with individual heterogeneity by grouping animals with similar capture probabilities into discrete classes for modeling purposes (Pledger 2000). For example, a 2-mixture distribution could group individuals into high and low capture probability classes. Similarly, covariates are variables thought to influence capture probability (and other demographic parameters) which, when added to capture probability models, may reduce unexplained heterogeneity and thereby improve parameter estimation (Pollock et al. 1984, Pollock 2002). Covariates may pertain to individual animals (e.g., age, sex, mass), in which case they are generally assumed constant over time for modeling purposes, or to the environment (e.g., temperature, precipitation), in which case they are generally assumed constant for all animals over a specified time span, such as 24 hours, for modeling purposes (Pollock et al. 1984, Pollock 2002).

In Program MARK, design matrices were coded to allow sites to be treated both individually and as groups, based on common attributes such as island or habitat. Capture and recapture probability were primarily modeled across these groups to increase statistical efficiency (i.e., reduce estimate variance) and allow abundance estimates to be generated from sites with few captures or recaptures (Bowden et al. 2003, White 2005, Conn et al. 2006, Converse et al. 2006). Models were specified using the logit link function to constrain parameter estimates to the range 0–1 and to allow the use of non-identity design matrices (Cooch and White 2005). Model building in Program MARK occurred in an iterative fashion, beginning with the traditional models (M_0, M_b, M_t, M_h, M_{tb}, M_{bh}, M_{th}, M_{tbh}) outlined in Otis et al. (1978), where subscripts indicate the type of capture probability variation dealt with by each model: b = behavioral variation, t = temporal variation, h = heterogeneity, and M_0 = constant capture probability. Models incorporating heterogeneity were specified as 2-mixture models, based on concerns that our dataset would not support a more parameterized model (Norris and Pollock 1996, Pledger 2000, Conn et al. 2006). Models were ranked based on AIC_c scores, with the top model being considered for further model development. If the top ranked model included a temporal component, a set of neophobia models were fit to the dataset. Neophobia models allowed capture probability to vary during the first (neo1) or first and second (neo2) sampling occasions, while holding capture probability constant for the remaining sampling occasions. The motivation for neophobia models came both from literature accounts of neophobia for introduced small mammals (Inglis et al. 1996, Thorsen et al. 2000, Clapperton 2006), as well as observations of an increase in captured individuals after the first or second sampling occasion at many of our sites. As before, the top ranked model was considered for further model development. The next subset of models added to the MARK analysis were parameterized to model capture probability, recapture probability, or both capture and recapture probability as a function of island, habitat, or site. This complexity was deemed necessary to investigate possible variation in capture and recapture probability across these groupings. We hypothesized that capture or recapture probability might differ between Guam (with brown treesnake predation pressure) and Rota, Saipan, and Tinian (without brown treesnake predation), so the island grouping was coded in 2 ways, with island[4] distinguishing between each island and island[2] distinguishing Guam from the combination of Rota, Saipan, and Tinian. Again, the top ranked model was used for further model development.

The final subset of models added to the MARK analysis contained combinations of 5 individual and 2 environmental covariates, beginning with the full model containing all covariates and proceeding through a series of more parsimonious models including only covariates important for explaining capture probability. Covariate importance was assessed by examining β values and 95% CIs, where covariates with non-zero overlapping 95% CIs were considered influential on capture probability. Model-averaged abundance estimates were then generated from this pool of models to account for model selection uncertainty, unless the top ranked model had a model weight > 0.90 (Burnham and Anderson 2002:150). Covariates under consideration included sex (male or female), age (adult or juvenile), reproductive status, body condition index, body size, rain previous night, and rain amount. Reproductive status (repstat) was a categorical variable that differentiated reproductively active adults from non-reproductive adults and juveniles; assignment of repstat class was based on mass and the presence of externally visible sexual characteristics such as descended testes for males and active lactation for females. Body condition index (bodycon) was calculated as the ratio between the observed and expected mass of an individual, where expected mass was determined from a linear regression of ln mass vs. ln head-body length. The expected mass regression was generated using mass and head-body measurements from all individuals (i.e., animals captured during both livetrapping and snaptrapping). For each species, variation in bodycon was modeled as a function of island[4], island[2], and habitat in an analysis of variance framework (Proc GLM, SAS Institute 2003; Table 3). A site-specific bodycon model was not considered because of sparse data for some sites, which might have biased bodycon estimates for individuals from those sites. Bodycon estimates from the top model (or the model-averaged bodycon estimate) for each species were included in MARK modeling. Body Size (size) was a species-specific composite variable created from a principle components analysis (Proc FACTOR, SAS Institute 2003) of mass, head-body length, tail length, hind foot length, and ear length measured for each captured individual. Rain previous night (rainprev) was a categorical measure of the presence or absence of rainfall during the night prior to each trap monitoring occasion. Finally, rain amount (rainamt) was a quantitative measure of the total rainfall (mm) at the center of the trapping grid during each 24-hour sampling occasion, with the exception of the first sampling occasion for which the rainfall measurement encompassed only a 12–16 hour period. Prior to including rainamt in MARK models, rainfall amounts for the 5 sampling occasions were examined for equality across sites. Based on overlapping 95% CIs, there was no effect of the abbreviated rainfall measurement period during the first sampling occasion (Table 4).

Because we did not know true abundance, mark-recapture and removal abundance estimates were compared against each other on a site-specific basis. Abundance estimates were evaluated based on the magnitude of coefficients of variation (CV) and the width of 95% confidence intervals (95% CI). We considered estimates with small CVs and narrow 95% CIs to be more informative than estimates with large CVs and wide 95% CIs. The sampling method producing the greatest proportion of informative site-specific estimates was then used to evaluate 2 common count-based indices of abundance generated from the livetrapping and snaptrapping data: the number of unique individuals captured (M_{t+1}; Otis et al. 1978) and the captures per unit effort (CPUE; White et al. 1982). We used the method described by Nelson and Clark (1973) to account for sprung traps when calculating sampling effort, and present CPUE as captures/100 corrected trap nights, where a trap night is defined as 1 trap active for 1 night. The relationship between the most informative abundance estimate and M_{t+1} or CPUE was investigated using regression analyses (Proc REG, SAS Institute 2003), with regressions constrained to pass through the origin. When evaluating constrained regressions, we calculated r^2 using the formula

$$r^2 = 1 - \frac{\mathrm{SSE}}{\mathrm{SST_c}},$$

where SSE = the sum of squared residuals and $\mathrm{SST_c}$ = the corrected total sum of squared deviations (Kvålseth 1985). This correction is necessary because many statistical packages (including SAS) calculate r^2 for constrained regressions by replacing $\mathrm{SST_c}$ in the previous equation with $\mathrm{SST_u}$, the uncorrected total sum of squares, resulting in artificially high r^2 values that are not directly comparable to r^2 values generated for unconstrained regressions (Kvålseth 1985, Cade and Terrell 1997). Because M_{t+1} and CPUE are frequently generated from short duration sampling events, we used indices from the first day of sampling, the first 3 days of sampling, and the full 5 day sampling period to investigate the effects of sampling duration on index performance. Indices were evaluated by comparing the width of 95% prediction intervals (95% PI), which are confidence intervals for an individual predicted value (Ott 1992:519). We considered indices with narrow 95% PIs to be better predictors of small mammal abundance.

We also compared livetrapping and snaptrapping based on the effectiveness of each method for capturing target species as well as the cost of implementing each sampling method. We evaluated trap effectiveness by comparing species-specific capture rates (captures/100 corrected trap nights) during livetrapping (Haguruma and Sherman live traps) and snaptrapping (Victor and Museum Special snap traps). Capture rate calculations included only sites where a species was captured. We investigated the cost of each sampling method by comparing the initial cost of supplies required to implement our sampling protocol, the mass and volume of those supplies, and the time required for site preparation and activities directly associated with sampling: trap baiting, trap monitoring, and the processing of captured animals. For comparative purposes, time requirements were standardized to person-hours to avoid possible bias resulting from unequal numbers of personnel participating in various activities.

RESULTS

We captured 681 *R. rattus*, 298 *S. murinus*, 154 *M. musculus*, 15 *R. exulans*, and 5 *R. norvegicus* in 14,915 trap nights (12,011.5 corrected trap nights) during livetrapping and 642 *R. rattus*, 255 *S. murinus*, 122 *M. musculus*, 14 *R. exulans*, and 3 *R. norvegicus* in 14,915 trap nights (8,952 corrected trap nights) during snaptrapping (Table 5). *S. murinus* was not captured or observed during 9 weeks spent on Rota, and is believed absent from that island. *R. exulans* and *R. norvegicus* were rarely captured with either sampling method, and are not considered further.

Modeling Capture and Recapture Probability

Modeling of *R. rattus* livetrapping and snaptrapping data revealed several common factors important for understanding capture probability, including temporal variation and the individual covariates sex and repstat. The top mark-recapture model for livetrapping data ($w_i = 0.871$) allowed neophobic temporal variation in capture probability (neo2) for each island (island[4]) as well as capture probability variation by sex, repstat, and rainamt, with recapture probability variation by island (island[4]), sex, repstat, and rainamt (Table 6). The top removal model for livetrapping data ($w_i = 0.860$) specified an identical parameter set to explain capture probability variation (Table 6). Model selection uncertainty increased for snaptrapping data; the top removal model ($w_i = 0.375$) allowed capture probability to vary by island[2], sex, age, repstat, and size (Table 6). All attempts to model unexplained heterogeneity in *R. rattus* sampling data using mixture models resulted in nonsensical parameter estimates. In contrast, individual and environmental covariates were useful for modeling unexplained heterogeneity; the top model without covariates had little support during either mark-recapture ($\Delta AIC_c = 17.97$) or removal modeling ($\Delta AIC_c = 19.98$) of livetrapping data or removal modeling of snaptrapping data ($\Delta AIC_c = 12.08$). Based on the top mark-recapture and removal models from livetrapping data described above, *R. rattus* capture probability was lower for males than for females, higher for reproductively mature individuals, and positively correlated with rainfall amount (Figures 3, 4; Table 7). The top removal model from snaptrapping data also indicated that capture probability was lower for males than for females and higher for reproductively mature individuals, while also suggesting that capture probability was higher for adults than juveniles, but lower for the largest individuals within age classes (Figure 5, Table 7).

There were fewer common factors in models of *S. murinus* livetrapping and snaptrapping data, although temporal variation in capture probability was always important. The top mark-recapture model for livetrapping data ($w_i = 0.994$) allowed both temporal variation and heterogeneity (2 mixtures) in capture and recapture probability, with the temporal recapture probability variation differing by island (island[4]; Table 8). Model selection uncertainty increased for removal modeling of livetrapping data; each of the closely ranked top models allowed neophobic temporal variation in capture probability (neo1; Table 8), with the top model ($w_i = 0.254$) also allowing capture probability variation by bodycon. *S. murinus* capture probability tended to increase with increasing bodycon ($\beta_{bodycon} = 4.33 \pm 2.96$, 95% CI = -1.47–10.13), although this relationship was weak as demonstrated by the 95% CI that asymmetrically overlapped zero. For snaptrapping data, the top removal model ($w_i = 0.836$) allowed neophobic temporal variation in capture probability (neo1) for each habitat (Table 8). Although heterogeneity was an important factor for mark-recapture modeling of livetrapping data, attempts to account for unexplained heterogeneity with mixture models during removal modeling of both livetrapping and snaptrapping data resulted in nonsensical parameter estimates. With the exception of bodycon, none of the covariates under consideration were useful for modeling *S. murinus* capture probability.

As with *S. murinus*, modeling of *M. musculus* livetrapping and snaptrapping data indicated that temporal variation was always an important factor for understanding capture probability. For livetrapping data there was considerable model selection uncertainty for both mark-recapture and removal models. The top mark-recapture model ($w_i = 0.349$) allowed both temporal variation and heterogeneity (2 mixtures) in capture and recapture probability, with the temporal variation in capture probability differing between Guam and the other islands (island[2]; Table 9). The top removal model for livetrapping data ($w_i = 0.414$) allowed neophobic temporal variation in capture probability (neo1; Table 9). For snaptrapping data, the top removal model ($w_i = 0.745$) allowed neophobic temporal variation in capture probability (neo1), with this temporal variation differing by island[2] (Table 9).

Although heterogeneity was an important factor for mark-recapture modeling of livetrapping data, attempts to account for unexplained heterogeneity with mixture models during removal modeling of both livetrapping and snaptrapping data resulted in nonsensical parameter estimates. Similarly, none of the covariates under consideration were useful for modeling *M. musculus* capture probability.

Abundance Estimates

In spite of differences in model structure, model-averaged mark-recapture and removal abundance estimates from livetrapping ($\hat{\bar{N}}_{\text{M-R Live}}$ and $\hat{\bar{N}}_{\text{REM Live}}$, respectively) and removal abundance estimates from snaptrapping ($\hat{\bar{N}}_{\text{REM Snap}}$) were correlated for each species. It should be noted that mark-recapture abundance estimates from livetrapping data were not model-averaged for *S. murinus* (top model w_i = 0.994; Table 8); however, the symbol $\hat{\bar{N}}_{\text{M-R Live}}$ will also be used for this species to avoid confusing notation. The strongest relationship was between $\hat{\bar{N}}_{\text{M-R Live}}$ and $\hat{\bar{N}}_{\text{REM Live}}$ for each species (*M. musculus*: $r^2 = 0.99$, *R. rattus*: $r^2 = 0.97$, and *S. murinus*: $r^2 = 0.99$). Weaker correlations were observed between $\hat{\bar{N}}_{\text{M-R Live}}$ and $\hat{\bar{N}}_{\text{REM Snap}}$ (*M. musculus*: $r^2 = 0.86$, *R. rattus*: $r^2 = 0.79$, and *S. murinus*: $r^2 = 0.58$) and between $\hat{\bar{N}}_{\text{REM Live}}$ and $\hat{\bar{N}}_{\text{REM Snap}}$ (*M. musculus*: $r^2 = 0.88$, *R. rattus*: $r^2 = 0.82$, and *S. murinus*: $r^2 = 0.56$). On a site-specific basis, abundance estimates were often qualitatively similar, although there was an overall trend of $\hat{\bar{N}}_{\text{REM Snap}}$ and $\hat{\bar{N}}_{\text{REM Live}} > \hat{\bar{N}}_{\text{M-R Live}}$ for *R. rattus* (Table 10) and $\hat{\bar{N}}_{\text{REM Snap}} > \hat{\bar{N}}_{\text{REM Live}}$ and $\hat{\bar{N}}_{\text{M-R Live}}$ for *S. murinus* (Table 11). $\hat{\bar{N}}_{\text{M-R Live}}$, $\hat{\bar{N}}_{\text{REM Live}}$, and $\hat{\bar{N}}_{\text{REM Snap}}$ were relatively analogous across most sites for *M. musculus* (Table 12).

In addition to these discrepancies in the magnitude of abundance estimates, we found that $\hat{\bar{N}}_{\text{REM Live}}$ and $\hat{\bar{N}}_{\text{REM Snap}}$ were generally less informative than $\hat{\bar{N}}_{\text{M-R Live}}$ based on comparison of CVs and 95% CIs. For *R. rattus*, the mean $\hat{\bar{N}}_{\text{M-R Live}}$ CV (0.24 ± 0.03, 95% CI = 0.17–0.30; n = 16) was lower than the mean CV for $\hat{\bar{N}}_{\text{REM Live}}$ (0.59 ± 0.04, 95% CI = 0.50–0.67; n = 16) or $\hat{\bar{N}}_{\text{REM Snap}}$ (0.40 ± 0.04, 95% CI = 0.32–0.48; n = 17). Further, 95% CIs were notably wider for $\hat{\bar{N}}_{\text{REM Live}}$ and $\hat{\bar{N}}_{\text{REM Snap}}$ than for $\hat{\bar{N}}_{\text{M-R Live}}$, and frequently overlapped zero (Table 10). For *S. murinus*, mean $\hat{\bar{N}}_{\text{M-R Live}}$ and $\hat{\bar{N}}_{\text{REM Live}}$ CVs were similar (0.30 ± 0.01, 95% CI = 0.28–0.32; n = 9 and 0.26 ± 0.01, 95% CI = 0.23–0.29; n = 9, respectively), with both being much lower than the mean $\hat{\bar{N}}_{\text{REM Snap}}$ CV (1.26 ± 0.01, 95% CI = 1.24–1.29; n = 10). $\hat{\bar{N}}_{\text{REM Snap}}$ 95% CIs were notably wider than 95% CIs for $\hat{N}_{\text{M-R Live}}$ or $\hat{\bar{N}}_{\text{REM Live}}$, and all $\hat{\bar{N}}_{\text{REM Snap}}$ 95% CIs overlapped zero (Table 11). For *M. musculus*, mean $\hat{\bar{N}}_{\text{M-R Live}}$ CV (0.34 ± 0.06, 95% CI = 0.19–0.48; $n = 8$) was lower than the mean CV for $\hat{\bar{N}}_{\text{REM Live}}$ (0.60 ± 0.05, 95% CI = 0.49–0.72; $n = 8$) or $\hat{\bar{N}}_{\text{REM Snap}}$ (0.75 ± 0.05, 95% CI = 0.62–0.88; $n = 7$). Again, 95% CIs were notably wider for $\hat{\bar{N}}_{\text{REM Live}}$ and $\hat{\bar{N}}_{\text{REM Snap}}$ than for $\hat{\bar{N}}_{\text{M}-\text{R Live}}$, and all $\hat{\bar{N}}_{\text{REM Live}}$ and $\hat{\bar{N}}_{\text{REM Snap}}$ 95% CIs overlapped zero (Table 12).

Evaluation of Count-Based Indices

Count-based indices of abundance generated from livetrapping and snaptrapping data were evaluated against our most information-rich abundance estimate, $\hat{\bar{N}}_{\text{M}-\text{R Live}}$. $M_{\text{t+1}}$ and CPUE from 1, 3, and 5 days of both livetrapping ($M_{\text{t+1 Live}}$ and CPUE$_{\text{Live}}$) and snaptrapping ($M_{\text{t+1 Snap}}$ and CPUE$_{\text{Snap}}$) were strong correlates ($r^2 \geq 0.8$) with $\hat{\bar{N}}_{\text{M}-\text{R Live}}$ in 10 of 12 comparisons (83%) for *R. rattus* and 9 of 12 comparisons (75%) for *M. musculus* (Table 13). In contrast, strong correlations were observed between indices and $\hat{\bar{N}}_{\text{M}-\text{R Live}}$ in only 6 of 12 comparisons (50%) for *S. murinus* (Table 13). In all cases, regression slope coefficients were > 1.0 (Table 13).

The utility of $M_{\text{t+1}}$ and CPUE as predictors of small mammal abundance differed depending on the sampling method and sampling duration the index was generated from. For *R. rattus*, $M_{\text{t+1 Live}}$ 95% PIs improved with additional sampling occasions (Figure 6). For example, a mid-range $M_{\text{t+1 Live}}$ value from 1 day of sampling (10 individuals) predicts $\hat{\bar{N}}_{\text{M}-\text{R Live}}$ of approximately 20–185 individuals, whereas a mid-range $M_{\text{t+1 Live}}$ value from 5 days of sampling (55 individuals) predicts $\hat{\bar{N}}_{\text{M}-\text{R Live}}$ of approximately 85–110 individuals (Figure 6). In contrast, there was little improvement in 95% PIs between a mid-range $M_{\text{t+1 Snap}}$ value from 1 day of sampling (15 individuals; 95% PI $\approx$ 45–150 individuals) and a mid-range $M_{\text{t+1 Snap}}$ value from 5 days of sampling (50 individuals; 95% PI $\approx$ 30–130 individuals; Figure 6). Similar patterns were evident for the 95% PIs of CPUE$_{\text{Live}}$ and CPUE$_{\text{Snap}}$ (Figure 7). For example, a mid-range CPUE$_{\text{Live}}$ value from 1 day of sampling (7 captures/100 corrected trap nights) predicts $\hat{\bar{N}}_{\text{M}-\text{R Live}}$ of approximately 5–170 individuals, whereas a mid-range CPUE$_{\text{Live}}$ value from 5 days of sampling (10 captures/100 corrected trap nights) predicts $\hat{\bar{N}}_{\text{M}-\text{R Live}}$ of approximately 75–125 individuals (Figure 7). In comparison, there was little improvement in predictive value between a mid-range CPUE$_{\text{Snap}}$ value (20 captures/100 corrected trap nights) from 1 day of sampling (95% PI $\approx$ 50–150 individuals) or 5 days of sampling (95% PI $\approx$ 85–190 individuals; Figure 7).

For *S. murinus*, $M_{\text{t+1 Live}}$ and $\text{CPUE}_{\text{Live}}$ 95% PIs improved with additional sampling occasions, whereas the width of $M_{\text{t+1 Snap}}$ and $\text{CPUE}_{\text{Snap}}$ 95% PIs was relatively independent of sampling duration, and quite poor overall (Figures 8, 9). For example, a mid-range $\text{CPUE}_{\text{Live}}$ value from 1 day of sampling (8 captures/100 corrected trap nights) predicts $\hat{\bar{N}}_{\text{M−R Live}}$ of approximately 60–100 individuals, whereas the same mid-range $\text{CPUE}_{\text{Live}}$ value from 5 days of sampling predicts $\hat{\bar{N}}_{\text{M−R Live}}$ of approximately 70–80 individuals (Figure 9). In contrast, there was little improvement in $M_{\text{t+1 Snap}}$ and $\text{CPUE}_{\text{Snap}}$ 95% PIs with increasing sampling duration (Figures 8, 9). For example, a mid-range $\text{CPUE}_{\text{Snap}}$ value of 7 captures/100 corrected trap nights from 1 day of sampling predicts $\hat{\bar{N}}_{\text{M−R Live}}$ of approximately 0–120 individuals, whereas the same $\text{CPUE}_{\text{Snap}}$ value from 5 days of sampling predicts $\hat{\bar{N}}_{\text{M−R Live}}$ of approximately 0–110 individuals (Figure 9).

As with *R. rattus* and *S. murinus*, *M. musculus* $M_{\text{t+1 Live}}$ and $\text{CPUE}_{\text{Live}}$ 95% PIs improved with additional sampling occasions (Figures 10, 11). In contrast to both *R. rattus* and *S. murinus*, however, there was also improvement $M_{\text{t+1 Snap}}$ and $\text{CPUE}_{\text{Snap}}$ 95% PIs as sampling occasions increased, although they remained wider than those for $M_{\text{t+1 Live}}$ and $\text{CPUE}_{\text{Live}}$ (Figures 10, 11). For example, a 1 day $\text{CPUE}_{\text{Live}}$ value of 5 captures/100 corrected trap nights predicts $\hat{\bar{N}}_{\text{M−R Live}}$ of approximately 20–60 individuals, whereas the same $\text{CPUE}_{\text{Live}}$ value from 5 days of sampling predicts $\hat{\bar{N}}_{\text{M−R Live}}$ of approximately 45–55 individuals (Figure 11). In comparison, a 1 day $\text{CPUE}_{\text{Snap}}$ value of 5 captures/100 corrected trap nights predicts $\hat{\bar{N}}_{\text{M−R Live}}$ of approximately 0–50 individuals, whereas the same $\text{CPUE}_{\text{Snap}}$ value from 5 days of sampling predicts $\hat{\bar{N}}_{\text{M−R Live}}$ of approximately 35–60 individuals (Figure 11).

Comparison of Livetrapping and Snaptrapping Capture Rates

R. rattus, *S. murinus*, and *M. musculus* were each captured in both available trap types during livetrapping and snaptrapping. However, the effectiveness (mean captures/100 corrected trap nights) of live and snap traps differed for each species, after controlling for the number of sites where each species was captured (Figure 12). During livetrapping, Haguruma traps were much more effective for capturing *R. rattus* (27.35 ± 6.35, 95% CI = 13.88–40.82; $n = 17$) than were Sherman traps (1.85 ± 0.87, 95% CI = 0.01–3.69; $n = 17$). In contrast, *S. murinus* captures were greater in Sherman traps (5.69 ± 1.73, 95% CI = 1.77–9.61; $n = 10$) than in Haguruma traps (0.60 ± 0.23, 95% CI = 0.08–1.12; $n = 10$). *M. musculus* captures were also greater in Sherman traps (3.37 ± 1.06, 95% CI = 0.92–5.82; $n = 9$) than in Haguruma traps (0.93 ± 0.61, 95% CI = 0–2.34; *n* = 9), although this difference was not significant based on overlapping 95% CIs. During snaptrapping, Victor traps were much more effective for capturing *R. rattus* (27.92 ± 7.13, 95% CI = 12.81–43.02; *n* = 17) than were Museum Special traps (2.63 ± 0.93, 95% CI = 0.67–4.60; *n* = 17). In contrast, *M. musculus* captures were generally greater in Museum Special traps (3.24 ± 1.22, 95% CI = 0.43–6.05; *n* = 9) than in Victor traps (0.33 ± 0.18, 95% CI = 0–0.73; *n* = 9), although this difference was not significant based on overlapping 95% CIs. Victor (4.81 ± 1.49, 95% CI = 1.44–8.18; *n* = 10) and Museum Special traps (5.84 ± 2.13, 95% CI = 1.02–10.66; *n* = 10) were equally effective for capturing *S. murinus*.

Comparison of Livetrapping and Snaptrapping Cost and Effort

The total cost of any sampling method is the sum of the cost of necessary supplies, the cost of transporting those supplies, and the labor costs associated with conducting sampling, including site preparation. Based on our sampling protocol, minimum initial supply cost was much lower for snaptrapping than for livetrapping (Table 14). Victor and Museum Special snap traps were also smaller and lighter than Haguruma and Sherman live traps (Table 15). More importantly, snaptrapping activities required less time (19.7 ± 0.9 person-hours, 95% CI = 17.8–21.5; *n* = 19) than activities associated with livetrapping (31.8 ± 2.4 person-hours, 95% CI = 26.7–36.8; *n* = 19; Table 16). Closer examination of these data revealed a more complex relationship, however, as time requirements increased with increasing small mammal captures for livetrapping, but not for snaptrapping. For example, livetrapping required nearly twice as much time on Rota, Saipan, and Tinian (37.7 ± 2.1 person-hours, 95% CI = 33.1–42.4; *n* = 12) as on Guam (21.5 ± 2.4 person-hours, 95% CI = 15.6–27.4; *n* = 7), whereas snaptrapping time requirements varied little between Rota, Saipan, and Tinian (19.7 ± 1.1 person-hours, 95% CI = 17.3–22.0; *n* = 12) and Guam (19.6 ± 1.6 person-hours, 95% CI = 15.7–23.5; *n* = 7; Table 16). Further, the time required for sampling activities was generally less than the time required to prepare sampling grids in the dense vegetation and rugged terrain of the Mariana Islands (48.6 ± 6.3 person-hours, 95% CI = 35.2–62.0; *n* = 18). Overall, the time required for site preparation and sampling activities was not markedly different between livetrapping (81.0 ± 6.3 person-hours, 95% CI = 67.8–97.2; *n* = 18) and snaptrapping (67.9 ± 6.4 person-hours, 95% CI = 54.3–81.4; *n* = 18; Table 16).

DISCUSSION

Comparison of Mark-Recapture and Removal Abundance Estimates

Using data collected during consecutive livetrapping and snaptrapping events, we demonstrate that mark-recapture abundance estimates generated from livetrapping data ($\hat{N}_{\text{M}-\text{R Live}}$) were more precise than removal abundance estimates generated from either livetrapping ($\hat{N}_{\text{REM Live}}$) or snaptrapping ($\hat{N}_{\text{REM Snap}}$) data. On a site-specific basis, $\hat{N}_{\text{REM Live}}$ and $\hat{N}_{\text{REM Snap}}$ were generally greater (often much greater) than $\hat{N}_{\text{M}-\text{R Live}}$, especially for *R. rattus* and *S. murinus*. Unfortunately, without knowledge of true abundance, we can not evaluate the accuracy or precision of these estimates. In practice, true abundance is rarely known and researchers must rely on abundance estimates to make conservation and management decisions. In that framework, the generally high variance of $\hat{N}_{\text{REM Live}}$ and $\hat{N}_{\text{REM Snap}}$, as demonstrated by large CVs (e.g., >0.30) and wide 95% CIs, would severely limit the utility of these estimates for any foreseeable conservation or management purpose. In fact, many of our $\hat{N}_{\text{REM Live}}$ and $\hat{N}_{\text{REM Snap}}$ had little or no informational value based on 95% CIs spanning or exceeding the plausible range of abundance we might encounter during sampling of wild populations of these species (e.g., 0–1000 *R. rattus* or 0–400 *S. murinus*). In contrast, the majority of $\hat{N}_{\text{M}-\text{R Live}}$ had reasonable CVs (e.g., ≤0.30) and 95% CIs, such that these estimates could be used to detect biologically significant differences in small mammal abundance across space or time.

We suspect that the high variance of $\hat{N}_{\text{REM Live}}$ and $\hat{N}_{\text{REM Snap}}$ is largely attributable to non-declining captures over successive sampling occasions. For example, livetrapping captures of new individuals declined over successive sampling occasions at only 1 of 16 sites with >5 *R. rattus* captures, 0 of 9 sites with >5 *S. murinus* captures, and 1 of 6 sites with >5 *M. musculus* captures. Similarly, snaptrapping captures of new individuals declined at only 4 of 14 sites with >5 *R. rattus* captures, 1 of 9 sites with >5 *S. murinus* captures, and 2 of 5 sites with >5 *M. musculus* captures. The most obvious explanation for non-declining captures over time is a failure of population closure (i.e., births, deaths, emigration, or immigration occurring during sampling). Of these possibilities, neither deaths nor emigration can explain non-declining captures over successive sampling occasions, as each would decrease the number of animals on the sampling area over time. In contrast, both births and immigration would add animals to the sampling area and contribute to non-declining captures over time. While births themselves are probably unimportant for short-duration sampling events, the maturation of juvenile animals could be important. Indeed, *R. rattus* juvenile captures increased from 17% of total captures during livetrapping to 35% of total captures during snaptrapping. Similarly, *M. musculus* juvenile captures increased from just 2% of total captures during livetrapping to 12.5% of total captures during snaptrapping. These increases could result from recently born individuals maturing and becoming available for capture, although it seems unlikely that this would be an important factor across multiple sites, habitats, and islands sampled at different times.

A more plausible explanation is that the removal of dominant adults during snaptrapping altered the behavior of non-dominant juveniles, thereby increasing their capture probability (Summerlin and Wolfe 1973). The removal of animals during snaptrapping also increases the likelihood of immigration, as territorial vacancies are created which may attract animals from outside the sampling area (Stickel 1946b, Fitzgerald et al. 1981, White et al. 1982). Immigration seems less probable during livetrapping, although it is possible that the use of bait in traps might attract animals into the sampling area (White et al. 1982). If immigration were occurring, we might expect the majority of new individuals to be captured on the perimeter of the sampling area, especially during later sampling occasions. However, a post-hoc analysis of captures in perimeter (defined as the 2 outer “rings” of traps) and interior traps revealed little evidence of immigration during either livetrapping or snaptrapping. Observed captures in perimeter and interior traps were generally within 5–10% of expected captures in each segment of the sampling area, and never exceeded expected captures by >13% during sampling occasions 3–5 (Appendix 2A).

Evaluation of Count-Based Indices

Using $\hat{N}_{\mathrm{M-R\ Live}}$ as our best measure of small mammal abundance, we found that count-based indices generated from livetrapping data ($M_{\mathrm{t+1\ Live}}$ and CPUE $_{\mathrm{Live}}$) generally had narrower 95% PIs than indices generated from snaptrapping data ($M_{\mathrm{t+1\ Snap}}$ and CPUE $_{\mathrm{Snap}}$). The predictive value of $M_{\mathrm{t+1\ Live}}$ and CPUE $_{\mathrm{Live}}$ increased with increasing sampling duration for all species. In contrast, there was little apparent benefit to increased sampling duration on the predictive value of $M_{\mathrm{t+1\ Snap}}$ and CPUE $_{\mathrm{Snap}}$, except for *M. musculus*. *M. musculus* snaptrapping captures may have been suppressed by *R. rattus* or *S. murinus* during early sampling occasions (Brown et al. 1996, Weihong et al. 1999) at some sites, such that indices derived from short-duration sampling had inflated 95% PIs. There was little difference between the predictive value of $M_{\mathrm{t+1\ Live}}$ vs. CPUE $_{\mathrm{Live}}$ or $M_{\mathrm{t+1\ Snap}}$ vs. CPUE $_{\mathrm{Snap}}$, perhaps not surprising as CPUE is an extension of $M_{\mathrm{t+1}}$. Nonetheless, $M_{\mathrm{t+1}}$ has limited utility because it provides no information about sampling effort and therefore does not facilitate comparisons between samples.

Indices such as $M_{\mathrm{t+1}}$ and CPUE are commonly used when relative, rather than absolute, measures of abundance are thought to be adequate to answer conservation or management questions, under the assumption that these methods require less costly data collection and analysis methods than those required for abundance estimation (Engeman 2003, 2005). Further, proponents of indices frequently reference the restrictive assumptions of abundance estimation methods as the primary argument for the use of indices (Engeman 2003, 2005) and, in effect, imply that indices are subject to fewer or less restrictive assumptions. It is important to note, however, that when an index is used to monitor populations across space or time, an assumption is made that the relationship between the index and true abundance is monotonic and spatially and temporally constant (Nichols 1992; Anderson 2001, 2003); in classic population modeling terminology this assumption is analogous to the constant capture probability ($p_.$) model (M_0; Otis et al. 1978). Without knowledge of true abundance, we can not directly evaluate this relationship, but we can use information provided by our modeling of livetrapping and snaptrapping data to evaluate the validity of the underlying assumption of constant capture probability. Although we observed a monotonic relationship, modeling of livetrapping ($\hat{N}_{\mathrm{M-R\ Live}}$ and $\hat{N}_{\mathrm{REM\ Live}}$) and snaptrapping ($\hat{N}_{\mathrm{REM\ Snap}}$) data indicated no support for the $p_.$ model for *R. rattus* ($\Delta\mathrm{AIC_c} = 105.12$, 51.71, and 15.09, respectively) and at best limited support for the $p_.$ model for *S. murinus* ($\Delta\mathrm{AIC_c} =$ 49.30, 4.08, and 11.90, respectively) and *M. musculus* ($\Delta\mathrm{AIC_c} = 35.91$, 4.04, and 3.85, respectively). Instead, capture probability varied over time and between habitats or islands for each species. Further, modeling of *R. rattus* sampling data identified several covariates (sex, age, reproductive status, size, and rainfall amount) that influenced capture probability. Similarly, capture probability heterogeneity (in the form of mixture models) was also found during mark-recapture modeling of *S. murinus* and *M. musculus* livetrapping data. As noted in Table 1, count-based indices of abundance are invalid when capture probability varies over space, time, or between individuals (heterogeneity). Thus, for these data, the assumptions intrinsic to the application of indices were clearly not met.

In practice, spatial, temporal, or individual variation in capture probability seem likely during any population study. Nonetheless, many have argued that when data are sparse or capture probabilities are low, the known negative bias (unless capture probability = 1) of count-based indices of abundance may be preferred (McKelvey and Pearson 2001, Engeman 2005) over the instability of model selection procedures, unknown bias, and large standard errors associated with abundance estimates derived under these conditions (Otis et al. 1978, Menkens and Anderson 1988, Pollock et al. 1990, Manning et al. 1995, Stanley and Burnham 1998, McKelvey and Pearson 2001). While this may have been true in the past, recent advances in population modeling procedures address many of these concerns. For example, Program MARK allows sites to be grouped based on common characteristics such as island or habitat to increase sample size for capture probability estimation, thereby lessening the negative impacts of sparse data or low capture probability at some sites (Bowden et al. 2003, White 2005, Conn et al. 2006, Converse et al. 2006). Information theoretic-based model selection procedures offer vast improvements over earlier techniques (Burnham and Anderson 2002), such as the much-maligned model selection procedure implemented in Program CAPTURE (Otis et al. 1978, White et al. 1982). Further, information theoretic-based model selection allows the generation of model-averaged abundance and variance estimates (Burnham and Anderson 2002). The ability to incorporate covariates into population modeling procedures has also improved the flexibility and biological relevancy of models, and can result in improved parameter estimation accuracy and precision (White 2005).

We were concerned that these results might be an artifact of our use of $\hat{\bar{N}}_{\mathrm{M-R\ Live}}$ as a proxy for true abundance, rather than being representative of the true predictive value of indices. If this were the case, then we might expect contradictory results from a comparison of count-based indices with $\hat{\bar{N}}_{\mathrm{REM\ Snap}}$. Indeed, a post-hoc investigation revealed that $M_{\mathrm{t+1\ Snap}}$ and CPUE $_{\mathrm{Snap}}$ had narrower 95% PIs than $M_{\mathrm{t+1\ Live}}$ and CPUE $_{\mathrm{Live}}$ when regressed against $\hat{\bar{N}}_{\mathrm{REM\ Snap}}$ (Appendix 2B). It is notable that 95% PIs were generally greater in these regressions than for our original $\hat{\bar{N}}_{\mathrm{M-R\ Live}}$ regressions, especially for indices generated from 1 or 3 days of sampling data (Appendix 2B). Regardless of which estimate we used, 95% PI width tended to decrease with increasing sampling duration.

Our evaluation of abundance estimates and index performance is somewhat limited because we did not know the true abundance of the small mammal populations we sampled. We note, however, that few comparisons of estimator or index performance involving known-abundance small mammal populations exist (e.g., Manning et al. 1995, Parmenter et al. 2003, Conn et al. 2006). These controlled studies, in turn, may have reduced applicability towards field-based sampling of small mammal populations. For example, it is not clear that populations of a single species in small enclosures (e.g., 0.02 ha: Conn et al. 2006; 0.2 ha: Manning et al. 1995), provided with supplemental food (Conn et al. 2006), are analogous to wild small mammal populations. Further, the effort involved in studying enclosed populations necessarily limits sample size, such that the range of observed population size (e.g., 2,700–14,700 *M. musculus*/ha: Conn et al. 2006; 0–20.6/ha across 12 species: Parmenter et al. 2003) may not be comparable to the potential density range of the same species in wild situations. Similarly, simulation-based evaluations (e.g., Otis et al. 1978, Menkens and Anderson 1988, McKelvey and Pearson 2001, Conn et al. 2004) of estimator or index performance, while valuable for testing robustness to violations of major assumptions, may not represent the full suite of conditions encountered during sampling of wild small mammal populations.

Comparison of Livetrapping and Snaptrapping Capture Rates

We found variable capture rates in our live and snap traps. During livetrapping, *R. rattus* capture rates were much greater in Haguruma traps than in Sherman traps. It is possible that the relatively open, wire mesh design of the Haguruma trap elicits a lesser avoidance response than the enclosed Sherman trap for *Rattus* species, which are commonly thought to be neophobic (Temme and Jackson 1979, Inglis et al. 1996, Thorsen et al. 2000, Priyambodo and Pelz 2003, Clapperton 2006). In contrast, both *S. murinus* and *M. musculus* were captured infrequently in Haguruma traps. Based on limited observations, we believe the mixed effectiveness of Haguruma traps results from a combination of body size and feeding behavior. Unlike *R. rattus*, which often attempted to remove bait from the trap, thereby triggering Haguruma traps, *S. murinus* and *M. musculus* tended to nibble at the bait without attempting to remove it. This behavior, coupled with the low mass of these species, might allow *S. murinus* and *M. musculus* to enter a Haguruma trap, sample the bait without disturbing the trigger, and then exit the trap without releasing the door. In contrast, the design of Sherman traps, which requires an animal to walk across the treadle to reach the bait, is more suitable for capturing these small, low mass species. Several other studies comparing wire mesh traps (Haguruma or others) with box-type traps (Sherman or Elliot traps) have noted similar results, with wire mesh traps being more effective for *Rattus* and other large species and box-type traps being more effective for small species (O'Farrell et al. 1994, Gragg 2004, Wilson et al. 2007). During snaptrapping, *R. rattus* capture rates were much greater in Victor traps than in Museum Special traps, most likely because the smaller Museum Special traps do not consistently kill captured *R. rattus*, some of which may then escape from the trap. The variable capture rate of *M. musculus* in Victor and Museum Special traps may again be related to the small size of this species. There is significant tension on Victor trap treadles, such that *M. musculus* may not consistently trigger these traps, even if they disturb the bait. Further, if *M. musculus* does trigger a Victor trap, it may be missed by the trap bale (or perhaps captured by only the tail). The relatively equivalent captures of *S. murinus* in Victor and Museum Special snap traps indicates that that this species, intermediate in size and mass between *R. rattus* and *M. musculus*, is efficiently captured in either trap. Other studies comparing various snap traps have noted similar results, with Victor rat traps being most effective for large species, and Museum Special (or Victor mouse traps) being effective for smaller species (Wiener and Smith 1972, Pendleton and Davison 1982, Perry et al. 1996). The mixed effectiveness of live and snap traps suggests that trap effectiveness for target species should be assessed prior to large-scale sampling activities; in many situations the use of multiple trap types may be beneficial, especially if resident small mammal species vary greatly in foraging behavior or size. As an added benefit, the use of multiple trap types with mixed effectiveness for target species may decrease the risk of capture suppression of non-dominant species (e.g., suppression of *M. musculus* by *Rattus* species; Brown et al. 1996, Weihong et al. 1999, Gragg 2004).

Comparison of Livetrapping and Snaptrapping Cost and Effort

We found that initial supply cost was approximately 4 times lower for snaptrapping than for livetrapping, based on our protocol. Victor and Musuem Special snap traps were smaller (~ 15 times less volume) and lighter (~ 5 times lighter) than Haguruma and Sherman live traps, which could provide a definite advantage for research conducted in remote areas, rugged terrain, or dense vegetation. Trap baiting, trap monitoring, and animal processing required less time during snaptrapping, although this difference was only pronounced at sites with abundant small mammal populations. In other words, there was no time penalty for livetrapping relative to snaptrapping at sites with few small mammal captures.

More importantly, when we included site preparation in time calculations, there was on average only a 16% time savings for snaptrapping relative to livetrapping. This difference is relatively small in comparison to the observed difference in the informational value between $\hat{\bar{N}}_{\mathrm{M-R\ Live}}$ and $\hat{\bar{N}}_{\mathrm{REM\ Snap}}$ generated from our sampling. It is important to note, however, that dense vegetation and rugged terrain at many of our sites increased site preparation time. The contribution of site preparation time to total time requirements would likely be much lower in other locations, increasing the time differential between livetrapping and snaptrapping.

Management Implications

Knowledge of small mammal population size is often necessary for implementation and evaluation of conservation and management practices (Lancia et al. 2005). Limited resources are available for collecting small mammal population data (Witmer 2005), and researchers frequently rely on sampling and data analysis methods assumed to be fast and cheap (Slade and Blair 2000, McKelvey and Pearson 2001), such as snaptrapping and count-based indices of abundance. We demonstrate, however, that there is limited utility in methods that typically produce unreliable and non-informative results (Nichols 1992; Anderson 2001, 2003), especially when they may not offer significant cost or time savings. Nonetheless, these methods may have value after validation against a more rigorous sampling or estimation procedure (Eberhardt and Simmons 1987). To be effective, however, validation should occur across the full breadth of conditions from which index data will be collected. We suggest that the resources required for thorough and repeated validation efforts might be better invested in implementing rigorous and robust sampling methods and population abundance estimation procedures.

LITERATURE CITED

Akaike, H. 1973. Information theory as an extension of the maximum likelihood principle. Pages 267–281 *in* B.N. Petrov and F. Csáki, editors. Second International symposium on information theory. Akademiai Kiado, Budapest, Hungary.

Anderson, D.R. 2001. The need to get the basics right in wildlife field studies. Wildlife Society Bulletin 29:1294–1297.

Anderson, D.R. 2003. Response to Engeman: index values rarely constitute reliable information. Wildlife Society Bulletin 31:288–291.

Baker, R.H. 1946. A study of rodent populations on Guam, Mariana Islands. Ecological Monographs 16:393–408.

Barbehenn, K.R. 1969. Responses of rodents and shrews to patterns of removal trapping. Pages 247–252 *in* D.W. Parrack, editor. Indian rodent symposium. Calcutta, India.

Barbehenn, K.R. 1974. Estimating density and home range size with removal grids: the rodents and shrews of Guam. Acta Theriologica 19:191–234.

Bellows, A.S., J.F. Pagels, and J.C. Mitchell. 2001. Macrohabitat and microhabitat affinities of small mammals in a fragmented landscape on the upper coastal plain of Virginia. American Midland Naturalist 146:345–360.

Bohlin, T., and B. Sundstrom. 1977. Influence of unequal catchability on population estimates using the Lincoln Index and the removal method applied to electro-fishing. Oikos 28:123–129.

Bowden, D.C., G.C. White, A.B. Franklin, and J.L. Ganey. 2003. Estimating population size with correlated sampling unit estimates. Journal of Wildlife Management 67:1–10.

Brown, K.P., H. Moller, J. Innes, and N. Alterio. 1996. Calibration of tunnel tracking rates to estimate relative abundance of ship rats (*Rattus rattus*) and mice (*Mus musculus*) in a New Zealand forest. New Zealand Journal of Ecology 20:271–275.

Burnham, K.P., and D.R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Second Edition. Springer, New York, New York, USA.

Cade, B.S., and J.W. Terrell. 1997. Comment: cautions on forcing regression equations through the origin. North American Journal of Fisheries Management 17:225–227.

Carothers, A.D. 1973. Capture-recapture methods applied to a population with known parameters. Journal of Animal Ecology 42:125–146.

Clapperton, B.K. 2006. A review of the current knowledge of rodent behaviour in relation to control devices. Science for Conservation 263. New Zealand Department of Conservation, Wellington, New Zealand.

Christian, D.P., P.T. Collins, J.M. Hanowski, and G.J. Niemi. 1997. Bird and small mammal use of short-rotation hybrid poplar plantations. Journal of Wildlife Management 61:171–182.

Conn, P.B., A.D. Arthur, L.L. Bailey, and G.R. Singleton. 2006. Estimating the abundance of mouse populations of known size: promises and pitfalls of new methods. Ecological Applications 16:829–837.

Conn, P.B., L.L. Bailey, and J.R. Sauer. 2004. Indexes as surrogates to abundance for low abundance species. Pages 59–74 *in* W.L. Thompson, editor. Sampling rare and elusive species: concepts, designs, and techniques for estimating population parameters. Island Press, Washington, D.C., USA.

Converse, S.J., G.C. White, and W.M. Block. 2006. Small mammal responses to thinning and wildfire in ponderosa pine-dominated forests of the southwestern United States. Journal of Wildlife Management 70:1711–1722.

Cooch, E.G., and G.C. White. 2005. Program MARK: a gentle introduction. (http://www.phidot.org/software/mark/docs/book/).

Dowding, J.E. and E.C. Murphy. 1994. Ecology of ship rats (*Rattus rattus*) in a Kauri (*Agathis australis*) forest in Northland, New Zealand. New Zealand Journal of Ecology 18:19–28.

Eberhardt, L.L., and M.A. Simmons. 1987. Calibrating population indices by double sampling. Journal of Wildlife Management 51:665–675.

Ecke, F., O. Löfgren, and D. Sörlin. 2002. Population dynamics of small mammals in relation to forest age and structural habitat factors in northern Sweden. Journal of Applied Ecology 39:781–792.

Engeman, R.M. 2003. More on the need to get the basics right: population indices. Wildlife Society Bulletin 31:286–287.

Engeman, R.M. 2005. Indexing principles and a widely applicable paradigm for indexing animal populations. Wildlife Research 32:203–210.

Falanruw, M.V.C., T.G. Cole and A.H. Ambacher. 1989. Vegetation survey of Rota, Tinian, and Saipan, Commonwealth of the Northern Mariana Islands. Resource Bulletin PSW-27. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, California, USA.

Fitzgerald, B.M., B.J. Karl, and H. Moller. 1981. Spatial organization and ecology of a sparse population of house mice (*Mus musculus*) in a New Zealand forest. Journal of Animal Ecology 50:489–518.

Gannon, W.L., R.S. Sikes, and the Animal Care and Use Committee of the American Society of Mammalogists. 2007. Guidelines of the American Society of Mammalogists for the use of wild mammals in research. Journal of Mammalogy 88:809–823.

Gragg, J.E. 2004. Rodent reduction for enhanced control of brown treesnakes (*Boiga irregularis*). M.S. Thesis, Colorado State University, Fort Collins, Colorado, USA.

Huggins, R.M. 1989. On the statistical analysis of capture-recapture experiments. Biometrika 76:133–140.

Huggins, R.M. 1991. Some practical aspects of a conditional likelihood approach to capture experiments. Biometrics 47:725–732.

Hurvich, C.M., and C.-L. Tsai. 1989. Regression and time-series model selection in small samples. Biometrika 76:297–307.

Inglis, I.R., D.S. Shepherd, P. Smith, P.J. Haynes, D.S. Bull, D.P. Cowan, and D. Whitehead. 1996. Foraging behaviour of wild rats (*Rattus norvegicus*) towards new foods and bait containers. Applied Animal Behaviour Science 47:175–190.

Jolly, G.M. 1965. Explicit estimates from capture-recapture data with both death and immigration-stochastic model. Biometrika 52:225–247.

Kvålseth, T.O. 1985. Cautionary note about R^2. The American Statistician 39:279–285.

Lancia, R.A., W.L. Kendall, K.H. Pollock, and J.D. Nichols. 2005. Estimating the number of animals in wildlife populations. Pages 106–153 *in* C.E. Braun, editor. Techniques for wildlife investigations and management. Port City Press, Baltimore, Maryland, USA.

Lindsey, G.D., S.M. Mosher, S.G. Fancy, and T.D. Smucker. 1999. Population structure and movements of introduced rats in an Hawaiian rainforest. Pacific Conservation Biology 5:94–102.

Manning, T., W.D. Edge, and J.O. Wolff. 1995. Evaluating population-size estimators: an empirical approach. Journal of Mammalogy 76:1149–1158.

McKelvey, K.S., and D.E. Pearson. 2001. Population estimation with sparse data: the role of estimators versus indices revisited. Canadian Journal of Zoology 79:1754–1765.

Menkens, G.E., Jr., and S.H. Anderson. 1988. Estimation of small-mammal population size. Ecology 69:1952–1959.

Metteler, L.S. 1986. Pacific island names: a map and name guide to the new Pacific. Bernice P. Bishop Museum Miscellaneous Publication 34, Honolulu, Hawaii, USA.

Mitchell, M.S., K.S. Karriker, E.J. Jones, and R.A. Lancia. 1995. Small mammal communities associated with pine plantation management of pocosins. Journal of Wildlife Management 59:875–881.

Mueller-Dombois, D. and F.R. Fosberg. 1998. Vegetation of the tropical Pacific islands. Springer, New York, New York, USA.

Musser, G.G., and M.D. Carleton. 2005. Family Muridae. Pages 1189–1531 *in* D.E. Wilson, and D.M. Reeder, editors. Mammal species of the world: a taxonomic and geographic reference. Third Edition. Smithsonian Institute Press, Washington, D.C., USA.

Nelson, L., Jr., and F.W. Clark. 1973. Correction for sprung traps in catch/effort calculations of trapping results. Journal of Mammalogy 54:295–298.

Nichols, J.D. 1992. Capture-recapture models. BioScience 42:94–102.

Norris, J.L., III., and K.H. Pollock. 1996. Nonparametric MLE under two closed capture-recapture models with heterogeneity. Biometrics 52:639–649.

O'Farrell, M.J., W.A. Clark, F.H. Emmerson, S.M. Juarez, F.R. Kay, T.M. O'Farrell, and T.Y. Goodlett. 1994. Use of a mesh live trap for small mammals: are results from Sherman live traps deceptive? Journal of Mammalogy 75:692–699.

Otis, D.L., K.P. Burnham, G.C. White, and D.R. Anderson. 1978. Statistical inference from capture data on closed animal populations. Wildlife Monographs 62:1–135.

Ott, R.L. 1992. An introduction to statistical methods and data analysis. Fourth Edition. Wadsworth Publishing Company, California, USA.

Parmenter R.R., T.L. Yates, D.R. Anderson, K.P. Burnham, J.L. Dunnum, A.B. Franklin, M.T. Friggens, B.C. Lubow, M. Miller, G.S. Olson, C.A. Parmenter, J. Pollard, E. Rexstad, T.M. Shenk, T.R. Stanley, and G.C. White. 2003. Small-mammal density estimation: a field comparison of grid-based vs. web-based density estimators. Ecological Monographs 73:1–26.

Pendleton, G.W., and R.P. Davison. 1982. Relative efficiency of three small-mammal traps in prairie wetlands. Prairie Naturalist 14:9–12.

Perry, R.W., P.A. Tappe, D.G. Peitz, R.E. Thill, M.A. Melchoirs, and T.B. Wigley. 1996. A comparison of snap traps for evaluating small mammal populations. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 50:280–286.

Pledger, S. 2000. Unified maximum likelihood estimates for closed capture-recapture models using mixtures. Biometrics 56:434–442.

Pollock, K.H. 1991. Modeling capture, recapture, and removal statistics for estimation of demographic parameters for fish and wildlife populations: past, present, and future. Journal of the American Statistical Association 86:225–238.

Pollock, K.H., J.E. Hines, and J.D. Nichols. 1984. The use of auxillary variables in capture-recapture and removal experiments. Biometrics 40:329–340.

Pollock, K.H., J.D. Nichols, C. Brownie, J.E. Hines. 1990. Statistical inference for capture-recapture experiments. Wildlife Monographs 107:1–97.

Pollock, K.H. 2002. The use of auxiliary variables in capture-recapture modeling: an overview. Journal of Applied Statistics 29:85–102.

Pollock, K.H., J.D. Nichols, T.R. Simons, G.L. Farnsworth, L.L. Bailey, and J.R. Sauer. 2002. Large scale wildlife monitoring studies: statistical methods for design and analysis. Environmetrics 13:105–119.

Priyambodo, S. and H. Pelz. 2003. Studies on neophobic behaviour in Norway rats (*Rattus norvegicus* Berkenhout, 1769) from farms in Germany. Pages 155–158 *in* G.R. Singleton, L.A. Hinds, C.J. Krebs, and D.M. Spratt, editors. Rats, mice, and people: rodent biology and management. Australian Centre for International Agricultural Research, Canberra, Australia.

Roberts, M., and J.L. Craig. 1990. The demography of kiore, *Rattus exulans* in three habitats. New Zealand Journal of Zoology 17:43–53.

Robins, J.H., M. Hingston, E. Matisoo-Smith, and H.A. Ross. 2007. Identifying *Rattus* species using mitochondrial DNA. Molecular Ecology Notes 7:717–729.

Rodgers, J.D., M.F. Solazzi, S.L. Johnson., and M.A. Buckman. 1992. Comparison of three techniques to estimate juvenile coho salmon populations in small streams. North American Journal of Fisheries Management 12:79–86.

Rosenberger, A.E., and J.B. Dunham. 2005. Validation of abundance estimates from mark-recapture and removal techniques for rainbow trout captured by electrofishing in small streams. North American Journal of Fisheries Management 25:1395–1410.

SAS Institute. 2003. SAS/STAT software. Version 9.1. SAS Institute, Inc., Cary, North Carolina, USA.

Seber, G.A.F. 1965. A note on the multiple-recapture census. Biometrika 52:249–259.

Slade, N.A., and S.M. Blair. 2000. An empirical test of using counts of individuals captured as indices of population size. Journal of Mammalogy 81:1035–1045.

Stanley, T.R., and K.P. Burnham. 1998. Estimator selection for closed-population capture-recapture. Journal of Agricultural, Biological, and Environmental Statistics 3:131–150.

Stickel, L.F. 1946a. Experimental analysis of methods of measuring small mammal populations. Journal of Wildlife Management 10:150–159.

Stickel, L.F. 1946b. The source of animals moving into a depopulated area. Journal of Mammalogy 27:301–307.

Strecker, R.L. 1962. Population levels. Pages 74–79 *in* T.I. Storer, editor. Pacific island rat ecology: report of a study made on Ponape and adjacent islands 1955–1958. Bernice P. Bishop Museum Bulletin 225, Honolulu, Hawaii, USA.

Sullivan, T.P., D.S. Sullivan, D.B. Ransome, and P.M.F. Lindgren. 2003. Impact of removal-trapping on abundance and diversity attributes in small-mammal communities. Wildlife Society Bulletin 31:464–474.

Summerlin, C.T., and J.L. Wolfe. 1973. Social influences on trap response of the cotton rat, *Sigmodon hispidus*. Ecology 54:1156–1159.

Temme, M., and W.B. Jackson. 1979. Criteria for trap evaluation. American Society of Testing Materials Special Technical Publication 680:58–67.

Thorsen, M., R. Shorten, R. Lucking, and V. Lucking. 2000. Norway rats (*Rattus norvegicus*) on Frégate Island, Seychelles: the invasion; subsequent eradication attempts and implications for the island's fauna. Biological Conservation 96:133–138.

U.S. Geological Survey. 1999a. Rota [map]. 1:25000. 7.5 Minute Series. Washington, D.C., USA.

U.S. Geological Survey. 1999b. Saipan [map]. 1:25000. 7.5 Minute Series. Washington, D.C., USA.

U.S. Geological Survey. 1999c. Tinian [map]. 1:25000. 7.5 Minute Series. Washington, D.C., USA.

U.S. Geological Survey. 2000. Guam [map]. 1:24000. 7.5 Minute Series. Washington, D.C., USA.

Weihong, J, C.R. Veitch, and J.L. Craig. 1999. An evaluation of the efficiency of rodent trapping methods: the effect of trap arrangement, cover type, and bait. New Zealand Journal of Ecology 23:45–51.

White, G.C. 2005. Correcting wildlife counts using detection probabilities. Wildlife Research 32:211–216.

White, G.C., D.R. Anderson, K.P. Burnham, and D.L. Otis. 1982. Capture-recapture and removal methods for sampling closed populations. LA-8787-NERP. Los Alamos National Laboratory, Los Alamos, New Mexico, USA.

White, G.C., and K.P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46(Supplement):120–138.

Wiener, J.G., and M.H. Smith. 1972. Relative efficiencies of four small mammal traps. Journal of Mammalogy 53:868–873.

Wiewel, A.S. 2004a. Systematic rodent monitoring project update: bait and trap preference. Report to USGS, Fort Collins, Colorado, USA.

Wiewel, A.S. 2004b. Systematic rodent monitoring project: WAPA Sherman trap comparison. Report to USGS, Fort Collins, Colorado, USA.

Wiewel, A.S. 2005. Systematic rodent monitoring protocol. Report to USGS, Fort Collins, Colorado, USA.

Williams, B.K., J.D. Nichols, and M.J. Conroy. 2002. Estimating abundance for closed populations with mark-recapture methods. Pages 289–332 *in* Analysis and management of animal populations. Academic Press, San Diego, California, USA.

Wilson, D.J., M.G. Efford, S.J. Brown, J.F. Williamson, and G.J. McElrea. 2007. Estimating density of ship rats in New Zealand forests by capture-mark-recapture trapping. New Zealand Journal of Ecology 31:47–59.

Witmer, G.W. 2005. Wildlife population monitoring: some practical considerations. Wildlife Research 32:259–263.

Yang, K.-C., C.J. Krebs, and B.L. Keller. 1970. Sequential live-trapping and snap-trapping studies of *Microtus* populations. Journal of Mammalogy 51:517–526.

Zippin, C. 1956. An evaluation of the removal method of estimating animal populations. Biometrics 12:163–189.

Zippin, C. 1958. The removal method of population estimation. Journal of Wildlife Management 22:82–90.

TABLES

Table 1. Comparison of the robustness of selected count-based indices and closed population abundance estimates to violation of population closure and potential forms of capture probability variation.

	Count-based Index		Abundance Estimate	
	M_{t+1}	CPUE	Removal	Mark-Recapture
population closure violation	invalid; unclear how violation would be recognized	invalid; unclear how violation would be recognized	invalid; difficult to recognize violation	problematic; possible to recognize violation
capture probability variation by:				
space	invalid across space	invalid across space	valid if modeled	valid if modeled
time	invalid across time	invalid across time	problematic; reduced accuracy and precision	valid if modeled
behavior	valid; only first capture data are utilized	valid; only first capture data are utilized	valid; only first capture data are utilized	valid if modeled
individual heterogeneity	invalid unless stratified	invalid unless stratified	problematic; reduced accuracy and precision	valid if modeled

Table 2. Introduced small mammal sampling site coordinates and dates on Guam, Rota, Saipan, and Tinian, 2005–2006. Coordinates indicate the site centroid, and are presented in decimal degrees (WGS 84, UTM Zone 55).

Site	Habitat	Dates Sampled	Latitude	Longitude
Guam				
MSRG	grassland	Jun 6–17, 2005	13.542	144.912
ASMF	*Leucaena* forest	May 30–Jun 10, 2005	13.512	144.870
GSYF	*Leucaena* forest	Nov 6–17, 2006	13.437	144.659
PAGO	*Leucaena* forest	Jun 20–Jul 1, 2005	13.417	144.783
GAHF	mixed	Oct 23–Nov 3, 2006	13.491	144.795
NMAR	native forest	May 16–27, 2005	13.378	144.672
RITL	native forest	Apr 18–29, 2005	13.648	144.863
Rota				
SABA	grassland	Jan 23–Feb 3, 2006	14.140	145.191
GAON	*Leucaena* forest	Jan 30–Feb 10, 2006	14.115	145.199
RAPF	mixed	Apr 10–21, 2006	14.170	145.240
ASAK	native forest	Apr 3–14, 2006	14.154	145.170
Saipan				
ACHU	grassland	Sep 19–30, 2005	15.238	145.773
OBYT	*Leucaena* forest	Sep 26–Oct 7, 2005	15.108	145.729
SAEN	mixed	Aug 22–Sep 2, 2006	15.127	145.727
SPOR	mixed	Aug 15–26, 2006	15.227	145.744
LATT	native forest	Sep 12–23, 2005	15.251	145.798
Tinian				
KAST	grassland	Oct 24–Nov 4, 2005	14.951	145.651
ABLE	*Leucaena* forest	Nov 7–18, 2005	15.076	145.640
LSUS	native forest	Oct 31–Nov 11, 2005	15.043	145.629

Table 3. Model selection results from analysis of variance of multiple models explaining variation in *Rattus rattus*, *Suncus murinus*, and *Mus musculus* body condition index (bodycon) on Guam, Rota, Saipan, and Tinian, 2005–2006, as a function of island[4] (each island modeled separately), island[2] (Guam vs. Rota, Saipan, and Tinian combined), and habitat. Results include the number of model parameters (K), relative Akaike's Information Criterion corrected for small sample size (ΔAIC_c), and Akaike weight (w_i).

	K	ΔAIC_c	w_i
R. rattus			
Bodycon(island[4])	6	0.00	1.000
Bodycon(island[2])	4	49.34	0.000
Bodycon(habitat)	6	70.70	0.000
S. murinus			
Bodycon(habitat)	6	0.00	0.999
Bodycon(island[4])	5	21.11	0.001
Bodycon(island[2])	4	23.90	0.000
M. musculus			
Bodycon(island[2])	6	0.00	0.656
Bodycon(island[4])	4	1.33	0.378
Bodycon(habitat)	6	9.38	0.006

Table 4. Mean ($\overline{X}$) rainfall (mm), standard error (SE), and 95% confidence intervals (95% CI) measured during livetrapping and snaptrapping on Guam, Rota, Saipan, and Tinian during 2005–2006 (n = 19 sites). Average rainfall measurement period was 12–16 hours on Occasion 1 and 24 hours on Occasions 2–5.

	Livetrapping			Snaptrapping		
	$\overline{X}$	SE	95% CI	$\overline{X}$	SE	95% CI
Occasion 1	3.8	2.3	0–8.3	3.9	1.7	0.5–7.3
Occasion 2	5.5	1.8	2.0–9.1	3.5	1.4	0.7–6.2
Occasion 3	4.1	1.2	1.6–6.5	3.7	1.7	0.3–7.1
Occasion 4	6.9	2.1	1.6–10.9	4.2	1.7	0.9–7.5
Occasion 5	5.4	2.7	0–10.8	2.3	0.6	1.1–3.5

Table 5. Number of individual *Mus musculus*, *Rattus exulans*, *R. norvegicus*, *R. rattus*, and *Suncus murinus* captured (M_{t+1}) and captures per unit effort (CPUE) during livetrapping (Live) and snaptrapping (Snap) of grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Rota, Saipan, and Tinian, 2005–2006.

		M. musculus		*R. exulans*		*R. norvegicus*		*R. rattus*		*S. murinus*	
Site	Habitat	Live	Snap	Live	Snap	Live	Snap	Live	Snap	Live	Snap
Guam											
MSRG	grassland	15 (2.2)	19 (2.8)	1 (0.2)	1 (0.2)			22 (3.2)	14 (2.1)	14 (2.0)	19 (2.8)
ASMF	*Leucaena* forest							5 (0.7)	1 (0.2)		
GSYF	*Leucaena* forest							13 (2.5)	3 (0.7)		
PAGO	*Leucaena* forest										
GAHF	mixed							1 (0.1)	1 (0.2)		5 (1.0)
NMAR	native forest										
RITL	native forest								1 (0.2)		
Rota											
SABA	grassland	25 (4.1)	24 (4.9)					88 (14.3)	82 (16.9)		
GAON	*Leucaena* forest	19 (3.6)	9 (2.2)	13 (2.5)	12 (3.0)			42 (7.9)	20 (5.0)		
RAPF	mixed	32 (6.1)	15 (3.8)					106 (20.3)	79 (19.9)		
ASAK	native forest	1 (0.2)						11 (1.9)	4 (1.0)		
Saipan											
ACHU	grassland	51 (8.1)	52 (8.4)					41 (6.5)	32 (5.2)	19 (3.0)	12 (1.9)
OBYT	*Leucaena* forest	2 (0.3)		1 (0.2)	1 (0.2)	2 (0.3)	1 (0.2)	50 (7.9)	63 (14.7)	43 (6.8)	80 (18.7)
SAEN	mixed		1 (0.2)			1 (0.1)		8 (1.1)	15 (2.6)	47 (6.6)	20 (3.4)
SPOR	mixed					1 (0.2)	1 (0.2)	29 (4.8)	34 (8.3)	9 (1.5)	14 (3.4)
LATT	native forest							24 (4.0)	28 (7.0)	19 (3.2)	27 (6.7)
Tinian											
KAST	grassland	9 (1.4)	2 (0.5)			1 (0.2)	1 (0.2)	106 (16.5)	145 (35.4)	11 (1.7)	6 (1.5)
ABLE	*Leucaena* forest							55 (9.0)	41 (9.1)	93 (15.2)	62 (13.7)
LSUS	native forest							80 (12.8)	79 (19.7)	43 (6.9)	10 (2.5)

Table 6. Model selection results for mark-recapture and removal modeling of capture (*p*) and recapture (*c*) probability for *Rattus rattus* livetrapping and snaptrapping data collected on Guam, Rota, Saipan, and Tinian, 2005–2006. Parenthetical terms indicate the nesting structure of the previous variable (e.g., neo2(island[4]) specifies separate neophobia effects for each island). All heterogeneity models (h) used 2 finite mixtures to approximate individual heterogeneity. Results include the number of model parameters (*K*), relative Akaike's Information Criterion corrected for small sample size ($\Delta \mathrm{AIC}_c$), and Akaike weight (w_i). Note that the exclusion of recapture data constrains $c = 0$ during removal modeling of livetrapping and snaptrapping data.

	K	$\Delta \mathrm{AIC}_c$	w_i
Mark-recapture modeling of *R. rattus* livetrapping data			
p neo2(island[4]) + sex + repstat + rainamt *c* island[4] + sex + repstat + rainamt	16	0.00	0.871
p neo2(island[4]) + repstat + rainamt *c* island[4] + repstat + rainamt	15	5.72	0.050
p neo2(island[4]) + sex + age + repstat + bodycon + size + rainprev + rainamt *c* island[4] + sex + age + repstat + bodycon + size + rainprev + rainamt	20	6.97	0.027
p neo2(island[4]) + sex + rainamt *c* island[4] + sex + rainamt	15	7.12	0.025
p neo2(island[4]) + rainamt *c* island[4] + rainamt	14	7.56	0.020
p neo2(island[4]) + sex + repstat *c* island[4] + sex + repstat	15	9.61	0.007
Removal modeling of *R. rattus* livetrapping data			
p neo2(island[4]) + sex + repstat + rainamt	12	0.00	0.860
p neo2(island[4]) + sex + age + repstat + bodycon + size + rainprev + rainamt	16	4.58	0.087
p neo2(island[4]) + sex + repstat	11	5.70	0.050
p neo2(island[4]) + sex + rainamt	11	12.18	0.003
Removal modeling of *R. rattus* snaptrapping data			
p(island[2]) + sex + age + repstat + size	6	0.00	0.375
p(island[2]) + age + repstat + size	5	2.24	0.122
p(island[2]) + sex + repstat	4	2.35	0.116
p(island[2]) + sex + age + repstat	5	2.77	0.094
p(island[2]) + sex + age + repstat + bodycon + size + rainprev + rainamt	9	2.86	0.090
p(island[2]) + sex	3	3.94	0.052
p(island[2]) + sex + repstat + size	5	4.09	0.049
p(island[2]) + age + repstat	4	4.55	0.038
p(island[2]) + repstat + size	4	5.44	0.025
p(island[2]) + age	3	6.50	0.015
p(island[2]) + sex + age	4	7.78	0.008
p(island[2]) + age + size	4	8.23	0.006
p(island[2]) + size	3	9.40	0.003
p(island[2]) + sex + age + size	5	9.69	0.003
p(island[2]) + sex + size	4	10.39	0.002
p(island[2])	2	12.08	0.001
p(island[4])	4	13.18	0.001

Table 7. Covariate effect sizes (β), standard errors (SE), and 95% confidence intervals (95% CI) from the top-ranked *Rattus rattus* models identified by mark-recapture and removal modeling of livetrapping data and snaptrapping data (Table 6) collected on Guam, Rota, Saipan, and Tinian, 2005–2006. Missing entries indicate that a particular covariate was not present in the top model.

	Livetrapping Data						Snaptrapping Data		
	Mark-Recapture Analysis			Removal Analysis			Removal Analysis		
Covariate	β	SE	95% CI	β	SE	95% CI	β	SE	95% CI
sex	-0.44	0.15	-0.75– -0.14	-1.69	0.61	-2.89– -0.50	-0.73	0.35	-1.41– -0.05
age							2.33	0.99	0.38–4.28
repstat	0.47	0.15	0.17–0.77	1.47	0.52	0.45–2.48	1.66	0.61	0.46–2.85
size							-0.92	0.40	-1.71– -0.13
rainamt	0.02	0.01	0.01–0.04	0.02	0.01	0.01–0.04			

Table 8. Model selection results for mark-recapture and removal modeling of capture (*p*) and recapture (*c*) probability for *Suncus murinus* livetrapping and snaptrapping data collected on Guam, Saipan, and Tinian, 2005–2006. Parenthetical terms indicate the nesting structure of the previous variable (e.g., neo2(island[4]) specifies separate neophobia effects for each island). All heterogeneity models (h) used 2 finite mixtures to approximate individual heterogeneity. Results include the number of model parameters (*K*), relative Akaike's Information Criterion corrected for small sample size (ΔAIC_c), and Akaike weight (w_i). Note that the exclusion of recapture data constrains *c* = 0 during removal modeling of livetrapping and snaptrapping data.

	K	ΔAIC_c	w_i
Mark-recapture modeling of *S. murinus* livetrapping data			
$p_{\text{t + h}}$ $c_{\text{t(island[4]) + h}}$	10	0.00	0.994
$p_{\text{t + h + sex + repstat + bodycon + size + rainprev + rainamt}}$ $c_{\text{t(island[4]) + h + sex + repstat + bodycon + size + rainprev + rainamt}}$	16	10.33	0.006
Removal modeling of *S. murinus* livetrapping data			
$p_{\text{neo1 + bodycon}}$	3	0.00	0.254
p_{neo1}	2	0.17	0.233
$p_{\text{neo1(island[2])}}$	3	0.26	0.223
$p_{\text{neo1(island[4])}}$	4	1.24	0.137
p_{neo2}	3	2.15	0.087
$p_{.}$	1	4.08	0.033
$p_{\text{neo1(habitat)}}$	5	4.58	0.025
$p_{\text{neo1 + sex + repstat + bodycon + size + rainprev + rainamt}}$	8	7.00	0.008
Removal modeling of *S. murinus* snaptrapping data			
$p_{\text{neo1(habitat)}}$	5	0.00	0.836
$p_{\text{neo1(island[2])}}$	3	4.16	0.105
$p_{\text{neo1(island[4])}}$	4	5.94	0.043
p_{neo1}	2	7.94	0.016

Table 9. Model selection results for mark-recapture and removal modeling of capture (*p*) and recapture (*c*) probability for *Mus musculus* livetrapping and snaptrapping data collected on Guam, Rota, Saipan, and Tinian, 2005–2006. Parenthetical terms indicate the nesting structure of the previous variable (e.g., neo2(island[4]) specifies separate neophobia effects for each island). All heterogeneity models (h) used 2 finite mixtures to approximate individual heterogeneity. Results include the number of model parameters (*K*), relative Akaike's Information Criterion corrected for small sample size (ΔAIC_c), and Akaike weight (w_i). Note that the exclusion of recapture data constrains $c = 0$ during removal modeling of livetrapping and snaptrapping data.

	K	ΔAIC_c	w_i
Mark-recapture modeling of *M. musculus* livetrapping data			
$p_{\text{t(island[2]) + h}}$ $c_{\text{t + h}}$	11	0.00	0.349
$p_{\text{t(island[2]) + h}}$ $c_{\text{t(island[2]) + h}}$	13	0.10	0.331
$p_{\text{t(island[4]) + h}}$ $c_{\text{t + h}}$	19	1.20	0.192
$p_{\text{t + h}}$ $c_{\text{t(island[2]) + h}}$	9	4.36	0.039
$p_{\text{t(island[4]) + h}}$ $c_{\text{t(island[4]) + h}}$	23	4.65	0.034
$p_{\text{t + h}}$ $c_{\text{t + h}}$	7	5.01	0.029
$p_{\text{neo1 + h}}$ c_{h}	4	6.79	0.012
$p_{\text{t + h}}$ $c_{\text{t(habitat) + h}}$	11	7.94	0.007
$p_{\text{neo2 + h}}$ c_{h}	5	8.82	0.004
$p_{\text{t(island[2]) + h + sex + repstat + bodycon + size + rainprev + rainamt}}$ $c_{\text{t + h + sex + repstat + bodycon + size + rainprev + rainamt}}$	17	9.23	0.003
Removal modeling of *M. musculus* livetrapping data			
p_{neo1}	2	0.00	0.414
$p_{.}$	1	0.90	0.264
$p_{\text{neo1(island[4])}}$	5	1.79	0.170
$p_{\text{neo1(island[2])}}$	3	2.01	0.152
Removal modeling of *M. musculus* snaptrapping data			
$p_{\text{neo1(island[2])}}$	3	0.00	0.745
p_{neo1}	2	3.61	0.123
$p_{.}$	1	3.85	0.109
$p_{\text{neo1(habitat)}}$	4	6.98	0.023

Table 10. *Rattus rattus* model-averaged mark-recapture and removal abundance estimates, standard errors (SE), coefficients of variation (CV), and 95% confidence intervals (95% CI) generated from livetrapping ($\hat{\bar{N}}_{\text{M−R Live}}$ and $\hat{\bar{N}}_{\text{REM Live}}$, respectively) and snaptrapping ($\hat{\bar{N}}_{\text{REM Snap}}$) data collected in grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Rota, Saipan, and Tinian, 2005–2006.

Site	Habitat	$\hat{\bar{N}}_{\text{M−R Live}}$	SE	CV	95% CI	$\hat{\bar{N}}_{\text{REM Live}}$	SE	CV	95% CI	$\hat{\bar{N}}_{\text{REM Snap}}$	SE	CV	95% CI
Guam[a]													
MSRG	grassland	41.1	9.4	0.23	22.8–59.5	76.6	45.9	0.60	0–166.5	17.5	5.2	0.30	7.3–27.7
ASMF	*Leucaena* forest	6.6	1.7	0.26	3.3–9.9	8.2	3.3	0.40	1.8–14.6	1.1	0.5	0.45	0.2–2.1
GSYF	*Leucaena* forest	22.9	5.8	0.25	11.5–34.3	34.6	19.8	0.57	0–73.3	3.7	1.4	0.38	1.1–6.4
GAHF	mixed	1.8	1.2	0.67	0–4.1	2.6	2.4	0.92	0–7.3	1.0	0.1	0.10	0.8–1.2
RITL	native forest									1.1	0.2	0.18	0.6–1.5
Rota													
SABA	grassland	142.4	22.8	0.16	97.6–187.1	229.8	114.2	0.50	6.0–453.7	210.0	107.0	0.51	0.3–419.6
GAON	*Leucaena* forest	70.0	12.9	0.18	44.7–95.2	100.9	45.4	0.45	11.9–189.9	34.9	9.2	0.26	16.9–52.9
RAPF	mixed	186.4	31.0	0.17	125.7–247.2	387.6	252.0	0.65	0–881.5	237.0	115.6	0.49	10.5–463.6
ASAK	native forest	17.8	4.3	0.24	9.4–26.3	24.3	10.8	0.44	3.1–45.4	9.1	5.2	0.57	0–19.3
Saipan													
ACHU	grassland	72.2	13.9	0.19	44.9–99.5	98.1	36.4	0.37	26.7–169.4	65.0	25.4	0.39	15.1–114.8
OBYT	*Leucaena* forest	90.6	17.4	0.19	56.4–124.7	180.9	115.2	0.64	0–406.6	151.5	53.0	0.35	47.6–255.4
SAEN	mixed	15.0	4.7	0.31	5.9–24.1	31.1	25.9	0.83	0–81.8	26.3	8.5	0.32	9.6–42.9
SPOR	mixed	54.8	11.7	0.21	31.9–77.7	134.8	100.4	0.74	0–331.5	82.5	38.6	0.47	6.8–158.1
LATT	native forest	47.1	11.1	0.24	25.4–68.8	77.0	44.0	0.57	0–163.3	55.4	16.6	0.30	22.8–87.9
Tinian													
KAST	grassland	194.4	34.5	0.18	126.8–262.1	374.3	230.7	0.62	0–826.5	474.8	268.6	0.57	0–1001.2
ABLE	*Leucaena* forest	85.6	14.1	0.16	58.0–113.2	131.5	59.8	0.45	14.4–248.6	122.8	79.3	0.65	0–278.2
LSUS	native forest	146.1	26.3	0.18	94.6–197.6	288.1	178.2	0.62	0–637.3	230.6	120.6	0.52	0–467.1

[a] Zero *R. rattus* captured at 2 sites (1 *Leucaena* forest and 1 native forest).

Table 11. *Suncus murinus* mark-recapture and model-averaged removal abundance estimates, standard errors (SE), coefficients of variation (CV), and 95% confidence intervals (95% CI) generated from livetrapping ($\hat{N}_{\text{M}-\text{R Live}}$ and $\hat{\bar{N}}_{\text{REM Live}}$, respectively) and snaptrapping ($\hat{\bar{N}}_{\text{REM Snap}}$) data collected in grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Rota, Saipan, and Tinian, 2005–2006. Note that mark-recapture estimates from livetrapping were not model-averaged (Table 8.). *S. murinus* was not captured or observed on Rota and is believed absent from that island.

Site	Habitat	$\hat{N}_{\text{M}-\text{R Live}}$	SE	CV	95% CI	$\hat{\bar{N}}_{\text{REM Live}}$	SE	CV	95% CI	$\hat{\bar{N}}_{\text{REM Snap}}$	SE	CV	95% CI
Guam[a]													
MSRG	grassland	20.3	5.8	0.29	8.9–31.7	23.3	6.5	0.28	10.6–35.9	80.9	97.0	1.20	0–271.0
GAHF	mixed									27.9	37.2	1.33	0–100.7
Saipan													
ACHU	grassland	28.8	8.5	0.30	12.1–45.5	32.3	8.7	0.27	15.2–49.4	53.1	65.7	1.24	0–181.8
OBYT	*Leucaena* forest	67.8	20.1	0.30	28.4–107.2	72.8	17.4	0.24	38.7–106.9	436.8	550.0	1.26	0–1514.7
SAEN	mixed	70.6	19.2	0.27	33.0–108.3	78.3	17.9	0.23	43.1–113.4	114.8	147.5	1.28	0–403.9
SPOR	mixed	13.6	4.5	0.33	4.9–22.4	15.5	5.1	0.33	5.6–25.4	80.3	103.8	1.29	0–283.8
LATT	native forest	30.0	9.4	0.31	11.5–48.4	31.0	7.9	0.25	15.5–46.5	135.4	168.8	1.25	0–466.2
Tinian													
KAST	grassland	17.3	5.8	0.34	5.9–28.7	18.5	5.4	0.29	7.9–29.0	26.6	33.6	1.26	0–92.5
ABLE	*Leucaena* forest	143.1	39.7	0.28	65.4–220.8	152.7	33.2	0.22	87.6–217.8	338.9	427.2	1.26	0–1176.3
LSUS	native forest	63.7	17.0	0.27	30.5–97.0	71.4	16.5	0.23	39.0–103.8	49.9	63.2	1.27	0–173.7

[a] Zero *S. murinus* captured at 5 sites (3 *Leucaena* forest and 2 native forest).

Table 12. *Mus musculus* model-averaged mark-recapture and removal abundance estimates, standard errors (SE), coefficients of variation (CV), and 95% confidence intervals (95% CI) generated from livetrapping ($\hat{\bar{N}}_{\text{M}-\text{R Live}}$ and $\hat{\bar{N}}_{\text{REM Live}}$, respectively) and snaptrapping ($\hat{\bar{N}}_{\text{REM Snap}}$) data collected in grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Rota, Saipan, and Tinian, 2005–2006.

Site	Habitat	$\hat{\bar{N}}_{\text{M}-\text{R Live}}$	SE	CV	95% CI	$\hat{\bar{N}}_{\text{REM Live}}$	SE	CV	95% CI	$\hat{\bar{N}}_{\text{REM Snap}}$	SE	CV	95% CI
Guam[a]													
MSRG	grassland	17.5	4.0	0.23	9.6–25.3	30.8	16.7	0.54	0–63.5	35.2	20.7	0.59	0–75.7
Rota													
SABA	grassland	41.5	10.1	0.24	21.6–61.3	51.6	27.1	0.53	0–104.7	54.4	37.8	0.69	0–128.4
GAON	*Leucaena* forest	32.0	8.3	0.26	15.7–48.3	39.2	20.9	0.53	0–80.1	20.5	14.8	0.72	0–49.5
RAPF	mixed	53.2	12.7	0.24	28.4–78.1	66.0	34.4	0.52	0–133.5	34.0	23.9	0.70	0–80.9
ASAK	native forest	1.7	1.2	0.71	0–4.0	2.1	1.9	0.90	0–5.7				
Saipan[b]													
ACHU	grassland	80.5	17.4	0.22	46.5–114.6	104.1	54.2	0.52	0–210.3	117.9	80.7	0.68	0–276.1
OBYT	*Leucaena* forest	3.2	1.6	0.50	0.2–6.3	4.1	3.0	0.73	0–10.0				
SAEN	mixed									2.3	2.3	1.00	0–6.8
Tinian[c]													
KAST	grassland	14.6	4.4	0.30	5.9–23.3	18.6	10.5	0.56	0–39.1	4.5	3.9	0.87	0–12.2

[a] Zero *M. musculus* captured at 6 sites (3 *Leucaena* forest, 1 mixed habitat, and 2 native forest).

[b] Zero *M. musculus* captured at 2 sites (1 mixed habitat and 1 native forest).

[c] Zero *M. musculus* captured at 2 sites (1 *Leucaena* forest and 1 native forest).

Table 13. Linear regression slope coefficients (β), standard errors (SE), and squared correlation coefficients[a] (r^2) relating the number of individuals captured (M_{t+1}) and captures/100 corrected trap nights (CPUE) derived from 1-, 3-, and 5-days of livetrapping ($M_{t+1\ \mathrm{Live}}$, $\mathrm{CPUE}_{\mathrm{Live}}$) and snaptrapping ($M_{t+1\ \mathrm{Snap}}$, $\mathrm{CPUE}_{\mathrm{Snap}}$) to model-averaged mark-recapture abundance estimates generated from 5-day livetrapping data ($\hat{\bar{N}}_{\mathrm{M-R\ Live}}$) or, for S. murinus, non-model-averaged mark-recapture estimates from livetrapping ($\hat{N}_{\mathrm{M-R\ Live}}$;Table 8). All regressions were constrained to pass through the origin.

	1 Day			3 Day			5 Day		
	β	SE	r^2	β	SE	r^2	β	SE	r^2
R. rattus (*n* = 19)									
$\hat{\bar{N}}_{\mathrm{M-R\ Live}}$ vs. $M_{t+1\ \mathrm{Live}}$	10.41	1.09	0.66	2.70	0.11	0.94	1.76	0.02	0.99
$\hat{\bar{N}}_{\mathrm{M-R\ Live}}$ vs. $M_{t+1\ \mathrm{Snap}}$	6.50	0.42	0.86	2.43	0.12	0.91	1.67	0.10	0.88
$\hat{\bar{N}}_{\mathrm{M-R\ Live}}$ vs. $\mathrm{CPUE}_{\mathrm{Live}}$	13.22	1.48	0.63	9.74	0.46	0.92	10.36	0.27	0.98
$\hat{\bar{N}}_{\mathrm{M-R\ Live}}$ vs. CPUE	5.32	0.31	0.89	5.97	0.32	0.90	7.01	0.45	0.86
S. murinus (*n* = 15)									
$\hat{N}_{\mathrm{M-R\ Live}}$ vs. $M_{t+1\ \mathrm{Live}}$	7.24	0.30	0.96	2.21	0.05	0.99	1.53	0.01	0.99
$\hat{N}_{\mathrm{M-R\ Live}}$ vs. $M_{t+1\ \mathrm{Snap}}$	7.86	1.64	0.40	2.37	0.41	0.53	1.49	0.24	0.58
$\hat{N}_{\mathrm{M-R\ Live}}$ vs. $\mathrm{CPUE}_{\mathrm{Live}}$	9.78	0.44	0.96	8.47	0.23	0.98	9.64	0.12	0.99
$\hat{N}_{\mathrm{M-R\ Live}}$ vs. CPUE	7.10	1.34	0.47	5.97	1.10	0.49	6.41	1.12	0.52
M. musculus (*n* = 19)									
$\hat{\bar{N}}_{\mathrm{M-R\ Live}}$ vs. $M_{t+1\ \mathrm{Live}}$	6.32	0.59	0.82	2.25	0.11	0.94	1.60	0.02	0.99
$\hat{\bar{N}}_{\mathrm{M-R\ Live}}$ vs. $M_{t+1\ \mathrm{Snap}}$	3.64	0.59	0.57	2.04	0.23	0.75	1.68	0.14	0.84
$\hat{\bar{N}}_{\mathrm{M-R\ Live}}$ vs. $\mathrm{CPUE}_{\mathrm{Live}}$	8.43	0.72	0.84	8.35	0.32	0.97	9.49	0.16	0.99
$\hat{\bar{N}}_{\mathrm{M-R\ Live}}$ vs. CPUE	3.79	0.54	0.65	7.02	0.59	0.85	9.88	0.54	0.93

[a] Squared correlation coefficients for constrained regressions were calculated as $r^2 = 1 - \frac{SSE}{SST_c}$, where SSE = the sum of squared residuals and $\mathrm{SST_c}$ = the corrected total sum of squared deviations.

Table 14. Initial cost (US$) of livetrapping and snaptrapping conducted on Guam, Rota, Saipan, and Tinian, 2005–2006, based on the purchase of minimal supplies necessary for sampling activities.

	n	Unit Cost	Total Cost
Livetrapping			
Haguruma trap [a]	36	14.99	539.64
Sherman trap [b]	121	16.15–17.65 [c]	1954.15–2135.65 [c]
1005-1 ear tags (*Rattus* species) [d]	1400	0.077–0.133 [e]	107.80–186.20 [e]
Application pliers for 1005-1 tags	1	25.00	25.00
Roestenburg ear tags (*Mus* and *Suncus*) [f]	900	0.33	297.00
Application pliers for Roestenburg tags	1	40.00	40.00
Total			2963.59–3223.49
Snaptrapping			
Victor trap [g]	36	1.63	58.68
Museum Special trap [h]	121	5.26	636.46
Total			695.14

[a] Standard Trading Co., Honolulu, HI
[b] Models LFG, LFAHD, and LFATDG, H.B. Sherman Traps, Inc., Tallahassee, FL
[c] Sherman cost varies depending on the metal (aluminum or galvanized) chosen for trap construction.
[d] National Band and Tag Co., Newport, KY
[e] Ear tag cost dependent on quantity ordered.
[f] S. Roestenberg, Riverton, UT
[g] Model M201, Woodstream Corporation, Lititz, PA
[h] Woodstream Corporation, Lititz, PA

Table 15. Volume (m^3) and mass (kg) of live and snap traps used during small mammal sampling on Guam, Rota, Saipan, and Tinian, 2005–2006.

	n	Unit Volume	Total Volume [a]	Unit Mass	Total Mass
Livetrapping					
Haguruma trap [b]	36	~ 0.0084	~ 0.302	0.460	16.6
Sherman trap [c]	121	~ 0.0015	~ 0.187	0.227–0.363 [d]	27.5–43.9 [d]
Total			~ 0.489		44.1–60.5
Snaptrapping					
Victor trap [e]	36	~ 0.00041	~ 0.015	0.132	4.8
Museum Special trap [f]	121	~ 0.00015	~ 0.018	0.049	5.9
Total			~ 0.033		10.7

[a] Represents minimum estimate of total volume; realized volume is greater due to inefficiency when packing traps for transport.

[b] Standard Trading Co., Honolulu, HI

[c] Models LFG, LFAHD, and LFATDG, H.B. Sherman Traps, Inc., Tallahassee, FL

[d] Sherman trap mass dependent on metal choice: aluminum (light weight) or galvanized (heavy weight).

[e] Model M201, Woodstream Corporation, Lititz, PA

[f] Woodstream Corporation, Lititz, PA

Table 16. Total time (person-hours) required for site preparation, 5-day livetrapping activites, and 5-day snaptrapping activities in grassland, *Leucaena* forest, mixed, and native forest habitats on Guam, Rota, Saipan, and Tinian, 2005–2006. Livetrapping and snaptrapping times include time required for trap baiting, trap monitoring, and processing of captured animals.

		Time Required		
	Habitat	Site Preparation	Livetrapping	Snaptrapping
Guam				
MSRG	grassland	19.0	34.4	22.5
ASMF	*Leucaena* forest	40.5	16.7	13.3
GSYF	*Leucaena* forest	52.1	21.5	16.3
PAGO	*Leucaena* forest	37.3	24.3	21.3
GAHF	mixed	60.5	17.5	18.3
NMAR	native forest	41.8	16.5	19.5
RITL [a]	native forest		19.5	26.2
Rota				
SABA	grassland	17.0	41.5	18.8
GAON	*Leucaena* forest	63.2	34.7	16.4
RAPF	mixed	64.2	50.3	21.5
ASAK	native forest	76.5	27.3	17.9
Saipan				
ACHU	grassland	8.5	48.3	15.0
OBYT	*Leucaena* forest	34.0	37.0	28.3
SAEN	mixed	67.5	27.0	15.3
SPOR	mixed	119.0	33.4	23.3
LATT	native forest	51.0	32.5	19.3
Tinian				
KAST	grassland	13.5	42.7	20.3
ABLE	*Leucaena* forest	41.0	38.8	21.6
LSUS	native forest	67.8	39.4	18.3

[a] Site preparation time not recorded.

FIGURES

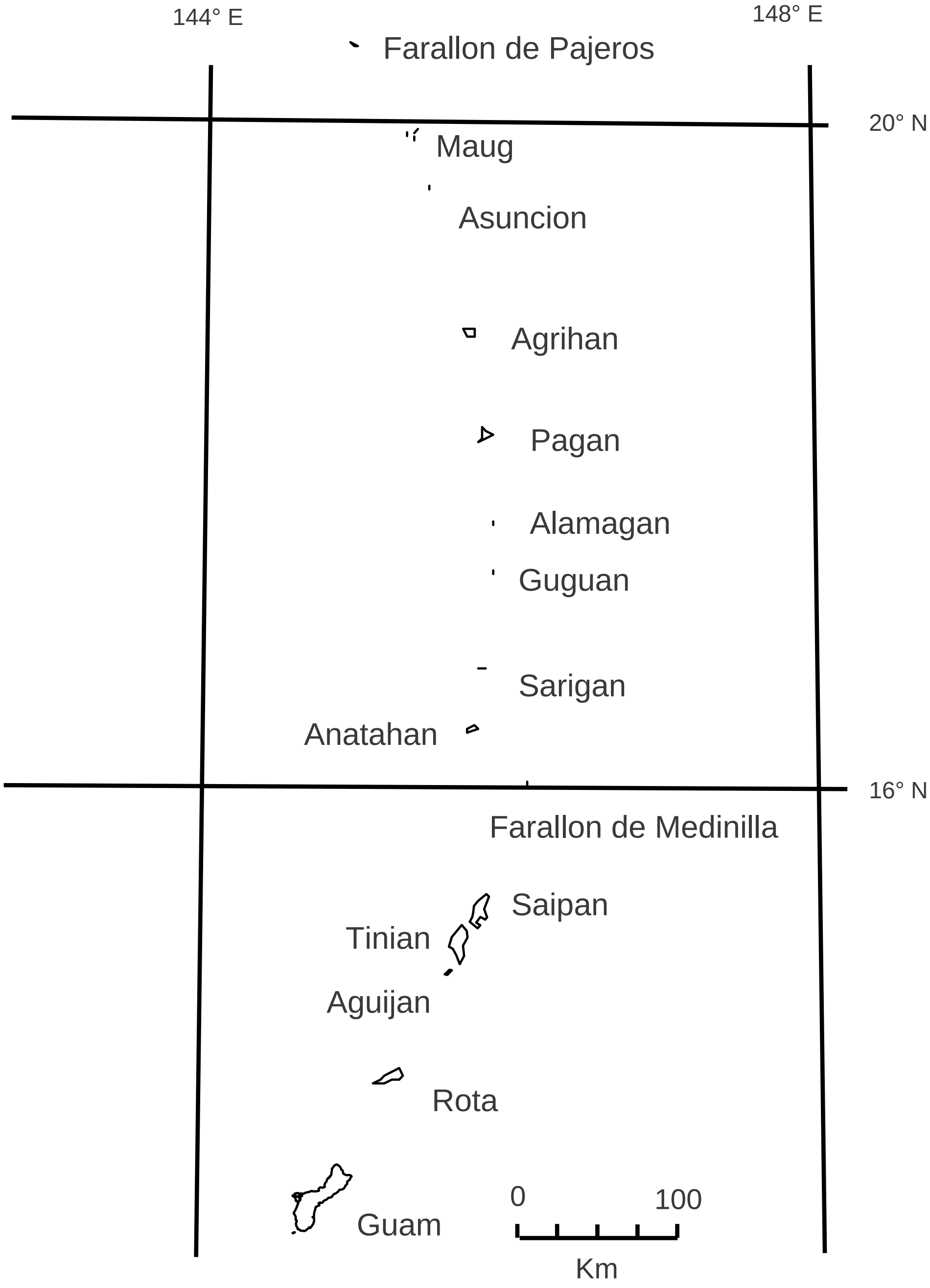


Figure 1. Map of the principal Mariana Islands.

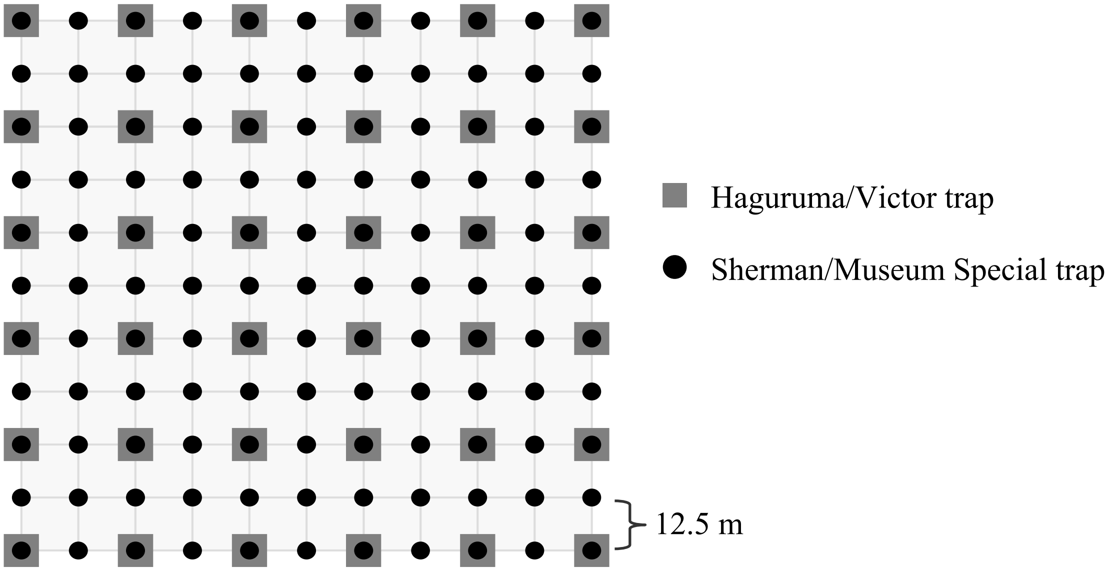


Figure 2. Schematic diagram of 11 x 11 grid (nominal area = 1.56 ha) used during livetrapping (Haguruma and Sherman traps) and snaptrapping (Victor and Museum Special traps) on Guam, Rota, Saipan, and Tinian, 2005–2006.

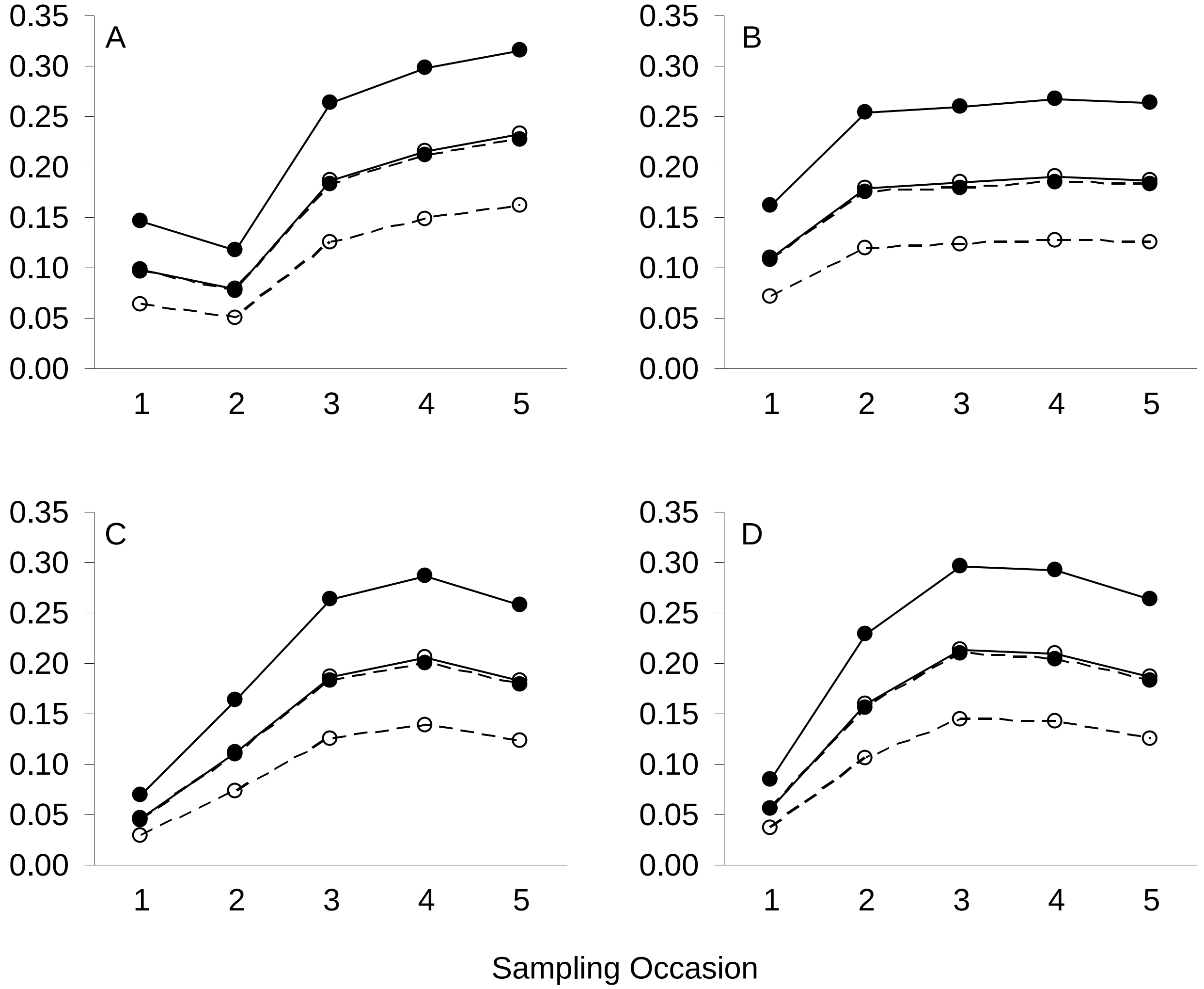


Figure 3. Effect of sex (female = •, male = ○) and reproductive status (reproductively active = solid line, non-reproductive = dashed line) on *Rattus rattus* capture probability generated from mark-recapture modeling of livetrapping data collected on Guam (A), Rota (B), Saipan (C), and Tinian (D), 2005–2006.

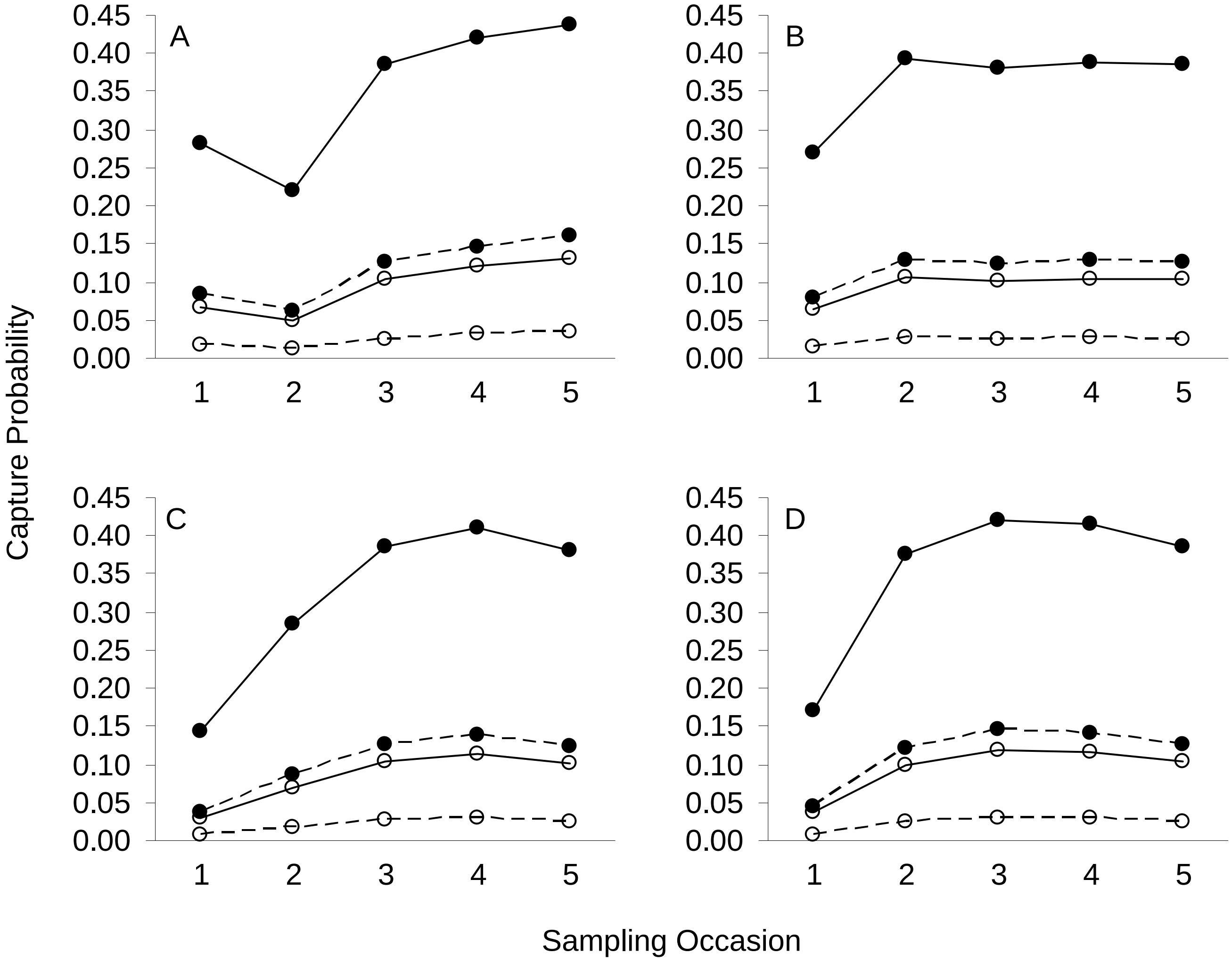


Figure 4. Effect of sex (female = •, male = ○) and reproductive status (reproductively active = solid line, non-reproductive = dashed line) on *Rattus rattus* capture probability generated from removal modeling of livetrapping data collected on Guam (A), Rota (B), Saipan (C), and Tinian (D), 2005–2006.

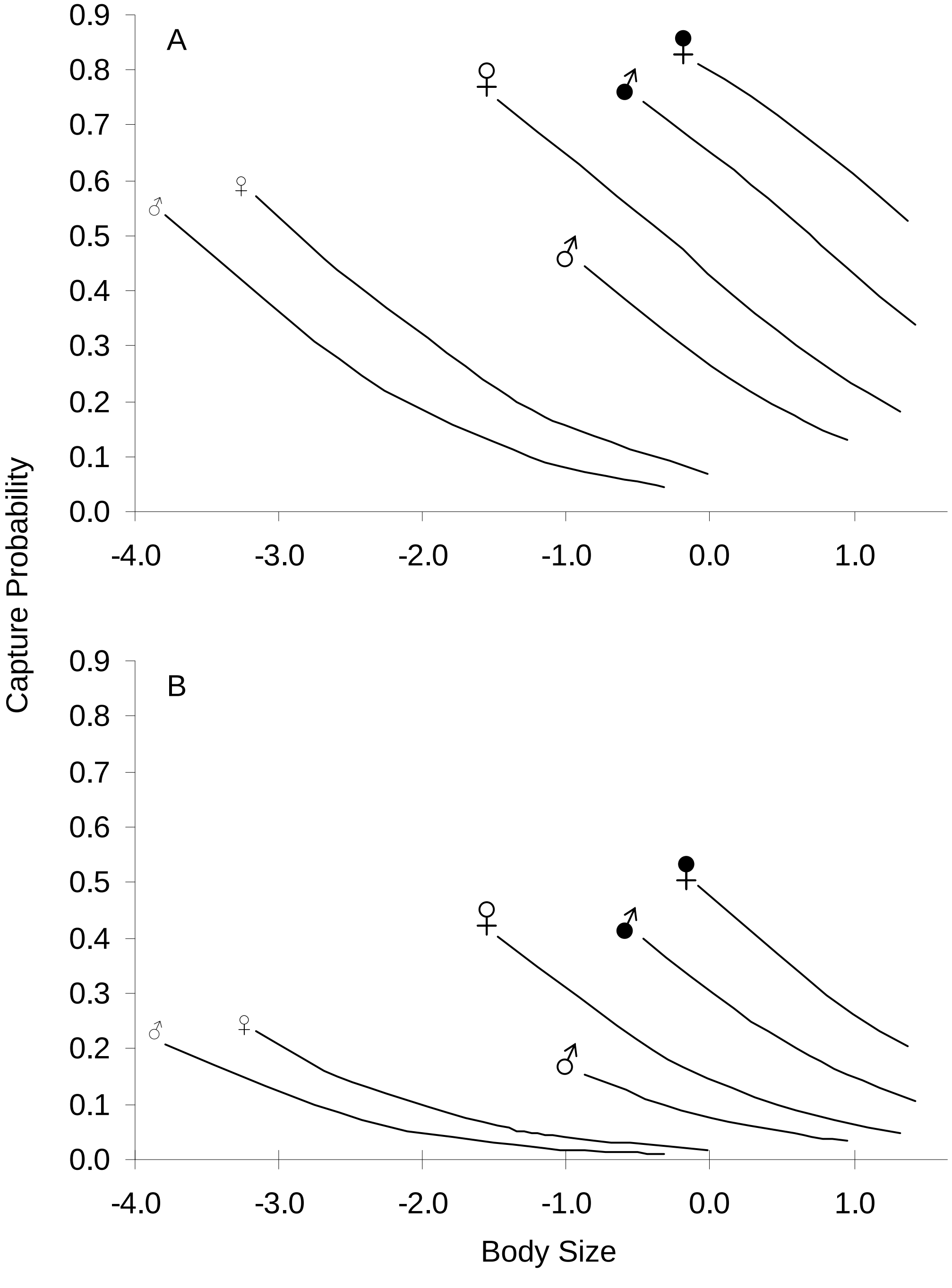


Figure 5. *Rattus rattus* snaptrapping capture probability as a function of sex, age (adult = large symbol, juvenile = small symbol), reproductive status (reproductively active = closed symbol, non-reproductive = open symbol), and body size (a composite variable created from a principle components analysis of mass, head-body length, tail length, hind foot length, and ear length, where size increases from left to right on the x-axis) for Guam (A) and the combination of Rota, Saipan, and Tinian (B).

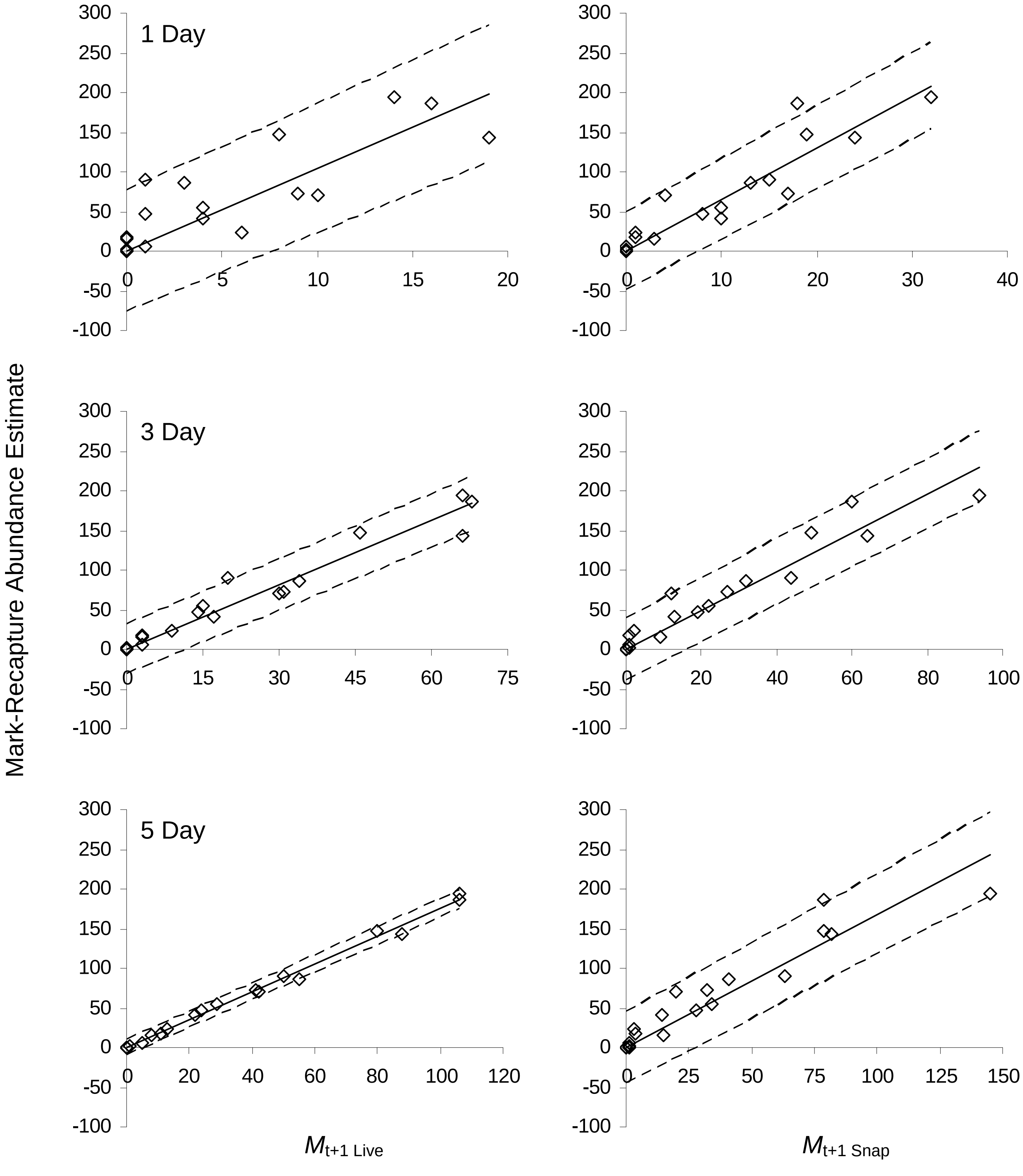


Figure 6. Relationship between *Rattus rattus* mark-recapture abundance estimates generated from livetrapping data and M_{t+1} from 1 day, 3 days, and 5 days of livetrapping ($M_{t+1\ Live}$) and snaptrapping ($M_{t+1\ Snap}$) conducted at 19 sites on Guam (*n* = 7), Rota (*n* = 4), Saipan (*n* = 5), and Tinian (*n* = 3), 2005–2006. Solid lines indicate the best-fit line, constrained to pass through the origin; dashed lines indicate 95% confidence intervals for an individual predicted value. Note the change in M_{t+1} scale as sampling duration increases.

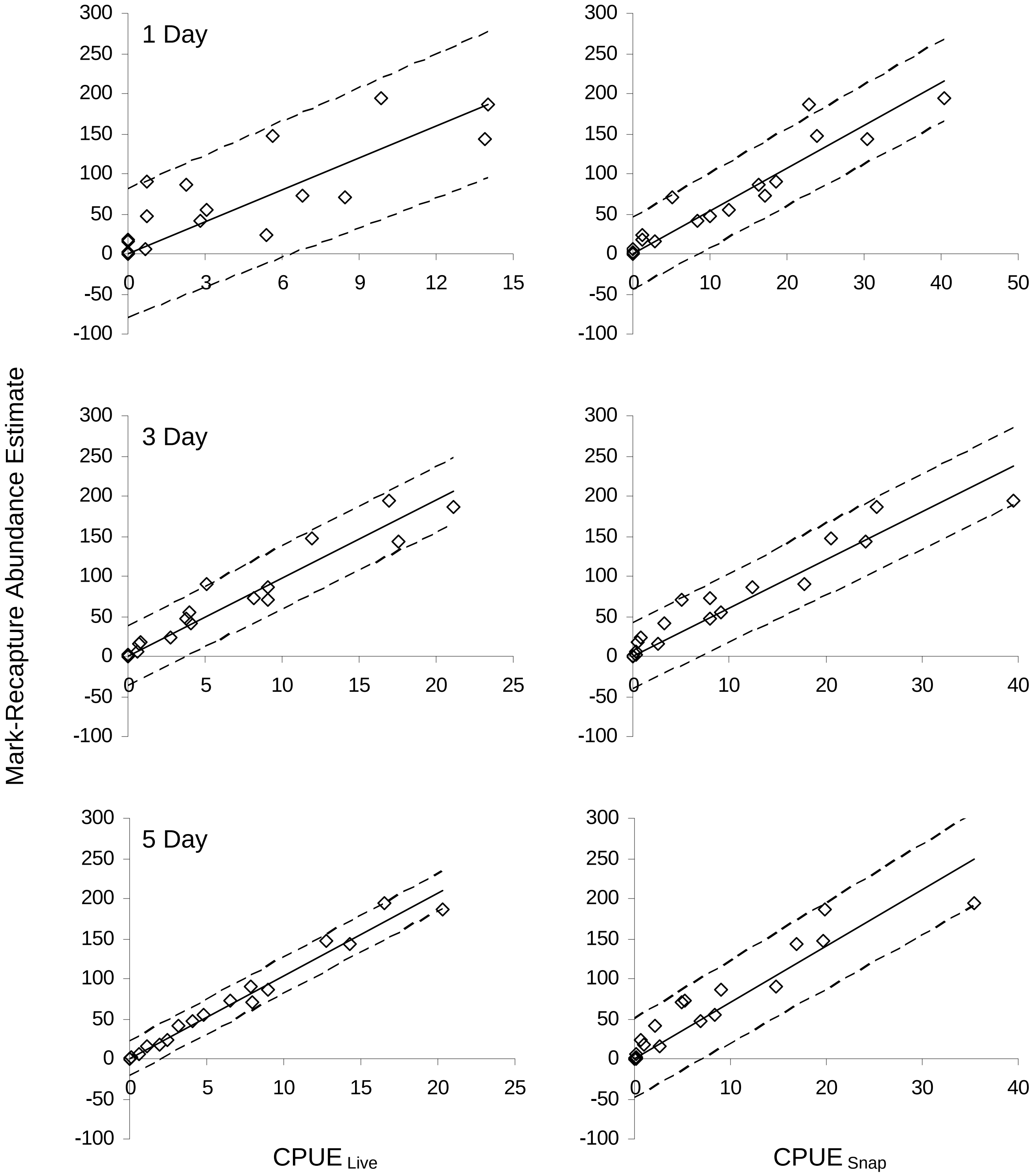


Figure 7. Relationship between *Rattus rattus* mark-recapture abundance estimates generated from livetrapping data and CPUE (captures / 100 corrected trap nights) from 1 day, 3 days, and 5 days of livetrapping (CPUE $_{\text{Live}}$) and snaptrapping (CPUE $_{\text{Snap}}$) conducted at 19 sites on Guam (*n* = 7), Rota (*n* = 4), Saipan (*n* = 5), and Tinian (*n* = 3), 2005–2006. Solid lines indicate the best-fit line, constrained to pass through the origin; dashed lines indicate 95% confidence intervals for an individual predicted value.

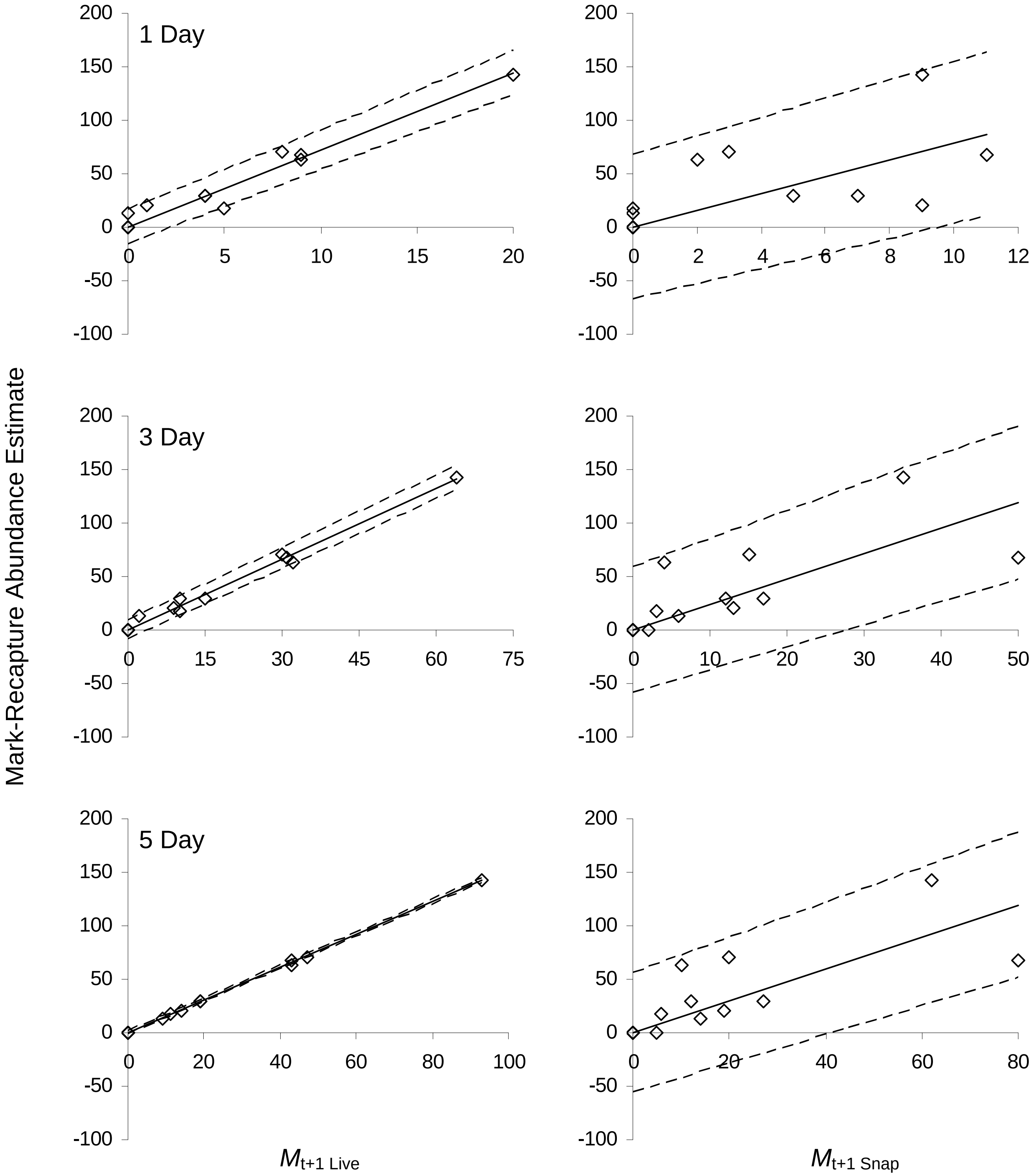


Figure 8. Relationship between *Suncus murinus* mark-recapture abundance estimates generated from livetrapping data and M_{t+1} from 1 day, 3 days, and 5 days of livetrapping ($M_{t+1\ Live}$) and snaptrapping ($M_{t+1\ Snap}$) conducted at 19 sites on Guam ($n = 7$), Rota ($n = 4$), Saipan ($n = 5$), and Tinian ($n = 3$), 2005–2006. *S. murinus* was not captured on Rota and the 4 sites from this island are not included. Solid lines indicate the best-fit line, constrained to pass through the origin; dashed lines indicate 95% confidence intervals for an individual predicted value. Note the change in M_{t+1} scale as sampling duration increases.

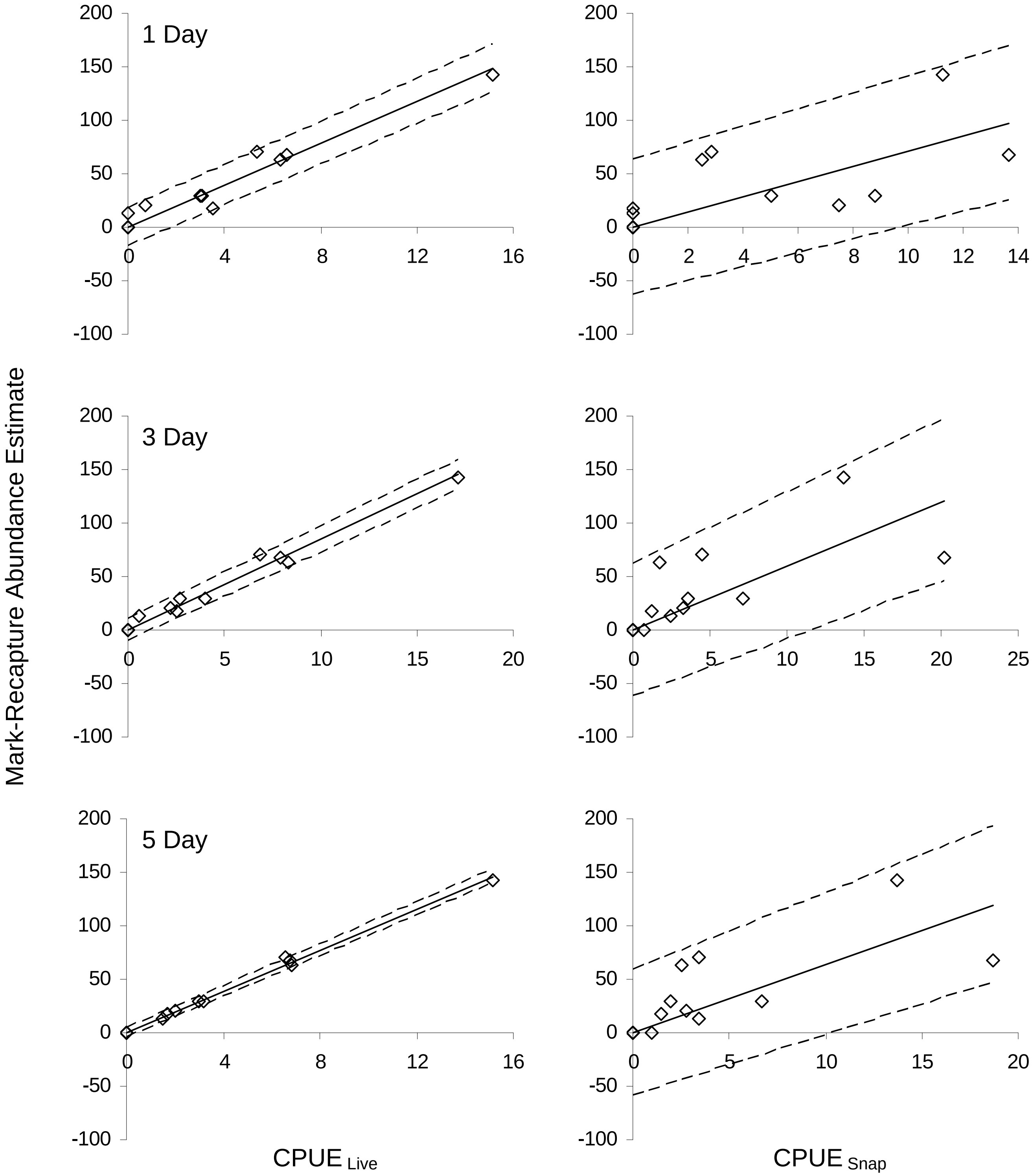


Figure 9. Relationship between *Suncus murinus* mark-recapture abundance estimates generated from livetrapping data and CPUE (captures / 100 corrected trap nights) from 1 day, 3 days, and 5 days of livetrapping (CPUE $_{\text{Live}}$) and snaptrapping (CPUE $_{\text{Snap}}$) conducted at 19 sites on Guam (*n* = 7), Rota (*n* = 4), Saipan (*n* = 5), and Tinian (*n* = 3), 2005–2006. *S. murinus* was not captured on Rota and the 4 sites from this island are not included. Solid lines indicate the best-fit line, constrained to pass through the origin; dashed lines indicate 95% confidence intervals for an individual predicted value.

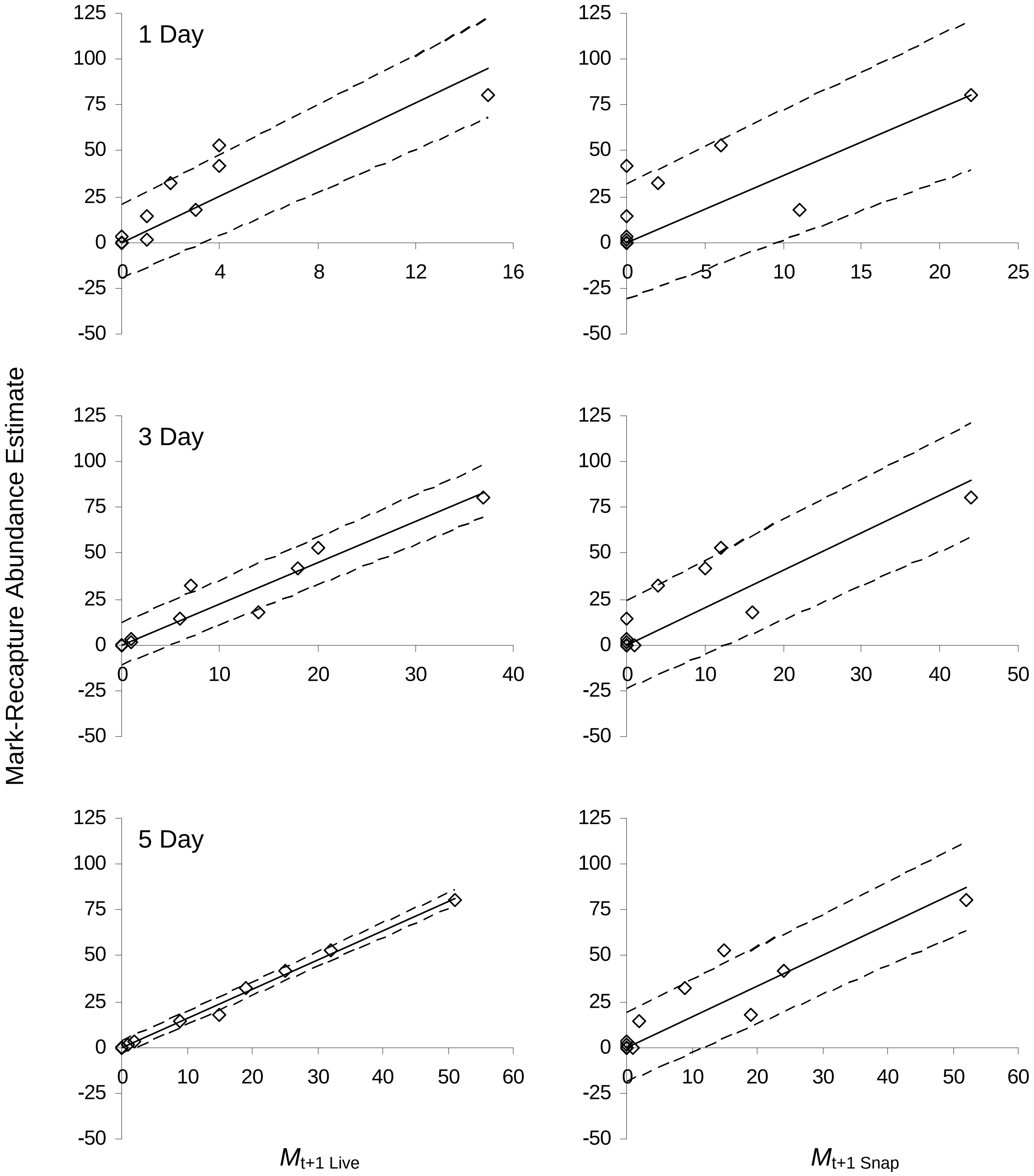


Figure 10. Relationship between *Mus musculus* mark-recapture abundance estimates generated from livetrapping data and M_{t+1} from 1 day, 3 days, and 5 days of livetrapping ($M_{t+1\ Live}$) and snaptrapping ($M_{t+1\ Snap}$) conducted at 19 sites on Guam (n = 7), Rota (n = 4), Saipan (n = 5), and Tinian (n = 3), 2005–2006. Solid lines indicate the best-fit line, constrained to pass through the origin; dashed lines indicate 95% confidence intervals for an individual predicted value. Note the change in M_{t+1} scale as sampling duration increases.

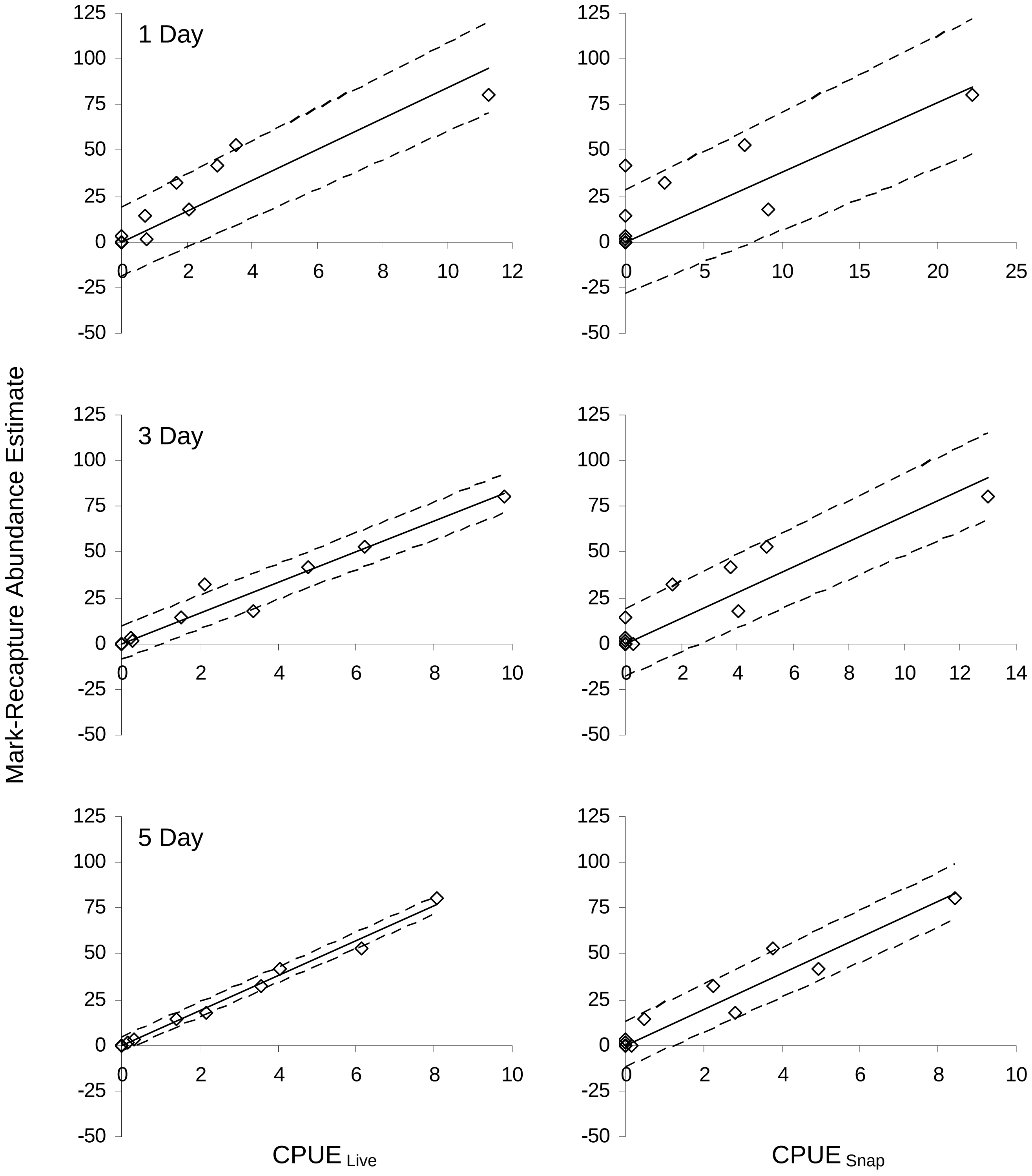


Figure 11. Relationship between *Mus musculus* mark-recapture abundance estimates generated from livetrapping data and CPUE (captures / 100 corrected trap nights) from 1 day, 3 days, and 5 days of livetrapping ($CPUE_{Live}$) and snaptrapping ($CPUE_{Snap}$) conducted at 19 sites on Guam (*n* = 7), Rota (*n* = 4), Saipan (*n* = 5), and Tinian (*n* = 3), 2005–2006. Solid lines indicate the best-fit line, constrained to pass through the origin; dashed lines indicate 95% confidence intervals for an individual predicted value.

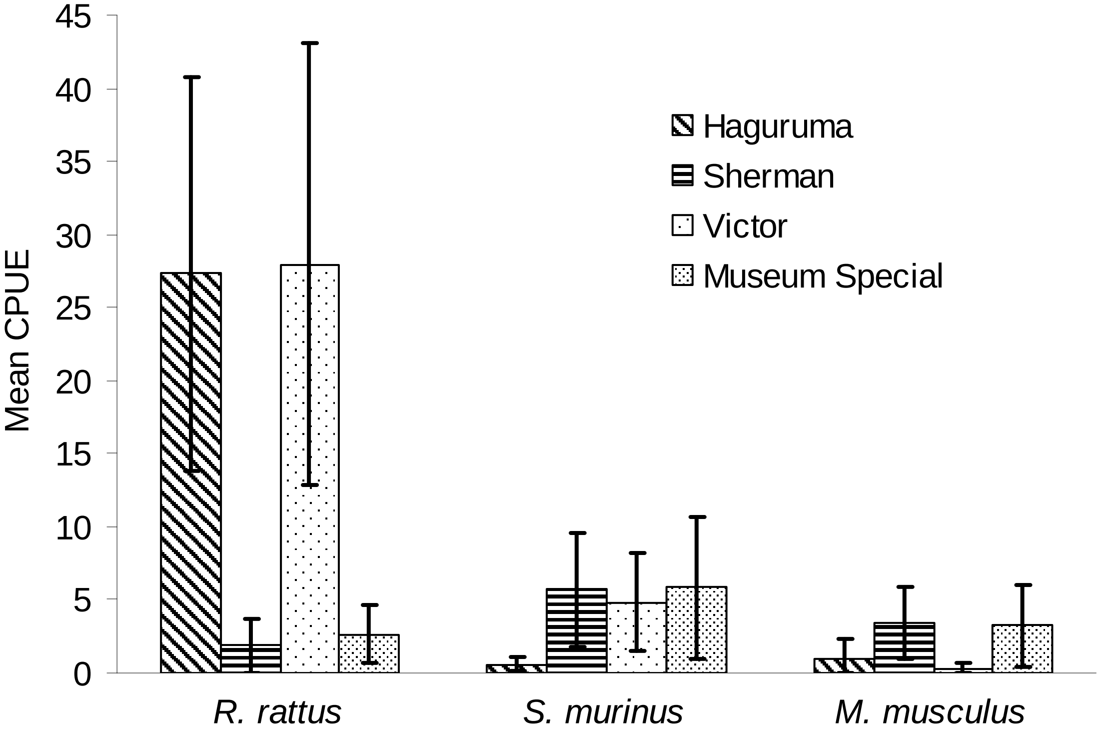


Figure 12. Mean *Rattus rattus*, *Suncus murinus*, and *Mus musculus* CPUE (captures / 100 corrected trap nights) in Haguruma and Sherman live traps (livetrapping) and Victor and Museum Special snap traps (snaptrapping) during small mammal sampling on Guam, Rota, Saipan, and Tinian, 2005–2006. Sampling effort in CPUE calculations includes only sites where a species was captured: *R. rattus n* = 17, *S. murinus n* = 10, and *M. musculus n* = 9. Bars indicate 95% confidence intervals.

APPENDIX 2A. Post-hoc investigation of possible geographic closure violations during small mammal sampling on Guam, Rota, Saipan, and Tinian, 2005–2006

Attempts to generate removal abundance estimates from small mammal livetrapping and snaptrapping data collected on Guam, Rota, Saipan, and Tinian, 2005–2006, were largely unsuccessful, possibly because of a lack of population closure. Failure of population closure (births, deaths, emigration, or immigration) is problematic for all closed population estimation techniques, but especially so for removal abundance estimation methods which also assume that captures decline over successive sampling occasions (Otis et al. 1978, White et al. 1982, Pollock 1991). If we consider only short-duration sampling events, births and deaths are unlikely to impact population closure in a significant fashion. Similarly, it seems unlikely that significant numbers of animals would move away from the sampling area (emigration) during a short-duration sampling event. Further, if emigration did occur during sampling, it would decrease the number of animals in the sampling area and facilitate a decline in new captures. It is therefore unlikely that we would be able to distinguish emigration from the desired effect of declining captures of new individuals over successive sampling occasions. Thus, the most plausible avenue for a failure of population closure, and the only one likely to account for non-declining captures of new individuals, is immigration. While immigration generally refers to the physical movement of new individuals into the sampling area, we might also consider situations where resident animals with very low capture probability (e.g., juvenile or otherwise non-dominant individuals) become more trappable over time as a form of immigration. This might occur as juveniles mature (unlikely to be important during short-duration sampling) or as the social structure of the sampling area is disrupted by the removal of dominant individuals, resulting in increasing social status (and perhaps increasing capture probability) for formerly non-dominant individuals (Summerlin and Wolfe 1973).

If physical immigration were to occur during sampling, we would expect this failure of population closure to be manifested as higher than expected captures in perimeter traps (defined as the 2 outer "rings" of traps; Figure A.1) of the sampling grid, especially during later sampling occasions (unfortunately, changes in social status can not be investigated in this way). Instead, the average deviation in *R. rattus* perimeter captures (observed - expected) during livetrapping was -6% (i.e., fewer perimeter captures than expected), with a maximum daily deviation of only 2% (Figure A.2). Similarly, the averaged deviation in *S. murinus* and *M. musculus* perimeter captures during livetrapping was 6.9% and 0.4%, respectively, with maximum daily deviations of 10% for *S. murinus* and 13% for *M. musculus* (Figure A.2). Immigration seems more likely during snaptrapping, as the physical removal of animals creates territorial vacancies which might attract animals into the sampling area, even over relatively short time spans (Stickel 1946, Fitzgerald et al. 1981, White et al. 1982). It is not surprising, then, that both the average and maximum daily deviation in perimeter captures were greater during snaptrapping for *R. rattus* (1.6% and 9%, respectively), *S. murinus* (7.5% and 17%, respectively), and *M. musculus* (6.1% and 28%, respectively; Figure A.3). It is notable that the maximum daily deviation in perimeter captures for *S. murinus* and *M. musculus* occurred on the first and second day of sampling, rather than during later sampling occasions as might be expected if significant numbers of animals were moving into the sampling area (Figure A.3).

Unfortunately, there is no defined level of increased perimeter captures which might be considered sound evidence either for or against immigration and failure of population closure. Further, one of the inherent issues with grid-based sampling is that more animals are exposed to perimeter traps than interior traps, because perimeter traps are available to animals with territories within the grid as well as animals with territories intersecting the perimeter of the grid, whereas interior traps are only available to animals with territories within the grid (Dice 1938, Stickel 1954, Otis et al. 1978). Thus, a slight positive deviation in perimeter trap captures might be expected as a result of perimeter trap captures of animals with home ranges only partially within the sampling grid (and therefore not likely to be captured in interior traps). Overall, although these results do not seem suggestive of significant immigration, especially during livetrapping, they do not provide sufficient evidence to rule out a failure of population closure due to immigration at any individual study site.

Literature Cited

Dice, L.R. 1938. Some census methods for mammals. Journal of Wildlife Management 2:119–130.

Fitzgerald, B.M., B.J. Karl, and H. Moller. 1981. Spatial organization and ecology of a sparse population of house mice (*Mus musculus*) in a New Zealand forest. Journal of Animal Ecology 50:489–518.

Otis, D.L., K.P. Burnham, G.C. White, and D.R. Anderson. 1978. Statistical inference from capture data on closed animal populations. Wildlife Monographs 62:1–135.

Pollock, K.H. 1991. Modeling capture, recapture, and removal statistics for estimation of demographic parameters for fish and wildlife populations: past, present, and future. Journal of the American Statistical Association 86:225–238.

Stickel, L.F. 1946. The source of animals moving into a depopulated area. Journal of Mammalogy 27:301–307.

Stickel, L.F. 1954. A comparison of certain methods of measuring home range of small mammals. Journal of Mammalogy 35:1–15.

Summerlin, C.T., and J.L. Wolfe. 1973. Social influences on trap response of the cotton rat, *Sigmodon hispidus*. Ecology 54:1156–1159.

White, G.C., D.R. Anderson, K.P. Burnham, and D.L. Otis. 1982. Capture-recapture and removal methods for sampling closed populations. LA-8787-NERP. Los Alamos National Laboratory, Los Alamos, New Mexico, USA.

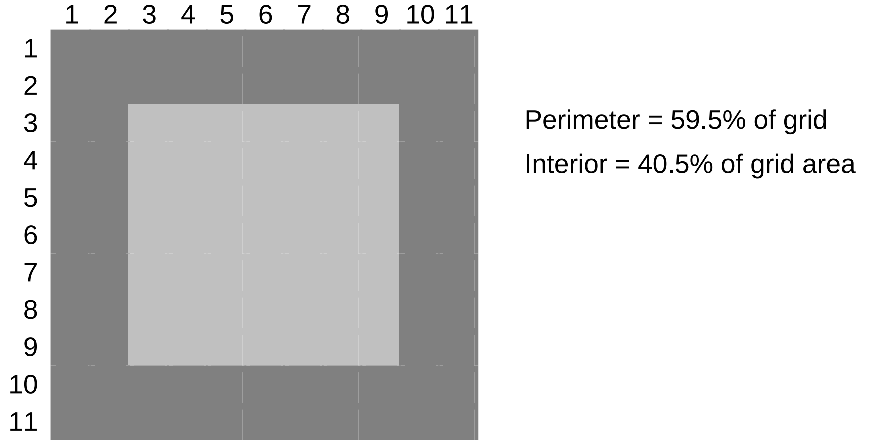


Figure A.1. Schematic representation of perimeter and interior trap assignment for evaluation of geographic closure during livetrapping and snaptrapping.

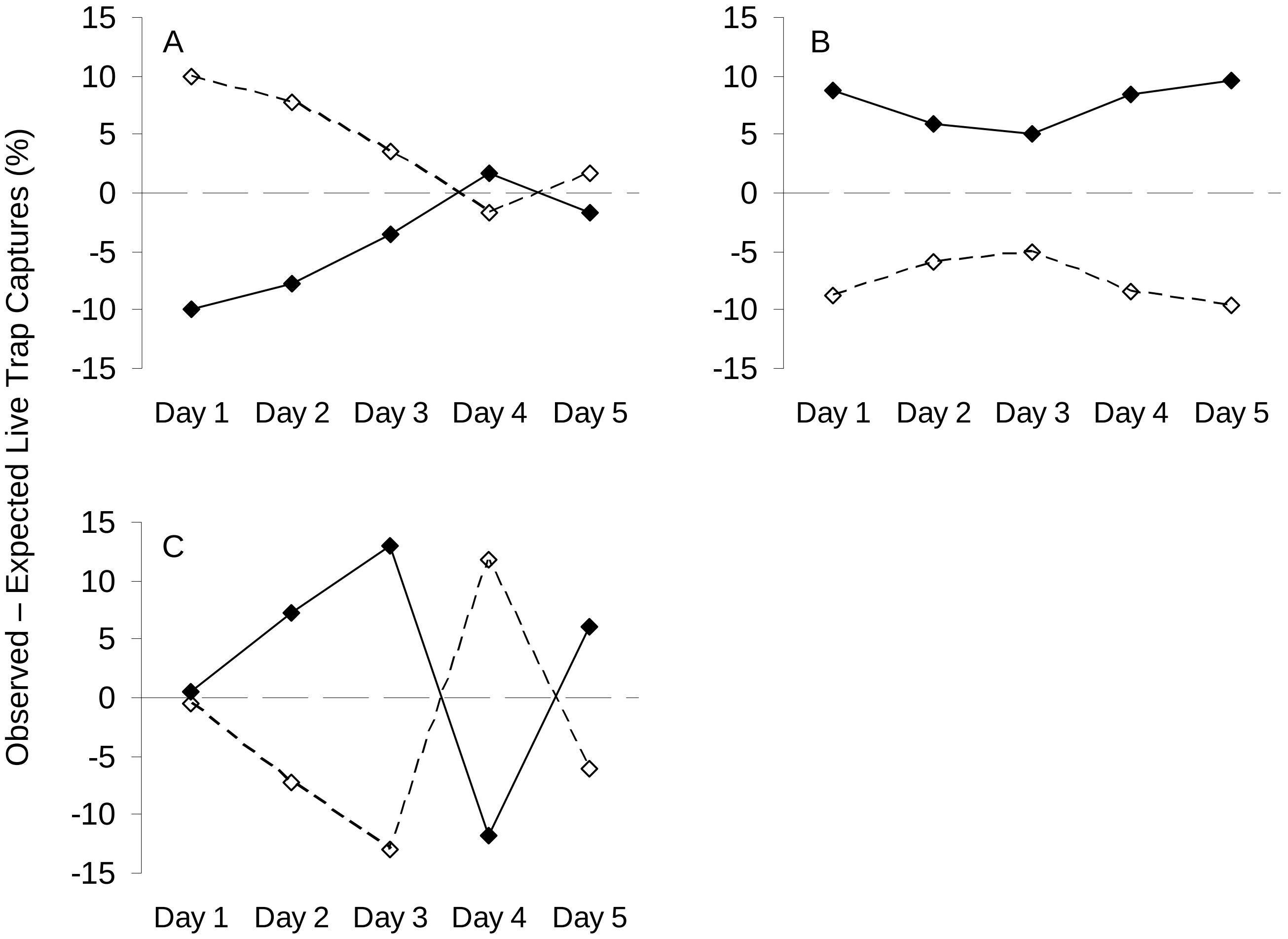


Figure A.2. Percent deviation in observed live trap captures of *Rattus rattus* (A), *Suncus murinus* (B), and *Mus musculus* (C) in interior (◊ and dashed line) and perimeter traps (♦ and solid line), relative to expected live trap captures based on the grid area encompassed by interior and perimeter traps, as specified in Figure A.1.

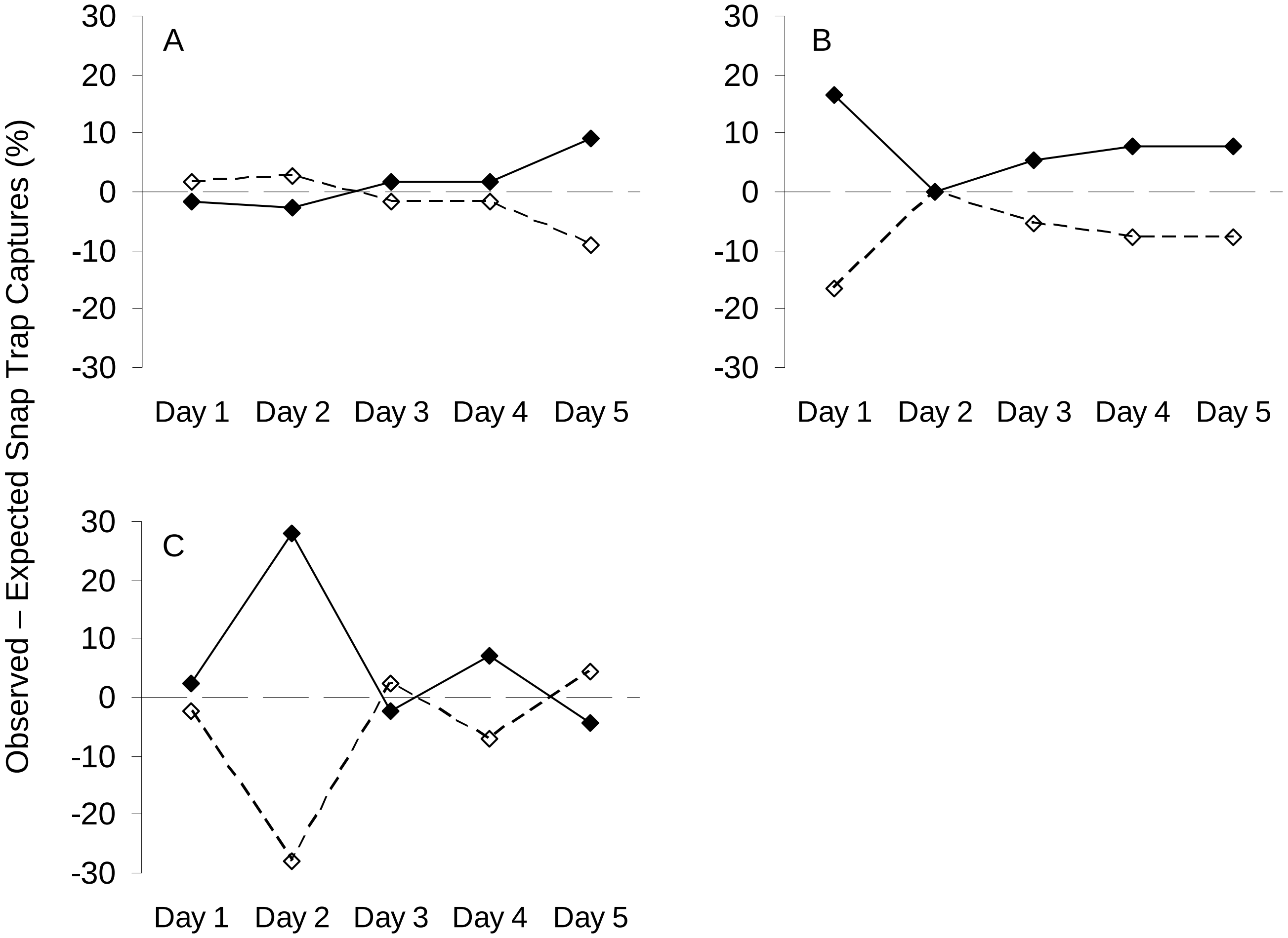


Figure A.3. Percent deviation in observed snap trap captures of *Rattus rattus* (A), *Suncus murinus* (B), and *Mus musculus* (C) in interior (◊ and dashed line) and perimeter traps (♦ and solid line), relative to expected snap trap captures based on the grid area encompassed by interior and perimeter traps, as specified in Figure A.1.

APPENDIX 2B. Post-hoc evaluation of relationship between count-based indices and removal abundance estimates from snaptrapping data

Due to concerns that our evaluation of M_{t+1} and CPUE from livetrapping ($M_{t+1\ \mathrm{Live}}$ and CPUE$_{\mathrm{Live}}$) and snaptrapping ($\mathrm{M}_{\mathrm{t+1\ Snap}}$ and CPUE$_{\mathrm{Snap}}$) may have been biased by our use of mark-recapture abundance estimates from livetrapping ($\hat{\hat{N}}_{\mathrm{M-R\ Live}}$) as a proxy for true abundance, we reevaluated the predictive utility of these indices using removal abundance estimates from snaptrapping ($\hat{\hat{N}}_{\mathrm{REM\ Snap}}$). Note that this change was not justified based on our evaluation of these abundance estimates, as $\hat{\hat{N}}_{\mathrm{M-R\ Live}}$ were clearly more informative (e.g., smaller CVs and narrower 95% CIs) than $\hat{\hat{N}}_{\mathrm{REM\ Snap}}$.

Indices generated from 1, 3, and 5 days of livetrapping and snaptrapping data were strong correlates ($r^2 \geq 0.8$) with $\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ in 7 of 12 comparisons (58%) for *R. rattus*, 5 of 12 comparisons (42%) for *S. murinus*, and 11 of 12 comparisons (92%) for *M. musculus* (Table B.1). In all cases, regression coefficients were > 1.0 (Table B.1).

The utility of M_{t+1} and CPUE as predictors of small mammal abundance differed depending on the sampling method (livetrapping vs. snaptrapping) and sampling duration the index was generated from. For *R. rattus*, $M_{t+1\ \mathrm{Snap}}$ and CPUE$_{\mathrm{Snap}}$ were better predictors of $\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ than $M_{t+1\ \mathrm{Live}}$ and CPUE$_{\mathrm{Live}}$ (Figures B.1, B.2). In all cases, the predictive value of indices improved with increased sampling duration, with the narrowest 95% prediction intervals (95% PIs) observed for indices generated from 5 days of sampling data. For example, a mid-range $M_{t+1\ \mathrm{Snap}}$ value from 1 day of sampling (15 individuals) predicts $\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ of approximately 50–275 individuals, whereas a mid-range $M_{t+1\ \mathrm{Snap}}$ value from 5 days of sampling (60 individuals) predicts $\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ of approximately 130–220 individuals (Figure B.1). In contrast, there was less improvement in predictive value between a mid-range $M_{t+1\ \mathrm{Live}}$ value from 1 day of sampling (10 individuals; 95% PL $\approx$ 0–350 individuals) and a mid-range $M_{t+1\ \mathrm{Live}}$ value from 5 days of sampling (50 individuals; 95% PL $\approx$ 25–250 individuals (Figure B.1). Similar patterns were evident in the predictive value of CPUE. For example, a mid-range CPUE$_{\mathrm{Snap}}$ value from 1 day of sampling (20 captures/100 corrected trap nights) predicts $\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ of approximately 90–290 individuals, whereas the same mid-range CPUE$_{\mathrm{Snap}}$ value from 5 days of sampling predicts $\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ of approximately 200–290 individuals (Figure B.2). Again, 95% PIs were wider for CPUE$_{\mathrm{Live}}$, ranging from approximately 0–350 individuals for a mid-range CPUE$_{\mathrm{Live}}$ value from 1 day of sampling (7 individuals/100 corrected trap nights) to approximately 25–300 individuals for a mid-range CPUE $_{\text{Live}}$ value from 5 days of sampling (Figure B.2).

For *S. murinus*, the predictive value of $M_{\text{t+1 Snap}}$ and CPUE $_{\text{Snap}}$ improved with additional sampling occasions, whereas the predictive value of $M_{\text{t+1 Live}}$ and CPUE $_{\text{Live}}$ was relatively independent of sampling duration, and quite poor overall (Figures B.3, B.4). For example, a mid-range CPUE $_{\text{Snap}}$ value from 1 day of sampling (7 captures/100 corrected trap nights) predicts $\hat{\hat{N}}_{\text{REM Snap}}$ of approximately 50–300 individuals, whereas a mid-range CPUE $_{\text{Snap}}$ value from 5 days of sampling (10 captures/100 corrected trap nights) predicts $\hat{\hat{N}}_{\text{REM Snap}}$ of approximately 200–275 individuals (Figure B.4). In contrast, a mid-range CPUE $_{\text{Live}}$ value (8 captures/100 corrected trap nights) predicts $\hat{\hat{N}}_{\text{REM Snap}}$ of approximately 0–400 individuals, independent of sampling duration (Figure B.4).

For *M. musculus*, the predictive value of $M_{\text{t+1 Snap}}$ and CPUE $_{\text{Snap}}$ improved with additional sampling occasions, although this effect is likely an artifact of an anomalous observation at a single site on the first day of sampling. At this site, where $\hat{\hat{N}}_{\text{REM Snap}} = 54$ individuals, there were 0 *M. musculus* captures on the first day of sampling which inflated the 1-day 95% PIs for both $M_{\text{t+1 Snap}}$ and CPUE $_{\text{Snap}}$ (Figures B.5, B.6). Without this outlier, 95% PIs from the first day of sampling were similar to 95% PIs from 3 and 5 days of sampling, suggesting that the predictive value of $M_{\text{t+1 Snap}}$ and CPUE $_{\text{Snap}}$ was relatively constant, with a mid-range CPUE $_{\text{Snap}}$ value (5 captures/100 corrected trap nights) from 5 days of sampling predicting $\hat{\hat{N}}_{\text{REM Snap}}$ of approximately 50–75 individuals (Figure B.6). In contrast, the predictive value of $M_{\text{t+1 Live}}$ and CPUE $_{\text{Live}}$ seemed to decrease slightly with additional sampling occasions (Figures B.5, B.6). For example, a CPUE $_{\text{Live}}$ value of 5 captures/100 corrected trap nights from 1 day of sampling predicts $\hat{\hat{N}}_{\text{REM Snap}}$ of approximately 40–70 individuals, whereas the same CPUE $_{\text{Live}}$ value from 5 days of sampling predicts $\hat{\hat{N}}_{\text{REM Snap}}$ of approximately 30–85 individuals (Figure B.6).

After reevaluating $M_{\text{t+1}}$ and CPUE, it is clear that our evaluation is biased towards whichever abundance estimate ($\hat{\hat{N}}_{\text{REM Snap}}$ or $\hat{N}_{\text{M}-\text{R Live}}$) is used as a proxy for true small mammal abundance. This is a troubling result, as it complicates any conclusions we might draw from our evaluation of count-based indices of abundance. We can conclude that the predictive value of count-based indices is related to sampling duration, with indices generated from 1 or 3 days of sampling data often having poor predictive value (i.e., wide 95% PIs). Further, we found little difference in the predictive value of $M_{\text{t+1}}$

$_{\text{Live}}$ vs. CPUE $_{\text{Live}}$ or $M_{\text{t+1 Snap}}$ vs. CPUE $_{\text{Snap}}$, perhaps not surprising as CPUE is an extension of $\mathrm{M}_{\text{t+1}}$. We also note that although $M_{\text{t+1 Snap}}$ and CPUE $_{\text{Snap}}$ were better predictors of $\hat{\bar{N}}_{\text{REM Snap}}$ than were $M_{\text{t+1 Live}}$ and CPUE $_{\text{Live}}$, there was little difference in the width of 95% PIs between $M_{\text{t+1 Snap}}$ and CPUE $_{\text{Snap}}$ for predicting $\hat{\bar{N}}_{\text{REM Snap}}$ or $\hat{\bar{N}}_{\text{M}-\text{R Live}}$, especially for indices generated from 1 or 3 days of sampling data (e.g., compare 95% PIs between Figure B.1 and Figure 6 in main body of Chapter 2, Figure B.2 and Figure 7 in main body of Chapter 2, etc). In other words, $M_{\text{t+1 Snap}}$ and CPUE $_{\text{Snap}}$ were no better for predicting $\hat{\bar{N}}_{\text{REM Snap}}$ than they were for predicting $\hat{\bar{N}}_{\text{M}-\text{R Live}}$, except for *S. murinus* 5 day $M_{\text{t+1 Snap}}$ and CPUE $_{\text{Snap}}$ and *M. musculus* 5 day $M_{\text{t+1 Snap}}$. It is not entirely clear, however, if this result is a product of the high variance of $\hat{\bar{N}}_{\text{REM Snap}}$, or is related to some characteristic of $M_{\text{t+1 Snap}}$ and CPUE $_{\text{Snap}}$.

Table B.1. Linear regression slope coefficients (β), standard errors (SE), and squared correlation coefficients [a] (r^2) relating the number of individuals captured (M_{t+1}) and captures/100 corrected trap nights (CPUE) derived from 1-, 3-, and 5-days of livetrapping ($M_{t+1\ \mathrm{Live}}$, CPUE $_{\mathrm{Live}}$) and snaptrapping ($M_{t+1\ \mathrm{Snap}}$, CPUE $_{\mathrm{Snap}}$) to model-averaged removal abundance estimates generated from 5-day snaptrapping data ($\hat{\hat{N}}_{\mathrm{REM\ Snap}}$). All regressions were constrained to pass through the origin.

	1 Day			3 Day			5 Day		
	β	SE	r^2	β	SE	r^2	β	SE	r^2
R. rattus (*n* = 19)									
$\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ vs. $M_{t+1\ \mathrm{Live}}$	16.00	2.63	0.49	4.30	0.44	0.75	2.83	0.24	0.82
$\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ vs. $M_{t+1\ \mathrm{Snap}}$	10.90	0.89	0.83	4.18	0.20	0.94	2.93	0.09	0.97
$\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ vs. CPUE $_{\mathrm{Live}}$	19.88	3.60	0.42	15.25	1.77	0.70	16.40	1.65	0.76
$\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ vs. CPUE $_{\mathrm{Snap}}$	9.02	0.60	0.89	10.36	0.37	0.97	12.37	0.31	0.98
S. murinus (*n* = 15)									
$\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ vs. $M_{t+1\ \mathrm{Live}}$	19.07	3.36	0.54	5.82	0.99	0.56	4.06	0.67	0.58
$\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ vs. $M_{t+1\ \mathrm{Snap}}$	27.68	3.77	0.69	8.71	0.31	0.97	5.39	0.07	0.99
$\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ vs. CPUE $_{\mathrm{Live}}$	25.90	4.51	0.55	22.36	3.82	0.56	25.61	4.21	0.59
$\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ vs. CPUE $_{\mathrm{Snap}}$	25.26	2.60	0.81	22.44	0.62	0.98	23.80	0.52	0.99
M. musculus (*n* = 19)									
$\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ vs. $M_{t+1\ \mathrm{Live}}$	8.36	0.38	0.96	2.81	0.14	0.94	1.91	0.14	0.89
$\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ vs. $M_{t+1\ \mathrm{Snap}}$	5.00	0.55	0.78	2.77	0.14	0.94	2.23	0.03	0.99
$\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ vs. CPUE $_{\mathrm{Live}}$	10.99	0.53	0.95	10.20	0.66	0.91	11.06	1.04	0.83
$\hat{\hat{N}}_{\mathrm{REM\ Snap}}$ vs. CPUE $_{\mathrm{Snap}}$	5.06	0.53	0.80	9.17	0.38	0.96	12.53	0.45	0.97

[a] Squared correlation coefficients for constrained regressions were calculated as $r^2 = 1 - \frac{SSE}{SST_c}$, where SSE = the sum of squared residuals and SST_c = the corrected total sum of squared deviations.

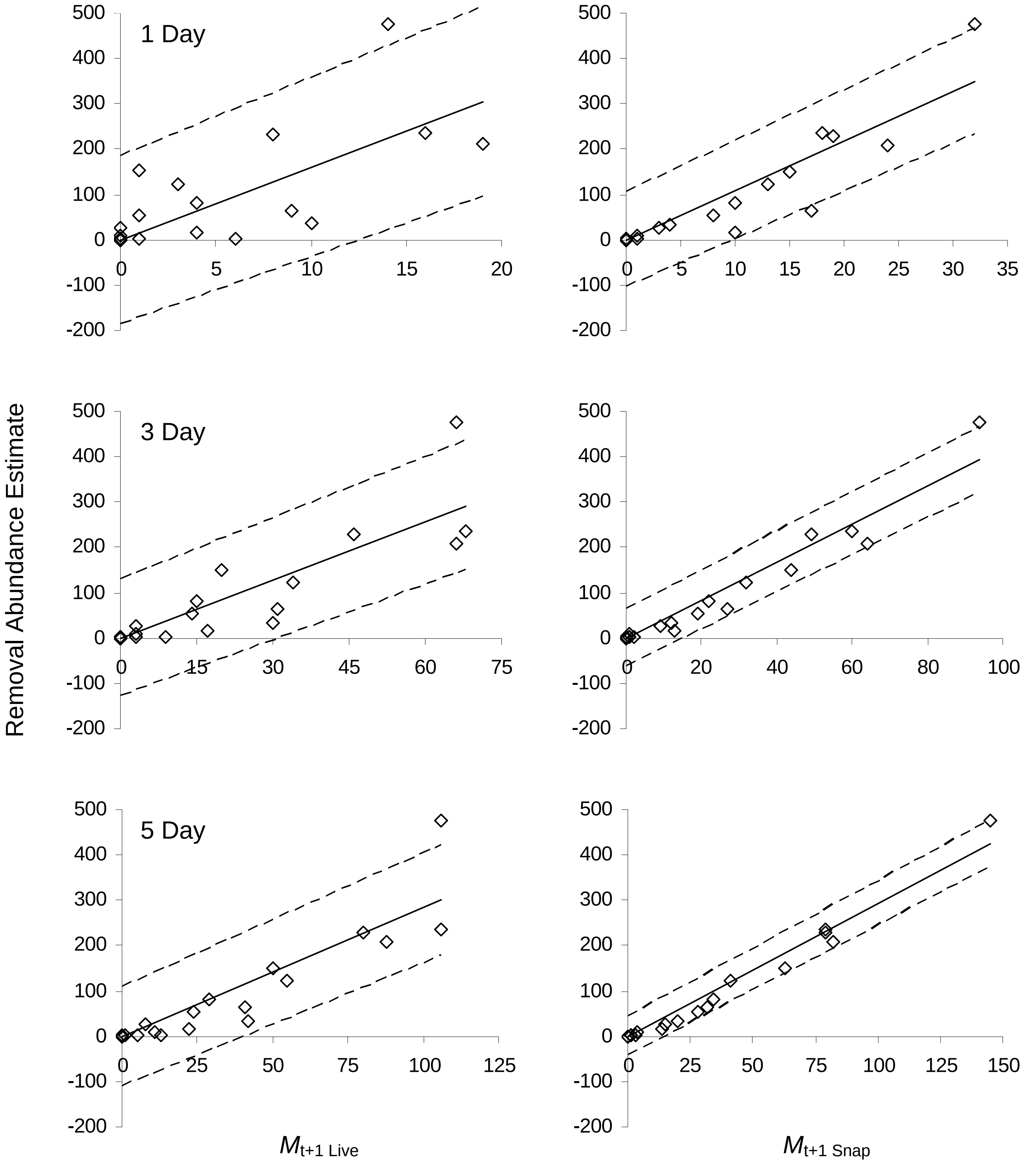


Figure B.1. Relationship between *Rattus rattus* removal abundance estimates generated from snaptrapping data and M_{t+1} from 1 day, 3 days, and 5 days of livetrapping ($M_{t+1\ Live}$) and snaptrapping ($M_{t+1\ Snap}$) conducted at 19 sites on Guam (*n* = 7), Rota (*n* = 4), Saipan (*n* = 5), and Tinian (*n* = 3), 2005–2006. Solid lines indicate the best-fit line, constrained to pass through the origin; dashed lines indicate 95% confidence intervals for an individual predicted value. Note the change in M_{t+1} scale as sampling duration increases.

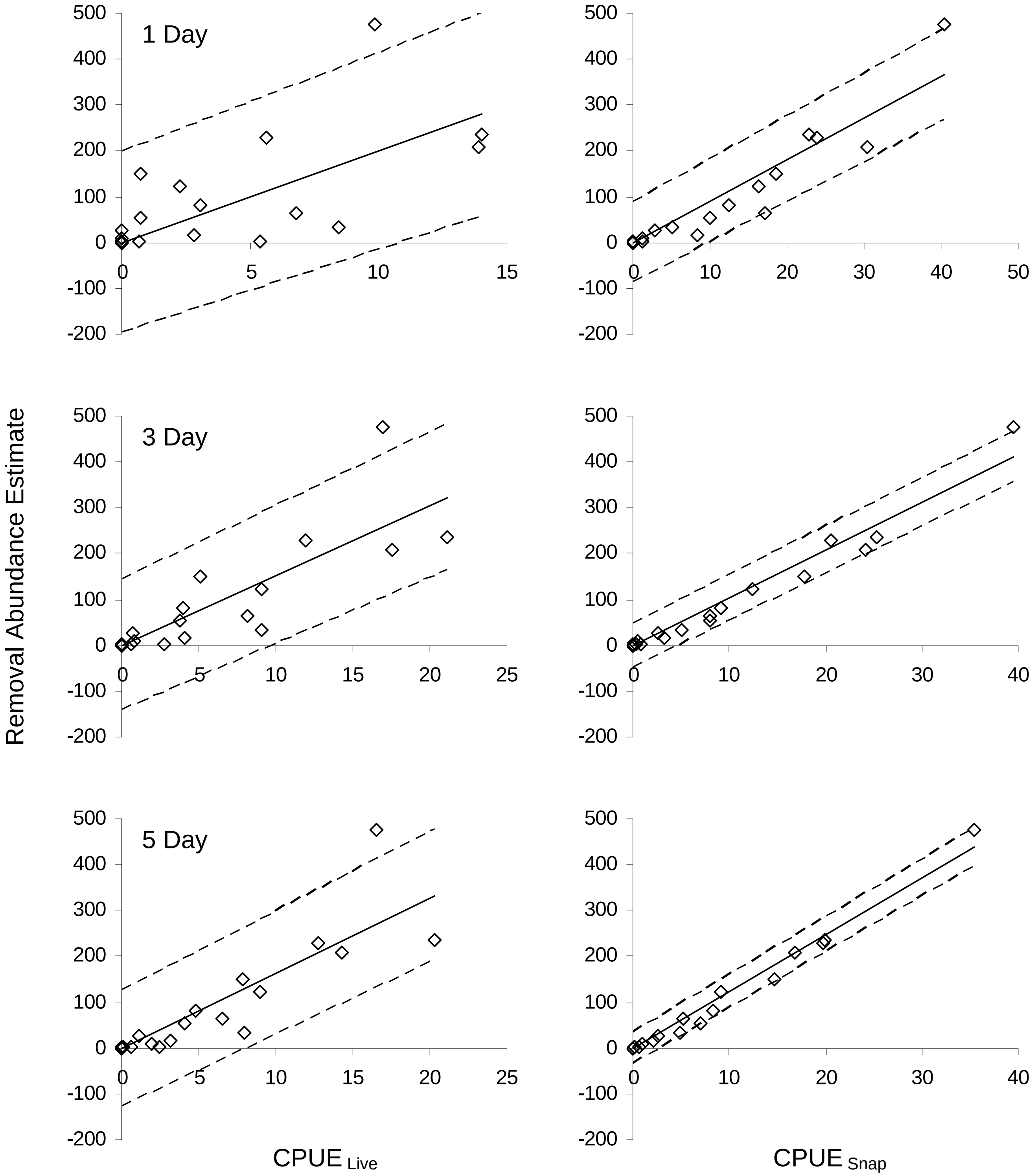


Figure B.2. Relationship between *Rattus rattus* removal abundance estimates generated from snaptrapping data and CPUE (captures / 100 corrected trap nights) from 1 day, 3 days, and 5 days of livetrapping (CPUE $_{\text{Live}}$) and snaptrapping (CPUE $_{\text{Snap}}$) conducted at 19 sites on Guam (*n* = 7), Rota (*n* = 4), Saipan (*n* = 5), and Tinian (*n* = 3), 2005–2006. Solid lines indicate the best-fit line, constrained to pass through the origin; dashed lines indicate 95% confidence intervals for an individual predicted value.

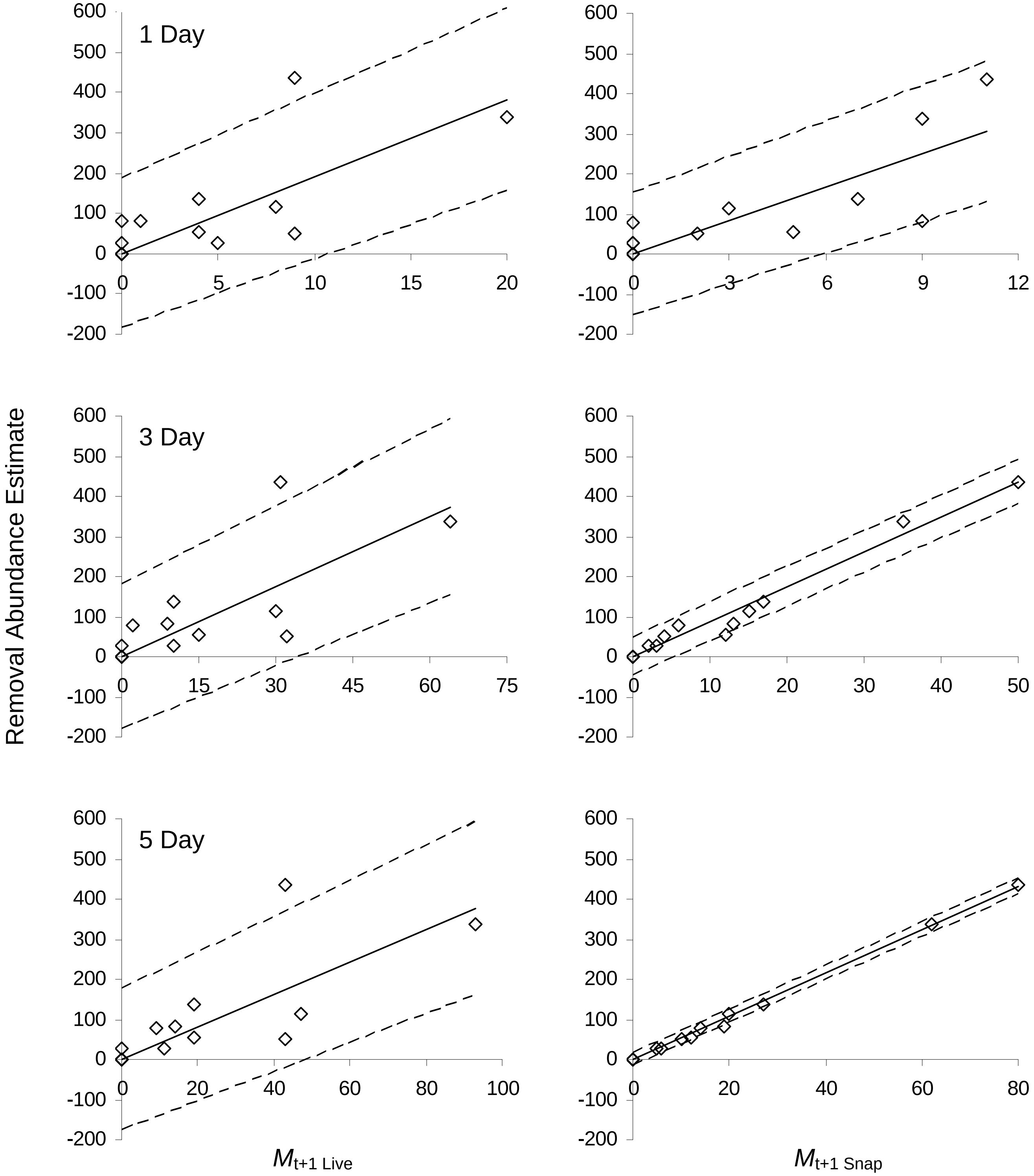


Figure B.3. Relationship between *Suncus murinus* removal abundance estimates generated from snaptrapping data and M_{t+1} from 1 day, 3 days, and 5 days of livetrapping ($M_{t+1\ Live}$) and snaptrapping ($M_{t+1\ Snap}$) conducted at 19 sites on Guam (*n* = 7), Rota (*n* = 4), Saipan (*n* = 5), and Tinian (*n* = 3), 2005–2006. *S. murinus* was not captured on Rota and the 4 sites from this island are not included. Solid lines indicate the best-fit line, constrained to pass through the origin; dashed lines indicate 95% confidence intervals for an individual predicted value. Note the change in M_{t+1} scale as sampling duration increases.

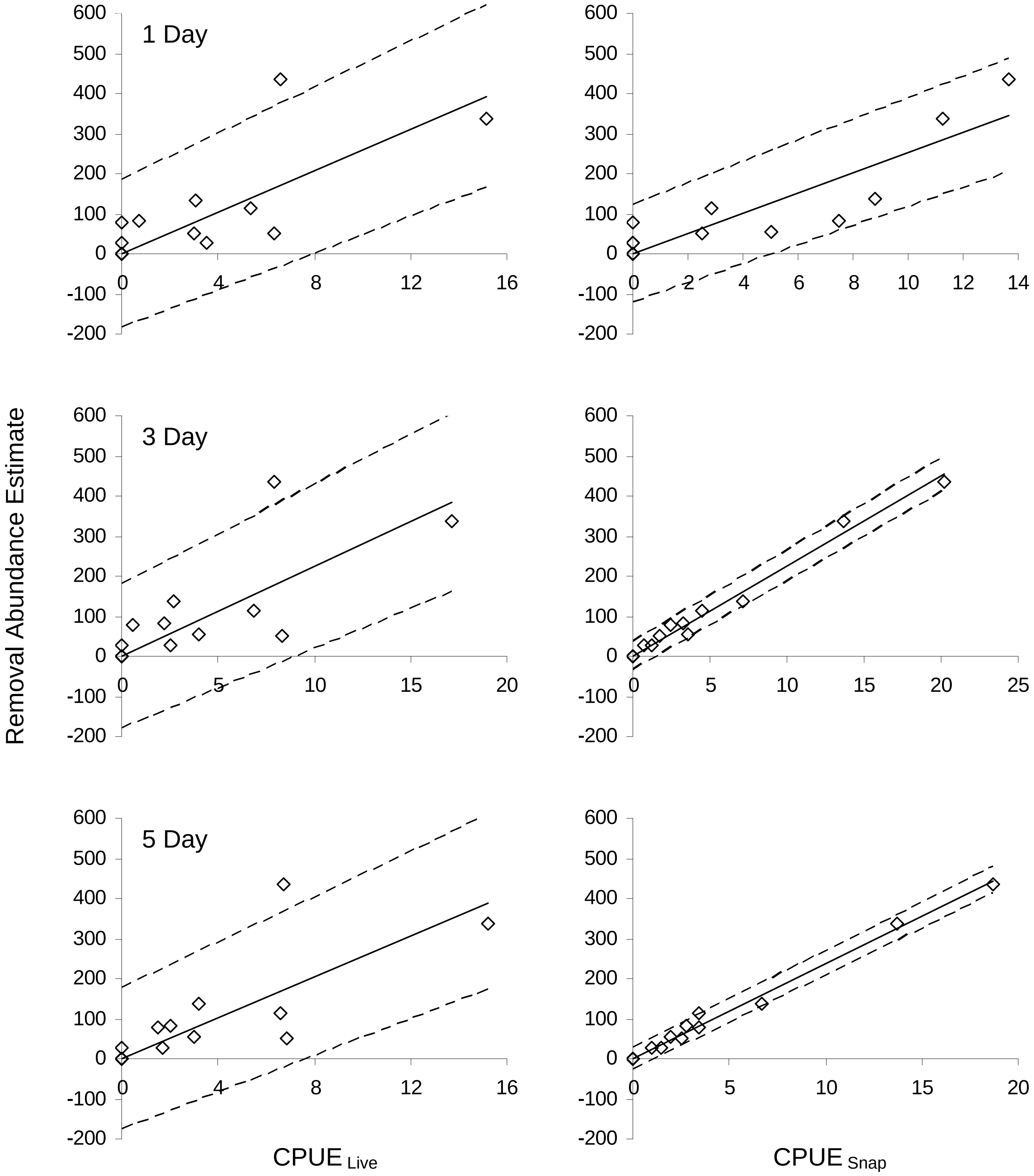


Figure B.4. Relationship between *Suncus murinus* removal abundance estimates generated from snaptrapping data and CPUE (captures / 100 corrected trap nights) from 1 day, 3 days, and 5 days of livetrapping (CPUE $_{\text{Live}}$) and snaptrapping (CPUE $_{\text{Snap}}$) conducted at 19 sites on Guam (*n* = 7), Rota (*n* = 4), Saipan (*n* = 5), and Tinian (*n* = 3), 2005–2006. *S. murinus* was not captured on Rota and the 4 sites from this island are not included. Solid lines indicate the best-fit line, constrained to pass through the origin; dashed lines indicate 95% confidence intervals for an individual predicted value.

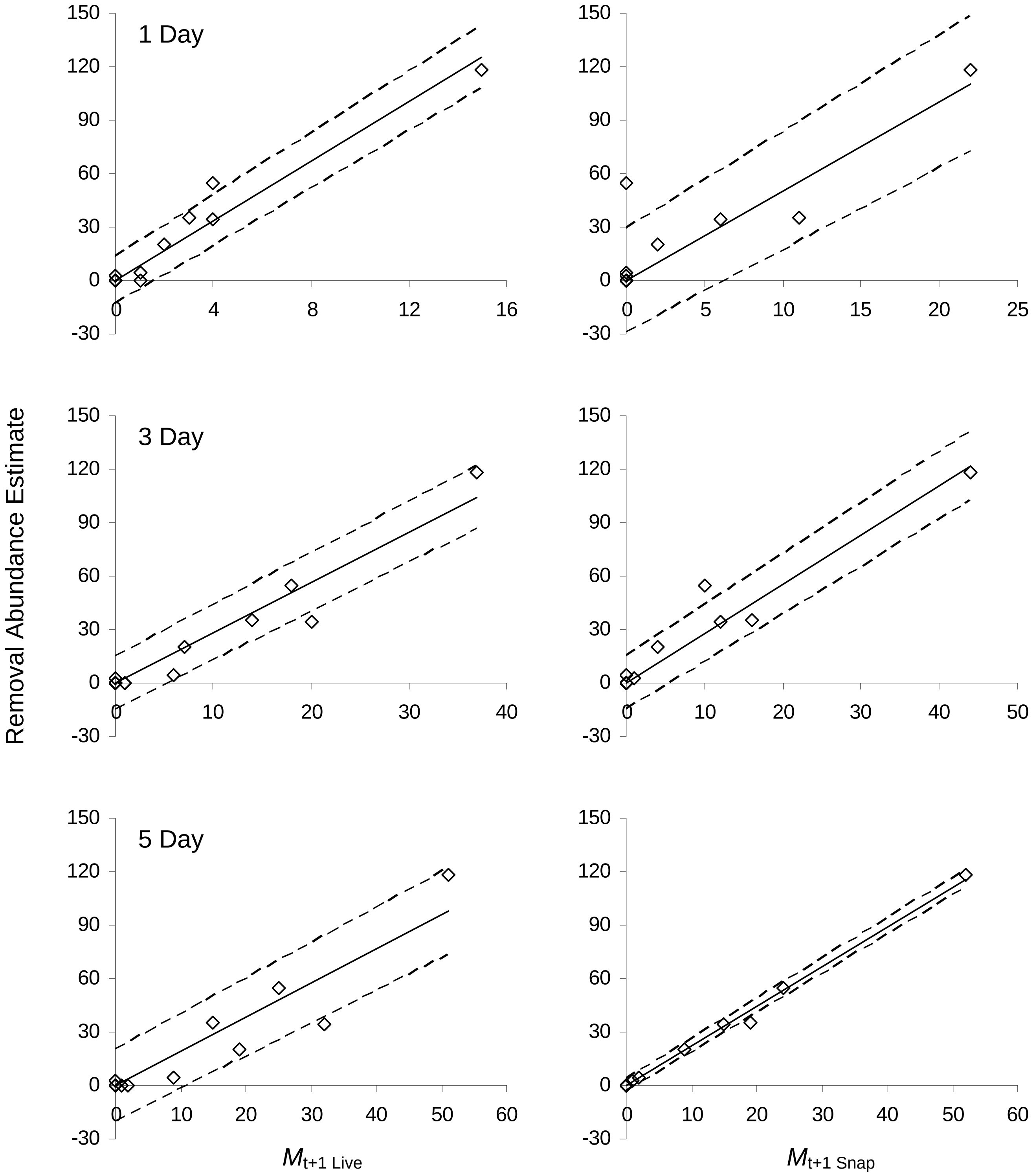


Figure B.5. Relationship between *Mus musculus* removal abundance estimates generated from snaptrapping data and M_{t+1} from 1 day, 3 days, and 5 days of livetrapping ($M_{t+1\ \mathrm{Live}}$) and snaptrapping ($M_{t+1\ \mathrm{Snap}}$) conducted at 19 sites on Guam (n = 7), Rota (n = 4), Saipan (n = 5), and Tinian (n = 3), 2005–2006. Solid lines indicate the best-fit line, constrained to pass through the origin; dashed lines indicate 95% confidence intervals for an individual predicted value. Note the change in M_{t+1} scale as sampling duration increases.

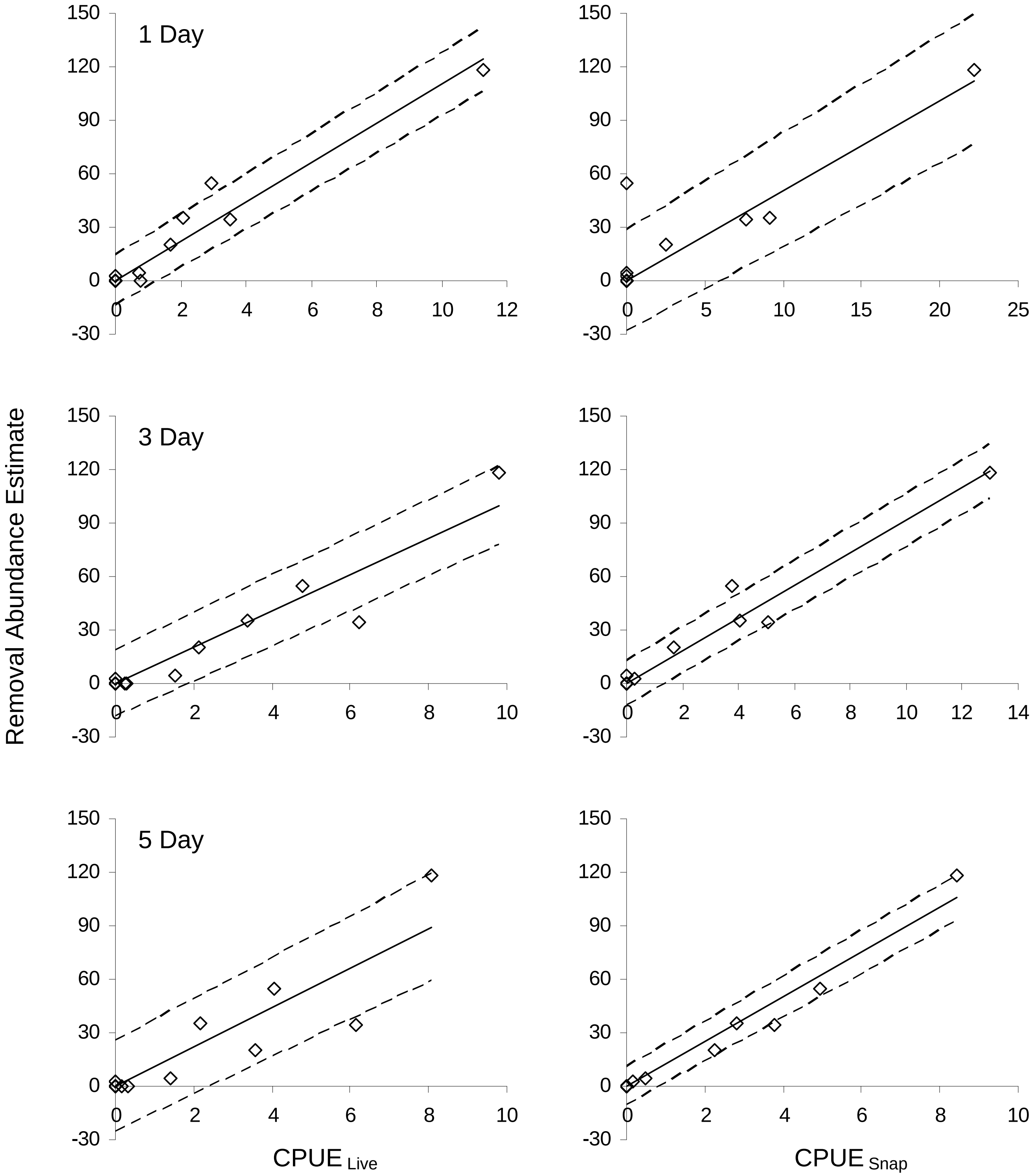


Figure B.6. Relationship between *Mus musculus* removal abundance estimates generated from snaptrapping data and CPUE (captures / 100 corrected trap nights) from 1 day, 3 days, and 5 days of livetrapping (CPUE $_{\text{Live}}$) and snaptrapping (CPUE $_{\text{Snap}}$) conducted at 19 sites on at 19 sites on Guam (*n* = 7), Rota (*n* = 4), Saipan (*n* = 5), and Tinian (*n* = 3), 2005–2006. Solid lines indicate the best-fit line, constrained to pass through the origin; dashed lines indicate 95% confidence intervals for an individual predicted value.

2009

ASSESSMENT FOR PACIFIC SHEATH-TAILED BATS (*EMBALLONURA SEMICAUDATA ROTENSIS*) ON AGUIGUAN, COMMONWEALTH OF THE NORTHERN MARIANA ISLANDS

Compiled by Thomas J. O’Shea and Ernest W. Valdez

Draft

Contents

Section I. Executive Summary 3

Section II. Introduction and Objectives........ 8

Section III. Current and Past Population Status and Use of Caves by Pacific Sheath-Tailed Bats (*Emballonura semicaudata rotensis*) on Aguiguan, Commonwealth of the Northern Mariana Islands.. .. 12

Section IV. Habitat Occupancy and Detection of the Pacific Sheath-Tailed Bat (*Emballonura semicaudata rotensis*) on Aguiguan, Commonwealth of the Northern Mariana Isands 52

Section V. Food Habits of the Pacific Sheath-Tailed Bat (*Emballonura semicaudata rotensis*) on Aguiguan, Commonwealth of the Northern Mariana Islands 75

Section VI. Capture, Morphometrics, Museum Specimens, and Other Sampling and Observations of Pacific Sheath-Tailed Bats (*Emballonura semicaudata rotensis*) on Aguiguan, Commonwealth of the Northern Mariana Islands 86

Section VII. Reproduction of Pacific Sheath-Tailed Bats (*Emballonura semicaudata rotensis*) on Aguiguan, Commonwealth of the Northern Mariana Islands 116

Section VIII. Sampling Guano for Organochlorine Insecticides and Other Contaminants 122

Section IX. Assessment for Pacific Sheath-Tailed Bats (*Emballonura semicaudata rotensis*) on Tinian, Commonwealth of the Northern Mariana Islands 127

Section X. Considerations for Future Management, Monitoring, and Research 133

Author Affiliations and Addresses 136

Section I. Executive Summary

The subspecies of the Pacific sheath-tailed bat that once occurred throughout the Mariana Islands (*Emballonura semicaudata rotensis*) has not been well studied biologically, despite its declining status. It is a small insectivorous bat, and in the Mariana Islands it is known only to roost in caves. All available data indicate that it now occurs only as a single remnant population on Aguiguan. Overall the species is categorized as Endangered by the International Union for the Conservation of Nature and Natural Resources. The subspecies is protected by the Commonwealth of the Northern Mariana Islands (CNMI) law, and is considered a Category 3 candidate for listing under the U.S. Endangered Species Act. This categorization under U.S. law is based on the imminence and magnitude of threats, but further actions have not had the highest priority possible in part because the remaining population on Aguiguan has been considered to be a subspecies of a more widely found species. However, a thorough, modern quantitative morphometric and molecular genetic analysis is needed to verify if the subspecific level in the taxonomic hierarchy is accurate or if full species designation may be warranted for the population in the Marianas Islands.

In this report we document results from a biological assessment for Pacific sheath-tailed bats carried out in 2008 on Aguiguan and Tinian, CNMI. The field work was done by a team consisting of a former Guam Division of Aquatic and Wildlife Resources biologist with past experience surveying for this species and four bat biologists from the U.S. Geological Survey (USGS) Fort Collins Science Center and the USGS Pacific Island Ecosystems Research Center. The assessment consisted of determining present abundance and use of caves on Aguiguan by these bats and interpreting these data in comparison with a synthesis of the literature and past unpublished data; establishing baseline site occupancy models of spatial foraging habitat use through monitoring of ultrasonic echolocation calls; determining basic aspects of diet through analysis of fecal material; sampling bats through capture to obtain new data on reproduction and body size, as well as to collect samples for future genetic analysis; and determining characteristics of temperature and humidity in caves. We conducted a review of specimens available in research museums, and obtained samples from guano deposits that may be useful in analysis for contaminants in comparison with analysis of guano from other islands where these bats have become extinct. We also conducted a limited survey for the presence of these bats on Tinian.

Our report summarizes previously unpublished results on numbers of Pacific sheath-tailed bats roosting in caves on Aguiguan in 1995 and 2003, and compares past results with findings from new surveys conducted in 2008. Overall, we examined the abundance, roosting behavior, and distribution of Pacific sheath-tailed bats on Aguiguan by searching caves and hollow trees for roosting bats during the day. Counts of bats at caves show that a small population of Pacific sheath-tailed bats continues to exist on Aguiguan, with a range of 359-466 individuals counted at five of 41 caves in 2008. Comparison with past counts suggests that this population has increased over the last 13 years. Bats appeared to prefer roosting in larger caves and displayed fidelity toward five of the seven roosts found occupied in the study.

Occupied caves were larger than most unoccupied caves but had similar conditions of temperature and humidity. In 2008 one cave consistently housed the largest colony, with a range of 308–382 bats counted, whereas counts at other occupied caves on Aguiguan yielded 1–64 individuals. Slight variability occurred in replicate counts on different dates during the 2008 survey. We found no evidence of hollow tree trunks being used as roosts. It is possible that a small number of colonies of these bats may remain undiscovered at inaccessible caves on Aguiguan.

Evaluation of trends in colony sizes of cave bats throughout the world generally relies on count data that are uncalibrated index values, which can be difficult to interpret. Therefore this assessment also sought to utilize a recently developed quantitative approach to establish a baseline site-occupancy model of spatial occurrence of foraging Pacific sheath-tailed bats on Aguiguan. This method uses detection of bat ultrasonic calls to assess presence-absence of foraging bats at night in relation to various habitat attributes. Thirty-one echolocation stations were deployed across Aguiguan between 25 June and 14 July 2008. Twenty-one of the 31 stations recorded ultrasonic pulses from sheath-tailed bats over a period of 19 days, with 35,858 calls recorded. Ten percent of the calls were characterized as peak activity, 40% as moderate activity, and 50% as brief passes. Analyses show that peak activity and occurrence is related to canopy cover, vegetation stature, and distance to known roosts. Native limestone forest is preferred foraging habitat. Echolocation calls of Pacific sheath-tailed bats were characterized for the first time, and search-phase calls were similar to those of other emballonurid bats that use a narrow bandwidth and short pulse duration to forage in cluttered vegetation.

There has been no prior information on the food habits of the Pacific sheath-tailed bat anywhere in the species' range. Herein we reported on new findings from analysis of fecal material from this bat on Aguiguan. We collected and analyzed 200 fecal pellets of bats from two roosts (Guano Cave and Crevice Cave). The diet of the Pacific sheath-tailed bat was diverse, but mostly consisted of small-sized prey ranging from 1.7 to 6.4 mm in length. Overall hymenopterans (ants, wasps, and bees), lepidopterans (moths), and coleopterans (beetles) were the three major food items in the diet of bats from both roosts. However, the ranking of volumes of each insect order consumed varied between roosts. At Guano Cave, hymenopterans made up 64% of the diet, followed by coleopterans (10%), and lepidopterans (8%). At Crevice Cave, lepidopterans made up 45% of the diet, followed by hymenopterans (41%), and coleopterans (10%). Within Hymenoptera, most of the prey items belonged to ichneumondoidea (parasitoid wasps), followed by formicids (ants belonging to Formicinae and Ponerinae; i.e., trap-jaw ants). Because alates (= winged adults) of ants and termites (isopterans) found in fecal samples generally have wings only when they are reproductive or establishing new colonies, it is likely that Pacific sheath-tailed bats take advantage of seasonal food sources. In other areas the occurrences of these winged forms are often present during the onset of rains; we sampled guano at the onset of the rainy season on Aguiguan (late June to early July). Lepidopterans, specifically microlepidopterans, likely were another seasonally abundant prey item. Silken fungus beetles and leaf beetles identified in the guano appear to be forest-dependent species and were a consistent component of the bats' diet. Not only do these and other prey items indicate that these bats forage mainly in forest habitat during late June and early July, but that they also capture prey near (above and below) the canopy. From these diet analyses, we categorize the Pacific sheath-tailed bat as an aerial insectivore or hawker, similar to other emballonurids around the world.

We also collected various other samples and obtained information on the biology and natural history of Pacific sheath-tailed bats on Aguiguan. We used standard means to capture Pacific sheath-tailed bats in mist nets while they dispersed or foraged through the forest, but these attempts were largely unsuccessful because these bats were highly maneuverable and easily avoided mist nets on close approach. We successfully captured 12 adult bats and one attached suckling young by using hand nets on bats in flight in the forest, or mist nets set in or near caves used as roosts. Both methods have logistical problems and limitations: in addition to the high maneuverability of the bats precluding use of mist nets in standard configurations, considerable time is required to accrue multiple captures using hand nets. Caves where bats roost are co-occupied by endangered Mariana swiftlets. Thus capturing bats at caves has the potential to disturb both the bats and the swiftlets. We found that these bats can be very sensitive to initial handling, but stress can be reduced by placing bats individually in cloth bags promptly after capture and before examining them. We determined body mass, length of forearm, and reproductive condition of the 12 adult bats. In addition to qualitative features of skull morphology, length of forearm has been given as a characteristic distinguishing between some subspecies of *E*. *semicaudata*. However, these new forearm measurements show that there is considerable overlap in body size between *E. semicaudata rotensis* and the other three subspecies of Pacific sheath-tailed bats. We also collected small wing biopsies from12 bats prior to release for some basic preliminary genetic analyses to ascertain genetic diversity of the population on Aguiguan and the depth of division of this subspecies based on comparison with published data on genetics of *E. s. semicaudata* from Fiji. This work will be carried out by USGS geneticists in 2009. We also prepared two museum voucher specimens of *E. s. rotensis*, increasing the number of known specimens from the Mariana Islands available in United States museums from two to four. We reviewed the literature and queried a limited number of online databases to compile updated information on specimens of Pacific sheath-tailed bats that might be available for taxonomic study. Considerable numbers of specimens including other subspecies are available worldwide (over 380), and about 22 additional specimens from the Marianas Islands (including Guam) are housed in museums in France and Japan. Expanded study of museum specimens and comparative genetic analyses are needed to fully ascertain the systematic status of the Pacific sheath-tailed bat population on Aguiguan.

There is limited information on reproduction in Pacific sheath-tailed bats in the CNMI or elsewhere. Six female bats captured by Wiles and others on Aguiguan late in the rainy season of 2003 were apparently not reproductive. In contrast, seven of the eight female bats we captured in June and July 2008 were either pregnant or lactating. We also observed 11 pups at roosts in caves during June and July 2008; all were singletons. None of the bats we captured were volant young of the year. The presence of reproductive females and pups or embryos in June and July but no volant young suggests the hypothesis that Pacific sheath-tailed bats on Aguiguan may have a diffuse seasonality in reproduction, such that the period of late gestation, lactation, and maturation of young coincides with the late June to early November rainy season. We observed one large embryo in a female dissected in June 2008, as was also observed in a female dissected by Lemke in June 1984. These dissections and the observations of 11 apparent singleton pups suggest a litter size of one. If reproduction occurs only once per year and litter size is one, then the capacity for population growth in Pacific sheath-tailed bats will be very limited. All bats that we captured at caves in 2008 and by others in years past were females, whereas 4 bats captured at dusk dispersing along a steep rocky hillside, not near any known colony, were males. This suggests that perhaps males may form bachelor colonies apart from roosts occupied primarily by females, as is known for other Old World species in the genus *Emballonura*. Elaborate social behavior patterns were also suggested by the audible communication sounds produced by bats that we observed foraging and dispersing through the forest and flying into caves.

The scientific literature includes speculation that the extinction of Pacific sheath-tailed bats on other islands may have been attributable at least in part to past use of organochlorine insecticides. However, there is no chemical or toxicological evidence that bears directly on this speculation. Analyses based on other species of insectivorous bats have shown that concentrations of organochlorine insecticides in bat guano can provide diagnostic evidence of mortality and population declines. Aguiguan has been mostly uninhabited since the use of organochlorines became widespread elsewhere in the world. Thus guano samples from sheath-tailed bats on Aguiguan could provide comparative baselines with which to compare contamination of guano from islands where these bats have become extinct (e.g. Guam). Therefore we used contaminant-free sampling approaches to obtain guano at 3 different depth levels (i.e., surface, 10 and 20 cm below surface) from two areas of a guano pile beneath roosting bats at Guano Cave. These samples are stored in the USGS laboratory at the Fort Collins Science Center and can be made available for future chemical analysis. However, because this guano was deposited over many years, the material also likely includes particles of guano from Mariana swiftlets. The degree of mixing of guano from these two sources should be estimated using microscopic techniques prior to chemical analysis.

Pacific sheath-tailed bats are only known from Tinian based on prehistoric deposits in caves. During the last 4 days and nights of our study we made an effort to document the presence of Pacific sheath-tailed bats on Tinian using echolocation detectors. We also queried knowledgeable individuals, and watched for bats and listened for audible calls during the echolocation surveys. We felt that our best chance for success in documenting bats on Tinian would be echolocation-based sampling in limestone forest areas because of their heavy use of this habitat for foraging on Aguiguan. We deployed two monitoring stations that sampled continuously all night long, both set out for one night in a forest in the Mount Lasso area and for a second night in the Kastiyu Forest. We also sampled for one night at each of these sites using ad hoc walking transects and echolocation detectors during the first part of the night, corresponding to peak times of bat echolocation activity on Aguiguan. No bats were detected. However, this survey was far from exhaustive, and additional effort using echolocation detectors over wider areas of forest and searches of caves will be needed to rule out the possibility that a small remnant population of these bats may still exist on Tinian. Similar echolocation-detector based surveys would also be useful on two other islands in the CNMI (Anatahan and Maug) where tentative sightings were reported in the early 1980s but never subsequently confirmed.

Draft

A number of considerations for future activities stem from the findings of this assessment. These are best characterized as activities related to management for conservation, monitoring, and research. Considerations for management for conservation include limiting disturbance of and access to caves used by roosting bats; and increasing the extent of native limestone forest, decreasing existing stands of invasive plants, and eliminating or avoiding actions that would reduce the amount of native limestone forest on Aguiguan. Considerations for future monitoring of sheath-tailed bats on Aguiguan include periodic monitoring of numbers of bats utilizing key caves, and monitoring the use of foraging habitat with echolocation detectors and site occupancy models. Considerations for research include searching the more inaccessible areas on Aguiguan for the presence of additional colonies that may occupy caves requiring technical climbing and caving skills to reach; increasing the foundation of ecological knowledge of this species pertinent to its conservation and management, including investigations into seasonal aspects of reproduction, roosting, and foraging biology; conducting a modern analysis of the taxonomic status of *Emballonura semicaudata* and its subspecies using combined quantitative morphometric and molecular genetic approaches; and further assessing the possible occurrence of Pacific sheath-tailed bats on Tinian and other islands.

Section II. Introduction and Objectives

Thomas J. O’Shea and Ernest W. Valdez

The subspecies of the Pacific sheath-tailed bat that once occurred throughout the Mariana Islands (*Emballonura semicaudata rotensis*) has not been well studied biologically, despite its declining status. It is a small insectivorous bat and in the Mariana Islands it is known only to roost in caves. Once found throughout the southern Mariana Islands, all available data indicate that it now only occurs as a single remnant population that roosts in a few caves on Aguiguan (e.g. Lemke 1986, see also Section III of this administrative report). There are three other subspecies of Pacific sheath-tailed bats distributed sporadically across southwestern Oceania (Koopman 1997, Helgen and Flannery 2002). However, there is little information available on basic biology of the species anywhere in its range. Reports on population status (summarized in Section III of this administrative report) suggest that in many areas it has seriously declined in abundance. A variety of factors have been hypothesized to be responsible for this decline, but no single cause has been pinpointed that is applicable to all areas. The most widely cited published assessment of the status of the Pacific sheath-tailed bat in the Mariana Islands is that of Lemke (1986), who reported seeing only a few bats on Aguiguan. Lemke’s (1986) assessment prompted Nowak (1994, p. 92) to speculate that “this subspecies may thus rank as one of the world’s most critically endangered mammals.”

Several accounts are available that summarize the distribution, history of its status, and known aspects of the biology of Pacific sheath-tailed bats (e.g. , Bonaccorso and Allison 2008, Hutson and others 2001, U.S. Fish and Wildlife Service 2007; see also other sections of this administrative report). The declining status of Pacific sheath-tailed bat populations has caused it to be placed in various protected categories by different organizations and governments. Overall the species is categorized as Endangered by the International Union for the Conservation of Nature and Natural Resources (Hutson and others 2001, Bonaccorso and Allison 2008). The subspecies *E. s. rotensis* in the Commonwealth of the Northern Mariana Islands (CNMI) is protected by CNMI law, and is considered a Category 3 candidate for listing under the U.S. Endangered Species Act of 1973 (U.S. Fish and Wildlife Service 2007). This categorization under U.S. law is based on the imminence and magnitude of threats, but further actions have not had the highest priority possible in part because the remaining population on Aguiguan is currently understood to be a subspecies of a more widely found species (U.S. Fish and Wildlife Service 2007, 2008). The currently accepted subspecies designation, however, is based on examination of a small series of specimens by Yamashina (1943) and a qualitative judgement with little systematic documentation by Koopman (1997). No thorough, modern quantitative morphometric or molecular genetic analyses have been conducted on this species throughout its range to determine if the subspecific level in the taxonomic hierarchy is accurate for the population in the Marianas Islands, or if full species designation may be warranted.

Given the lack of substantial background biological information pertinent to the conservation and management for Pacific sheath-tailed bats, our study had multiple objectives. In this report we grouped our results under seven separate topics or groups of topics. Each of the remaining sections treats these topics or groups of topics with separate statements of introductory information, materials and methods, results and discussion, and references cited. A final section summarizes considerations for future research and management for sheath-tailed bats on Aguiguan and elsewhere if pertinent.

Our primary effort was dedicated towards summarizing all past information on use of caves as roosts by these bats on Aguiguan, combined with results of new efforts in 2008 to re-visit these caves and obtain new counts for an up-to-date assessment of trends in abundance. Bats pose major challenges for arriving at valid estimates in population trends, many of which are reviewed in a series of papers in the volume edited by O'Shea and Bogan (2003) and summarized by O'Shea and others (2003). There are as yet no well-established, standard methods for estimating bat abundance or colony sizes with a statistically sound theoretical basis. Most bat researchers therefore rely on visual counts of bats emerging at dusk as an index to population size ("emergence counts" of Kunz 2003). As described in Section III of this report, such counts were the basis of the past estimates of colony sizes on Aguiguan and were duplicated in 2008. The surveys in 2008 were further supplemented by using a night vision device and infrared light to make daytime counts in one accessible cave, and by replicate counting to qualitatively assess variability in these counts. Information from the 2008 surveys were then combined with previously unpublished reports and data from the literature to synthesize all past and present knowledge on the distribution and status of this subspecies.

There has been one very recent major statistical advance in sampling bats for trend information. This has been the adapting of the newly developed site occupancy modeling approach in wildlife studies (e.g. MacKenzie and others 2002, 2006) to bats based on presence-absence data obtained through monitoring potential foraging habitat for their ultrasonic echolocation pulses. This combined approach was first used in studies of the endangered Hawaiian hoary bat (*Lasiurus cinereus semotus*) by Gorresen and others (2008). It was adapted for the assessment of habitat use by sheath-tailed bats on Aguiguan in Section IV of this report by Gorresen and others, based in part on earlier demonstrations that bat echolocation activity can be detected in Aguiguan's native limestone forest by Esselstyn and others (2004). This carefully designed monitoring of bat-produced ultrasounds was intended to provide a baseline for future monitoring of sheath-tailed bat habitat use, an improved understanding of the areas these bats use for foraging on Aguiguan, and new information on characteristics of Pacific sheath-tailed bat echolocation calls and activity patterns.

Subsequent sections of this report deal with some basic biological and natural history information and samples pertinent to the conservation of sheath-tailed bats that were also obtained during the course of fieldwork on Aguiguan in parts of June and July 2008. These include the first description of insect prey of importance in the diet of these bats based on fecal analyses, results of the first attempts to capture sheath-tailed bats in foraging and dispersal areas, new information pertinent to understanding vital parameters of reproduction, obtaining of samples of guano for assessment of contaminant concentrations and biopsies for genetic studies in the future, new information on body size (of relevance to subspecies characteristics), and an updated summary of museums specimens of all subspecies now available for study at research museums around the world. We also report on a pilot study that attempted to assess the presence of echolocating sheath-tailed bats in likely foraging habitat on Tinian, where there has been no evidence of occurrence since pre-historic times.

Acknowledgments

The effort in 2008 was carried out through an interagency agreement between the U.S. Fish and Wildlife Service (FWS), Ecological Services field office, Honolulu, Hawaii and the U.S. Geological Survey (USGS) Fort Collins Science Center, based on a larger joint program in the CNMI by FWS and the U.S. Navy. Curt Kessler of FWS provided guidance and coordination, logistic support, and loan of equipment. Patricia Stevens of the USGS Fort Collins Science Center provided encouragement and oversight, as did Loyal Mehrhoff of the USGS Pacific Island Ecosystems Research Center. All phases of the project were enabled by the previous ground-breaking work on this subspecies by Gary Wiles and his colleagues, and by Wiles' guidance and knowledge of Aguiguan in the field. We thank the following individuals for review of various sections of this report: Frank Bonaccorso, Paul Cryan, Marcos Gorresen, Daniel Neubaum, Melissa Neubaum, T. Rodhouse, and Gary Wiles. Work in the field would not have been possible without the substantial assistance of Elvin Masga, Jess Omar, and Fabi Muna of the CNMI Division of Fish and Wildlife. Surveys on Tinian were aided by Ton Castro. Additional assistance was given by David Evangelista, Greg Camacho, John Salas, Mike Palacios, Rodney Camacho, and Jay Camacho. Marcos Gorresen and Frank Bonaccorso were supportive in the field and gave helpful suggestions throughout. Anne Brooke of the U.S. Navy provided initial suggestions in planning. Sylvan Igisomar, Director of the CNMI Division of Fish and Wildlife, gave enthusiastic approval, and the work was carried out under authority of Scientific Research Permit 01048-08 issued by the Division of Fish and Wildlife, Commonwealth of the Northern Mariana Islands. Any use of trade, product, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the U.S. Government.

References Cited

Bonaccorso, F., and Allison, A., 2008, *Emballonura semicaudata*: In: IUCN 2008, 2008 IUCN Red List of Threatened Species, www.iucnredlist.org.

Esselstyn, J. A., Wiles, G.J., and Amar, A., 2004, Habitat use of the Pacific sheath-tailed bat (*Emballonura semicaudata*) on Aguiguan, Mariana Islands: Acta Chiropterologica, vol. 6, p. 303-308.

Gorresen, P. M., Miles, A. C., Todd, C. M. , Bonaccorso, F. J., and Weller, T. J., 2008, Assessing bat detectability and occupancy with multiple automated echolocation detectors: Journal of Mammalogy, vol. 89, p. 11-17.

Helgen, K. M., and Flannery, T. F., 2002, Distribution of the endangered Pacific sheath-tailed bat *Emballonura semicaudata*: Australian Mammalogy, vol. 24, 209-212.

Hutson, A. M., Mickleburgh, S. P., and Racey, P. A. (compilers), 2001, Microchiropteran bats: global status survey and conservation action plan. IUCN/SSC Chiroptera Specialist Group, International Union for the Conservation of Nature and Natural Resources, Gland, Switzerland and Cambridge, United Kingdom.

Koopman, K. F., 1997, The subspecies of *Emballonura semicaudata* (Chiroptera: Emballonuridae): Journal of Mammalogy, vol. 78, p. 358-360.

Kunz, T.H., 2003, Censusing bats: challenges, solutions, and sampling biases, in O'Shea, T. J., and Bogan, M. A.,eds., *Monitoring trends in bat populations of the United States and territories: problems and* *prospects*: Washington, D.C., U.S. Geological Survey, Biological Resources Discipline, Information and Technology Report, USGS/BRD/ITR--2003-003, p. 9-19.

Lemke, T. O., 1986, Distribution and status of the sheath-tailed bat (*Emballonura semicaudata*) in the Mariana Islands: Journal of Mammalogy, vol. 67, p. 743-746.

MacKenzie, D. I., Nichols, J. D. , Lachman, G. B., Droege, S., Royle, J. A., and Langtimm, C.A., 2002, Estimating site occupancy when detection probabilities are less than one: Ecology, vol. 83, p. 2248–2255.

MacKenzie, D. I., Nichols, J. D., Royle, J. A., Pollock, K. H., Bailey, L. L., and Hines, J. E., 2006, *Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence*. Elsevier, San Diego, USA

Nowak, R. M., 1994, Walker's bats of the world: Baltimore, Maryland, The Johns Hopkins University Press, 287 p.

O'Shea, T. J., and Bogan, M. A., eds., 2003, Monitoring trends in bat populations of the United States and territories: problems and prospects: U.S. Geological Survey, Biological Resources Discipline, Information and Technology Report, USGS/BRD/ITR--2003-003, 274 p.

O'Shea, T. J., Bogan, M. A., and Ellison, L. E., 2003, Monitoring trends in bat populations of the United States and territories - status of the science and recommendations for the future: Wildlife Society Bulletin, vol. 31, p. 16-29.

Yamashina, M. Y., 1943, On a new subspecies of *Emballonura semicaudata* from the Mariana Islands: Bulletin of the Biogeographical Society of Japan, vol. 13, p. 99-100. (In Japanese, cited by Lemke 1986).

U.S. Fish and Wildlife Service, 2007, 50 CFR Part 17; Endangered and Threatened Wildlife and Plants; Review of Native Species That Are Candidates for Listing as Endangered or Threatened; Annual Notice of Findings on Resubmitted Petitions; Annual Description of Progress on Listing Actions; Proposed Rule. Federal Register (December 6, 2007), vol. 72 (no. 234), p. 69033-69106.

U.S. Fish and Wildlife Service, 2008, 50 CFR Part 17; Endangered and Threatened Wildlife and Plants; Review of Native Species That Are Candidates for Listing as Endangered or Threatened; Annual Notice of Findings on Resubmitted Petitions; Annual Description of Progress on Listing Actions; Proposed Rule. Federal Register (December 10, 2008), vol. 73 (no. 238), p. 75175-75244.

Section III. Current and Past Population Status and Use of Caves by Pacific Sheath-Tailed Bats (*Emballonura semicaudata rotensis*) on Aguiguan, Commonwealth of the Northern Mariana Islands

Gary J. Wiles, David J. Worthington, Jacob A. Esselstyn, Thomas J. O'Shea, and Ernest W. Valdez

ABSTRACT

The Mariana Islands subspecies of the Pacific sheath-tailed bat (*Emballonura semicaudata rotensis*) declined greatly in abundance and distribution during the 20th century. The island of Aguiguan now supports the only persisting population. We examined the abundance, roosting behavior, and distribution of this population by searching caves and hollow trees for roosting bats during surveys in 1995, 2003, and 2008. The only roosts of bats we found were in caves. Counts at caves suggest a substantial increase in abundance during the course of the study, with 98 bats recorded at five of 85 caves in 1995, 333–348 bats recorded at six of 57 caves in 2003 (including the discovery of one large colony previously unknown), and 359-466 bats recorded at five of 41 caves in 2008. Bats appeared to prefer roosting in larger caves and displayed significant fidelity toward five of the seven caves found occupied during the study. One cave consistently held the largest colony, with a range of 308–382 bats (mean [± SD] = 333 ± 33.6, n = 4) counted at emergence in 2008. Other caves served as roosts for 1–64 individuals. Most departures from roosts began 3.2 ± 8.7 min before sunset and ended 7.1 ± 8.1 min after sunset. We found no evidence of hollow tree trunks being used as roosts. As of 2008, the population of *E. s. rotensis* on Aguiguan probably numbered around 450–600 bats. Related research shows that the population relies heavily on native forest, regeneration of which is severely limited by feral goats (*Capra hircus*). Eradication of these goats combined with a reforestation program could increase and enhance foraging habitat of bats. Existing evidence supports the current designation of *E. s. rotensis* by the U.S. Fish and Wildlife Service as a candidate for listing as an endangered species under the U.S. Endangered Species Act.

INTRODUCTION

The Pacific sheath-tailed bat (*Emballonura semicaudata*) is distributed across much of Oceania, being found in the Mariana and Caroline Islands, Samoa, Tonga, Fiji, and Vanuatu (Flannery 1995, Koopman 1997, Helgen and Flannery 2002). These bats appear to be abundant at some locations, especially in the Caroline Islands (Bruner and Pratt 1979, Wiles and others 1997; G. Wiles, pers. observ.), but populations have inexplicably declined on many other islands (Lemke 1986, Grant and others 1994, Flannery 1995, Hutson and others 2001, Tarburton 2002, Palmeirim and others 2007). In the Mariana Islandss, where the endemic subspecies *E. s. rotensis* occurs (Koopman 1997), populations of sheath-tailed bats on Guam, Rota, and Saipan disappeared between the late 1940s and 1970s (Lemke 1986, Wiles and others 1995). The species occurred in recent prehistoric times on Tinian (Steadman 1999), but there are no historic records for this island. Lemke (1986) reported possible occurrences on Anatahan and Maug, but populations have not been confirmed on these islands, despite a number of visits by biologists to both islands since 1983. Although there are no previously published records of *E. s. rotensis* from Saipan, P. Krutzsch (in litt.) made several sightings of a few small insectivorous bats on this island in about 1945. These are considered to represent *E. s. rotensis* because no other microchiropteran bats occur in the archipelago.

The only known remaining population of this subspecies of Pacific sheath-tailed bat is on the island of Aguiguan (also known as Aguijan, Agiguan, and Goat Island). Biologists first recorded *E. s. rotensis* on Aguiguan in 1984 and 1985, when “three or four” bats were found in each of two caves (Lemke 1986). Subsequent observations from 1987 to 1992 documented up to 13 bats at one of the caves, but none was found at any of the few other caves examined, suggesting that the overall population was small (Rice and Taisacan 1993). The bats on Aguiguan were the only ones found in a search of 12 islands or island groups in the Northern Mariana Islands in 1983-1985 (Lemke 1986). Our main objective during this study was to conduct a more complete assessment of the Pacific sheath-tailed bat population on Aguiguan, and to synthesize all available past information pertinent to the status of this population. We developed a descriptive inventory and catalog of all caves that were searched for evidence of roosting sheath-tailed bats, including results based on field work in 2008 as well as unpublished data from our visits in 1995, 2002, and 2003. We also provide results of counts of numbers of bats using the caves that were determined to be occupied by bats. We interpret our results in relation to past findings of others as reported in the literature, explore possible causes for declines, and discuss possible measures that could enhance conservation of this population. Additional biological findings from field studies in 2008 are also provided in other sections of this administrative report.

STUDY AREA

Aguiguan (14°51'N, 145°33'E) is located in the southern Mariana Islands in western Micronesia and is administered by the U.S. Commonwealth of the Northern Mariana Islands (CNMI). The island is small (7.2 km^2) and comprised entirely of raised limestone karst, making it geologically similar to the neighboring islands of Tinian, Saipan, Rota, and much of Guam (Butler 1992, Stafford and others 2004, Jenson and others 2006). A large central plateau dominates the terrain and is bordered by a series of narrow terraces falling to the ocean. Coastal escarpments 10–40 m tall surround most of the shoreline. Maximum elevation is 166 m. Morphology of the island’s caves is discussed in Stafford (2003) and Stafford and others (2004). Aguiguan’s climate is tropical, with mean daily temperatures ranging from 24 to 32°C. Annual rainfall probably averages somewhat less than 2,000 mm, which is the approximate mean for Saipan 32 km to the north (Butler 1992, Lander 2004). Most rain occurs from July through November.

Aguiguan was mostly covered with native limestone forest until 1936 or 1937, when Japanese colonists began clearing large sections of the main plateau and larger terraces for sugar cane cultivation (Butler 1992). Former crop fields occupy 43% of the island and are now largely revegetated by weedy thickets of introduced plants, primarily *Lantana camara*, *Chromolaena odorata*, *Mikania scandens*, *Tridax* *procumbens*, and several grasses (Engbring and others 1986, Butler 1992, Rice 1993*a*). Limestone forest remains on about 45% of Aguiguan and grows on smaller terraces and steeper slopes (Esselstyn and others 2004, see Section IV of this report). Common tree species include *Guamia mariannae*, *Cynometra ramiflora*, *Pisonia grandis*, *Ochrosia mariannensis*, *Aglaia mariannensis*, *Ficus prolixa*, *Cerbera dilatata*, *Premna obtusifolia*, *Drypetes dolichocarpa*, *Erythrina variegata*, and *Psychotria mariana* (Chandran and others 1992; G. J. Wiles, pers. obs.), and canopy height is 7-15 m tall. Goats (*Capra hircus*) were introduced to the island in the mid-1800s (Butler 1992). Decades of overbrowsing by sizable feral goat populations have created an open forest understory dominated by two unpalatable species, *C. ramiflora* and *G. mariannae*, with little ground cover present. Goats have undoubtedly altered the species composition of the forest. Groves of secondary forest comprised of the introduced trees *Acacia confusa*, *Leucaena leucocephala*, *Triphasia trifolia*, and *Casuarina equisetifolia* occur at a number of disturbed sites. This habitat covers about 10% of the island and frequently contains some native trees (e.g., *O. mariannensis*, *G. mariannae*, and *Melanolepis multiglandulosa*) (Esselstyn and others 2004). Grassy and shrubby coastal strand occupies the remainder of the island. A control program greatly reduced goat numbers in 1989–1990 (Rice 1991, 1993*a*), but failed to eradicate them. Goats remained uncommon in 1995, but were again abundant from 2002 through 2008. The island has been uninhabited by people since the end of World War II, but is regularly visited by goat and coconut crab (*Birgus latro*) hunters from Tinian.

MATERIALS AND METHODS

Pacific sheath-tailed bats were surveyed during four trips to Aguiguan on 21–27 March 1995, 30 May–7 June 1995, 9–19 September 2003, and 19 June–15 July 2008. A few additional observations were made from 15–20 March 2002. Our study emphasized finding bats at roosting sites, thus we visited most caves previously known based on efforts of past researchers (Lemke 1986, Rice and Taisacan 1993), and also made extensive searches throughout the island for additional caves, rock crevices, and hollow trees that might serve as roosts for these bats. However, searches still remain incomplete because there are undoubtedly caves located in places that are inaccessible without technical climbing skills. We also did not find three small caves as reported by Butler (1992) and 13 caves (10 small, 3 possibly medium-sized) as reported by Stafford (2003), although Stafford (personal communication) reported no sightings of bats in these caves.

All accessible caves were entered and examined for bats or evidence of their occupation, including echolocation calls and guano. Colonies of Mariana swiftlets (*Aerodramus bartschi*), a cave-dwelling aerial insectivore, resided in at least nine of the island's caves. The presence of swiftlet guano often complicated our efforts because it can be difficult to distinguish from bat guano after it ages. However, we attempted to look for the distinctively shaped pellets of recent guano produced by insectivorous bats on walls and floors of caves in places apart from swiftlet nesting areas. Direct counts of bats roosting inside caves during the day were made whenever possible. In 2008, these were aided at some caves by the use of an infrared night vision device (model ATN NVM-14-3A, American Technologies Network Corporation, San Francisco, California) and an infrared illuminator. The interior dimensions of many caves were measured with a hip chain or visually estimated. Most of the entrances to larger caves were also measured. Interior size of caves was categorized as small, medium, or large. Small caves were generally $\leq$ 15 m long and 50 m^2 in floor area. Most consisted of low rock overhangs, narrow vertical crevices, or various cavities that were usually located at the bases of cliffs or underneath large boulders. Medium-sized caves generally had 50–100 m^2 of floor space and often had wider rooms than small caves. Large caves were usually >100 m^2 in floor size, with ceiling heights usually reaching 5–30 m. We also visually inspected hollow trees during the 1995 surveys to look for roosting sheath-tailed bats and guano deposits.

Evening emergence counts (Kunz 2003) of sheath-tailed bats were conducted at a number of caves that were potentially suitable for bats. Observers positioned themselves near the cave's opening, either inside or outside, to obtain the best possible vantage point for counting exiting bats. Observers remained quiet and motionless to avoid disturbing bats, and counts ended at total darkness (these bats emerge at early dusk, see below) or at least 15 minutes after the last bat emerged in cases where bats were present. Total numbers of bats exiting a cave were determined by subtracting the number of individuals entering from those departing. Observers used ultrasonic detectors (in 2003, model D-100, Pettersson Elektronik AB, Uppsala, Sweden; in 2008, AnaBat SD1 CF, Titley Electronics, Ballina, New South Wales, Australia) to assist in the detection of emerging bats at some sites. In 2003 and 2008 we used a mist net to capture a small number of bats flying inside one cave (Guano Cave, described below) on two days when counts at emergence were not conducted. In 2008 no counts were made after this potential disturbance took place.

RESULTS

Searches of Caves and Counts of Bats

We inventoried and cataloged 114 caves during the study, including sites recorded by Butler (1992) and Stafford (2003) but not visited by us (Appendix III-A). These represented 18 large, 9 medium, 74 small, and 13 undefined caves. We visited 85 caves in 1995; 57 caves in 2003, including nine not found in 1995; and 41 caves in 2008, including four not found in 1995 or 2003. Caves were distributed throughout much of Aguiguan, almost always in association with cliffs or fault lines. The two largest sites (Krisidu and Dangkolo Caves) featured single main rooms that were $\geq$ 50 m in length, were 15–20 m wide, and had ceiling heights of 15–20 m.

During surveys in 1995, we inspected 78 caves and conducted emergence counts at 10 caves, including two that could not be entered. Ninety-eight bats were recorded at five of the caves (Table III-1). The other five caves had no bats. Guano Cave was the only cave with bats visited in 1995 that was previously known to support bats (Table III-1). The four caves first documented in 1995 as having roosting bats were Cliff Cave, Pillar Cave, East Black Noddy Cave, and Crevice Cave. The largest colony numbered 69 bats at East Black Noddy Cave, with aggregations of 2–17 animals recorded at the other sites (Table III-1). In 2003, we inspected 52 caves and made emergence counts at eight caves, including three that were not entered. A total of 333–348 bats was counted at six caves, with bats present at the same five sites that were occupied in 1995 as well as at one newly discovered site (Fault Line Cave 1) that had a single bat (Table III-1). East Black Noddy Cave again held the largest colony, with an emergence count tallying 296 bats on 18 September. Other caves held up to 35 bats (Table III-1). Bat numbers were also larger at Guano and Pillar Caves in 2003 than in 1995.

In 2008, we visited 41 caves, inspected 34 caves internally, and made emergence counts at 18 caves, including seven that were not entered. Using minimum and maximum counts at occupied caves, a range of 359-466 bats was counted at five caves, with bats present at four of the six sites used in 1995 or 2003 and one new site, New Cave 1 (Tables III-1 and III-2). East Black Noddy Cave continued to hold the largest colony, with four emergence counts ranging from 308-382 bats (mean [± SD] = 333 ± 33.6). Internal counts at Guano Cave on six dates using the night vision device varied from 43-64 bats (mean = 55 ± 7.0). Other occupied caves held 2-12 bats. Compared to 2003, counts in 2008 were higher at East Black Noddy, Guano, and Cliff Caves, about the same at Crevice Cave, and declined to zero at Pillar Cave and Fault Line Cave 1 (Table III-1). All occupied caves were used throughout the survey period except New Cave 1, which held at least five bats on 4 July 2008, but none during visits on three other occasions in 2008 (Table III-Table III-III-2). Ten additional caves where multiple surveys were made showed no daytime use by bats (Table III-Table III-III-3).

From our observations it appears that Pacific sheath-tailed bats on Aguiguan prefer larger caves as diurnal roosts (Table III-4). Using our size criteria for caves, bats have been recorded in six large caves, one medium-sized cave, and one of unknown size. Variation in main entrance size of occupied caves ranged from 4 m wide by 25 m tall at Crevice Cave to about 1 m wide by 0.5 m tall at the upper entrance of East Black Noddy Cave (Table III-4). Swiftlets were present in seven of the eight caves inhabited by bats. We examined the trunk cavities of 22 hollow trees (21 *Pisonia grandis*, 1 *Psychotria mariana*), but found no evidence of occupation by bats. *Pisonia* hollows were typically 1–4 m tall and 20-45 cm in diameter, with their openings usually located near the base of the trunk.

Variability in Counts of Bats at Caves

Counts indicate that minor movement of bats among caves might take place, in some cases perhaps daily. We have no evidence that such movements will add a great deal of variability to the counts. Bats were always present at the two largest colonies, whereas caves with high variability in the presence or absence of bats during different visits all held small numbers of bats at any one time. Landing Cave was visited 11 times between 1984 and 2008, with bats seen on only two visits, both times in small numbers (≤ 4; Table III-1). Seasonal changes in attendance at roosts were not indicated by the results at Landing Cave: bats were present on 22 June in 1984 and 23 May 1992, whereas negative findings were recorded during June on three other visits in different years, as well as in months of January, March, and September (Table III-1). Similarly, Pillar Cave had no bats on five counts in five different years from 1985-1995 during the months of January, March, and June. However, from 2-10 bats were counted at Pillar Cave on three dates in March, June, and September in 1995-2003; none used Pillar Cave during the day on two dates in June and July 2008, but 16-21 bats entered the cave after dusk from elsewhere. Day-to-day differences were also noted within years and within field trips: Fault Line Cave 1 had one bat during the day in 2003, but none was present during the day on seven dates the cave was visited in 2008; five bats were observed at New Cave 1 on 4 July, but none on 25 June, 5 July, or 10 July in 2008; Cliff Cave had no bats in March 1995 but seven in June, and Pillar Cave had none in March 1995 but two in June (Tables III-1 and III-2).

Three roosts consistently had bats on each visit from 1995-2008: Guano Cave, East Black Noddy Cave complex, and Crevice Cave. The Guano and East Black Noddy caves have the largest colonies known, whereas Crevice Cave had only 2-3 adult bats on each count. Counts made during the day inside Guano Cave using a night vision device on six dates between 20 June and 7 July in 2008 (Table III-2) were different on each date, ranging from 43 to 64 bats (coefficient of variation 12.8 %). Counts made during flyouts at dusk at East Black Noddy Cave on four dates also varied similarly (coefficient of variation 10.1 %), and ranged between 308 and 382 bats. It is likely that some of this variation is due to movements of bats among different roosts, as is more clearly seen in the caves that vary from none to small numbers of bats. However, counts are also subject to a number of other sources of variability, including observer variability, environmental effects, and bias from unknown sources. Even using the night vision device with supplemental infrared illumination at Guano Cave, a single observer recording three separate tallies on each visit on five dates in 2008 had variability in counts within each day. On three of the five days the three separate counts only varied by a single bat each day. On two dates however, the three counts varied by a range of seven bats on one day and by 11 bats on the other. The range was primarily attributable to variability in the ability to discern large pups that roosted next to their mothers.

Observations of Sheath-Tailed Bat Behavior at Caves

Roosting behavior of bats was observed only at Guano and Crevice Caves. At Guano Cave in 1995, all bats roosted in one area of the cave spaced apart about 7–20 cm on the open dome-like ceiling in a small chute at the rear of the cave's dark main room. The roosting area was the ighest point in the cave. This same dome also was used in 2003 and 2008. In 2008 the night vision equipment allowed more detailed observation. The bats roosted singly (or a female next to young), were spaced about 5-30 cm apart on the ceiling and upper walls of the dome, appeared to have most of their ventral surfaces appressed to the rock surface with heads facing downwards, and were dispersed in a pattern intermediate between a regular and random spacing. This cave was also occupied by about 250 or more cave swiftlets. Most swiftlets did not roost in the dome occupied by the bats, but in adjoining areas of the large main room that were nearer the cave entrance. The few swiftlets that also roosted in the chamber were lower than the bats and against the walls of the dome at Crevice Cave in 1995, three bats roosted about 30 cm apart while roosting prone against the vertical wall of a side chamber. Although the room was in the darkest portion of the cave, the animals remained in dim twilight. This location was also occupied in 2003 and 2008. Because emergence and roost counts were similar at both caves, we are confident that few if any uncounted bats were hidden in cracks or crevices (we found no evidence that Pacific sheath-tailed bats at Aguiguan roost in narrow crevices or cracks in rock, and this is generally consistent with the literature for the species elsewhere). Bats were not detected during inspections of the main room at East Black Noddy Cave in 1995 and by K. Stafford (pers. comm.) in 2003, and therefore presumably roosted in an unexamined 7-m-long passage leading in from the upper entrance. We did not observe roosting bats at Pillar Cave, suggesting that roosting occurred in an inaccessible area. We were unable to enter Cliff Cave, and the bats at Fault Line Cave 1 and New Cave 1 were disturbed before they were observed.

Overall, total duration of sheath-tailed bat emergences from roosting caves averaged 18.3 ± 11.7 (SD) min, with most exits occurring during a mean period of 10.4 ± 5.6 min (Table III-5). On average, most departures began 3.2 ± 8.7 min before sunset and ended 7.1 ± 8.1 min after sunset. There was little or no overlap of activity between sheath-tailed bats and swiftlets during most (6 of 8) counts, with the period of most bat exits ending an average of 3.6 ± 6.6 min before the period of most swiftlet entries (Table III-5). However, scheduling of the emergence periods varied with colony size, with larger colonies emerging over longer periods, beginning earlier in relation to sunset, and ending closer to the time when swiftlets began entering the cave to roost for the night (Table III-6). Bats sometimes emitted audible high-pitched calls prior to emerging, presumably as they flew about inside the cave. Most individuals exited their caves by flying straight from the entrance, but some circled briefly outside before leaving the vicinity. At East Black Noddy and Guano Caves, a few emerging bats made one or several circling forays 5–15 m outside the caves' entrances, and then re-entered the cave or departed. Bats at smaller colonies exited singly, but at East Black Noddy Cave, some departed in groups of two to four. On 17 September 2003, a light rain shower several minutes after all bats had departed East Black Noddy Cave caused 50–75 bats to return immediately to the cave. This suggested that some bats linger in the general vicinity of the cave after emerging. Inspections of Guano Cave after counting ended in 2003 and 2008 showed that no bats remained inside immediately after the evening emergence was completed. We captured six bats inside Guano Cave in 2003, all of which were females.

In 2008, post-emergence use of caves as night roosts was detected at three caves, none of which were found to be used as day roosts during field work this year. We recorded 13 bats entering Pillar Cave from 18:47 to 19:02 h on 21 June, with chattering vocalizations heard from inside the cave several times afterwards, indicating that roosting had occurred. At least one of the calls came from the cave's outer room. Inspection of the cave with night vision equipment from 20:00 to 20:15 h found no bats present. On 7 July, during our only other evening count at this site, 21 bats entered from 18:41 to 18:56 h and appeared to remain inside. On 24 June we observed a single bat circling repeatedly inside the main entryway of Cave 63 (a small cave) at 1937 hr. On 11 July at 1847 we observed similar behavior by a bat that flew into the main opening of Fault Line Cave 1 and made audible vocalizations (short "chirps") as it remained inside for a few minutes. It then exited through a small opening at the ceiling of the cave after one of us entered at the main opening.

One other notable observation was that of an early evening (18:41-19:20 h) passage of at least 43-47 sheath-tailed bats flying singly past the vicinity of Caves 66-68, 94, 95, and 101 on several nights (see also Section VI of this administrative report). All of the bats followed a similar route through the forest understory, coming uphill from undetermined source to the west and continuing to the south or east. None came from any of the caves in the immediate area. Use of bat detectors revealed that few if any of the bats remained in the area after 19:20 to at least 22:00.

DISCUSSION

Pacific Sheath-Tailed Bats on Aguiguan

Our surveys in 1995, 2003, and 2008 represent the only attempts to assess the status of the Pacific sheath-tailed bat population on Aguiguan since 1984 (Lemke 1986). Over this period of time counts increased. We counted totals of 98 bats in 1995, 333–348 bats in 2003, and 359–466 bats in 2008. Bats occurred in only seven of the 95 caves examined (i.e., those entered or surveyed using emergence counts), even though many unoccupied sites appeared suitable as roosts. Despite good coverage of the island's inland cave system during the study, a few undocumented caves occupied by colonies of unknown size may be present. By comparison, most of the coastal cliffs ringing the island could not be surveyed for caves because they are difficult and dangerous to access. Several large caves are known in these escarpments, thus further assessment for their use as roosts by bats is needed, especially because *E. semicaudata* inhabits sea caves elsewhere in its range (Grant and others 1994). With the exception of one area, acoustic surveys conducted across the island in 2003 and 2008 did not detect concentrations of bat activity away from areas with known colonies (Esselstyn and others 2004, Section IV of this administrative report). Substantial early evening bat activity was noted in 2008 and to a lesser extent in 2003 at an acoustic station near the island's northeast coast, suggesting the presence of an undocumented colony in that general area. However, Pacific sheath-tailed bats are known to commute distances exceeding 5 km to reach foraging sites in Palau (Wiles and others 1997), thus it may be possible that the bats at this station originated from East Black Noddy Cave, which is located 1.7 km to the west. Based on the likelihood that small numbers of additional colonies may remain undetected, it is possible that the total current sheath-tailed bat population on Aguiguan numbers more than our maximum count of 466.

Our surveys documented larger numbers of sheath-tailed bats in 1995 than those counted by previous observers (Lemke 1986, Rice and Taisacan 1993), but much of this can be attributed to improved survey coverage. However, data indicate that marked population growth had occurred since 1995. Colony size grew at three of five caves from 1995 to 2003, with numbers expanding more than four fold at the largest colony. Additional but more modest growth continued from 2003 to 2008. Bat numbers at Guano Cave are particularly illustrative and have increased from four in 1985 (Lemke 1986) to about 55 in 2008. Perhaps the population of sheath-tailed bats on Aguiguan has been increasing since the mid-1980s after a past bottleneck. Typhoons can be a major source of bat mortality on other Pacific Islands (see below), and at least two major typhoons struck the island during this time (Lander 2004; C. Kessler, personal communication). However, the increase in numbers suggests that the sheath-tailed bat population on Aguiguan was not severely impacted by these typhoons. Although our results show some variability in counts that likely indicate movements of small numbers of bats among roosts, and we also note potential biases in variation in counts from other sources, such variation seems unlikely to explain the large increases in colony sizes we have documented at some caves. Instead we think our results reflect true growth in the populations using the major caves. Surveys at East Black Noddy (upper entrance) and Guano Caves included counts made by the same person (GJW) in all three surveys and were in close agreement with simultaneous counts made by the rest of us, thus major inter-observer variability should not be an important source of error in our results.

An increasing trend on Aguiguan since 1995 is especially noteworthy given the many reported declines elsewhere in the species' range in recent decades (Lemke 1986, Grant and others 1994, Flannery 1995, Hutson and others 2001, Tarburton 2002, Palmeirim and others 2007). Is it theoretically plausible that the changes in counts we observed are due to intrinsic population growth? A basic underlying model often used in analyses of population trend data is $N_t = N_0\lambda^t$, where N_0 is the initial population size, N_t is population size at time t, and λ is the population growth rate (*e.g.* Eberhardt 2002). An estimate of λ can be calculated using the equation $\ln (N_t) = \ln (N_0) + t \ln(\lambda)$. Although this model is based on a number of assumptions that are unknown for Pacific sheath-tailed bats and is usually applied for multiple time series, we use it in a simplified way to determine if it is plausible for the observed increases to be due to intrinsic population growth. The model yields an estimated λ = 1.13 at Black Noddy Cave and λ = 1.09 at Guano Cave between 1995 and 2008. There have been few empirically derived estimates for annual population growth rates in insectivorous bats, but those that have been calculated for growing populations of other species of bats using more sophisticated models and accurate vital parameter data are consistent with the calculations of λ for these two largest colonies of Pacific sheath-tailed bats at Aguiguan. These estimates for other species of insectivorous bats with seasonal breeding and litter sizes of one range from λ = 1.03-1.22 (Frick and others 2005, Pryde and others 2005). We do not suggest that the specific growth rates we calculated above should be considered accurate for this population, but only that they crudely demonstrate that it is indeed plausible that the magnitudes of the observed changes in counts may be due to population growth. The future trend of the population of sheath-tailed bats on Aguiguan is impossible to predict, but numbers of bats counted in 2003 and 2008 resemble those of the ecologically similar Mariana swiftlet, whose surveyed numbers regularly exceed 400 birds (Cruz and others 2008; G. J. Wiles, unpubl. data). This current similarity in abundance may indicate that bat numbers are reaching their upper limit on the island.

The colony of up to 382 bats at East Black Noddy Cave is by far the largest ever recorded for *E. s. rotensis*. Counts at other roosts on Aguiguan ranged from 1-64 individuals. Our capture of six females and no males at Guano Cave in 2003 and five females and no males at this cave in 2008 (see Section VI of this administrative report) suggests that this colony may have been comprised mostly of females. Aggregations of fewer than 25 bats and segregation of the sexes are common roosting traits among bats in the genus *Emballonura* (Flannery 1995, Bonaccorso 1998, Nowak 1999). Nevertheless, larger colonies with up to several hundred bats may have once been common in the Mariana Islands, as has been found with *E. s. palauensis* in Palau (Wiles and Conry 1990, Wiles and others 1997). Roosts of this size are also known for *E. s. sulcata* in Chuuk (Bruner and Pratt 1979). Amerson and others (1982) documented a cave in American Samoa with perhaps as many as 10,000 *E. s. semicaudata*.

Aguiguan's bats appear to roost exclusively in caves, which resembles the behavior of *E. s. semicaudata* and *E. s. palauensis* (Grant and others 1994, Wiles and others 1997). By contrast, some *E. s. sulcata* colonies select tree cavities as day roosts on Pohnpei (D. W. Buden, pers. comm.). Most other members of the genus reside in caves (e.g. Goodman and others 2006, Bonaccorso 1998), especially in or near the twilight zones of smaller caves, but several species also roost in foliage, hollow logs, and human-built structures (Flannery 1995, Nowak 1999, Kingston and others 2006). Results of surveys on Aguiguan (Table III-1) indicate that sheath-tailed bats may exhibit strong fidelity to some of these caves. Guano Cave, for example, appears to have been occupied since at least 1985, while three other caves (Crevice, East Black Noddy, and Cliff caves) were occupied during each of our visits in 1995, 2003, and 2008. Other sites, such as Pillar, Landing, Fault Line 1, and New 1 Caves, may be inhabited temporarily. We caution that we cannot vouch for the accuracy of counts made prior to May 1995 when none of us was present, especially those at Cliff and Pillar Caves. Surveys in 2008 were the first to attempt replicate counts at the same roosts on different days. Results from East Black Noddy and Guano Caves indicate that numbers of bats roosting at these sites will vary over periods of several days. Emergence counts like those made at East Black Noddy Cave can be susceptible to observer error, which may account for some of the variation recorded. However, the direct counts of roosting bats made at Guano Cave with night vision equipment should be more accurate. We recommend that future surveys at these two important roosts routinely incorporate counts on multiple dates to better assess their variation. The lack of measures of variation in counts of emerging bats is a common problem in studies of most species of bats throughout the U.S. and territories, but is critical for assessing trends in abundance (Ellison and others 2003).

This study is the first to document the use of caves as night roosts by *E. semicaudata*. Night-roosting behavior has many potential functions in bats (Ormsbee and others 2007). Each of the three caves observed to be used at night in this study was occupied at dusk or shortly thereafter, suggesting that food digestion was not a goal of the animals involved. Based on the presence of multiple animals and vocalizations, Pillar Cave may have served as a site for social interaction, such as mating or information transfer.

Decline of *E. s. rotensis* in the Marianas Islands

Causes for the overall decline of *E. s. rotensis* in the southern Marianas are unclear. Extirpations of sheath-tailed bats on Rota, Saipan, and Guam roughly coincided with declines or population losses in Mariana swiftlets, suggesting that both species experienced common threats, perhaps because of their similar roosting and feeding habits. Swiftlets no longer occur on Rota, but persist in low to moderate numbers on Saipan and Guam (Engbring and others 1986; Cruz and others 2008; G. Wiles, unpubl. data).

For example, human occupation and warfare during World War II heavily impacted many caves in the Marianas, when Japanese troops used caves as defensive fortifications. Grenades and flame-throwers were commonly used by the U.S. military to eliminate Japanese soldiers using these fortified caves. Such disturbance presumably harmed numerous bat and swiftlet colonies, but unless populations were completely eliminated should have been a temporary effect that would have subsided after the war.

Since then, visitation of caves by hunters, vandals, hikers, and guano miners has continued (U.S. Fish and Wildlife Service 1992), but has probably not been extensive enough at most sites since the 1980s to be problematic. On Aguiguan, several caves (including Guano Cave and Fault Line Cave 1) show evidence of extensive use by the Japanese before or during the war, and it is unlikely that bats occupied the caves at that time. However, there was no combat or destructive use of munitions in caves on Aguiguan (Butler 1992). Guano Cave has also been used for small-scale guano mining (U.S. Fish and Wildlife Service 1992). Many caves on other islands show similar signs of disturbance (U.S. Fish and Wildlife Service 1992; G. Wiles, pers. obs.). By contrast, Cliff Cave has probably never been entered by humans because of its high cliffside location. East Black Noddy Cave, which holds the largest documented number of bats on Aguiguan, has also probably received very little disturbance from people since the war because it is difficult to access. Entry of caves by introduced ungulates is also potentially disruptive, especially on Aguiguan, where feral goats habitually seek shelter in many caves.

Historical pesticide contamination may have posed significant problems for sheath-tailed bats in the Marianas, but this has not been thoroughly investigated (see Section VIII of this administrative report). Liberal use of compounds such as DDT and malathion is known to have occurred between the 1940s and 1970s (Baker 1946, Townes 1946, Drahos 1977, Jenkins 1983). Applications were most intense on Guam, Saipan, and Tinian (Townes 1946) because of their larger human populations and the presence of American military bases. Residues of the break-down product DDE have been found in swiftlet tissues and guano samples from Guam (Drahos 1977, Grue 1985), but tests by Grue (1985) yielded no evidence to support the hypothesis that poisoning by DDT or DDE had caused declines among Guam’s avifauna. Concentrations of DDT and DDE in swiftlet guano measured in 1981 were much less than those associated with avian mortality or reproductive failure, and an order of magnitude less than concentrations in bat guano that can be linked to mortality in insectivorous bats (Clark and others, 1982, 1995; Clark and Shore 2001). Concentrations of contaminants have never been measured in sheath-tailed bat guano or tissues. Additionally, the carbamate and organophosphate insecticides that were also used are not persistent in tissues or guano, and exposure of bats or swiftlets to these compounds has not been assessed. Some of these compounds were likely responsible for deaths of bats in the United States in the 1960s (Clark and Shore 2001). It is also possible that sheath-tailed bats and swiftlets could have been at risk if they or their insect prey base were more susceptible to pesticide contamination than other animals because of physiological differences in organochlorine accumulation or differential vulnerability at various stages in their life history (Clark and Shore 2001). However, DDT was used extensively in Palau in the 1940s (Baker 1946) and probably thereafter, but sheath-tailed bats remain abundant there (Wiles and others 1997). Development of DDT as an insecticide did not occur until the 1940s (Metcalf 1973). It is unknown if other pesticides were used by Japanese colonists on sugar plantations at Aguiguan prior to World War II. Aguiguan was neither populated nor used agriculturally after the war, and thus it is unlikely that significant amounts of pesticides were applied to the island during the period when they were in use elsewhere in the Marianas. A likely absence of the intensive use of pesticides on Aguiguan could have contributed to the persistence of sheath-tailed bats there. The apparently low numbers of bats on Aguiguan in the 1980s in comparison with 2008 (e.g. at Guano Cave) are enigmatic in relation to effects of contaminants. Samples of guano from Aguiguan were taken in 2008 (see Section VIII of this administrative report) and their analysis for contaminants might be instructive in this regard.

Because *E. s. rotensis* forages almost exclusively in forests (Esselstyn and others 2004; see Section IV of this administrative report), it seems likely that extensive deforestation in the southern Marianas has contributed to reduced populations of sheath-tailed bats. From the 1920s to early 1940s, Japanese colonists cleared from 75% to as much as 98% of Saipan, Tinian, and Rota (Bowers 1951) and about 43% of Aguiguan (Engbring and others 1986) for agriculture and other activities. Construction of major American military installations on Guam, Saipan, and Tinian during and after the war caused additional habitat loss, as did extensive civilian development on Saipan and Guam in subsequent decades. On Aguiguan, few of the fields cleared before the onset of World War II have returned to forest cover. This means that the sheath-tailed bat population is largely restricted to feeding within the remaining 4 km^2 of forested land available on the island. There was no use of munitions in caves on Aguiguan during World War II (Butler 1992), and the large number of unoccupied inland caves with suitable temperatures (see also Section VI) suggests that roost site availability does not limit the population of sheath-tailed bats on Aguiguan. Deforestation is likely the major current limiting factor, and is known to be a major cause of bat declines and losses elsewhere in the world (e.g., Brosset and others 1996, Lane and others 2006; Wiles and Brooke, in press).

We found no evidence of predation on sheath-tailed bats on Aguiguan. Monitor lizards (*Varanus indicus*), which were likely introduced to the Marianas after European contact (Pregill 1998), are abundant on Aguiguan and represent a potential predator of sheath-tailed bats. They are adept climbers and may be able to reach day-roosting bats in hollow tree trunks or smaller caves with low ceilings. Although such predation may currently be insignificant, it may have influenced the selection of the roost sites in larger caves now used by bats. Introduced rats and large geckos are common in some caves and have the potential to take young bats at roosts on occasion. Rats are unlikely to reach the high walls and ceilings used by bats in most caves, however, and adult bats are typically alert and will fly readily when threatened. Avian predation is probably limited to occasional owls in migration and the resident diurnal collared kingfisher (*Todirhamphus chloris*). Predation by the introduced brown tree snake (*Boiga irregularis*) has devastated native wildlife populations on Guam (Fritts and Rodda 1998) and currently prevents swiftlet recovery on the island (Wiles and others 2003). Brown tree snakes conceivably contributed to declines or caused losses in sheath-tailed bats in southern Guam during the 1950s and 1960s, but probably played no role in the bat's subsequent demise in northern Guam. Snakes did not invade northernmost Guam in significant numbers until the late 1970s (Savidge 1987, Wiles and others 2003), which was at least a decade after serious declines in *E. s. rotensis* numbers were noted there (Perez 1972) and 6-7 years after the last known sighting in 1972. Brown tree snakes do not occur on Aguiguan, but have the potential to be predators of sheath-tailed bats if they were to reach the island.

Grant and others (1994) identified a succession of severe typhoons as a possible contributing factor in the recent decline of *E. s. semicaudata* in Samoa. This cause is unlikely to be related to the overall decline throughout the Marianas. However, because of Aguiguan's small land area and bat population, it is conceivable that an unusually intense storm or series of storms could severely impact the species by precluding foraging during storms, damaging important foraging habitat by destroying vegetation that its prey depends upon, or flooding colonies in seaside caves. Direct mortality from high winds blowing into exposed caves is also possible, as seen on Guam, where Supertyphoon Pongsona killed at least 30 roosting swiftlets at a cave in 2002 (C. Clark, pers. comm.).

When viewed in conjunction with the extirpation of Pacific sheath-tailed bats on other Mariana Islands, this study suggests that *E. s. rotensis* is highly vulnerable to extinction, with probably no more than a few hundred of these bats restricted to one very small island. Preservation of this bat hinges on the maintenance of forested habitat and safe roosting sites. Eradication or strict management of goats is a key recommendation for conserving Aguiguan's forests. This will greatly improve regeneration of native tree species and insure the long-term stability of forests on the island. A reforestation program should be initiated to replant sizable areas of weedy fields with native trees. *Emballonura s. rotensis* and virtually all other native wildlife species would benefit from such treatment. Although not frequent, human visitation to Aguiguan occurs, primarily by hunters from Tinian. Interest in developing the island for ecotourism has also been expressed in the past (J. de Cruz, pers. comm.). If human visitation increases from increased hunting or ecotourism, bat colonies at some of the easily accessible and reasonably well known caves (e.g., Guano Cave) will be at risk from irresponsible visitation. Additionally, ongoing efforts to prevent the establishment of brown tree snakes in the CNMI are an obvious priority for protecting this bat population.

Expanded study of sheath-tailed bats on Aguiguan is needed to provide additional information on population size, vital parameters, basic ecology and natural history, genetics, and important limiting factors. A priority for future surveys should be improved assessment of use of coastal caves in areas that cannot be reached without technical climbing expertise, as well as additional efforts to re-survey areas with caves visited in the past. We recommend that additional paleontological work, similar to that conducted by Koopman and Steadman (1995) and Steadman (1999), be conducted to learn more about past use of caves by these bats on Aguiguan.

Despite its rarity, neither the Pacific sheath-tailed bat nor its habitat is afforded protection in the U.S. possessions where it presently occurs. The species is on the CNMI list of threatened or endangered species, but this law provides no protection to the bat or its habitat. Under U.S. federal law, the U.S. Fish and Wildlife Service has categorized the bat as a candidate species, meaning that sufficient information is available to consider listing it as threatened or endangered. However, lack of funding, its subspecific status, and other constraints have precluded proceeding with listing (U.S. Fish and Wildlife Service 2002, 2007). Greater protection could prohibit take of the bat and help enhance its habitat and conservation.

ACKNOWLEDGMENTS

Funding in 1995 and 2003 was provided by the U.S. Fish and Wildlife Service's Federal Aid to Wildlife Restoration Projects in the CNMI and Guam, and its Ecological Services office in Honolulu, Hawaii. Funding in 2008 was through an interagency agreement between the U.S. Fish and Wildlife Service and

U.S. Geological Survey as part of a larger joint effort by the Service and the U.S. Navy. Any use of trade, product, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the U.S. Government. We thank A. P. Marshall and C. C. Kessler for organizing the 1995 and 2008 surveys, respectively. For assistance with fieldwork and logistics, we are grateful to E. Masga, E. M. Taisacan, V. A. Camacho, P. M. Gorresen, R. S. Ulloa, D. Reyes, J. Omar, E. W. Campbell, A. P. Marshall, H. King, F. Muna, G. Camacho, M. Palacios, R. Camacho, D. Evangelista, J. Salas, T. Castro, J. Camacho, F. J. Bonaccorso, and D. J. Grout. Numerous other CNMI Division of Fish and Wildlife (DFW) staff have participated in trips to the island since 1983 and we acknowledge their contribution here. S. Igisomar of the DFW facilitated our work, which was carried out under authority of Scientific Research Permit 01048-08 issued by the Commonwealth of the Northern Mariana Islands. J. B. de Cruz and K. W. Stafford kindly supplied additional information. R. Crowe, M. Cunningham, and N. Kigure provided helicopter transport to the island. Helpful comments on the manuscript were provided by P. Cryan, M. Neubaum, and D. Neubaum. This paper is dedicated to the memory of David T. Aldan, former DFW biologist, whose diverse knowledge of the natural history of the Marianas benefited many.

References Cited

Amerson, A. B., Jr., Whistler, W. A. and Schwaner, T. D., 1982, Wildlife and wildlife habitat of American Samoa. Vol. II: Washington, D.C., U.S. Fish and Wildlife Service.

Baker, R. H., 1946, Some effects of the war on the wildlife of Micronesia: Transactions of the North American Wildlife Conference, vol. 11, p. 205-213.

Bonaccorso, F. J., 1998, Bats of Papua New Guinea: Washington, D.C.,Conservation International.

Bowers, N. M., 1951, Problems of resettlement on Saipan, Tinian, and Rota, Mariana Islands: Ann Arbor, Michigan, Ph.D. thesis, University of Michigan.

Brosset, A., Charles-Dominique, P., Cockie, A., Cosson, J.-F., and Masson, D., 1996, Bat communities and deforestation in French Guiana: Canadian Journal of Zoology, vol. 74, p. 1974-1982.

Bruner, P. L., and Pratt, H. D., 1979, Notes on the status and natural history of Micronesian bats: ‘Elepaio, vol. 40, p.1-4.

Butler, B. M., 1992, An archaeological survey of Aguiguan (Aguijan), Northern Mariana Islands: Saipan, Commonwealth of the Northern Mariana Islands, Division of Historic Preservation, Micronesian Archaeological Survey Report No. 29.

Chandran, R., Craig, R. J., Keys, Z., Sheu, C., and Dubrall, J., 1992, The structure and tree species composition of Aguiguan forests, *in* Craig, R. J., ed. The Aguiguan expedition: Saipan, Commonwealth of the Northern Mariana Islands, Proceedings Marianas Research Symposium, Vol. 1, Northern Marianas College.

Clark, D. R., Jr., and Shore, R. F., 2001, Chiroptera, *in* Shore, R. F., and Rattner, B. A., eds., Ecotoxicology of wild mammals: Chichester, West Sussex, United Kingdom, John Wiley & Sons, p. 159-214.

Clark, D. R., Jr., LaVal, R. K., and Tuttle, M. D., 1982, Estimating pesticide burdens of bats from guano analyses: Bulletin of Environmental Contamination and Toxicology, vol. 29, p. 214–220.

Clark, D. R.. Jr., Moreno-Valdez, A., and Mora, M. A., 1995, Organochlorine residues in bat guano from nine Mexican caves 1991: Ecotoxicology, vol. 4, p. 258–265.

Craig, R. J., and Chandran, R., 1992, Wildlife species recorded during the Aguiguan expedition: 20-25 May, 1992, *in* Craig, R. J., ed. The Aguiguan expedition: Saipan, Commonwealth of the Northern Mariana Islands, Proceedings Marianas Research Symposium, Vol. 1, Northern Marianas College, p. 1-7.

Cruz, J. B., Kremer, S. R., Martin, G., Williams, L. L., and Camacho, V. A., 2008, Relative abundance and distribution of Mariana swiftlets (Aves: Apodidae) in the Northern Mariana Islands: Pacific Science, vol. 62, p. 233-246.

Drahos, N. 1977. Status of the Vanikoro swiftlet on Guam (*Collocalia vanikorensis* Quoy and Gaimard): Mangilao, Guam, Division of Aquatic and Wildlife Resources.

Division of Fish and Wildlife, 2000, Summary of wildlife surveys, Aguiguan Island, March 31-April 8, 2000: Saipan, Commonwealth of the Northern Mariana Islands, Division of Fish and Wildlife.

Eberhardt, L.L., 2002, A paradigm for population analysis of long-lived vertebrates: Ecology, vol. 83, p. 2841-2854.

Ellison, L. E., O'Shea, T. J., Bogan, M. A., Everette, A. L., and Schneider, D. M., 2003, Existing data on colonies of bats in the United States: summary and analysis of the U.S. Geological Survey's bat population database, *in* O'Shea, T.J. , and Bogan, M. A., eds., Monitoring trends in bat populations of the United States and Territories: problems and prospects: Washington, D. C., U.S. Geological Survey Information and Technology Report USGS/BRD/ITR-20030-0003, p. 127-237.

Engbring, J., Ramsey, F. L., and Wildman, V. J., 1986, Micronesian forest bird survey, 1982: Saipan, Tinian, Agiguan, and Rota: Honolulu, Hawaii, U. S. Fish and Wildlife Service.

Esselstyn, J. A., Wiles, G. J., and Amar, A., 2004, Habitat use of the Pacific sheath-tailed bat (*Emballonura semicaudata*) on Aguiguan, Mariana Islands: Acta Chiropterologica, vol. 6, p. 303-308.

Flannery, T., 1995, Mammals of the South-West Pacific & Moluccan Islands: Ithaca, New York, Cornell University Press.

Frick, W.F., Rainey, W.E., and Pierson, E. D., 2007, Potential effects of environmental contamination on Yuma myotis demography and population growth: Ecological Applications, vol. 17, p. 1213-1222.

Fritts, T. H., and G. H. Rodda, G. H., 1998, The role of introduced species in the degradation of island ecosystems: a case history of Guam: Annual Review of Ecology and Systematics, vol. 29, p. 113-140.

Goodman, S. M., Cardiff, S. G., Ranivo, J., Russell, A. L., and Yoder, A.D., 2006, A New Species of *Emballonura* (Chiroptera: Emballonuridae) from the Dry Regions of Madagascar: American Museum Novitates, no. 3538, p. 1-24.

Grant, G. S., Banack, S. A., and Trail, P., 1994, Decline of the sheath-tailed bat *Emballonura semicaudata* (Chiroptera: Emballonuridae) on American Samoa: Micronesica, vol. 27, p. 133-137.

Grue, C. E., 1985, Pesticides and the decline of Guam's native birds: Nature, vol. 316, p. 301.

Helgen, K. M., and Flannery, T. F., 2002, Distribution of the endangered Pacific sheath-tailed bat *Emballonura semicaudata*: Australian Mammalogy, vol. 24, 209-212.

Hutson, A. M., Mickleburgh, S. P., and Racey, P. A., 2001, Microchiropteran bats: Global status survey and conservation plan: Gland, Switzerland and Cambridge, UK, IUCN/SSC Chiropteran Specialist Group, 258 p.

Jenkins, J. M., 1983, The native forest birds of Guam: Ornithological Monographs, vol. 31, p. 1-61.

Jenson, J. W., Keel, T. M., Mylroie, J. R., Mylroie, J. E., Stafford, K. W., Taboroši, D., and Wexel, C., 2006, Karst of the Mariana Islands: the interaction of tectonics, glacio-eustasy, and freshwater/seawater mixing in island carbonates, *in* Harmon, R. S., and Wicks, C., eds., Perspectives on karst geomorphology, hydrology, and geochemistry, Geological Society of America Special Paper, vol. 404, p. 129-138.

Kingston, T., Lim, B. L., and Akbar, Z., 2006, Bats of Krau Wildlife Reserve: Bangi, Malaysia, Penerbit Universiti Kebangsaan Malaysia.

Koopman, K. F., 1997, The subspecies of *Emballonura semicaudata* (Chiroptera: Emballonuridae): Journal of Mammalogy, vol. 78, p. 358-360.

Koopman, K. F., and D. W. Steadman, D. W., 1995, Extinction and biogeography of bats on ʻEua, Kingdom of Tonga: American Museum Novitates, No. 3125, p.1-13.

Kunz, T.H., 2003, Censusing bats: challenges, solutions, and sampling biases, in O'Shea, T. J., and Bogan, M. A.,eds., Monitoring trends in bat populations of the United States and territories: problems and prospects: Washington, D.C., U.S. Geological Survey, Biological Resources Discipline, Information and Technology Report, USGS/BRD/ITR--2003-003, p. 9-19.

Lander, M. A., 2004, Rainfall climatology for Saipan: distribution, return-periods, El Nino, tropical cyclones, and long-term variations: Mangilao, Guam, Technical Report 103, Water and Environmental Research Institute, University of Guam.

Lane, D. J. W., Kingston, T., and Lee, B. P. Y.-H., 2006, Dramatic decline in bat species richness in Singapore, with implications for Southeast Asia: Biological Conservation, vol. 131, p. 584-593.

Lemke, T. O., 1986, Distribution and status of the sheath-tailed bat (*Emballonura semicaudata*) in the Mariana Islands: Journal of Mammalogy, vol. 67, p.743-746.

Metcalfe, R.L., 1973, A century of DDT: Journal of Agricultural and Food Chemistry, vol. 21, p. 511-519.

Nowak, R. M., 1999, Walker's mammals of the world. Vol. 1. 6th ed.: Baltimore, Maryland, Johns Hopkins University Press.

Ormsbee, P. C., Kiser, J. D., and Perlmeter, S. I., 2007, Importance of night roosts to the ecology of bats: *in* Lacki, M. J., Hayes, J. P., and Kurta, A. eds., Bats in forests: conservation and management, Baltimore, Maryland, Johns Hopkins University Press, p. 129-151

Palmeirim, J. M., Champion, A., Naikatini, A., Niukula, J., Tuiwawa, M., Fisher, M., Yabaki-Gounder, M., Thorsteinsdóttir, S., Qalovaki, S., and Dunn, T., 2007, Distribution, status and conservation of the bats of the Fiji Islands: Oryx, vol. 41, p. 509-519.

Perez, G. S. A., 1972, Observations of Guam bats: Micronesica, vol. 8, p. 141-149.

Pregill, G., 1998, Squamate reptiles from prehistoric sites in the Mariana Islands: Copeia, vol. 1998, p. 64-75.

Pryde, M. A., O'Donnell, C. F. J., and Barker, R. J., 2005, Factors influencing survival and long-term population viability of New Zealand long-tailed bats (*Chalinolobus tuberculatus*): implications for conservation: Biological Conservation, vol. 126, p. 175-185.

Reichel, J. D., and Glass, P. O., 1988, Land and freshwater bird investigations, *in* Five-year progress report, fiscal years 1983-1987, Pittman-Robertson Federal Aid in Wildlife Restoration Program: Saipan, Commonwealth of the Northern Mariana Islands, Division of Fish and Wildlife, p. 106-130

Rice, C. G., 1991. Goat removal from Aguijan Island: lessons for future efforts: Transactions of the Western Section of the Wildlife Society, vol. 27, p. 42-46.

Rice, C. G., 1993*a*, Feral ungulate research, *in* Five-year progress report, fiscal years 1988-1992, Pittman-Robertson Federal Aid in Wildlife Restoration Program: Saipan, Commonwealth of the Northern Mariana Islands, Division of Fish and Wildlife, p. 303-319.

Rice, C. G. 1993*b*. Island swiftlet research, *in* Five-year progress report, fiscal years 1988-1992, Pittman-Robertson Federal Aid in Wildlife Restoration Program: Saipan, Commonwealth of the Northern Mariana Islands, Division of Fish and Wildlife, p. 241-255.

Rice, C. G., and E. Taisacan, 1993, Marianas fruit bat surveys, *in* Five-year progress report, fiscal years 1988-1992, Pittman-Robertson Federal Aid in Wildlife Restoration Program: Saipan, Commonwealth of the Northern Mariana Islands, Division of Fish and Wildlife, p. 35-47.

Savidge, J. A., 1987, Extinction of an island forest avifauna by an introduced snake: Ecology, vol. 68, p. 660-668.

Stafford, K. W. 2003. Structural controls on megaporosity in eogenetic carbonate rocks: Tinian, CNMI. M.S. thesis, Mississippi State University, Mississippi State, Mississippi.

Stafford, K., Mylroie, J., Taboroši, D., and Jenson, J., 2004. Eogenetic karst development on a small, tectonically active, carbonate island: Aguijan, Mariana Islands: Cave and Karst Science (Transactions of the British Cave Research Association), vol. 31, p. 101-108.

Steadman, D. W., 1999, The prehistory of vertebrates, especially birds, on Tinian, Aguiguan, and Rota, Northern Mariana Islands: Micronesica, vol. 31, p. 319-345.

Tarburton, M. K., 2002. Demise of the Polynesian sheath-tailed bat *Emballonura semicaudata* in Samoa: Micronesica, vol. 34, p. 105-108.

Townes, H. K., 1946, Entomology. Part I. Non-agricultural plants. Economic Survey of Micronesia, Vol. 12: Honolulu, Hawaii, U.S. Commercial Company.

U.S. Fish and Wildlife Service, 1992, Recovery plan for the Mariana Islands population of the Vanikoro swiftlet, *Aerodramus vanikorensis bartschi*: Portland, Oregon, U. S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service, 2002, , 50 CFR Part 17; Endangered and threatened wildlife and plants; review of plant and animal species that are candidates or proposed for listing as endangered or threatened, annual notice of findings on recycled petitions, annual description of progress on listing actions; notice of review. Federal Register 67(114):40657-40679.

U.S. Fish and Wildlife Service, 2007, 50 CFR Part 17; Endangered and threatened wildlife and plants; review of native species that are candidates for listing as endangered or threatened; annual notice of findings on resubmitted petitions; annual description of progress on listing actions; proposed rule. Federal Register (December 6, 2007)72 (234): 69033-69106.

Wiles, G. J., and Brooke, A. P., in press, Conservation threats to bats in the tropical Pacific islands and insular Southeast Asia, *in* Fleming, T. H., and Racey, P., eds., Island bats: ecology, evolution, and conservation: Chicago, Illinois, University of Chicago Press.

Wiles, G. J., and Conry, P. J., 1990, Terrestrial vertebrates of the Ngerukewid Islands Wildlife Preserve, Palau Islands: Micronesica, vol. 23, p. 41-66.

Wiles, G. J., Aguon, C. F., Davis, G. W., and Grout, D. J., 1995, The status and distribution of endangered animals and plants in northern Guam: Micronesica, vol. 28, p. 31-49.

Wiles, G. J., Bart, J., Beck, R. E. Jr., and Aguon, C. F., 2003, Impacts of the brown tree snake: patterns of decline and species persistence in Guam's avifauna: Conservation Biology, vol. 17, p. 1350-1360.

Wiles, G. J., Engbring, J., and Otobed, D., 1997, Abundance, biology, and human exploitation of bats in the Palau Islands: Journal of Zoology (London), vol. 241, p. 203-227.

Table III-1. Numbers of Pacific sheath-tailed bats recorded at seven caves on Aguiguan, Mariana Islands, from 1984-2008, as determined by direct counts of day-roosting bats or evening emergence counts. Fields with dashes indicate that no counts were made.

	Cave									
					East Black Noddy					
Date	**Landing**	**Guano**	**Cliff**	**Pillar**	**Upper Entrance**	**Middle Entrance**	**Crevice**	**Fault Line Cave 1**	**New Cave 1**	**References[a]**
22 June 1984	3-4[b]	-	-	-	-	-	-	-	-	1
January 1985	0[b,c]	4[b]	0[c]	0[c]	-	-	-	-	-	1, 2
18 July 1985	-	2[b]	-	-	-	-	-	-	-	3
28 Feb-1 March 1987	0[c]	3[b]	-	0[c]	-	-	-	-	-	3, 4, 5
6-9 June 1988	0[c]	5[b]	-	0[c]	-	-	-	-	0[c]	3, 4, 5
21 September 1989	-	13[b]	-	-	-	-	-	-	-	3, 5
March 1992	0[b]	9[b]	0[c]	0[c]	-	-	-	-	-	3, 4
23 May 1992	2[c]	-	-	-	-	-	-	-	-	6
23, 26 March 1995	0[b]	15[b], 16[b]	0[c]	0[c]	64[c]	5[c]	-	-	-	7
31 May-6 June 1995	0[b,c]	17[b,c], 16[b]	7[c]	2[c]	-	-	3[b,c]	-	-	7
15-20 March 2002	0[b]	15[c]	-	5[c]	-	-	-	-	-	7

13, 28 May 2003	-	-	-	-	-[d]	-[d]	2[b]	-	-	8
9-19 Sept 2003	0[b]	25-35[c]	4[e]	5-10[c]	280[c]	16[c]	2[c]	1[b]	0[b]	7
19 June-15 July 2008	0[b]	55[b,f]	6-12[c]	0[c]	277[c,g]	56[c,g]	2, 3[b,c]	0[b]	0, 5[b,c]	7

Note: Results from Cliff and Pillar Caves in 1985-1992 were obtained during evening arrival counts of Mariana swiftlets (Rice 1993*b*, Rice and Taisacan 1993). Bats were not detected during bat emergence counts at Dangkolo, Krisidu, West Black Noddy, New Cave 3, and No. 18 Caves in 1995; at Dangkolo, Krisidu, and E Caves in 2003; or at Dangkolo, New Caves 2 and 3, No. 26, 28, 64, 65, 66, 67, 95, and 102 Caves in 2008.

[a] References: 1, Lemke (1986); 2, Reichel and Glass (1988); 3, Rice and Taisacan (1993); 4, Rice (1993*b*); 5, unpublished CNMI Division of Fish and Wildlife trip reports; 6, Craig and Chandran (1992); 7, this study; and 8, K. W. Stafford (pers. comm.).

[b] Direct roost count.

[c] Emergence count.

[d] Bats were not detected and were likely missed.

[e] Partial emergence count.

[f] Number represents the mean of six direct roost counts made with a night vision device.

[g] Number represents the mean of multiple emergence counts (see Table III-2).

Table III-2. Count results at caves occupied by Pacific sheath-tailed bats on Aguiguan, Mariana Islands, from 19 June-15 July 2008.

Cave	Date	No. Bats Recorded	Survey Period	Survey Method	Notes
New Cave 1	25 June	0	Dusk	Emergence	Only the large entrance was counted
New Cave 1	4 July	5	Day	Internal	Bats were seen briefly only while in flight; caves walls not scanned for more bats
New Cave 1	5 July	0	Day	Internal	
New Cave 1	5 July	0	Dusk	Emergence	Both entrances counted
New Cave 1	10 July	0	Day	Internal	
Crevice Cave	23 June	3	Day	Internal	2 adults, 1 pup
Crevice Cave	23 June	2	Dusk	Emergence	
Crevice Cave	27 June	4	Dusk, night	Emergence, internal	3 adults departed, 1 pup remained
Crevice Cave	10 July	2	Day	Internal	2 adults
Guano Cave	20 June	56	Day	Internal	Night vision device used for count
Guano Cave	21 June	52	Day	Internal	Night vision device used for count
Guano Cave	25 June	64	Day	Internal	Night vision device used for count
Guano Cave	28 June	43	Day	Internal	Night vision device used for count
Guano Cave	30 June	58	Day	Internal	Night vision device used for count
Guano Cave	7 July	54	Day	Internal	Night vision device used for count
East Black Noddy Cave	22 June	308	Dusk	Emergence	270 bats counted at west entrance, 38 at middle entrance
East Black Noddy Cave	27 June	382	Dusk	Emergence	321 bats counted at west entrance, 61 at middle entrance
East Black Noddy Cave	1 July	323	Dusk	Emergence	260 bats counted at west entrance, 63 at middle entrance
East Black Noddy Cave	5 July	317	Dusk	Emergence	255 bats counted at west entrance, 62 at middle entrance

Table III-3. Survey efforts at caves visited more than once that were not occupied by day-roosting Pacific sheath-tailed bats on Aguiguan, Mariana Islands, from 19 June-15 July 2008.

Cave	Date	Survey Period	Survey Method	Notes
Dankolo Cave	23 June	Day	Internal	
Dankolo Cave	27 June	Day	Internal	
Dankolo Cave	27 June	Dusk	Emergence	
Dankolo Cave	10 July	Day	Internal	
Cave 28	23 June	Day	Internal	
Cave 28	23 June	Dusk	Emergence	
Stairway Cave	27 June	Day	Internal	
Stairway Cave	9 July	Day	Internal	
Cave 65	24 June	Dusk	Emergence	
Cave 65	26 June	Day	Internal	
Cave 66	24 June	Dusk	Emergence	
Cave 66	26 June	Day	Entrance	
Cave 66	26 June	Dusk	Emergence	
Cave 67	26 June	Day	Internal	
Cave 67	26 June	Dusk	Emergence	
Cave 68	24 June	Day	Internal	
Cave 68	27 June	Day	Internal	
Cave 68	3 July	Day	Internal	
Cave 68	13 July	Day	Internal	
Pillar Cave	21 June	Day	Internal	
Pillar Cave	21 June	Dusk	Emergence	16 bats entered cave at dusk to roost
Pillar Cave	7 July	Day	Internal	
Pillar Cave	7 July	Dusk	Emergence	21 bats entered cave at dusk to roost
Fault Line Cave 1	21 June	Day	Internal	
Fault Line Cave 1	24 June	Day	Internal	
Fault Line Cave 1	26 June	Dusk	Emergence	Only the rear entrance was counted
Fault Line Cave 1	30 June	Day	Internal	
Fault Line Cave 1	11 July	Day	Internal	
Fault Line Cave 1	11 July	Dusk	Internal	1 bat seen entering cave; cave inspection found no other bats
Fault Line Cave 1	13 July	Day	Internal	
Fault Line Cave 1	14 July	Day	Internal	
Cave 95	24 June	Day	Entrance	
Cave 95	26 June	Dusk	Emergence	

Table III-4. Descriptions of caves occupied by Pacific sheath-tailed bats on Aguiguan, Mariana Islands. (S) designates sites with swiftlet colonies.

Cave Name	**Description**
Landing Cave	Large. Main chamber is 23 m long, 15 m wide, and 16 m tall at highest point. Two smaller chambers extend roughly 35 and 13 m beyond rear of main chamber. Much of cave is well lit, but portions are completely dark. Cave is damp and algae grows on most surfaces. Entrance is about 8.5 m wide and 16 m high. (S)
Guano Cave	Large. Main chamber is 20 m long, 3-5.5 m wide, and 7-18 m tall, with nearly vertical walls and a dome-like ceiling. A smaller side chamber is 5 m long, 2-3 m wide, and 7-9 m tall. Both rooms are completely dark. Two entrances occur side by side and measure 7 m wide by 2 m tall and 1 m wide by 4 m tall. (S)
Cliff Cave	Size unknown, but probably medium-sized or large. Located high up a cliff; not entered. Cave has two entrances, with west opening being larger at about 1 m wide by 2.5 m tall. (S)
Pillar Cave	Large. A single tunnel. Front section is a large well-lit dome, 20 m long, 5-10 m wide, and 8-15 m tall. Rear section is narrow and dark, 30 m long, 0.7-2.5 m wide, and 3-10 m tall. Entrance is 10 m wide. (S)
East Black Noddy Cave	Large. Main room angles steeply upward and is 12 m long by 45 m wide, with a ceiling height of 3-5 m. Floor is a jumbled mass of boulders. A smaller room is at the bottom. Both rooms are completely dark. Three entrances exist. An upper western entrance, 1 m wide by 0.5 m tall, is located high up a cliff and connects to the main room via a 7-m-long and 2-m-wide passage. A lower middle entrance, 2 m wide by 3 m tall, is a nearly vertical shaft going upward about 12 m. A lower eastern entrance, 0.3 m wide by 0.6 m tall, connects to the smaller room. (S)
Crevice Cave	Large. Main section is a narrow crevice 30 m long, 1-2 m wide, and 25 m tall. It extends vertically to the terrace above and is open at the top along its entire

	length. Bats inhabit a small dimly-lit upward curving chamber off to side, which is 5 m long, 0.3-1.2 m wide, 8 m tall, and the darkest part of the cave. Main entrance is 4 m wide by 25 m tall. (S)
Fault Line Cave 1	Medium-sized. Main chamber is 15 m long, 5-6 m wide, 10 m tall, and dimly lit. Two main openings present, with largest being 0.6-1.0 m wide and 2 m tall.
New Cave 1	Large. Main chamber is 9-10 m long, 5 m wide, and 7-10 m tall, with two large entrance chambers connecting to it. The largest of these is 10 m long, 3-6 m wide, and 4-15 m tall, and full of boulders; the second is 15 m long, 2-5 m wide, and 2-5 m tall. At least two other smaller openings also present. (S).

Table III-5. Emergence times of Pacific sheath-tailed bats and arrival times of Mariana swiftlets during evening counts at caves on Aguiguan, Mariana Islands.

			Sheath-Tailed Bats		
Cave	**Date**	**Time of Sunset (hr)**	**Time of initial and Final Exits (hr)**	**Time of Most Exits (hr)**	**Arrival Times of Most Swiftlets (hr)**
East Black Noddy Cave	26 March 1995	1829	1828-1841	1830-1838	1840-1900
East Black Noddy Cave	17 Sept 2003	1818	1810-1828	1812-1823	1820-1845
East Black Noddy Cave	18 Sept 2003	1818	1808-1828	1810-1823	-[a]
East Black Noddy Cave	22 June 2008	1850	1833-1857	1835-1850	1852-1910
East Black Noddy Cave	27 June 2008	1851	1836-1912	1844-1902	1905-1915
East Black Noddy Cave	1 July 2008	1851	1835-1911	1840-1855	-
East Black Noddy Cave	5 July 2008	1852	1834-1905	1840-1900	-
Guano Cave	23 March 1995	1829	1839-1846	1839-1846	1840-1855
Guano Cave	31 May 1995	1844	1844-1855	1844-1855	1902-1919
Guano Cave	18 March 2002	1828	1833-1912	1833-1851	-
Guano Cave	10 Sept 2003	1824	1807-1825	1810-1820	-

Guano Cave	21 June 2008	1850	1837-1900	-	-
Cliff Cave	1 June 1995	1844	1840-1850	1840-1850	1900-1910
Cliff Cave	21 June 2008	1850	1852-1903	1852-1900	-
Pillar Cave	1 June 1995	1844	1846	1846	1900-1914
Pillar Cave	15 March 2002	1828	1845-1850	1845-1850	-
Crevice Cave	6 June 1995	1845	1837-1900	1837-1840	-
Crevice Cave	23 June 2008	1850	1844-1848	1844-1848	-

[a] - = times not recorded.

Table III-6. Characteristics of Pacific sheath-tailed bat emergence periods in relation to colony size. Specific times for emergence periods appear in Table III-5.

Characteristic	**Large Colonies (≥ 232 bats)**	**Small Colonies (≤ 69 bats)**
	Mean ± SD	Mean ± SD
Length of emergence period from 1st bat to last	27.5 ± 7.9 min (*n* = 6)	13.8 ± 10.6 min (*n* = 12)
Length of period when "most" bats emerged	15.3 ± 3.3 min (*n* = 6)	7.7 ± 4.6 min (*n* = 11)
Beginning time in relation to sunset	9.8 ± 3.4 min before (*n* = 6)	0.5 ± 8.6 min after (*n* = 11)
Ending time in relation to sunset	5.5 ± 3.7 min after (*n* = 6)	8.1 ± 9.8 min after (*n* = 11)
Ending of period when "most" bats emerged in relation to period of most swiftlet entries	0.7 ± 3.2 min before (*n* = 3)	5.4 ± 7.7 min before (*n* = 5)

Appendix III-A. Descriptive catalog of all caves visited on Aguiguan in March, May, and June 1995, September 2003, and June-July 2008. Caves were entered whenever possible and were categorized as small, medium, or large in overall size (see Methods section of this report). Cave and entrance dimensions (m) were measured or visually estimated for many of the caves and are reported with the following abbreviations: l, long; w, wide; and t, tall. Caves had no Pacific sheath-tailed bats, Mariana swiftlets, guano, or nests unless specifically mentioned. Coordinates are those taken in 2008, and are in datum WGS 84, 55 P.

No.	Other Name	Description and Comments
1.	Landing	Large. About 100 m north of the old boat landing site and about 30 m inland from ocean in an indentation in the cliff wall. Main chamber is 23 l x 15 w, and 16 t at highest point. Two smaller chambers extend about 35 l and 13 l beyond rear of main chamber. Entrance: about 8.5 w x 16 t. Curtain-like rock formations are present. Cave is damp with algae growing on most surfaces. Bats present in 1984 and 1992, swiftlets are currently present. A little human-made debris present.
2.	Guano	Large. Main chamber (20 l x 3-5.5 w x 7-18 t) branches to left with a smaller side chamber (5 l x 2-3 w x 7-9 t) on the right. Two entrances (7 w x 2 t, 1 w x 4 t) separated by a rock. All bats roost in main chamber. Swiftlets are present. Old boards and other human debris present. Coordinates 343039E, 1642089N
3.	-	Small. Cave-like hole under a huge boulder; 4 l, low ceiling, has a second opening on other side, small flowstone pillar inside. Cave is located in the "Crack."
4.	-	Small. Vertical crevice; 5 l x 0.3-0.6 w x 3 t. Small opening at end. Cave is located in the "Crack."
5.	-	Small. Horizontal hole beneath a large boulder; 9 l x 2 t. Cave is located in the "Crack" just below Cave 3.
6.	-	Small. Hole in cliff face.

7.	-	Small. Hole in cliff face.
8.	-	Small. Hole in cliff face.
9.	-	Small. Hole in cliff face.
10.	-	Small. Hole in cliff face.
11.	New 1	Large. Located 25-30 m southwest of New Cave 3 along the same small limestone ridge and at same elevation. Main chamber is 9-10 l x 5 w x 7-10 t, with two large entrance chambers leading into it and at least two other smaller openings also present. Largest entrance chamber is 10 l x 3-6 w x 4-15 t, full of boulders indicating past ceiling collapse, 2-3 large vertical *Ficus* roots present, not safe to enter main chamber via this entrance. Second entrance chamber is 15 l x 2-5 w x 2-5 t, main chamber accessible via this entrance. Both entrances used by swiftlets. Appears suitable for bats, swiftlets are present. Formerly known as #6 Cave when first discovered in the late 1980s; its location was incorrectly mapped in some CNMI field trip reports (e.g., Reichel and Camacho 1989). Coordinates 343187E, 1641656N
12.	E	Size unknown. Located along cliff face below old Japanese road. Single entrance is split into three parts by two boulders. These open into a nearly vertical shaft (20-25 m deep) that is too steep to descend without equipment. Chamber continues on out of sight at bottom. Two entrance openings measure: 1.5 w x 1 t, 0.3 w x 1 t. Perhaps suitable for bats.
13.	D	Small. Next to Cave B and near Caves A and C. Entrance splits off into two smallish chambers, with longest being 10 l. Most of cave is well lit. Entrance: 5 w x 1-2 t. Parts appear suitable for bats. Part of Orphan Kids Cave Complex in Stafford (2003). Coordinates 343205E, 1641432N.
14.	C	Small. Near Caves A, B, and D. Vertical crevice that is a true cave; 10 l x 0.5-1 w x 6 t.
15.	B	Medium. Next to Cave D and near Caves A and C. Main shaft slopes downward (35 l x 2-7 w), with a smaller one angling slightly upward (10 l). A smallish room exists midway just before shafts split. Main entrance: 1 w x 1 t.

		Upper shaft has a second tiny opening. Perhaps suitable for bats, but not swiftlets. Part of Orphan Kids Cave Complex in Stafford (2003).
16.	A	Medium. Near Caves B, C, and D. Room slopes down at 45-degree angle; 19 l x 4-12 w. Upper section is well lit. Two entrances: 5.2 w x 1.5 t, 2 w x 0.7 t. Part of Orphan Kids Cave Complex in Stafford (2003). Coordinates 343190E, 1641390N.
17.	Crevice	Large. Main portion of cave is a narrow crevice that appears to be open along the top; 30 l x 1-2 w x 25 t. Crevice is so tall that it extends to the terrace above. Entrance: 4 w x 25 t. Floor angles upward, with a 2-m tall step of loose rock midway through, which makes access to rear difficult. Most of cave is dimly lit, but a rear room has more light. Cave is reminiscent of a slot canyon in the southwestern U.S. A small upward curving side crevice is located 13 m from the entrance on the right wall; 5 l x 0.3-1.2 w x 8 t; several bats are present here, with a small amount of guano found on the floor at the entrance of this feature. Guano is absent from rest of cave. Insect Bat Cave in Stafford (2003). Coordinates 343541E, 1641526N
18.	-	Small. Vertical crevice about 50 m east of Crevice Cave. Tall but not too long. Extends upward to the terrace above, with potential for a hidden cave to be located much higher up the cliffside.
19.	Dangkolo	Large. Huge central room (52 l x 15 w x 20 t), with two deeper side chambers, one at each rear corner. Side chambers are 6-10 m deeper than main room. Entrance: 4 w x 0.9 t. Cave is extremely damp, with a faint mist hanging in air. Appears suitable for bats, swiftlets are present. Coordinates 343542E, 1641686N.
20.	-	Small. Vertical crevice with darkened ceiling; 3-10 l. Appear suitable for bats.
21.	-	Small. Vertical crevice with darkened ceiling; 3-10 l. Appear suitable for bats.
22.	-	Small. Vertical crevice with darkened ceiling; 3-10 l. Appear suitable for bats.
23.	-	Small. Low ceiling.
24.	-	Small. Low ceiling.

25.	-	Small. Located close to large natural window in top of cliff face.
26.	-	Large. An attractive cave, open and well lit; 18 l x 8 w x 4-10 t. Perhaps suitable for bats, but probably not for swiftlets. Hollow Column Cave in Stafford (2003).
27.	-	Small. Located about 4 m high in an indentation in the cliff; 4.5 l x 2 w x 6 t. Open and well lit. Goat sign.
28.	-	Medium. Vertical slot; 25 l x 2-3 w x 10 t. Fairly well lit, but two high ceiling domes appear dark enough to be suitable for bats and swiftlets; a small (0.6 w x 1.0 l) dome just inside the entrance look good for bats too. Large rocks cover much of floor part way inside. Large broken stalagmite leans across cave near entrance. Goat sign. Toppled Column Cave in Stafford (2003). Coordinates 343914E, 1641548N.
29.	-	Small. Located part way up cliff side. A crevice that may extend inward a fair distance. Did not enter; entry would require some effort to climb up the cliff. Perhaps suitable for bats. Part of Natural Arch Cave in Stafford (2003).
30.	-	Small. Located about 4 m high on a sheer rock wall. Entrance: 7-8 w x 1 t, with a pillar located near the middle. Did not enter. Interior appears dark. May be too shallow for bats, but should be checked. Part of Natural Arch Cave in Stafford (2003).
31.	-	Medium. Main room is 10 l x 9 w x 10 t, with a large open ledge on the right side. A smaller room (6 t) in the rear is located about 3.5 m above the floor and angles upward to a dome ceiling. The smaller room appears especially suitable for bats and swiftlets. Note: cliffs above this site and the last few previous caves contain a series of good ledges and formations at mid-level. Smaller and crevices could be present and may potentially hold bats.
32.	-	Large. Main room: 23 l x 6-8 w x 8-10 t. Large opening: 10 w. Two darker domes in the rear. Appears quite suitable for bats and swiftlets. Goat sign. Part of Diamond Cave in Stafford (2003).
33.	-	Small. Adjacent to previous cave, with 3 columns located in between them. Chamber is a tunnel: 15 l x 0.6-5 w x 2-6 t. Higher dome in rear appears suitable for bats. Goat sign. Part of Diamond Cave in Stafford (2003).

34.	-	Large. Must climb up about 4 m to enter cave. Main room is large and auditorium-like; 15 l x 31 w x 10 t. Open, airy, and well lit. Entrance: 31 w x 10 t. A smaller dark room (4 w) is located in the rear and is elevated 10 m above main floor. Could not enter small room, but no bats or swiftlets were present. Small room appears excellent for both species. Goat sign in main room. *Lantana* grows in front of main entrance. Goat Cave in Stafford (2003).
35.	-	Small. A small overhang; 3 l x 4 w x 1 t.
36.	Krisidu	Large. Huge main room (75 l x 20 w x 15 t), with a smaller but long tunnel extending farther inward from right rear corner. Entrance: 7 w x 1.3 t. Very moist inside. Appears suitable bats and swiftlets. Site No. 48 in Butler (1992); Liyang Atkiya in Stafford (2003).
37.	Stairway	Large. Located just west of the foot path leading down cliff side. Main room is large and open; 8 l x 30 w x 5-7 t. A small dark side chamber is on right side of main room; 20 l x 2-4 w x 1-2.2 t. Lots of boards and human debris in main room. Side room appears suitable for bats, swiftlets are present. Site No. 11 in Butler (1992). Coordinates 345682E, 1642066N.
38.	-	Small. Attractive cave, somewhat circular; 6 l x 8 w x 1-2 t. Two pillars inside. A 4-inch by 4-inch plank and some small pieces of wood are inside. Goat sign.
39.	-	Small. 4-6 l x 10 w x 1.2-2 t. Two entrances and two pillars. Goat sign.
40.	-	Small. Narrow horizontal crevice under rocks.
41.	-	Small. Narrow horizontal crevice under rocks.
42.	-	Small. Horizontal crevice under a rock; 5 l x 3-6 w x 0.8-1.3 t.
43.	-	Small. Comprised of two vertical crevices, 2.5-4 t, with dark areas.
44.	-	Small. Horizontal crevice under a boulder, with two rocks supporting the east side; 6 l x 8 w x 1-2 t.

45.	-	Small. Cave-like hole in lower cliff wall; 0.7-2 t.
46.	-	Small. Cave-like hole in lower cliff wall, with two low side rooms; 0.3-1 t. May be Site No. 44 in Butler (1992) and Waypoint Cave in Stafford (2003).
47.	-	Small. Cave-like hole in lower cliff wall, with two low side rooms; 0.3-1 t. May be Site No. 44 in Butler (1992) and Waypoint Cave in Stafford (2003).
48.	-	Small. Narrow vertical crevice in lower cliff; 10 l x <1 w x 10 t. South wall is a fallen rock slab.
49.	-	Medium. A two-leveled hole (one hole above the other) in lower cliff. Cannot reach the upper hole, which has dark entrance and is 2 w x 2 t. Rest of site is open. May be suitable for bats. Site No. 45 in Butler (1992); part of Tridacnid Cave Complex in Stafford (2003).
50.	-	Large. Very open and well lit; 20 l x 7-15 w. Entrance: 8 w x 6 t. A small hole in the right upper rear corner may be suitable for bats. It is high above the floor, but its interior is not visible from below. Much goat sign. Site No. 45 in Butler (1992); part of Tridacnid Cave Complex in Stafford (2003).
51.	-	Large. Attached by an elevated cliff side ledge to the previous cave. Comprised of two rooms (totaling 35 w), with the ceiling low (1 t) almost throughout, but reaching 2 t in part of the front room. Back room is completely dark. Two entrances: one is 4 w x 1 t, second is small. Unexploded bomb near entrance. Cave appears suitable for bats. Site No. 45 in Butler (1992); part of Tridacnid Cave Complex in Stafford (2003).
52.	-	Small. Located along bottom of cliff. May be Cabrito Cave in Stafford (2003).
53.	-	Small. Located along bottom of cliff. May be Cabrito Cave in Stafford (2003).
54.	-	Small. Located along bottom of cliff. Cave is: 10 l x 2.5-4 w x 2-3 t. Moderately lit. Discarded piece of Tupperware lies outside front of cave.
55.	-	Small. Circular main room (7 l x 5-6 m x 3-4 t), with small chamber on side (1 t). Entrance: 2.5 w x 3 t. Appears suitable for bats. Much goat sign inside.

56.	-	Small. Located at bottom of a cliff wall; 8 l x 3-7 w x 0.7-1 t. Rock fortification is piled on left side of entrance. Goat sign and skulls inside.
57.	-	Size unknown, perhaps large, viewed from long distance. A large vertical slot located in the upper third of a coastal cliff. Entrance is perhaps 3 w x 12-14 t. Cave depth difficult to judge, but may not be deep. Perhaps suitable for bats and swiftlets.
58.	-	Size unknown, perhaps small, viewed from long distance. A small vertical slot located near top of a coastal cliff. Can not judge cave depth. Perhaps suitable for bats and swiftlets.
59.	-	Size unknown, perhaps small, viewed from long distance. A horizontal cave located half way up a coastal cliff. Two or three dark openings present, not large. Can not judge cave depth. Perhaps suitable for bats.
60.	-	Small. Located at the opposite end of the same crevice holding the previous cave. A covered-over T-shaped crevice. One side is very narrow and cannot be entered. Not suitable for bats.
61.	-	Small. Located at end of a large fissure. Cave is 10 l x 2 w x 10 t. Dark at rear, with several small crevices at rear.
62.	-	Small. Could not enter. Deep vertical crevice; 10 l x 1 w x 16 t. Most of crevice is open at top, but several small holes lead out of sight and appear potentially suitable for bats.
63.	-	Small. Difficult to enter. A well-lit nearly vertical crevice (1.8 l x 5 w x 4 t) is the main room, with a small chamber (6 l x 1 w x 3 t) on east side and a narrow crevice (3 l x 0.4 w x 3 t) on the west side; entrance is 6 w x 1.5 l. Located beneath some boulders on edge of open forest and east side of karst rock associated with the “Fault Line”; located 10-15 m east of Cave 65. Part of the “Fault Line” cave complex. One bat seen circling inside entrance room after dark in 2008, but follow-up daytime visit 2 days later found no roosting bats.
64.	-	Large. An interesting cave with many different features, making description difficult. A large and fairly well lit central room is present, with a smaller room to the north that is accessed by crawling through a diagonal slot; combined size of rooms is 16 l x 8-10 w x 1-15 (?) t. A long narrow crevice (16 l x 0.3 w) extends from opposite end of main room. At least five entrances present; largest is 2 t x 0.4 w, another larger entrance is partially

		covered by several *Ficus* roots. Part of the "Fault Line" cave complex. No human debris or guano. Appears suitable for bats, but entrances may be too small for swiftlets. Coordinates 344844E, 1643303N.
65.	-	Small. Cave is a thin vertical side crevice along east side of a large open fissure; 9 l x 0.3 w x 3.5 t. Crevice is too narrow to continue after 2 m. Part of the "Fault Line" cave complex. Coordinates 344840E, 1643270N.
66.	-	Medium. At the end of a fissure. Could not get down to the cave floor or see the rear of the lower main room. Entrance: 4 w x 1-5 t, with a tall (11 t) slender crevice on left side. Part of the "Fault Line" cave complex. Perhaps suitable for bats.
67.	-	Small. At end of an open slanting fissure; 10 l x 1-1.2 w x 5-6 t. A true cave exists at rear, which is 5 l x 0.3-0.5 w x 1-5 t. Part of the "Fault Line" cave complex. Dark, but probably not suitable for bats. Not suitable for swiftlets.
68.	-	Medium. Comprised of a series of small low rooms, with multiple entrances, runs along base of hillside and is never more than a few m deep. Overall floor space is fairly large; about 40-50 m long, parts of cave are only 0.8-1.2 m tall. One room is 1.5-1.8 t and a side crevice is 3-4 t. Lighting is dim to almost dark. Old boards present. Looks suitable for bats. Site No. 6 in Butler (1992). Coordinates 344765E, 1643135N.
69.	-	Small. An overhang in a cliff face; 3 w x 4 t. One of David Steadman's excavation sites.
70.	-	Small. A vertical crevice; 6 l x 3 t.
71.	-	Small. A crevice turning into a small cave; 10-15 l x 2-3 t. Dimly lit inside.
72.	-	Small. Rockshelter under a large boulder; 5 l x 6 w x 2.5 t.
73.	-	Small. Rockshelter under a large boulder; several meters in all dimensions.
74.	-	Small. A vertical crevice; 5 l x 2 w x 15 t.
75.	-	Small. Rockshelter overhang; 4 l x 6 w x 0.8 t.

76.	East Black Noddy	Large. Located at east end of a broad circular indentation in cliff side along the north shore. Three entrances, as mapped by Stafford (2003), all of which are somewhat cryptic until closer inspection is made. Middle entrance (2 w x 3 t) is a nearly vertical shaft going upward about 12 m from ground level, but is too steep to climb safely. East entrance (0.3 w x 0.6 t) is a few meters to the left on an adjacent ledge. West (uppermost) entrance is circular in shape and about 0.6-1 m in diameter and is located about 11-13 m high on cliff face near a wavy rock formation; this is about 15 m west of middle entrance and about 3 m east of the rock pillar standing below on the ground. Human entry is easiest via the east entrance, which accesses a small first room attached to a much larger main room (12 l x 45 w x 3-5 t). This room angles steeply upward, but climbing is treacherous. Middle entrance is not visible from this room. Bats and swiftlets are present. Part of Swiftlet Cave in Stafford (2003). Coordinates 344004E, 1642923N.
77.		Cave number not in use.
78.	West Black Noddy	Large. Located at west end of a broad circular indentation in cliff side along the north shore. A single open room (20 l x 10 w x 12 t) with a large entrance (12 w x 12 t). Most of cave is dimly lit, but a small indentation on south wall may be dark enough to attract bats and swiftlets. Part of Swiftlet Cave in Stafford (2003).
79.	-	Small. A crevice among boulders, dimly lit, open on both ends; 5 l x 1.5 w x 3 t.
80.	-	Small. Rockshelter under a boulder; 10 l x 3 w x 1 t.
81.	Cliff	Size unknown, probably medium or large. Located high up cliff side and is too treacherous to reach without climbing gear. Cave has two entrances: west opening is largest, with two pillars inside giving the appearance of splitting the opening into three sections; may be 1 w x 2.5 t overall. East opening is a bit higher and has some rock extending down over part of the top. Bats and swiftlets are present. Viewed from coordinates 343432E, 1642715N.
82.	Pillar	Large. Long narrow cave, with a pillar inside near the entrance. Front section of cave is a large well-lit dome; 20 l x 5-10 w x 8-15 t. Rear section is a dark narrow tunnel; 30 l x 0.7-2.5 w x 3-7 t, but is 10 t in one area. Entrance: 10 w. Bats and swiftlets are present. Coordinates 343363E, 1642648N.

83.	-	Small. Rockshelter; 7 l x 5 w x 2 t. Fairly well lit inside.
84.	-	Small. Located in rock pile; 3 l x 0.3-0.7 w x 0.7-1.2 t. Dark at the end. May be Booney Bee Sink in Stafford (2003).
85.	-	Small. Upward angled rock overhang, narrow in rear; 5 l x 1.2-5 w x 1-4 t. Perhaps suitable for bats.
86.	-	Small. Overhang under boulder; 8 l x 5 w x 1 t. Three openings, dimly lit inside, well ventilated. Goat sign inside. Fortified, two rocks piles at entrances, with one entrance having four wooden posts present to support boulder. Probably Site No. 25 in Butler (1992). Coordinates 343408E, 1642122N.
89.	-	Small. Single room, 7 l x 3-5 w x 2 t. Probably moderately-well lit at times. Goat sign. Listed as Find Site 3 in Butler (1992).
91.		Cave number not in use.
92.	Elvin's	Small. Begins with a 4-m deep vertical entry shaft, floor then angles downward out of sight. Difficult to enter. No dimensions available. Located along the east wall of a long (> 100 m) fissure just west of New Cave 1.
93.	-	Large. Large open rockshelter-like site under a huge fallen boulder, with overall size being 20 l x 30 w x 1-5 t. Most of underhang is well lit, but two small dark areas are present at base of boulder. Site does not look suitable for bats. A smaller overhang occurs on backside of boulder.
94.	Fault Line 1	Medium. Main room is 15 l x 5-6 w x about 10 t overall. Lower half of room has a level floor, while the other half angles steeply upward. Roof is formed by a large fallen rock slab. Two main openings exist, with several small holes present at top of one side. Largest entrance is triangular shaped and is 2 t x 0.6-1 w. Second entrance is a diagonal crevice and is tight to squeeze through. Part of the "Fault Line" cave complex. Some old lumber and human debris occur inside. One bat present in 2003. May be appropriate for swiftlets. Included under Site No. 6 by Butler (1992).
95.	-	Small. Located adjacent to and below Fault Line Cave 1. Roof is formed by a fallen rock, with main room being 12 l x 1-2.5 w x 4-7 t. Mostly well-lit, but has a couple of darker areas. Three entrances, with the lower one being

		an open crevice that is 3-4 l x 0.3-1 w x 2-6 l. Part of the "Fault Line" cave complex. Probably not suitable for bats or swiftlets.
96.	-	Small. A narrow crevice, 10 l x 0.3 w x 6-7 t. Mostly open, but has some closed ceiling in rear. Located just west of the "Fault Line" near a natural arch in the rocks. Part of the "Fault Line" cave complex.
97.	-	Small. Located on lower side of a boulder next to cliff base; 2 l x 1.5 w x 0.5 t; *Ficus* roots cover part of the entrance.
98.	New 2	Small. Located in same karst hole as the main entrance to New Cave 1; found opposite this entrance perhaps 5-10 m away; also located about 25 m southwest of New Cave 3. Cave has one room, 10 l x 5 w x 2 t.
99.	New 3	Large. Located along the same small limestone ridge as New Caves 1 and 2, about 25 m to the northeast. Two entrances are known. The southwest entrance is 1 x 1.5 w and drops into a deep shaft that could not be entered; swiftlets mostly enter this hole. The northeast entrance (50 m northeast of the first entrance) is actually comprised of four entrances. These fall sharply downward about 20 m, but the shaft is too steep to descend. A rope is needed for access. The chamber at the bottom is fairly dark, extends inward out of sight, and could be large. Main opening at this entrance is 5 w x 2.5 t and occurs under a fairly flat rock; single small openings occur on both sides of the main entrance; swiftlets mostly exit this entrance. Many old boards and other human debris lay on the inner level below the entrance, but nothing is visible at the bottom. Appears suitable for bats, swiftlets are present. Site No. 55 in Butler (1992). NE entrance coordinates 343232E, 1641708N; SW entrance coordinates 343201E, 1641682N.
100.	-	Size unknown, probably small. Located partway down along the west wall of the "stairway" passage near Stairway Cave. A narrow crevice in the rock leads to a vertical shaft estimated at 30 m deep. The shaft and its entrance are triangular shaped and about 5 m wide on each side. Cannot see the bottom well enough to know whether rooms are present.

101.	-	Size unknown, probably small. Uppermost site in the “Fault Line” cave complex; located about 30 m uphill from Fault Line Cave 1. Has 3 main entrances that fall 15 m to a level floor; these are too steep to enter. Dimensions not estimated. Cannot see entire interior, thus may be larger than expected.
102.	-	Small. Located between Caves 63 and 66 at south end of the bottom “step” inside the “Fault Line.” Dimensions: 4 l x 1.5 w x 2.5 t. Part of the “Fault Line” cave complex.
103.	-	Medium. The entire feature is large in size, but most of it is uncovered by ceilings. A front vestibule (24 l x 4-8 w x 18 t) and two large side rooms (east room: 22 l x 8 w x 8-12 t; west room: 20 l x 8 w x 12-14 t) are all uncovered and therefore do not provide cave environments. Front vestibule empties out onto a sheer cliff, which falls about 30 m to the terrace below. The only true cave occurs in the middle of the south wall. It has an entry chamber (10 l x 2-3 w x 2-3 t) that goes straight in, with a side room (also 10 l x 2-3 w x 2-3 t) facing west near the rear. Looks suitable for bats. Entry is via the side of the front vestibule and requires a fairly risky climb down. Coordinates 344316E, 1642923N.

Other Caves Described in Butler (1992).

87.	-	Listed as Site No. 30. Located at cliff base. A rockshelter with a lower chamber (5 l) on east side leading upward to a larger overhang (3-4 l x 15 w) with a low ceiling. Larger overhang is 3 m above ground level; its floor has been leveled with stacked rocks. Human debris present.
88.	-	Listed as Site No. 29, with photograph presented. Located at cliff base. Small rockshelter; 3-4 l x 3 w. Vertical logs positioned across part of entrance.
90.	-	Listed as Site No. 56. Entrance opens into a “large” dark room that was not visited. Rocks stacked around the entrance.

<u>Other Caves Described in Stafford (2003).</u>

104. - Listed as Swarming Termites Cave. Located in the eastern region of the middle terrace. Small.

105. - Listed as Liyang Lomuk. Located in the north-central region of the middle terrace. Small.

106. - Listed as Lizard Cave. Located in the north-central region of the middle terrace. Small.

107. - Listed as Spider Cave. Located in the north-central region of the middle terrace. Small.

108. - Listed as Scorpion Cave. Located in the north-central region of the middle terrace. Medium-sized?

109. - Listed as Goat Fracture Cave. Located in the northwest region of the lower terrace. Small.

110. - Listed as Anvil Cave. Located in the northwest region of the upper terrace. Medium-sized?

111. - Listed as Dove Cave. Located in the northwest region of the upper terrace. Small.

112. - Listed as Almost Cave. Located in the northwest region of the upper terrace. Small.

113. - Listed as Screaming Bat Cave. Located in the northwest region of the upper terrace. Two small caves present. Named for several fruit bats vocalizing nearby at night (K. Stafford, pers. comm.).

114. - Listed as Biting Mosquitoes Cave. Located in the northwest region of the upper terrace. Small.

115. - Listed as Isotope Cave. Located in the northwest region of the upper terrace. Medium-sized?

116. - Listed as Pepper Cave. Located in the northwest region of the upper terrace. Small.

Section IV. Habitat Occupancy and Detection of the Pacific Sheath-Tailed Bat (*Emballonura semicaudata rotensis*) on Aguiguan, Commonwealth of the Northern Mariana Islands

P. Marcos Gorresen, Frank J. Bonaccorso and Corinna A. Pinzari

ABSTRACT

We used occupancy analysis to quantify Pacific sheath-tailed bat (*Emballonura semicaudata*, Emballonuridae) foraging activity and its relationship to forest structure and proximity to cave roosts on Aguiguan Island in the Commonwealth of the Northern Mariana Islands. Canopy cover, vegetation stature and distance to known roosts were covariates that best explained bat occurrence. Additionally, we provide quantitative descriptions of the echolocation calls of *E. semicaudata.* Search-phase calls were characterized by a relatively narrow bandwidth and short pulse duration typical of insectivores that forage within vegetative clutter. Two distinctly characteristic frequencies were recorded: 30.97 ± 1.08 kHz and 63.15 ± 2.20 kHz.

INTRODUCTION

The Pacific sheath-tailed bat (*Emballonura semicaudata*) was once common and widely distributed across the southwestern tropical Pacific. It is the only insectivorous bat recorded from much of this region (Koopman 1997), and four subspecies have been described: *E. s. rotensis* from the Mariana Islands (Guam and the U.S. Commonwealth of the Northern Mariana Islands [CNMI]), *E. s. palauensis* from Palau, *E. s. sulcata* from the Caroline Islands (Chuuk and Pohnpei), and *E. s. semicaudata* from Vanuatu, Fiji, Tonga, and Samoa (Independent and American). Although populations appear sizable and stable in some locations, mainly in the Caroline Islands (Wiles and others 1997), they have declined considerably in other areas, including the Mariana Islands, Fiji, Samoa, and possibly Tonga (Hutson and others 2001, Helgen and Flannery 2002). In the Marianas, the endemic subspecies *E. s. rotensis* formerly inhabited Guam, Rota, Aguiguan, Tinian, Saipan, and possibly Anatahan and Maug (Lemke 1986, Flannery 1995, Ellison and others 2003). Currently, it appears to be almost entirely extirpated from the Mariana archipelago, with a single remnant population of this subspecies occurring on the small uninhabited island of Aguiguan. The species is listed as Endangered by the World Conservation Union (Chiroptera Specialist Group 2000) and the Government of CNMI (Anonymous 1991). *Emballonura. s. rotensis* and *E. s. semicaudata* are category 3 candidates for listing under the U.S. Endangered Species Act (U.S. Fish and Wildlife Service 2007). The threats to the Aguiguan population primarily include habitat loss from past clearing of native forest for agriculture, with subsequent replacement by invasive vegetation (Esselstyn and others 2004), habitat degradation from feral goat browsing (goats were introduced in the mid-1800s; Esselstyn and others 2004), and a small population size with limited distribution that leaves it vulnerable to extirpation by typhoons (Chiroptera Specialist Group 2000).

Current status assessments of *E. s. rotensis* on Aguiguan indicate that the population numbers about 400-500 individuals based on counts of 359-466 bats at caves (see Section III of this administrative report), roosts are limited to caves (Section III of this administrative report), and the bat primarily uses forest habitat (Esselstyn and others 2004). Although critical for assessments of population status, the estimation of population size is complicated when bats use more than one roost and when the location of occupied caves are incompletely known, particularly if surveys of caves for bats are not completed simultaneously or in a short time period. Movement of bats among alternate caves may cause counts to be annually or seasonally variable even if population size remains fairly constant. Consequently, variability in counts can make assessments of bat population trends difficult (for more details on problems in estimating bat population size see papers in O'Shea and Bogan 2003).

Occupancy analysis is a fairly new technique only recently being applied to bat studies in which echolocation calls are used as a measure of occurrence and activity (Gorresen and others 2008). The technique corrects for bias in estimates of spatial occurrence by accounting for imperfect detection (i.e., bats present but not detected; MacKenzie and others 2002). It also generates metrics with associated variance estimates that permit comparative analyses (i.e., future assessments of occupancy and distribution over time). We used occupancy analysis primarily to quantify Pacific sheath-tailed bat foraging activity on Aguiguan. Secondary objectives included further study of the relationship of foraging activity to forest structure and land-cover composition and the temporal use of forest habitat and proximity to cave roosts. We also provide quantitative descriptions of the echolocation calls of *E. s. rotensis* because there is little published information on the calls of this or other species of the genus *Emballonura*.

MATERIALS AND METHODS

Study Area

Detailed descriptions of the environment on Aguiguan (14°51' N, 145°33' E) are available in Engbring and others (1986), Esselstyn and others (2004) and Wiles and others (Section III of this administrative report), and are summarized below from these sources. Aguiguan is located in the southern Mariana Islands in western Micronesia. It is a small (7.0 km^2) limestone island with a flat central plateau encircled by escarpments and terraces. A ridgeline along the northern edge of the island attains a maximum elevation of 166 m.

Landcover on Aguiguan is comprised of 4 general types of vegetation: native limestone forest; non-native forest; non-native shrubland; and coastal scrub and grassy areas. Native limestone forest occurs on about 49% of the island and is mostly found on steep slopes and terraces. The forest canopy reaches up to 15 m and intense browsing by feral goats (*Capra hircus*) has formed an open understory in most areas. Although Aguiguan is now uninhabited by people, the central plateau (making up about 42% of the island area) was cleared for agriculture between about 1936 and the early 1940s. This area is now primarily comprised of non-native secondary shrub and forest vegetation. Shrubland consists of dense 1-3 m tall thickets and most non-native forest occurs in small patches 5-10 m in stature. Coastal scrub, grass and unvegetated areas make up the remaining 9% of the island's landcover.

Study Design and Analysis

Thirty-one stations were surveyed for Pacific sheath-tailed bat activity between 25 June and 14 July 2008. Stations were established at or near locations initially sampled by Esselstyn and others (2004) and generally spaced at 370-m intervals. Sampling techniques and measures of bat activity and habitat use followed the approach developed by Gorresen and others (2008) for the endangered Hawaiian hoary bat (*Lasiurus cinereus semotus*) in which bat vocalizations (i.e., echolocation "calls") were recorded at a series of stations ("sites") on consecutive nights. Calls were recorded on Aguiguan with Anabat II detectors (Titley Electronics, Ballina, New South Wales, Australia) over 3-5 nights at each site, and were written to a compact flash card with a Zero-Crossings Analysis Interface Module (ZCAIM; Titley Electronics). Call files were processed with AnalookW software (version 3.3f; Titley Electronics; available at http://www.hoarybat.com; accessed July 2008) to filter ambient noise. Graphic files were visually inspected to ensure that residual noise was not interpreted as echolocation calls or call components ("pulses"). Descriptive characteristics of search-phase echolocation calls extracted by AnalookW included minimum (F_{min}), maximum (F_{max}) and characteristic (F_c) frequencies (kHz); frequency range (difference between F_{max} and F_{min}); pulse duration (ms); and time (T_c) from the start of pulse to F_c (ms). Parameters F_c and T_c were derived only from the "body" (i.e., flattest portion) of the pulse as defined AnalookW, whereas the entire pulse was used to characterize the other parameters. Detailed definitions of these call parameters are provided by Gannon and others (2004). *Emballonura semicaudata* is the only species of echolocating bat known from the Marianas Islands (Flannery 1995, Esselstyn and others 2004) and there were thus no questions about the species identity of the calls we recorded.

Occupancy analysis (MacKenzie and others 2002) was used to assess the relationship between habitat attributes and the proportion of occupied sites (Ψ), to adjust Ψ for a detection probability (*p*) of less than one, and to produce associated measures of uncertainty for comparative analyses. Occupancy analysis was performed with the software program PRESENCE (version 2.0; available at http://www.mbr-pwrc.usgs.gov/software.html; accessed September 2008).

We developed *a priori* models in which bat occupancy was a function of habitat covariates; i.e., Ψ (Cov)$p(\cdot)$. We used our constant parameter model, $\Psi(\cdot)p(\cdot)$, as a reference null model from which to compare habitat effects on occupancy. Because of small sample size, interactions between covariates were not examined. Weather conditions were uniform during the 3-week period of sampling, therefore no sampling covariates were used to adjust detection probabilities; i.e., $\Psi(\cdot)p$(Cov) models were not examined (e.g., where *p* could be a function of wind or rain).

Habitat attributes that were visually and qualitatively assigned into binary categories included understory clutter (open-uncluttered; closed-cluttered) and dominant vegetation (native; exotic). Attributes with more than 2 levels were quantified with indicator variables, and included stem diameter

(<10, 10-30 and >30 cm dbh), vegetation stature (<5; 5-10; >10 m height) and canopy cover (<30%; 30-70%; >70% closure). Proximity from each survey station to 7 known roost sites was calculated as the nearest neighbor distance in ArcGIS 9.2 (ESRI 2006). To improve model performance, proximity was standardized to have a mean of 0 and a standard deviation of 1 (Donovan and Hines 2007). The seven known roosts analyzed included Landing, Pillar, Guano, Crevice, Cliff, East Black Noddy, Fault Line 1 (as described n Section III of this administrative report).

Although habitat use may be defined simply by the occurrence of a species, this is a non-discriminating criterion because bats can commute through areas not used for foraging. Therefore we distinguished areas of higher and lower activity based on the number of echolocation pulses in recorded bat call files. In addition to the use of a “no threshold” of activity (i.e., all echolocation pulses were used), we identified observations where the total number of pulses within each 1-min period exceeded one of three nested series of thresholds: 50th (median), 70th and 90th percentile. These higher activity events were coded as 1 in matrices that tallied their incidence for each site and each night. Zeros were assigned to matrix cells for periods in which there was no recorded activity or pulse numbers were below the selected activity threshold. Detection probability and occupancy estimates for each survey site were calculated in program PRESENCE. The relationship of known roost proximity to bat arrival time (defined as the first hour with detections) was examined by correlation analysis.

Occupancy models were first ranked according to AIC values (Burnham and Anderson 2002). We subsequently used single-variable models within 2 AIC units of the best model as candidates for building 2-variable models. Models with a greater number of covariates were not considered because of the relatively small number of sites ($n = 31$). Interpretive results are presented only for the top-ranked models. Model goodness-of-fit was assessed with a parametric bootstrap procedure (MacKenzie and Bailey 2004), in which a Pearson X^2 test statistic p-value <0.05 and an estimated over-dispersion parameter <0.5 or >1.5 were measures indicative of a significant lack of model fit.

RESULTS

Bat Echolocation Calls

Pacific sheath-tailed bats were recorded at 22 of the 31 sites (Figure IV-1) sampled over a period of 19 nights on Aguiguan (for a total of 109 detector-nights). The distribution of pulses (38,858 pulses in 1,224 tallies of 1-minute duration) was highly skewed with a large proportion of filtered call files comprised of few pulses (Figure IV-2). One-half of all bat detections consisted of brief passes with less than 15 pulses per 1-minute interval. In contrast, 10% of observations were of peak activity events indicative of sustained foraging bouts with between 63 and 422 pulses. Another 40% of observations ($\geq$50th to <90th percentile) were of moderate activity in which total pulses numbered between 15 and 62.

The search-phase echolocation calls of *E. s. rotensis* were characterized by a fairly uniform narrowband, quasi-constant frequency (QCF) structure (Figure IV-3). The central shallow-modulated part of a QCF pulse was accompanied by a descending FM terminal element, and an ascending component to each pulse was also usually present. Two distinctly characteristic frequencies (F_c) were recorded: a 30.97 ± 1.08 kHz ("low") and 63.15 ± 2.20 kHz ("high") (Table IV-1). Despite greater atmospheric attenuation at higher frequencies under humid conditions (Lawrence and Simmons 1982), about three-quarters of the characteristic frequencies recorded were of the high harmonics (relative humidity was usually >80% on Aguiguan; T. O'Shea, USGS, pers. comm.). Although the Anabat echolocation system does not fully measure multi-harmonic information (Fenton and others 1999), the proportion of low versus high frequencies recorded may reflect shifts in the main energy from one dominant harmonic to another (e.g., "harmonic alternation"; Jung and others 2007), or the effects of microphone sensitivity (higher frequencies are more readily detected than lower frequencies) and the distance between a bat and detector (lower frequencies are less affected by distance; C. Corben, *in litt*.; http://users.lmi.net/corben/hrmncs.htm#Harmonics). No evidence of other harmonics was observed, but these may be present (e.g., 3rd and 4th harmonics; Ibáñez and others 2002) and "masked" by more dominant harmonics. The overall frequency range (i.e., difference between the maximum and minimum frequencies) was fairly narrow for both harmonics (low: 1.83 ± 1.10 kHz; high: 11.04 ± 4.05 kHz). Both low and high frequency pulses were of relatively short duration and the time to attain a characteristic frequency comprised most of the pulse extent (low: 1.44 ± 0.47 ms, T_c = 1.31 ± 0.37 ms; high: 2.75 ± 0.56 ms, T_c = 1.73 ± 0.46 ms).

Habitat, Occupancy and Detection Probability

Canopy cover, vegetation stature and distance (proximity to known roosts) were covariates that best explained bat occurrence across all threshold levels in models that accurately fit the data (Table IV-2). Each of these variables alone or in combination with one other accounted for up to 66% of AIC model weight.

Bat occupancy was related to canopy closure in a somewhat complex manner. It was highest at sites with high canopy closure and lowest at sites with moderate canopy closure, whereas it appeared intermediate at sites with low canopy closure (Table IV-3 and Figure IV-4). This may simply be due to the effects of small sample size on parameter estimation (*n* = 3 for low canopy closure sites). It may also reflect bat use of open canopied habitats adjacent to forest (all 3 sites were within 200 m of forest edge). Generally, occupancy in habitat characterized by high canopy closure was about 0.80 (e.g., $\hat{\Psi}$ = 0.84 ± 0.09 and 0.79 ± 0.11 in single-covariate 50th and 70th percentile models, respectively). Higher levels of other covariates acted to increase occupancy to over 0.90 (e.g., $\hat{\Psi}$ for high canopy closure sites near known roosts was 0.93 ± 0.06 for the 50th percentile model).

Vegetation stature exhibited a positive and direct relationship with occupancy, particularly in combination with other covariates. For example, occupancy in tall stature forest ranged between 0.55 ± 0.36 and 0.96 ± 0.06 depending on proximity to known roost caves, and 0.06 or less for mid- and low stature sites (90th percentile model; Table IV-3 and Figure IV-4).

Similarly, proximity to known roost caves consistently appeared as a significant covariate accounting for bat occupancy. Generally, occupancy at sites near roosts was about 0.85 (e.g., $\hat{\Psi}$ = 0.87 ± 0.09 and 0.83 ± 0.10 in single-covariate 50th and 70th percentile models, respectively; Table IV-3). Higher levels of other covariates also acted to further augment occupancy estimates. In addition to its effect upon the likelihood of bat use of particular habitats, roost proximity was also closely related to the timing of bat arrival at a site. The hour of first bat detections occurred significantly earlier at sites near caves (r = 0.64, *p*-value = 0.002; Figure IV-1). Detections at sites far from roost caves also tended to occur infrequently and at irregular intervals.

Bat occurrence was widespread on Aguiguan (observed Ψ = 0.71 when all bat detections were included; Table IV-4). As expected, peak bat activity was limited to a smaller proportion of sampled area than that indicated by simple presence alone. The 50th, 70th and 90th percentile null reference models (i.e., those with no habitat covariates) exhibited average $\hat{\Psi}$ of 0.62 ± 0.09, 0.59 ± 0.09 and 0.27 ± 0.08, respectively. In other words, whereas bats were detected across almost 3/4 of all sites, peak activity was observed at only 1/4 of the sampled landscape. Likewise, the probability of detecting bats was related to the activity threshold level. For example, *p* for all detections was 0.76 ± 0.05 but declined to 0.61 ± 0.09 for the 90th percentile threshold.

DISCUSSION

As first established by Esselstyn and others (2004), Pacific sheath-tailed bat activity was found to be closely associated with native limestone forest and proximity to known cave roosts. We also determined bat occupancy to be related to habitat characteristics typical of more structurally developed and mature forest; i.e., closed canopied and tall stature stands. However, scattered detections in open (non-forest) areas were notable because they indicate an ability to traverse and perhaps forage over such habitats. Moreover, the existence of at least one unknown roost is suggested by the early arrival and high activity of bats near several northeastern sample sites (stations "e" and "i"; Figure IV-1). Such roosts may contribute additional individuals to the current counts of 359-466 bats (Section III of this administrative report).

The high number of unoccupied but apparently suitable caves (Sections III and VI of this administrative report) suggests that the population size of *E. s. rotensis* may not be limited by roost availability. Instead, population size may be restricted by the small amount of mature native limestone forest (3.4 km^2) present on Aguiguan. On the other hand, the mobility of sheath-tailed bats (Wiles and others [1997] report commuting distances of at least 5 km in Palau) and their (albeit limited) use of exotic or less structurally complex vegetation is encouraging because it may mean that moderately more habitat is available than that solely provided by native limestone forest. Alternatively, it could also indicate that the population may be exceeding carrying capacity of the preferred habitat and that activity in areas with non-native vegetation represents a "spilling over" into suboptimal habitat.

The search-phase calls produced by *E. s. rotensis* were characterized by a relatively narrow bandwidth and short pulse duration typical of insectivores which forage close to and among vegetative clutter (Neuweiler 1989, Jung and others 2007). These call attributes are similar to other emballonurids that forage near vegetation such as *Rhynchonycteris naso* and *Balantiopteryx* spp. (O'Farrell and Miller 1997, Ibáñez and others 2002). This inference is supported from our direct observations and those by Esselstyn and others (2004) of bats flying slowly and "erratically" while foraging within forest between 1 m of the ground and up to tree-tops. Esselstyn and others (2004) also observed bats foraging above the forest canopy. Use of open areas near forest edges also was confirmed by our echolocation recordings.

The metrics generated by this study can serve as a quantitative baseline for future assessments of status following changes in habitat due to management activities (e.g., feral goat control) or other factors (e.g., typhoon impacts). For instance, our sites can be re-sampled and analyzed with multi-season models (MacKenzie and others 2002) to determine whether the proportion of occupied sites that exhibit peak activity have decreased or increased following loss or recovery of native limestone forest habitat. We also found the use of nested activity thresholds for quantifying peak bat activity to be useful in identifying high occupancy locations and making inferences about important habitat attributes.

Although relative variance (as measured by CV) of $\hat{\Psi}$ and *p* was generally greater at higher activity thresholds, standard errors were similar across thresholds (e.g., $\hat{\Psi}$ $(\overline{\mathrm{SE}})$ for all null models was about 0.08; Table IV-4). This means that future occupancy surveys may focus on sites with high expected activity. These "core" sites are generally in tall stature native limestone forest and are more easily traversed and sampled than the dense thickets of exotic shrub (primarily *Lantana camara*) that comprise about one-half the island's landcover.

The current study was designed to randomly resurvey as many as possible of the 50 sites established on a systematic grid by Esselstyn and others (2004). However, not all locations were accessible (because of impenetrable *Lantana camara* thickets and the brief 19-day period available to us), and the 31 sites actually surveyed do not represent a fully random subset of the original 50 locations. Departure from a probabilistic sampling design may bias $\hat{\Psi}$ and *p* (upward in our case since proportionally fewer sites in non-forest were available to us than initially available to Esselstyn and others 2004). Therefore, future surveys of Pacific sheath-tailed bat occupancy on Aguiguan should seek to fully sample the grid of 50 sites if island-wide characterization of habitat use is a primary objective. However, if monitoring bat activity in preferred habitat is the main objective, the $\hat{\Psi}$ and *p* variances produced by this study may be used as a guide to generate a revised sampling design following the methods presented by Bailey and others (2007). For example, a design comprised of 30 sites sampled for 4 nights (or alternatively, 25 sites for 5 nights) is needed to attain a desired CV of $\leq$0.05 given an expected Ψ of 0.93 and $p = 0.78$ (as observed for the median activity threshold in mature native limestone forest near known roosts; Tables IV-3 and IV-4).

The importance of native limestone forest to the persistence of *E. s. rotensis* on Aguiguan cannot be over-emphasized. Bat species specialized to forage near or within forest on average face a greater extinction risk than aerial insectivores or species with comparatively flexible foraging strategies (Kingston and others 2003, Safi and Kerth 2004, Lane and others 2006). Moreover, minimum area thresholds of species occurrence indicate that island occupancy by insectivorous bats may be strongly limited by resource requirements (Frick and others 2008). Given the island's very limited resource base and size, the extreme isolation of the population, its vulnerability to typhoons (e.g., Grant and others 1994), and the species' relatively narrow habitat preference and specialized foraging strategy, it is imperative that efforts to reverse the decline in native limestone forest on Aguiguan be implemented to ensure the long-term survival of the Pacific sheath-tailed bat.

ACKNOWLEDGEMENTS

Funding was provided by the U.S. Fish and Wildlife Service Ecological Services office in Honolulu, Hawai'i. We are indebted to C. Kessler and J. Salas for logistical support and to D. Evangelista, E. Masga and F. Muña for field assistance. We thank T. O'Shea, T. Rodhouse and G. Wiles for manuscript review. Any use of trade, product, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the U.S. Government.

REFERENCES CITED

Anonymous, 1991, Threatened or endangered species: Commonwealth of the Northern Mariana Islands, Commonwealth Register, vol. 13, p. 7537.

Bailey, L. L., Hines, J. E., Nichols, J. D., and MacKenzie, D. I., 2007, Sampling design trade-offs in occupancy studies with imperfect detection: examples and software: Ecological Applications, vol. 17, p. 281-290.

Burnham, K. P., and Anderson, D. R., 2002, Model selection and multimodel inference: a practical information-theoretic approach, second edition: New York, U.S.A., Springer-Verlag.

Chiroptera Specialist Group 2000. Emballonura semicaudata. In: IUCN 2007. 2007 IUCN Red List of Threatened Species. Downloaded on 02 September 2008 from http://www.iucnredlist.org.

Donovan, T. M., and Hines, J., 2007, Exercises in occupancy modeling and estimation. Downloaded on 04 September 2008 from http://www.uvm.edu/envnr/vtcfwru/spreadsheets/occupancy/occupancy.htm.

Ellison, L. E., O'Shea, T. J., Bogan, M. A., Everette, A. L., and Schneider, D. M., 2003. Existing data on colonies of bats in the United States: summary and analysis of the U.S. Geological Survey's bat population database, *in* O'Shea, T.J. , and Bogan, M. A., eds., Monitoring trends in bat populations of the United States and Territories: problems and prospects: Washington, D. C., U.S. Geological Survey Information and Technology Report USGS/BRD/ITR-20030-0003, p. 127-237.

Engbring, J., Ramsey, F. L., and Wildman, V. J., 1986, Micronesian forest bird survey, 1982: Saipan, Tinian, Agiguan, and Rota: Honolulu, Hawaii, U. S. Fish and Wildlife Service.

ESRI, 2006, ESRI ArcMap 9.2: Redlands, Calif.

Esselstyn, J. A., Wiles, G. J., and Amar, A., 2004, Habitat use of the Pacific sheath-tailed bat (*Emballonura semicaudata*) on Aguiguan, Mariana Islands: Acta Chiropterologica, vol. 6, p. 303-308.

Fenton, M. B., Rydell, J., Vonhof, M. J., Eklöf, J., and Lancaster, W. C., 1999, Constant-frequency and frequency-modulated components in the echolocation calls of three species of small bats (Emballonuridae, Thyropteridae, and Vespertilionidae): Canadian Journal of Zoology, vol. 77, p. 1891-1900.

Flannery, T., 1995, Mammals of the South-West Pacific & Moluccan Islands: Ithaca, NY, Cornell University Press.

Frick, W. E., Hayes, J. P., and Heady, P. A., 2008, Patterns of island occupancy in bats: influences of area and isolation on insular incidence of volant mammals: Global Ecology and Biogeography, vol. 17, p. 622-632.

Gannon, W. L., O'Farrell, M. J., Corben, C., and Bedrick, E. J., 2004, Call character lexicon and analysis of field recorded bat echolocation calls, *in* Thomas, J. A. , Moss, C. F., and Vater, M. , eds., Echolocation of bats and dolphins: Chicago, Illinois, University of Chicago Press, p. 478-484.

Gorresen, P. M., Miles, A. C., Todd, C. M. , Bonaccorso, F. J., and Weller, T. J., 2008, Assessing bat detectability and occupancy with multiple automated echolocation detectors: Journal of Mammalogy, vol. 89, p. 11-17.

Grant, G. S., Banack, S. A., and Trail, P., 1994, Decline of the sheath-tailed bat *Emballonura semicaudata* (Chiroptera: Emballonuridae) on American Samoa: Micronesica, vol. 27, p. 133-137.

Helgen, K. M., and Flannery, T. F., 2002, Distribution of the endangered Pacific sheath-tailed bat *Emballonura semicaudata*: Australian Mammalogy, vol. 24, 209-212.

Hutson, A. M., Mickleburgh, S. P., and Racey, P. A. (compilers), 2001, Microchiropteran bats: global status survey and conservation action plan. IUCN/SSC Chiroptera Specialist Group, International Union for the Conservation of Nature and Natural Resources, Gland, Switzerland and Cambridge, United Kingdom.

Ibáñez, C., Juste, J., López-Wilchis, R., Albuja V., L., and Núñez-Garduño, A., 2002, Echolocation of three species of sac-winged bats (*Balantiopteryx*): Journal of Mammalogy, vol. 83, p. 1049-1057.

Jung, K., Kalko, E. K. V., von Helversen, O., 2007, Echolocation calls in Central American emballonurid bats: signal design and call frequency alternation: Journal of Zoology, vol. 272, p. 125-137.

Kingston, T., C. M. Francis, Z. Akbar, and T. H. Kunz. 2003. Species richness in an insectivorous bat assemblage from Malaysia. Journal of Tropical Ecology 19:67-79.

Koopman, K. F. 1997. The subspecies of *Emballonura semicaudata* (Chiroptera: Emballonuridae). Journal of Mammalogy 78:358-360.

Lane, D. J. W., Kingston, T., and Lee, B. P. Y.-H., 2006, Dramatic decline in bat species richness in Singapore, with implications for Southeast Asia: Biological Conservation, vol. 131, p. 584-593.

Lawrence, B. D., and Simmons, J. A., 1982, Measurements of atmospheric attenuation at ultrasonic frequencies and the significance for echolocation by bats: Journal of the Acoustical Society of America, vol. 71, p. 585–590.

Lemke, T. O., 1986, Distribution and status of the sheath-tailed bat (*Emballonura semicaudata*) in the Mariana Islands: Journal of Mammalogy, vol. 67, p. 743-746.

MacKenzie, D. I., and Bailey, L. L., 2004, Assessing the fit of site-occupancy models: Journal of Agricultural, Biological, and Environmental Statistics, vol. 9, p. 300-318.

MacKenzie, D. I., Nichols, J. D. , Lachman, G. B., Droege, S., Royle, J. A., and Langtimm, C.A., 2002, Estimating site occupancy when detection probabilities are less than one: Ecology, vol. 83, p. 2248–2255.

Neuweiler, G., 1989, Foraging ecology and audition in echolocating bats: Trends in Ecology and Evolution, vol. 4, p. 160-166.

O'Farrell, M. J., and Miller, B. W., 1997, A new examination of echolocation calls of some Neotropical bats (Emballonuridae and Mormoopidae): Journal of Mammalogy, vol. 78, p. 954-963.

O’Shea, T. J., and Bogan, M. A., eds., 2003, Monitoring trends in bat populations of the United States and territories: problems and prospects: U.S. Geological Survey, Biological Resources Discipline, Information and Technology Report, USGS/BRD/ITR--2003-003, 274 p.

Safi, K., and Kerth, G., 2004, A comparative analysis of specialization and extinction risk in temperate-zone bats: Conservation Biology, vol. 18, p. 1293-1303.

U.S. Fish and Wildlife Service, 2007, 50 CFR Part 17; Endangered and Threatened Wildlife and Plants; Review of Native Species That Are Candidates for Listing as Endangered or Threatened; Annual Notice of Findings on Resubmitted Petitions; Annual Description of Progress on Listing Actions; Proposed Rule. Federal Register (December 6, 2007), vol. 72 (no. 234), p. 69033-69106.

Wiles, G. J., Engbring, J., and Otobed, D., 1997, Abundance, biology, and human exploitation of bats in the Palau Islands: Journal of Zoology (London), vol. 241, p. 203-227.

Table IV-1. Characteristics of "low" and "high" harmonics in the search-phase calls of *E. s. rotensis*. Variables measured include minimum (F_{min}) and maximum (F_{max}) frequencies (kHz); frequency range (F_{max} - F_{min}); characteristic (F_c) frequency (kHz); duration of entire pulse (ms); and time (T_c) from the start of pulse to F_c (ms). The number of call files and pulses examined were 33 and 245 for low harmonics and 45 and 509 for high harmonics, respectively.

Harmonic	Parameter	F_{min}	F_{max}	Range	F_c	Duration	Tc
"Low"	Minimum	26.85	31.01	0.24	27.03	0.49	0.49
	Maximum	32.79	33.33	5.39	33.06	3.11	2.36
	Mean	30.43	32.26	1.83	30.97	1.44	1.31
	SD	1.13	0.64	1.10	1.08	0.47	0.37
"High"	Minimum	40.20	58.82	1.43	50.63	0.26	0.28
	Maximum	64.00	67.23	23.29	66.12	4.07	3.80
	mean	53.33	64.37	11.04	63.15	2.75	1.73
	SD	4.12	1.22	4.05	2.20	0.56	0.46

Table IV-2. Summary of model selection for 4 activity threshold levels: none, and 50th, 70th and 90th percentiles of bat call pulse number. ΔAIC is the relative difference in AIC values from the model with the smallest AIC value; *w* is the AIC model weight; *k* is the number of parameters; X^2 is the test statistic for model fit; *p*-value is the probability of observing a test statistic ≥ X^2 based upon 999 parametric bootstraps; and ĉ is the estimated over-dispersion parameter. Estimated occupancy is presented in Table IV-3 only for models (indicated below in bold) for which ΔAIC ≤ 2 and model goodness-of-fit is adequate (*p*-value ≥ 0.05 and ĉ ranging between 0.5 and 1.5).

Threshold	**Model**	**ΔAIC**	***w***	***k***	**X^2**	***p*-value**	**ĉ**
none	canopy	0.00	0.82	4	147.2	0.024	2.62
none	distance	4.60	0.08	3	153.0	0.023	2.70
none	stature	5.66	0.05	4	171.0	0.022	3.08
none	null	6.93	0.03	2	172.8	0.030	2.94
none	native-exotic	8.07	0.01	3	181.3	0.018	3.21
none	understory	8.93	0.01	3	172.9	0.024	3.11
none	stem diameter	9.87	0.01	4	181.5	0.018	3.22
50	**canopy & distance**	0.00	0.45	5	55.5	0.357	1.01
50	**canopy**	1.02	0.27	4	50.2	0.386	0.96
50	**distance**	1.58	0.21	3	50.8	0.403	0.92
50	stature	5.64	0.03	4	50.4	0.391	0.89
50	null	6.78	0.02	3	48.4	0.435	0.85
50	native-exotic	7.40	0.01	4	48.5	0.398	0.86
50	stem diameter	7.70	0.01	3	48.9	0.370	0.90
50	understory	7.83	0.01	3	48.5	0.406	0.88
70	**canopy & stature**	0.00	0.35	6	40.4	0.705	0.74
70	**canopy & distance**	1.64	0.15	5	40.9	0.717	0.73
70	**canopy**	1.83	0.14	4	38.5	0.765	0.71
70	**distance**	1.85	0.14	3	38.6	0.761	0.69
70	distance & stature	3.07	0.08	5	39.3	0.720	0.71
70	stature	3.48	0.06	4	39.0	0.712	0.71
70	null	5.02	0.03	2	38.2	0.718	0.69
70	stem diameter	5.84	0.02	4	38.6	0.717	0.68
70	native-exotic	6.09	0.02	3	38.5	0.698	0.70
70	understory	7.02	0.01	3	38.1	0.735	0.69
90	**distance & stature**	0.00	0.66	5	82.8	0.099	1.52
90	stem diameter [1]	3.01	0.15	4	121.0	0.012	2.24
90	distance	4.56	0.07	3	68.8	0.238	1.25
90	stature	4.75	0.06	4	95.0	0.064	1.74
90	stem diam. & stature	4.93	0.06	6	111.1	0.014	2.05
90	native-exotic	11.87	0.00	3	132.7	0.027	2.37
90	understory	11.99	0.00	3	76.6	0.193	1.40
90	canopy	13.13	0.00	4	81.3	0.148	1.46
90	null	14.82	0.00	2	94.5	0.118	1.69

[1] Model “stem diameter & distance” failed to convergence and was excluded from output.

Table IV-3. Occupancy estimates ($\hat{\Psi}$) for existing combinations of variables. Results are shown only for top-ranked models (indicated in bold in Table IV-2) for each of 3 activity threshold levels: 50th, 70th, and 90th percentiles of bat call pulse number (the "no threshold" models did not meet goodness-of-fit criteria and are not presented). Estimates for the continuous variable "distance" are derived from mid-range examples of post hoc categories "near", "middle", and "far". Results are sorted by ascending occupancy estimate within each threshold and model.

Threshold	Model	$\hat{\Psi}$	SE	95% CI
50	canopy "middle", distance "far"	0.092	0.092	-0.088 – 0.272
50	canopy "middle", distance "middle"	0.323	0.158	0.012 – 0.633
50	canopy "low", distance "far"	0.422	0.349	-0.263 – 1.106
50	canopy "high", distance "far"	0.559	0.243	0.084 – 1.035
50	canopy "middle", distance "near"	0.562	0.239	0.093 – 1.031
50	canopy "low", distance "near"	0.746	0.257	0.242 – 1.249
50	canopy "high", distance "middle"	0.817	0.103	0.615 – 1.020
50	canopy "high", distance "near"	0.928	0.064	0.803 – 1.053
50	canopy "middle"	0.274	0.135	0.010 – 0.539
50	canopy "low"	0.667	0.272	0.133 – 1.201
50	canopy "high"	0.839	0.094	0.655 – 1.023
50	distance "far"	0.196	0.140	-0.079 – 0.471
50	distance "middle"	0.633	0.099	0.440 – 0.827
50	distance "near"	0.869	0.087	0.698 – 1.041
70	canopy "middle", stature "middle"	0.112	0.107	-0.097 – 0.321
70	canopy "low", stature "middle"	0.582	0.380	-0.163 – 1.327
70	canopy "high", stature "middle"	0.681	0.150	0.387 – 0.974
70	canopy "middle", stature "tall"	0.712	0.247	0.229 – 1.194
70	canopy "high", stature "short"	0.791	0.241	0.319 – 1.263
70	canopy "high", stature "tall"	0.977	0.036	0.906 – 1.047
70	canopy "middle", distance "far"	0.115	0.107	-0.095 – 0.326
70	canopy "middle", distance "middle"	0.322	0.157	0.014 – 0.629
70	canopy "low", distance "far"	0.463	0.351	-0.224 – 1.150
70	canopy "high", distance "far"	0.466	0.275	-0.073 – 1.005
70	canopy "middle", distance "near"	0.521	0.242	0.046 – 0.995
70	canopy "low", distance "near"	0.734	0.260	0.224 – 1.244
70	canopy "high", distance "middle"	0.750	0.121	0.513 – 0.986
70	canopy "high", distance "near"	0.884	0.087	0.713 – 1.055
70	canopy "middle"	0.277	0.137	0.010 – 0.545
70	canopy "low"	0.668	0.273	0.134 – 1.203
70	canopy "high"	0.792	0.107	0.583 – 1.001
70	distance "far"	0.210	0.147	-0.078 – 0.498
70	distance "middle"	0.506	0.109	0.293 – 0.720
70	distance "near"	0.832	0.103	0.629 – 1.034
90	distance "far", stature "short"	0.000	0.000	0.000 – 0.000
90	distance "near", stature "short"	0.000	0.000	0.000 – 0.000
90	distance "far", stature "middle"	0.006	0.014	-0.021 – 0.034
90	distance "middle", stature "middle"	0.060	0.067	-0.070 – 0.191

90	distance “middle”, stature “tall”	0.554	0.363	-0.157 – 1.265
90	distance “near”, stature “tall”	0.959	0.064	0.834 – 1.083

Table IV-4. Occupancy ($\hat{\Psi}$) and detection probability (*p*) estimates for null and top-ranked models (indicated in bold in Table IV-2) for each of 4 activity threshold levels: “none”, and 50th, 70th, and 90th percentiles of bat call pulse number. “Null” indicates models with no habitat covariates included. “Obs Ψ ” is the observed or “naïve” estimate for occupancy (i.e., not adjusted for detection probability). Mean, standard error and associated coefficient of variation (CV) were derived from parameter estimates for all sampled sites.

Threshold	**Model**	**Obs** Ψ	$\bar{\hat{\Psi}}$ ($\overline{\text{SE}}$)	**CV**	***p*** **(SE)**	**CV**
none	null	0.71	0.72 (0.08)	0.11	0.76 (0.05)	0.07
50	null	0.61	0.62 (0.09)	0.14	0.78 (0.05)	0.07
50	canopy & distance	0.61	0.63 (0.14)	0.22	0.78 (0.05)	0.07
50	canopy	0.61	0.57 (0.14)	0.24	0.78 (0.05)	0.07
50	distance	0.61	0.63 (0.11)	0.17	0.78 (0.05)	0.07
70	null	0.58	0.59 (0.09)	0.15	0.70 (0.06)	0.09
70	canopy & stature	0.58	0.58 (0.16)	0.27	0.70 (0.06)	0.09
70	canopy & distance	0.58	0.60 (0.15)	0.25	0.70 (0.06)	0.09
70	canopy	0.58	0.56 (0.15)	0.26	0.69 (0.06)	0.09
70	distance	0.58	0.60 (0.11)	0.19	0.70 (0.06)	0.09
90	null	0.26	0.27 (0.08)	0.31	0.59 (0.10)	0.17
90	distance & stature	0.26	0.27 (0.08)	0.32	0.61 (0.09)	0.16

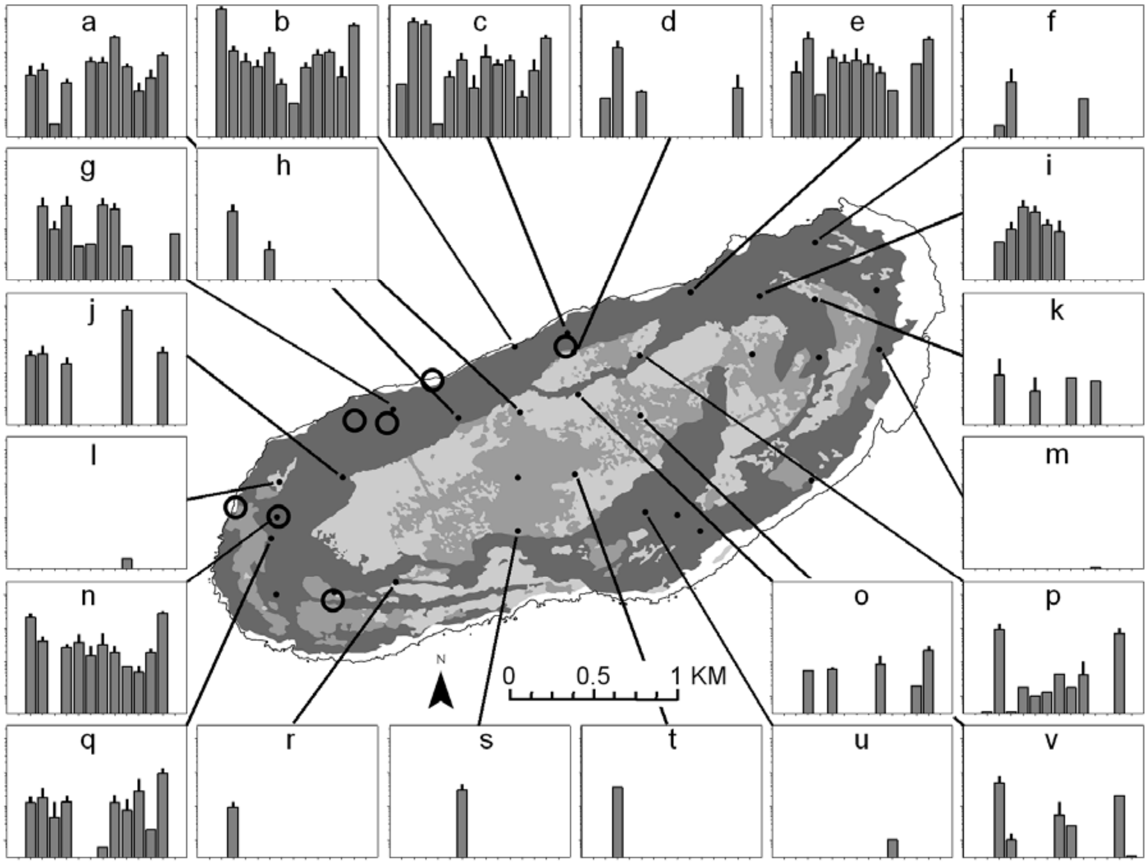


Figure IV-1. Distribution of survey stations and *E. s. rotensis* activity by hour of night for all sites with detections on Aguiguan. X axis on each inset bar graph is hour of night beginning at 1700 and ending at 0600. Y axis is natural log of mean pulse total (+ 1 SD; major tick marks range from 10^0 to 10^3). Bars with no SD whisker had only a single observation in the hour. Graph axes are shown in detail for each site (labeled a-v) in Appendix IV-1. Open circles on background image indicate known roost locations and points designate sample sites. Landcover classes shown include native limestone forest (dark gray), non-native forest (mid-tone gray), non-native shrubland (light gray), and coastal scrub, grass and unvegetated areas (white). Landcover map courtesy of Fred Amidon (USFWS, *in litt.*).

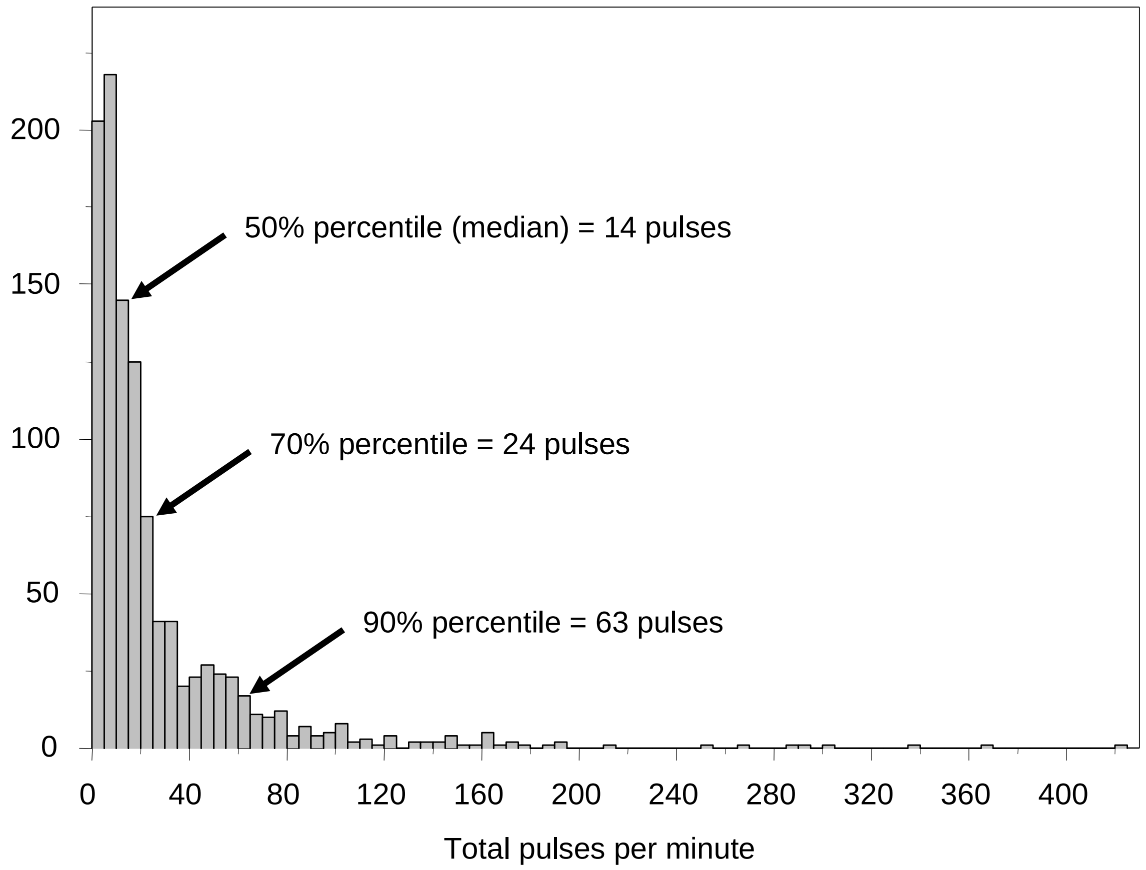


Figure IV-2. Distribution of total bat call pulses per 1-minute interval (*n* = 1,224). One-half of all bat detections consisted of brief passes with less than 15 pulses. Higher thresholds of activity indicative of sustained bouts of foraging were defined with 50th, 70th and 90th percentiles of pulse number.

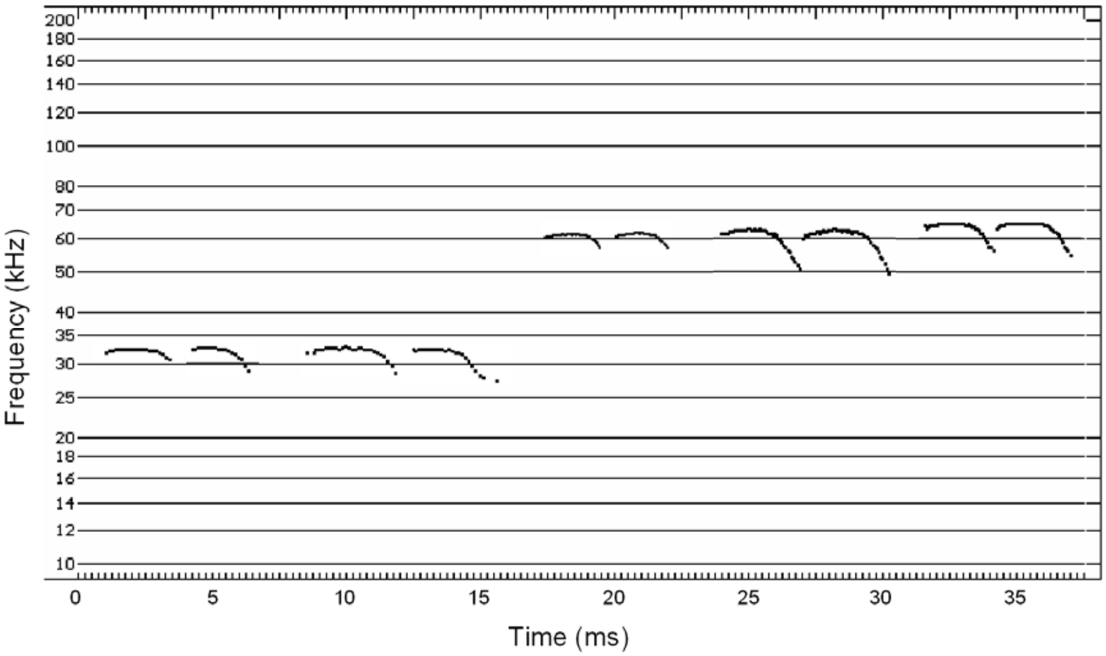
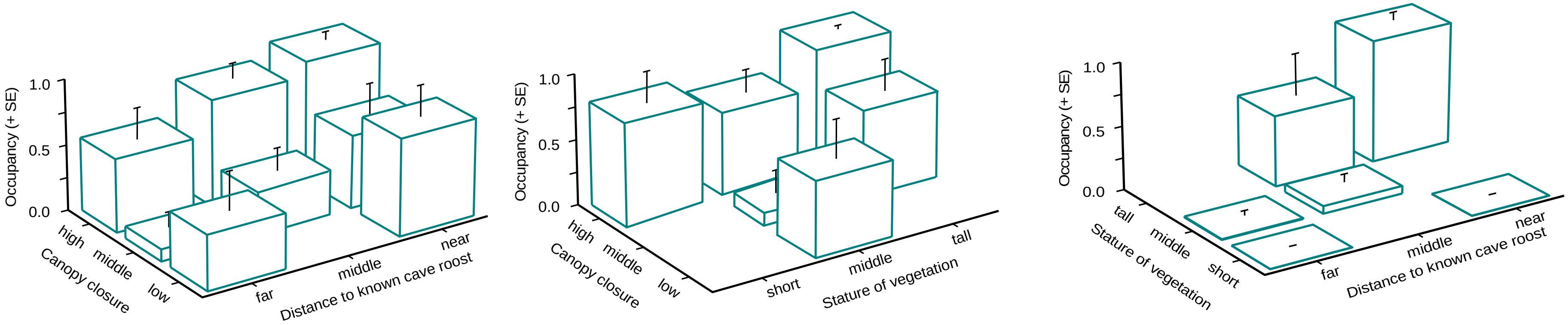


Figure IV-3. Examples of pairs of characteristic pulses in the search-phase calls of *E. s. rotensis*. Note that paired examples were derived from different Anabat call files, and the time between pulses was compressed to permit display of various pulses.



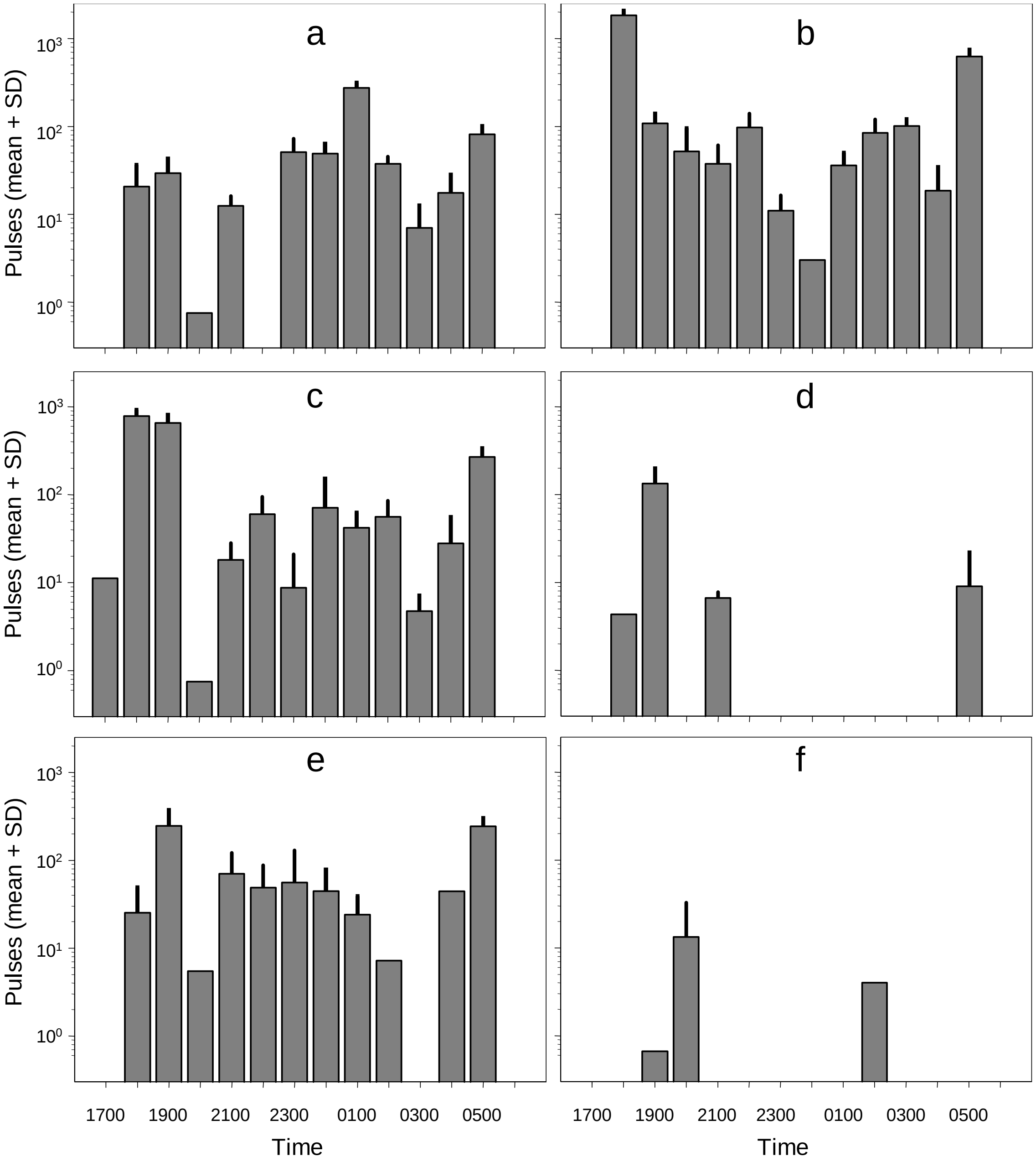
50th percentile threshold

70th percentile threshold

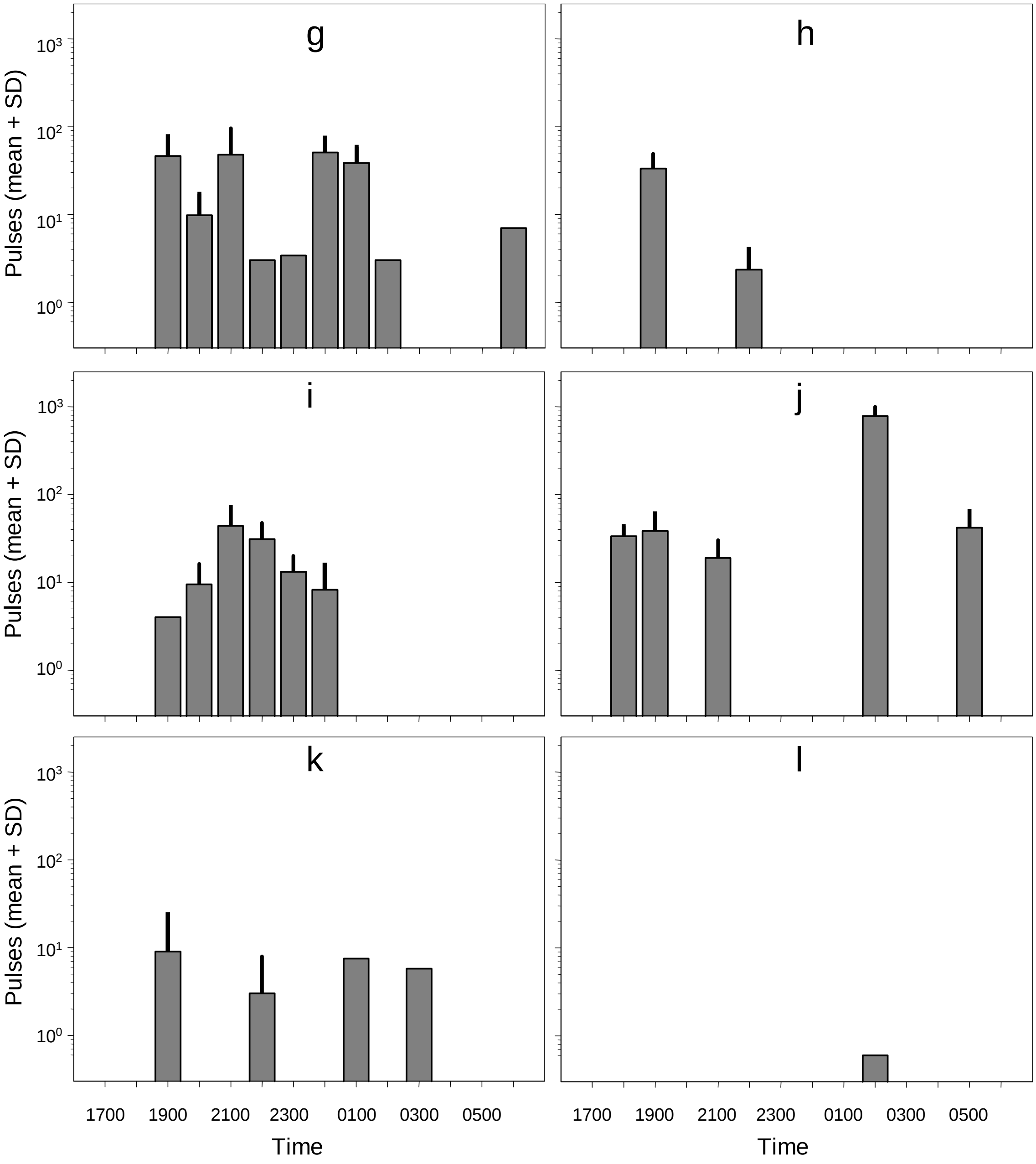
90th percentile threshold

Figure IV-4. Occupancy estimates (mean ± 1 SE) for existing combinations of habitat covariates at the 50th, 70th and 90th percentile thresholds of activity (results shown only for models with lowest AIC values and adequate goodness-of-fit).

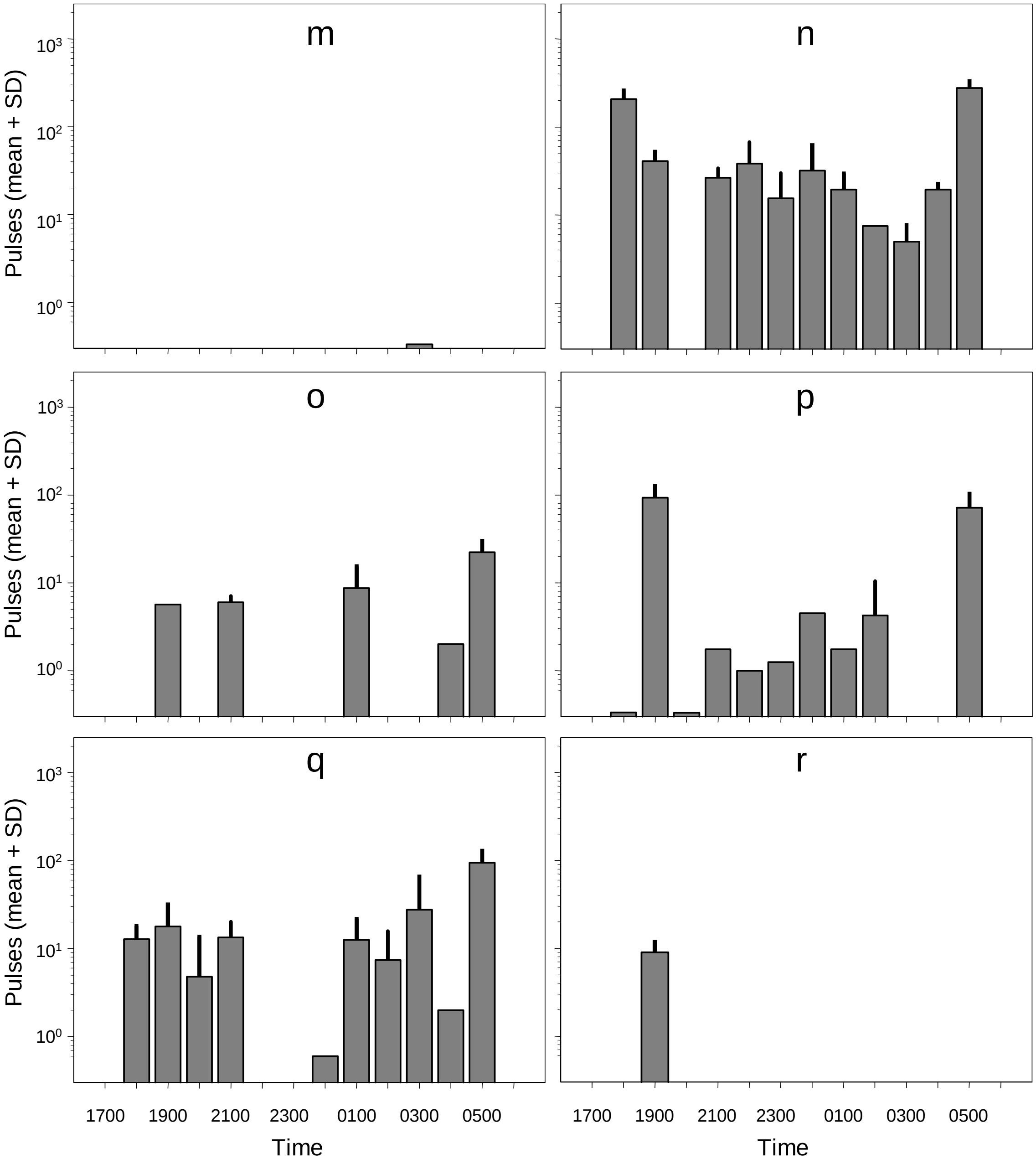
Draft



Appendix IV-1. *E. s. rotensis* activity by hour of night for all sites with detections on Aguiguan. X axis on each inset bar graph is hour of night beginning at 1700 and ending at 0600. Y axis is natural log of mean pulse total (+ 1 SD). Bars with no SD whisker had only a single observation in the hour. Graphs are shown as insets for each site (labeled a-v) in Figure IV-1.

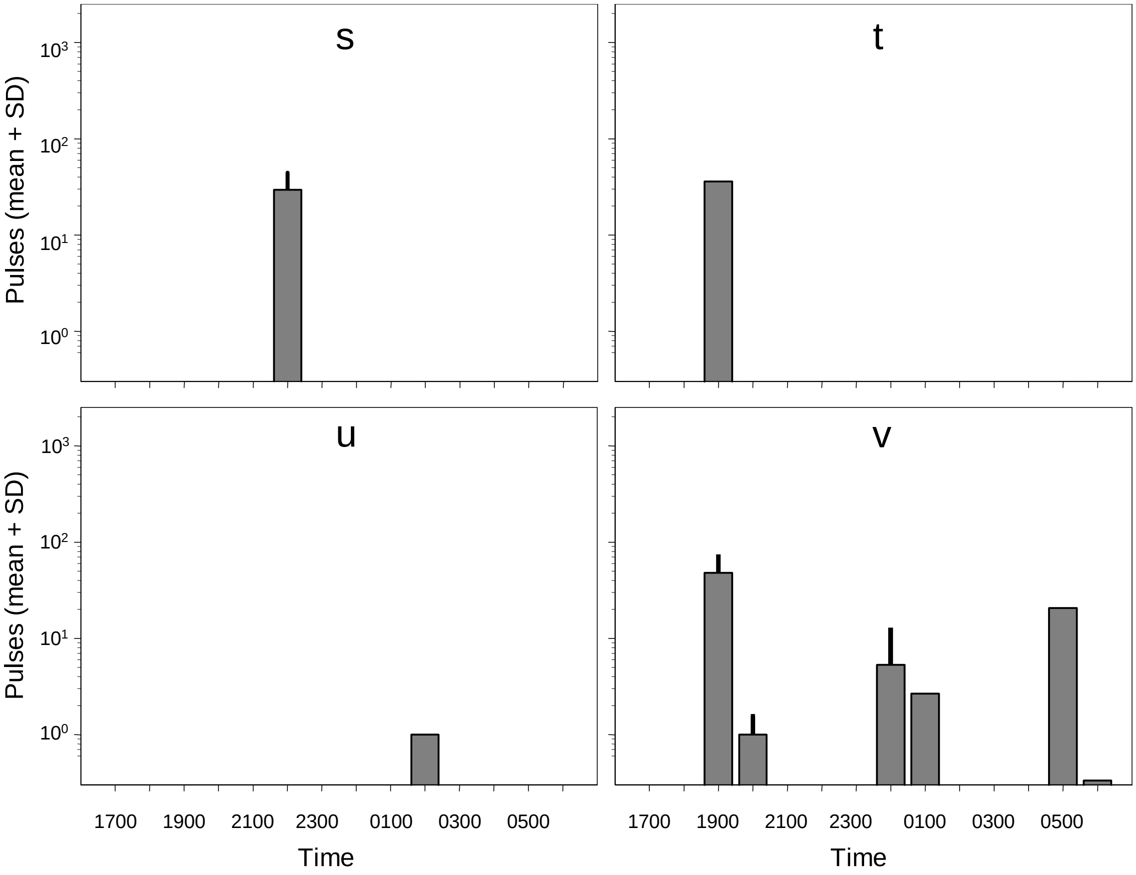


Appendix IV-1 continued.



Appendix IV- 1 continued.

Draft



Appendix IV-1 continued.

Section V. Food Habits of the Pacific Sheath-Tailed Bat (*Emballonura semicaudata rotensis*) on Aguiguan, Commonwealth of the Northern Mariana Islands

Ernest W. Valdez

ABSTRACT

*Emballonura semicaudata rotensis*is is an endemic subspecies of the Pacific sheath-tailed bat known only from the Mariana Islands. It is extinct on all of the Mariana Islands where it once occurred except for the small limestone island of Aguiguan that supports an isolated remnant population. There is no information about the food habits of Pacific sheath-tailed bats. In 2008, I conducted an investigation of food habits of *E. s. rotensis* based on analysis of fresh fecal pellets from bats roosting in Guano and Crevice Caves on Aguiguan. I analyzed 100 pellets from each roost and found that major orders of insects consumed by *E. s. rotensis* from Guano Cave (based on % volume) included hymenopterans (64%), coleopterans (10%), lepidopterans (8%), isopterans (8%), and psocopterans (5%). Major prey items of bats from Crevice Cave included lepidopterans (45%), hymenopterans (41%), coleopterans (10%), and isopterans (5%). Most of the identified hymenopterans found in the guano from both roosts belonged to ichneumondoidea, followed by prey items belonging to formicinae and ponerinae. Because alates of formicines and ponerines, as well as isopterans, generally have wings only when they are reproductive or establishing new colonies, often at the onset of rains, it is likely that these food items occur in the diet of *E. s. rotensis* seasonally. Microlepidopterans were another likely seasonally abundant prey item consumed by *E. s. rotensis* include were lepidopterans. Beetles (Coleoptera) that were likely forest-dependent species were a consistent component of the diet. Most insect prey items were small ranging from 1.7 to 6.4 mm in length. From observations and diet analyses, *E. s. rotensis* can be categorized as an aerial insectivore or hawker, similar to other emballonurids around the world.

INTRODUCTION

Emballonura semicaudata rotensis, the subspecies of the Pacific sheath-tailed bat unique to the Marianas Islands, is an insectivorous microchiropteran that occurred historically on multiple islands including Guam (Lemke 1986, Koopman 1997). This bat is now extinct throughout its range except for the small (7.2 km^2) uninhabited island of Aguiguan in the Commonwealth of the Northern Mariana Islands (CNMI, Hutson and others 2001). At present, this bat is listed as a Category 3 candidate for listing under the U.S. Endangered Species Act (U.S. Fish and Wildlife Service 2007) and is categorized as Endangered by the International Union for the Conservation of Nature and Natural Resources (Bonnacorso and Allison 2008, Hutson and others 2001). Exact reasons for its decline are unknown, but it is has been suggested that *E. s. rotensis* had succumbed to a series of events during a short period of time that eventually led to its demise throughout most of its historic range (Hutson and others 2001, U.S. Fish and Wildlife Service 2007, 2008, Bonaccorso and Allison 2008). Some of these events include:

disturbance of caves (especially during military operations in World War II), the only structures known to be used as roosts by these colonial bats in the CNMI (see Section III of this administrative report); loss or destruction of forest habitat used for by foraging bats during World War II and subsequent clearing for development; use of insecticides; typhoons; and perhaps invasive predators on some islands. All these factors are cited as possibly directly or indirectly impacting populations of this subspecies (Hutson and others 2001, Bonaccorso and Allison 2008, Cruz and others 2008). Although the use of insecticides and loss of forest habitat have been suggested as possible causes of declines, there is no baseline information about the kinds of insects that these bats eat. Understanding food requirements is an important componet for conservation and management of wildlife. Therefore, my objective was to determine what types of prey items are used to meet the dietary needs of Pacific sheath-tailed bats on Aguiguan.

MATERIALS AND METHODS

The food habits of insectivorous bats are typically determined by identification of chitinous fragments of insects in guano (Whitaker 1988). These bats masticate insects into small fragments that are best identified in guano rather than in stomach contents because the latter contain larger amounts of unidentifiable digestible material and require sacrificing individuals to obtain ingesta. Guano can be sampled non-invasively, which is of importance in studies of bats that are of conservation concern. On Aguiguan I sampled guano beneath daytime roosts of Pacific sheath-tailed bats at two caves. Guano Cave (Datum: WGS 84; 55P, 343039E, 1642089N, elev. 100m), the larger of the two roosts, houses one of the larger colonies of sheath-tailed bats on Aguiguan, with 43-64 bats counted in the roost in June and July of 2008 (see Section III of this administrative report). This cave is also shared by a colony of about 200 or more Mariana swiftlets (*Aerodramus bartschi*). Roosting bats were positioned about 15-20 m above the cave floor in a distinctive domed ceiling at the end of the deepest chamber of the cave, whereas most of the swiftlet nests were located closer to the outside opening on the inside wall of the same chamber, about 3-15 m above the cave floor. Most of the guano on the floor of the cave had accumulated over many years and had disintegrated into a fine powder, making it difficult to distinguish bat guano from swiftlet guano. To obtain comparative material, on 25 June 2008 I placed a 1 m x 1 m plastic sheet on the cave floor directly beneath the roosting bats and a similar sheet beneath the area where swiftlets roosted. On 30 June 2008, the sheets were retrieved from the roost and placed in plastic re-sealable bags.

We also sampled guano at a second roost that was not used by swiftlets. On 27 June 2008 I placed a 0.5 m x 0.5 m plastic sheet 1 m beneath a small group of bats (2-3 adults, 1 pup) in Crevice Cave (Datum: WGS 84; 55P, 343541E, 1641526N, elev. 72m). This roost can be described as a chute, chimney, or vertical solution tube in the back portion of the main cave. Because I observed only 3 bats using this roost, I left the plastic sheet under the bats for a longer period of time (13 days) to allow for a larger accumulation of guano.

During the period of guano sampling I also collected arthropods (mostly insects) as a reference collection for fecal analyses. Insects were collected mostly at night following techniques described by Kunz (1988), including use of a sweep net, beating of vegetation, and setting out a black light. I also attempted to collect insects using sticky-traps made of 76.2mm x 127mm index cards coated with an insect barrier (Tanglefoot Tree Pest Barrier ®). I attached these traps to twine and hung them vertically from the canopy, but abandoned this method after rain disintegrated the cards. Arthropods were placed in vials of 95% ethanol and identified at the Museum of Southwestern Biology, University of New Mexico, Albuquerque.

In the field I examined fecal material of bats and swiftlets to ascertain if these could be readily distinguished in the fresh samples. Intact fecal pellets produced by *E. s. rotensis* were elliptical and averaged about 4 mm long by 2 mm wide. Intact guano produced by swiftlets was globular, as observed for other small insectivorous birds, but varied in size. Swiftlet guano also could be differentiated by uric acid crystals combined with digested insects; uric acid crystals were not present in bat guano. Finally, microscopic inspection affirmed these gross differences: insect matter consumed by bats was always chewed into much smaller fragments than those found in swiftlet fecal matter. Using the aforementioned criteria for distinguishing bird and bat guano, I sorted formed guano pellets of *E. s. rotensis* from powdered guano, then grouped fecal material according to roost. Pellets were randomly sub-sampled and analyzed following techniques described by Whitaker (1988). Fecal pellets were placed in watch glasses with 95% ethanol and teased apart under a stereo-zoom microscope. Insect prey found in the guano were identified to the lowest taxonomic level, usually to family, using Chujo (1970), Borror and White (1970), White (1983), Whitaker (1988), Arnett (2000), Arnett and Thomas (2001), Arnett and others (2002), and Triplehorn and Johnson (2005) as guides for identification.

A single pellet represented one sample, and a total of 200 intact fecal pellets (100 from each roost) were analyzed. Percent volume and frequency were calculated for each prey item (Whitaker 1988). In addition to fecal analyses, I used digital calipers (Mitutoyo ®) to measure length x width, in mm to the nearest 0.01, of a single representative from my reference collection of arthropods that were similar in appearance to matched prey items found in the diet of *E. s. rotensis*.

RESULTS

The major food items consumed by *E. s. rotensis* from Guano Cave were hymenopterans at 64% volume and 95% occurrence (Table V-1). Prey items belonging to Ichneumonoidea (parasitic wasps) had the greatest percent volume (25%) and percent occurrence (45%) among identified hymenopterans. Prey items in the Formicidae (ants) were also identified within the Hymenoptera, including ants belonging to Formicinae (7%, 12%) and Ponerinae (2%, 12%, Table V-1). I identified the ponerines (i.e., trap-jaw ants) by the distinct shape of their mandibles, antennae, and fragments of head capsules, and believe that the individuals consumed by *E. s. rotensis* belong to the genus *Anochetus*. Other key prey items found in the guano of *E. s. rotensis* from Guano Cave included coleopterans (beetles) at 10% volume and 73% occurrence, followed by microlepidopterans (moths, 8%, 38%), isopterans (termites, 8%, 10%), and psocopterans (bark lice, 5%, 26%, Table V-1). All other identified prey items had values of 1% volume or less and did not occur frequently (Table V-1).

Pacific sheath-tailed bat fecal samples from Crevice Cave primarily contained microlepidopterans (45% volume, 86%occurrence) and hymenopterans (41%, 82%; Table VI-1). Ichneumonoids had the greatest percent volume (31%) and percent occurrence (46%) values among identified hymenopterans. Formicines contributed to 7% of the volume and were encountered in 26% of the samples (Table V-1). Coleopterans contributed to 10% of the volume consumed by the bats at Crevice Cave and were encountered in 68% of the samples examined (Table V-1). Within Coleoptera, beetles belonging to the Cryptophagidae (silken fungus beetles) accounted for 3% of the volume and were encountered in 18% percent of the samples (Table V-1). Isopterans contributed to 5% of the volume and were occurred in 6% of the samples from Crevice Cave (TableV-1). All other identified prey items had values less than 1% for percent volume and 5% or less for percent occurrence.

Measurements of arthropods matched with those consumed by Pacific sheath-tailed bats ranged in size from the smallest, a scolytine at 1.72 mm x 0.85 mm to the largest, a ponerine, at 7.6 mm x 1.63 mm (Table V-2). Isopterans were the next largest prey item at 6.13 mm x 1.55 mm, and could be considered the longest (11.87 mm) if wings are included in the measurements. I did not have a voucher specimen of an ichneumonoid from Aguiguan in my reference collection. Therefore, I used the size of the ichneumonoid wings found in the guano to estimate total size of the prey item, and then measured a formicid of similar size to provide an approximate length and width (Table V-2).

DISCUSSION

Results from this study represent the first documented information on the food habits of Pacific sheath-tailed bats. From observations of dusk and night-time flights of these bats in the forest on Aguiguan (described in Section VI of this administrative report) and those of Esselstyn et. al. (2004), foraging activity occurs near (above and below) the canopy of the native forest Sampling of echolocation activity on Aguiguan during the same period when I collected guano (see Section IV of this administrative report) also indicated that Pacific sheath-tailed bat activity occurred mostly in stands of native limestone forest. The activity observations and the types of food items determined from analyses of guano demonstrate that Pacific sheath-tailed bats on Aguiguan share the same feeding behavior (i.e., aerial insectivore or hawker) as noted for other members of Emballonuridae (Bonaccorso 1998, Lim and Engstrom, 2001). However, prey items found in this study differ slightly from those of other emballonurids elsewhere (Bradbury and Vehrencamp 1976).

Results from my analyses indicate that sizes of prey items or related insects consumed by Pacific sheath-tailed bats on Aguiguan were small (ranging from 1.7 mm to 6.4 mm in length). Hymenopterans, coleopterans, and lepidopterans were the three major groups of insects consumed by bats from both roosts. Interestingly, ranking of orders by greatest percentages of volumes and frequency of occurrence differed between roosts. Although there were noticeable differences for hymenopterans at both roosts, the greatest observed differences were for the percent volumes of lepidopterans, with a greater consumption of moths in samples from Crevice Cave. The overall percentages of volumes and frequencies of occurrence for coleopterans in the diet of *E. s. rotensis* were nearly equal in guano sampled at the two roosts.

Within the Hymenoptera, I found that the percentages of volumes and occurrences of ichneumonoids consumed by bats from both roosts were similar. These parasitic wasps seemed to be a consistent prey item of Pacific sheath-tailed bats during the time I sampled guano. Based on samples from Guano Cave, I believe that formicids found in the diet of these bats may have been taken opportunistically during periods when alates form large swarms; usually during breeding or establishment of new colonies (Triplehorn and Johnson 2004). This also holds true for isopteran alates present in the bat guano (Triplehorn and Johnson 2004). Often swarms of ant and termite alates are associated with the onset of seasonal rains (Triplehorn and Johnson 2004). Rebello and Martius (1994) noted that in the Amazon peak periods of isopteran alate abundance occurred at the beginning and end of rainy seasons, but were lowest during the height of the rainy season. I collected guano on Aguiguan during the early part of the rainy season (late June to early July) and the presence of these ant and termite alates may be consistent with opportunistic feeding on these insects by Pacific sheath-tailed bats during the onset of the rains. Opportunistic feeding on winged formicids or isopterans by insectivorous bats is not uncommon and has been reported for other species elsewhere (Razakarivony and others 2005, Rakotoarivelo and others 2007).

It is interesting to note that even though Crevice and Guano Caves were only about 500 m apart there was a slightly higher presence of lepidopterans over hymenopterans in guano from Crevice Cave. I suspect that the higher abundance of lepidopterans at Crevice Cave may reflect greater availability and abundance of these insects, as noted for ant and termite alates. The intraspecific partitioning of resources and territories among individual bats has been noted for other emballonurids (Bradbury and Vehrencamp 1976) and could be a contributing factor for difference in food habits of bats sampled at different roosts.

From analyses of guano, I found fragments of coleopterans in some fecal pellets that had been broken down into a paste-like texture, likely by chitinase produced by bacteria in the gastrointestinal tracts of the bats. Whitaker and others (2004) noted that during summer feeding by bats in North America, harder and larger pieces of insects often pass through the digestive tracts of bats undigested. In part this is related to the fast (~30-60 min) transit time in the digestion of food by insectivorous bats (e.g., *Myotis lucifugus*; Buchler 1976). However, Whitaker and others (2004) also suggested that after an insect has been chewed into small pieces, the presence of chitinase helps break down connective tissue, making it easier to digest prey items. Whitaker and others (2004) also found that during winter months, chitinase activities are lower because bats are in torpor and often have little amounts of food. Because chitin remains in the gut for a longer period of time, this allows for a longer period for the breakdown of insect parts by chitinases. Because of the presence of highly digested beetle parts in the guano, I believe that some Pacific sheath-tailed bats may digest certain food items longer (i.e., beetles) with chitinase, especially during periods when food resources are low or preferred items are unavailable.

The concurrent analyses by Goressen and others (Section IV in this administrative report) show that native limestone forest habitat is a key component to the foraging behavior of Pacific sheath-tailed bats on Aguiguan. Coleopterans identified from fecal analyses also provide specific information on the importance of native forests. For example, cryptophagids feed on fungi and vegetation matter at different levels of decay, and many scolytines feed on dying trees (Triplehorn and Johnson 2005). Because of their ecology and fidelity to dying or decaying trees the presence of these beetles in the diet of Pacific sheath-tailed bats on Aguiguan provides additional evidence that the bats are foraging in and near the forests. Although beetles are not present in large volumes, they occur in a large proportion of the fecal pellets examined and are thus an important part of the diet..

My results suggest that during the onset of the rainy season (late June to early July) mature, native forests are important in providing food resources for Pacific sheath-tailed bats. I suggest that forest management emphasize practices that are not likely harmful to populations of small insects for future conservation of this bat on Aguiguan. For example, use of herbicides to control invasive vegetation must account for likely effects on native plants that support insect populations, and use of insecticides in native forests could alter the prey base of the bats. This study only represents a snap-shot of time in understanding what prey items are consumed by Pacific sheath-tailed bats on Aguiguan. I believe that the diet of this bat could be complex, likely includes other unidentified food items, and may change with variation in seasonal availability of prey, environmental variation, or reproductive conditions of the bats. Therefore, to better manage forest and perhaps other habitat needed for foraging by bats, I suggest that future studies investigate the feeding ecology and behavior of these bats during other periods of the year.

ACKNOWLEDGMENTS

I thank T. O’Shea, E. Masga, F. Muna, G. Wiles, M. Gorresen, F. Bonaccorso, D. Evangalista, Ton Castro, J. Omar, G. Comacho, J. Salas and the rest of the field crew for their help in the field. I thank Sylvan Igisomar, C. Kessler, and Division of Land and Natural Resource offices from Saipan and Tinian for their logistic support while in the CNMI. I thank K. Miller, N. Lord, A. Johnson, and S. Brantley for their insight and assistance on arthropod identification. Comments on previous drafts were made by P. Cryan, T. O’Shea, and G. Wiles. Special thanks go to G. Wiles for preparing me for island work and making sure I didn’t do anything stupid while I was there. Any use of trade, product, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the U.S. Government.

REFERENCES CITED

Arnett, R. H., Jr., 2000, American insects: A handbook of insects north of Mexico, Second ed. : Boca Raton, Fla., CRC Press, 1024 p.

Arnett, R. H., Jr., and Thomas, M. C., 2001, American beetles, Archostemata, Myxophaga, Adephaga, Polyphaga: Staphyliniformia, Vol. 1. : Boca Raton, Fla., CRC Press, 443 p.

Arnett, R. H, Jr., Thomas, M. C., Skelley, P. E., and Frank, J. H., 2002, American beetles, Polyphaga: Scarabaeoidea through Curculionidea, Vol. 2. : Boca Raton, Fla., CRC Press, 861 p.

Bonaccorso, F. J., 1998, Bats of Papua New Guinea: Washington, D.C., Conservation International, 489 p.

Bonaccorso, F., and Allison, A., 2008, *Emballonura semicaudata*: In: IUCN 2008, 2008 IUCN Red List of Threatened Species, www.iucnredlist.org.

Borror, D. J., and White, R. E., 1970, A field guide to insects, America north of Mexico: The Peterson field guide series: 404 p.

Bradbury, J. W., and Vehrencamp, S. L., 1976, Social organization and foraging in emballonurid bats: Behavioral Ecology and Sociobiology, vol. 1, p. 337-381.

Buchler, E. R., 1975, Food transit time in *Myotis lucifugus* (Chiroptera: Vespertilionidae): Journal of Mammalogy, vol. 56, p. 252-255.

Chujo, M., 1970, Coleoptera: Cryptophagidae and Mycetophagidae: Insects of Micronesia, vol. 16, p. 271-278.

Cruz, J B., Kremer, S. R., Martin, G., Williams, L. L., and Camacho, V. A., 2008, Relative abundance and distribution of Mariana swiftlets (Aves: Apodidae) in the Northern Mariana Islands: Pacific Science, vol. 62, p. 233-246.

Gorresen, P. M., Bonaccorso, F. J., and Pinzari, C. A., 2008, Habitat occupancy and detection of the Pacific sheath-tailed bat (*Emballonura semicaudata*) on Aguiguan, Mariana Islands: U.S. Geological Survey, Fort Collins, Colorado, this administrative report.

Hutson, A. M., Mickleburgh, S. P., and Racey, P. A., 2001, Microchiropteran bats: Global status survey and conservation plan: IUCN/SSC Chiropteran Specialist Group: IUCN, Gland, Switzerland and Cambridge, UK x + 258pp.

Koopman, K. F., 1997, The subspecies of *Emballonura semicaudata* (Chiroptera: Emballonuridae): Journal of Mammalogy, vol. 78, p. 358-360.

Kunz, T. H., 1988, Methods of assessing the availability of prey to insectivorous bats, in Kunz, T. H. ed., Ecological and behavioral methods for the study of bats: Washington, D.C. , Smithsonian Institution Press, p. 191-210.

Lemke, T. O., 1986, Distribution and status of the sheath-tailed bat (*Emballonura semicaudata*) in the Mariana Islands: Journal of Mammalogy, vol. 67, p. 743-746.

Lim, B. K., and Engstrom, M. D., 2001, Bat community structure at Iwokrama Forest, Guyana: Journal of Tropical Ecology, vol. 17, p. 647-665.

Rakotoarivelo, A. A., Ranaivoson, N., Ramilijaona, O. R., Kofoky, A. F., Racey, P. A., and Jenkins, R. K. B., 2007, Seaonal food habits of five sympatric forest microchiropterans in western Madagascar: Journal of Mammalogy, vol. 88, p. 959-966.

Razakarivony, V. R., Rajemison, B. and Goodman, S. M., 2005, The diet of Malagasy Microchiroptera based on stomach contents: Mammalian Biology, vol. 70, p. 312-316.

Rebello, A. M. C., and Martius, C., 1994, Dispersal flights of termites in Amazonian forests (Isoptera): Sociobiology, vol. 24, p. 127-146.

Triplehorn, C. A., and Johnson, P. J., 2005, Borror and Delong's introduction to the study of insects, Seventh ed.: Brooks Cole Publishing, 864 p.

U.S. Fish and Wildlife Service, 2008, 50 CFR Part 17; Endangered and Threatened Wildlife and Plants; Review of Native Species That Are Candidates for Listing as Endangered or Threatened; Annual Notice of Findings on Resubmitted Petitions; Annual Description of Progress on Listing Actions; Proposed Rule. Federal Register (December 10, 2008), vol. 73 (no. 238), p. 75175-75244.

U.S. Fish and Wildlife Service, 2007, 50 CFR Part 17; Endangered and Threatened Wildlife and Plants; Review of Native Species That Are Candidates for Listing as Endangered or Threatened; Annual Notice of Findings on Resubmitted Petitions; Annual Description of Progress on Listing Actions; Proposed Rule. Federal Register (December 6, 2007) 72 (234): 69033-69106.

White, R. E., 1983, A field guide to the beetles of North America: The Peterson field guide series: 368 p.

Whitaker, J. O., Jr., 1988, Food habits of insectivorous bats, in Kunz, T. H. ed., Ecological and behavioral methods for the study of bats: Washington, D.C., Smithsonian Institution Press, p. 171-189.

Whitaker, J. O., Jr., Dannelly, H. K., and Prentice, D. A., 2004, Chitinase in insectivorous bats: Journal of Mammalogy, vol. 85, p. 15-18.

Wiles, G. J., Worthington, D. J. , Esselstyn, J. A., O'Shea, T. J., and Valdez, E. W., 2008, Population assessments of Pacific sheath-tailed Bats (*Emballonura semicaudata*) on Aguiguan, Mariana Islands: U.S. Geological Survey, Fort Collins, Colorado, this administrative report.

Table V-1. Percent volumes and percent occurrences of food items of the Pacific sheath-tailed bat (*Emballonura semicaudata rotensis*) from Aguiguan, as determined by fecal analyses. Sample sizes from each cave are noted in parentheses. Values in bold represent overall percentages for volumes and occurrences of each order.

	Guano Cave (n=100)		**Crevice Cave (n=100)**	
Prey Item	**% Vol.**	**% Occurrence**	**% Vol.**	**% Occurrence**
Ichneumonoidea	25	45	31	46
Formicinae	7	12	7	26
Ponerinae	2	12		
Unkn. Hymenoptera	31	63	3	18
HYMENOPTERA	**64**	**95**	**41**	**82**
Curculionidae	<1	2		
Scolytinae	1	9		
Cryptophagidae	<1	4	3	18
Mordellidae	<1	3	<1	6
Chrysomelidae			1	1
Unkn. Coleoptera	8	60	6	53
COLEOPTERA	**10**	**73**	**10**	**68**
Microlepidoptera	8	38	45	86
LEPIDOPTERA	**8**	**38**	**45**	**86**

ISOPTERA	**8**	**10**	**5**	**6**
Pseudocaeciliidae	5	26	<1	2
PSOCOPTERA	**5**	**26**	**<1**	**2**
Cicadellidae	<1	4	<1	2
Unkn. Auchenorhyncha	<1	7	<1	2
HEMIPTERA: Auchenorhyncha	**<1**	**11**	**<1**	**4**
DIPTERA	**<1**	**1**	**<1**	**5**
UNKNOWN INSECT	**1**	**10**	**<1**	**20**
FEATHER FRAGMENT	**1**	**24**		
UNKNOWN	**2**	**4**		

Table V-2. Measurements (length and width) of a single representative from some of the prey items consumed by Pacific sheath-tailed bats on Aguiguan were recorded to the nearest 0.01 mm. Lengths are measured from tip of head or mandible, whichever extends furthest, to the end of body; lengths in parentheses represent length of head to posterior tip of wings. Widths are measured at the widest point of the insect, including head or body, but not legs. Missing values are marked with hyphens. Asterisks denote the estimated length and width of consumed Ichneumonoidea, based on measurements of a formicid of similar wing size.

Insect	**Length in mm**	**Width in mm**
Ichneumonoidea*	3.38 (4.15)	0.73
Formicinae	5.10 (6.07)	1.20
Ponerinae	7.60 (-)	1.63
Curculionidae	2.37	1.20
Scolytinae (large)	2.57	0.95
Scolytinae (small)	1.72	0.85
Cryptophagidae	3.28	1.50
Mordellidae	3.32	1.26
Chrysomelidae	6.07	3.91
Microlepidoptera	2.69 (3.14)	0.94
Isoptera	6.13 (11.87)	1.55
Pseudocaeciliidae	2.75 (3.36)	0.98
Cicadellidae	2.89 (3.76)	1.36

Section VI. Capture, Morphometrics, Museum Specimens, and Other Sampling and Observations of Pacific Sheath-Tailed Bats (*Emballonura semicaudata rotensis*) on Aguiguan, Commonwealth of the Northern Mariana Islands

Thomas J. O'Shea and Ernest W. Valdez

ABSTRACT

In 2008, we used standard means to capture Pacific sheath-tailed bats in mist nets on Aguiguan while they dispersed or foraged through the forest, but these attempts were largely unsuccessful because the bats were highly maneuverable and easily avoided mist nets on close approach. We successfully captured 12 adult bats and one attached suckling young by using hand nets on bats in flight in the forest, or mist nets set in or near caves used as roosts. Both methods have logistical problems and limitations: in addition to the high maneuverability of the bats precluding use of mist nets in standard configurations, considerable time is required to accrue multiple captures using hand nets, and caves where bats roost are co-occupied by endangered Mariana swiftlets. We also found that these bats can be very sensitive to initial handling. We discuss suggestions for capturing and handling bats in future studies. Despite limited numbers of bats captured, forearm measurements show for the first time that there is considerable overlap in body size with the other three subspecies of *Emballonura semicaudata*. In addition to variation in skull morphology, size was previously thought to be another trait that may vary with subspecies. We also collected small wing biopsies from these 12 bats prior to release for some basic preliminary genetic analyses to ascertain genetic diversity of the population on Aguiguan, and the depth of division of this subspecies based on comparison with published data on genetics of *E. semicaudata semicaudata* from Fiji. Although not part of the original proposal, laboratory phases of the genetics analyses are planned for 2009 by the U.S. Geological Survey. We also prepared two museum voucher specimens of *Emballonura semicaudata rotensis*, increasing the number of specimens from the Mariana Islands available in United States museums from two to four. Considerable numbers of specimens of the other subspecies are available worldwide (over 300), and about 22 additional specimens from the Marianas Islands (including Guam) are housed in museums in France and Japan. Expanded study of museum specimens and comparative genetic analyses would be needed to fully ascertain the systematic status of the Pacific sheath-tailed bat population in the Mariana Islands. All bats captured at caves by us in 2008 and by others in years past were females, whereas the 4 bats we captured at dusk dispersing along a steep rocky hillside, not near any known colony, were males. This tentatively suggests that perhaps males may form bachelor colonies apart from roosts occupied primarily by females, as is known for other species of Old World species in the genus *Emballonura*. Elaborate social behavior patterns were also suggested by the audible communication sounds produced by bats that we observed foraging and dispersing through the forest and flying into caves. Thermal

characteristics of caves on Aguiguan show little variability, and relative humidity is high. Cave temperatures are similar to those used by other tropical emballonurids elsewhere, and do not suggest that the availability of caves with special thermal or humidity characteristics is a limiting factor for Pacific sheath-tailed bats on Aguiguan.

Introduction

There have been few attempts to capture Pacific sheath-tailed bats for scientific research in the Commonwealth of the Northern Mariana Islands or Guam. Lemke (1986) summarized the early literature documenting the taking of two bats for museum voucher specimens on Aguiguan in 1984, 14 bats for taxonomic study on Rota in 1932, and six bat specimens on Guam in 1887, all obtained by unspecified techniques. We were interested in capturing bats on Aguiguan for a variety of purposes, particularly at locations other than roosts. We hoped to gain additional data pertaining to body size, reproduction, and population sex and age composition. There is very little prior information on these biological attributes of this subspecies. We also sought to (1) obtain biopsy samples that could be used in future genetics research aimed at determining both the genetic diversity and degree of genetic distinctiveness in this population; (2) examine and sample the bats for ectoparasites; (3) sample fresh guano for food habits analysis; and (4) deploy miniature radio transmitters to assess movements and possibly determine the locations of previously unknown roosts. In separate sections of this administrative report we provide results of findings on reproduction (Section VII) and food habits (Section V). Herein, we provide a summary of all other efforts at capturing sheath-tailed bats on Aguiguan and resulting data and observations. We also provide a summary of available museum records for *Emballonura semicaudata* from throughout its range that may be useful to future researchers, and a descriptive summary of the temperature regimes and humidity in some of the caves on Aguiguan. The purpose of describing temperature and humidity regimes of caves was to determine if enough variability in these factors existed to support a hypothesis that numbers of suitable caves might be a limiting factor for this population. We also offer suggestions for future researchers regarding the capture of sheath-tailed bats on Aguiguan and how the risks of capture stress can be minimized.

MATERIALS AND METHODS

Capturing, Handling, and Sampling Bats

In most studies elsewhere, small insectivorous bats are typically captured at roosts or as they fly close to open sources of freshwater in pools and ponds to drink or feed or at constricted “flyways” along edges or through vegetation (Kunz and others 1996). There were no sources of open freshwater on Aguiguan and we did not observe natural flyways that would facilitate capture of bats. Additionally, sampling directly at roosts in this study was complicated by the presence of endangered cave swiftlets at nearly all roost locations and their overlapping times of ingress and egress with bats at dusk; it was also complicated by the potential of creating disturbance to the extent that bats might abandon roosts.

Therefore we initially attempted to capture bats in flight in the forest as they were foraging or dispersing from roosts, or at caves on return flights from foraging after swiftlets had settled in for the night. We risked disturbance by capturing bats within a roost during the day only once at Crevice Cave and once at Guano Cave, both instances took place near the end of the study.

We attempted to capture bats in the following seven ways, which met with only limited or no success. These were: (1) placing mist nets in the forest a short distance below Guano Cave with the intent to capture bats as they returned from foraging, after swiftlets had gone to roost; (2) placing mist nets within the outer opening of East Black Noddy Cave with the intent to capture bats as they returned from foraging (after swiftlets had gone to roost and most bats had emerged); (3) placing mist nets in the forest below the canopy on the first forested terrace above East Black Noddy Cave to capture them as they foraged or dispersed to forage; (4) placing mist nets in the forest below the canopy at various other locations in native limestone forest to capture them as they foraged; (5) setting a short mist net across the opening of the outer, shallow, chamber of Guano Cave to intercept bats taking refuge after we intentionally disturbed them, a few at a time, during the day in their regular roost within the same cave; (6) similarly using a short mist net and insect sweep net at Crevice Cave during the day; (7) using a hand insect sweep net and stationary mist nets to capture bats in flight as they dispersed up the hillside among the boulders and cliffs near Fault Line Cave 1 (Cave 94). Locations and descriptions of caves are provided in Section III of this administrative report.

Captured bats were examined to determine sex and age (adult or volant juvenile based on the degree of closure of the phalangeal epiphyses following Anthony 1988), and weighed on a Pesola spring balance calibrated in the field (to 0.1 g). Wings, ears, uropatagia, and fur of bats were examined with a 20x magnification visor and fine-point forceps for visible ectoparasites. Prior to field preparation of museum vouchers, specimens were also examined under a 14-60 x stereo-zoom microscope for ectoparasites. We also measured forearm length (to 0.1 mm) with dial calipers, and took small (3 mm diameter) circular punches of the wing membrane in the proximal plagiopatagium using skin biopsy tools after sterilization of the wing with a general antiseptic. Biopsies were immediately placed in 95% ethanol in the field and were stored at -80˚C after return to the laboratory in Fort Collins in late July. Sampling of wing tissues following this methodology is a standard procedure in bat genetics research (e.g. Worthington Wilmer and Barratt 1996, Neubaum and others 2007). Other methods also are standard procedures in bat field research (see Kunz 1988).

Records of Specimens in Museums

We tabulated records of *Emballonura semicaudata* in various museum collections around the world to provide an estimate of the range of localities and number of specimens that may be available for future studies of the taxonomy of these bats. We tabulated sources from the literature that refer to specimens, and also queried the online database of the Global Biodiversity Information Facility (http://www.gbif.org/). The results are probably a minimum number because some major museums do not have searchable databases online. However, there also may be some overlap between specimens referenced in publications and those retrieved online, although we avoided duplicate counting of these when obvious to us.

Temperature and Humidity of Caves

We sampled relative humidity manually using a sling psychrometer at about 1.5 m above ground, recording wet and dry bulb temperatures. We sampled temperatures using Thermocron iButton data loggers (model DS1921, Dallas Semiconductor Corporation) that are factory calibrated at a level of precision of ± 1 °C. We programmed the loggers to record temperature hourly. Two data loggers were placed together at each sampling point to insure redundancy in obtaining temperature data in the event one logger failed. When both loggers were functional we took the average of the two readings (most differences were low and none at caves exceeded 1.5° C, see results). We calculated summary statistics for each station based on time periods that ran for complete 24 hour cycles to avoid including biases from any particular time of day. For logistic reasons we could not run all loggers simultaneously for the same number of days. Therefore we also provide and compare summary statistics for all stations between 7 July and 10 July, a period when all loggers were operating simultaneously. In most cases we took temperature readings at the rock surface because these bats roost singly appressed to the rock walls or ceilings (see Section III of this administrative report). We had access to two roosts that were regularly occupied by bats. We did not attempt to measure temperatures at the precise places where bats roosted because we did not want to risk disturbing them. We placed dataloggers at two locations in Crevice Cave after the bats left to forage at dusk, with the highest about 0.5 m from where the bats roosted. At Guano Cave we recorded temperatures directly below the area used by the roosting bats (see Section III of this administrative report) up to a height of 6 m, about 4-6 m directly below the area occupied by roosting bats. Here, at a more shallow domed area closer to the mouth of Guano Cave, and at Fault Line Cave 1 we took readings at multiple heights above the cave floor by taping data loggers to tall poles we propped against the cave wall. Ambient temperatures for the study period were taken at base camp on the Aguiguan central plateau (coordinates 344803E and 1642496N, WGS 84 55 P). At camp paired data loggers were suspended on a cord in mottled shade at heights that were 1 m, 2 m, and 3 m above cleared level ground. We viewed gathering of data on temperature and humidity within caves as a preliminary, exploratory attempt to characterize the amount of variability in these attributes. Our analysis of the temperature data from these caves is limited to calculation of summary statistics for hourly temperature readings at each station, with a qualitative discussion of their attributes and variability in relation to knowledge about cave environments used by emballonurids and other tropical bats elsewhere.

RESULTS AND DISCUSSION

Capture and Handling

We attempted to capture bats on 13 different dates during our field work on Aguiguan (Table VI-1). We captured 12 adult bats, plus an attached single young of one female (Table VI-2). None of these had any visible ectoparasites. The first three bats we captured appeared stunned by handling, and one of these died. In the latter case, the bat was held for about 10 minutes while photographed using a flash, measured, and examined under illumination of a headlamp. This handling routine was also followed in the case of the two stunned bats; however, these two bats were placed in small cloth bags after handling, then later sampled for wing biopsies prior to release. During subsequent captures we immediately placed captured bats individually in small cloth bags with minimal handling or shining of lights before taking measurements or inspecting them closely. When thus handled the bats seemed less stunned and flew readily upon release. It appears that to avoid stunning or death, sheath-tailed bats should not be subject to intense handling immediately following the initial shock of capture. Instead the bats should be placed individually in cloth bags and left undisturbed and out of the beams of headlamps for 10 minutes or more before resuming handling (10 minutes was the shortest interval we measured between placing a bat in a cloth bag and its removal and release without signs of stunning).

We abandoned our original intent to apply radio transmitters to Pacific sheath-tailed bats in part because of the possible detrimental effects of the lengthy handling (20-40 minutes or more) prior to release that tagging would require. This extra time would be necessary to allow the colostomy cement used for attachment of the tag to dry under the high humidity at Aguiguan. We intended to apply tags primarily to bats captured during foraging to try to locate unknown roosts, but such capture attempts were only minimally successful (see below) until the last few days of our field work when subsequent tracking would be infeasible. However, considering the capture successes on 11-13 July (Tables VI-1 and VI-2) when no mortality and almost no stunning occurred after quickly placing bats in cloth bags and not handling them for at least 10 mins after capture, future researchers should not be discouraged completely from using radio telemetry as a tool to answer specific biological questions about sheath-tailed bats. All of the bats captured within Guano Cave on 12 July flew readily on release after being held in cloth bags from 10 mins to up to 2 hours (1 case) and then handled for about 5-10 minutes additional time after being held in bags. None flew outside of the cave in the daylight, and all returned to the main roosting chamber. Future researchers should use the approach of initially holding the bats in cloth bags with minimal disturbance prior to handling, and cautiously attempt to attach radio tags to a small number of bats to further determine if radiotagging will be a feasible tool for study. Radiotagging to locate roosts may have other limitations: one tag we activated and placed within Guano Cave below roosting bats had the signal severely attenuated by the rock and was only detectable at the cave mouth.

We saw Pacific sheath-tailed bats easily avoid mist nets on multiple occasions when we tended nets in the forest, and in the rocks along likely dispersal routes and foraging areas. These bats are light and maneuverable fliers. Other species of *Emballonura* elsewhere in the Paleotropics are also known to be highly acrobatic flyers, adapted to foraging in the understory for aerial and foliage-perched insects (Bonaccorso 1998). Pacific sheath-tailed bats on Aguiguan easily detected and avoided nets in open areas, turning to fly over or along the lengths of nets when within 1 m or less of the mesh (some of these observations are listed in Table VI-1). We found that the best method for capturing bats under these circumstances was to use a hand-held insect collecting net, sweeping flying bats into it. This seemed to be especially efficient when bats were flying close to mist nets. Unfortunately, this method is most effective only during the short interval at dusk, such as when we captured bats dispersing in the vicinity of fault Line Cave 1 (Cave 94). Few bats were observed in this area after the first 30-60 minutes after dark. Using sweep nets will only yield a small number of bats, caught one at a time during a short period each night. Obtaining larger samples will require considerable effort on multiple nights using more than one observer with sweep nets. Use of sweep nets might be suitable, for example, in capturing small numbers of bats for possible radiotagging to attempt to locate unknown roosts or foraging areas. Capturing bats at roosts may be more efficient for obtaining larger sample sizes for assessment of reproduction, sex, age and morphometrics. Attempts in the future will need to devise ways to sample bats at roosts without unintentional capturing of cave swiftlets which occupy the same caves used by the bats. Development of novel methods for capturing these bats should also be considered.

Observations of Feeding and Dispersing Bats, and Sex and Age Composition of Captured Bats

Our attempts to capture bats in the vicinity of Cave 94 (Tables VI-1 through VI-3) were based on observations made early in the study during searches for previously unknown roosts along the fault line that is a dominant feature of the area (see Section III of this administrative report). We saw Pacific sheath-tailed bats dispersing uphill and foraging at various heights under the canopy, ranging from about 1.5 to 5 m, while we were standing near openings of inaccessible caves to observe possible emergences of bats at dusk (none emerged). Some of these bats would pause to forage in a characteristic "beat" (sensu Vaughan 1959) flying back and forth in elliptical patterns about 20 m long for a few (e.g. 2-4) minutes, including insect pursuit phases heard on echolocation detectors, whereas most seemed to fly directly through the area heading uphill. Multiple bats were seen foraging and dispersing through this area on several nights (Table VI-4). This minor concentration of bats led us to believe that there is an undiscovered roost nearby. This is suggested by the following lines of evidence: 1) The first observations of bats in flight ranged from 18:31 to 18:50 h, overlapping with the times of initial exits of sheath-tailed bats we observed at caves used as roosts (Section III of this administrative report); 2) This much activity was not observed at dusk below the forest canopy at a distance of about 90 m above and 115 m inland from the largest known colony at East Black Noddy Cave (see below); 3) Echolocation activity at dusk at a point near the sea cliffs below the fault line was low when monitored with a hand-held bat detector on 2 July, indicating that not many bats had followed the cliff edges from Black Noddy Cave to enter the forest and fly up the fault line at this point on the island; 4)We captured four bats as they dispersed through the area, and all were adult male (this was the only place males were captured), whereas all 12 bats we captured at Guano Cave and East Black Noddy Cave were adult females, as were 6 and 2 bats captured at Guano Cave in 2003 and1984, respectively (see Lemke 1986). This suggests that the unknown roost may harbor a bachelor colony. Such predominantly male colonies are known for some other insectivorous bat species, as well as in other species of *Emballonura* in the paleotropics (Bonaccorso 1998). All these observations are preliminary, however, and more dedicated field work will be necessary to determine if males form bachelor colonies and if undiscovered roosts exist in the "Fault Line" area (near Caves 66-68, 94, 95, and 101) of Aguiguan. These bats have been observed commuting distances of 5 km during evening dispersal (Wiles and others 1997), so it is also possible that the observations were of bats originating at a more distant site than the immediate area where they were observed. Sex composition of bats using caves could also vary seasonally and with mating or breeding behavior, but this also will require additional detailed field research to determine.

A few other noteworthy observations were made of sheath-tailed bats at dusk or while foraging or dispersing. As noted above, we set up mist nets under the canopy in the forest on the first terrace above East Black Noddy Cave on 1 July (Table VI-1). Although 323 bats were observed exiting the cave that night between 18:35 and 19:11 h (Section III of this administrative report), only 2 bats were observed in the forest about 115 m away and 90 m above the cave (see coordinates in Table VI-1) during and after the emergence period (Table VI-1). This suggests that the bats using this cave either disperse widely soon after exiting, or do not enter below the canopy until they are farther from the cave.

Interesting observations were made at Pillar Cave at dusk on 21 June and 7 July (see also Section III of this report) when 13 and 21 bats, respectively, entered the cave from elsewhere and used multiple audible communication sounds between 1841 h and 1902 h. No bats were observed roosting in this cave during the day in 2008, but smaller numbers (2-10) had used the cave in 1995, 2002, and 2003 (Section III of this administrative report). These and other observations suggest that these bats use some caves at night for social reasons. We also heard social calls given by some bats dispersing at dusk in the vicinity of Fault Line Cave 1 (Cave 94), and by one bat that entered this cave at dusk on 13 July. It entered the cave while emitting single audible chirps with 1-5 sec pauses between chirps, until it exited the cave through a small opening in the ceiling. Audible calls of these bats were high pitched chirps that varied from about 1 to 5 syllables, and to our ears reminded us of directive calls of pallid bats (*Antrozous pallidus*) heard in the desert southwestern United States (e.g. Orr 1954, Brown 1976, O'Shea and Vaughan 1977). Communication calls of sheath-tailed bats were also heard as these bats dispersed and foraged in the forest. On 2 July we tended nets set below the canopy in the forest and noted a few bats dispersing through the area up until 19:00 h, but then there was no notable activity of bats in flight. Beginning around 22:15 h we began to observe bats flying between the forest floor and canopy producing audible social calls, and once saw three bats flying in a group. On the following night we saw bats in the same area earlier, from 19:00-22:50 h, heard audible calls, and observed one bat chasing another near but below the canopy. Bat social activity and behavior may change from place to place and from night to night as well as seasonally.

Museum Specimens, Morphometrics, and Subspecies Designations

The female bat that died during handling and a subsequently captured male that appeared stunned were saved as museum study skins and skeletons, with soft tissues (heart, kidney, liver, spleen, lungs, and muscle) preserved in 95% ethanol. These two specimens are a small but important addition to the few specimens of this subspecies available in museum collections and have been deposited in the Museum of Southwestern Biology (MSB) at the University of New Mexico, Albuquerque (MSB 125656 and 125657). Soft tissues are cataloged as NK104278 and 104279.Historic specimens were reviewed by Lemke (1986) and are briefly repeated here. There are six specimens in France's National Museum of Natural History in Paris. These were collected in 1887 on Guam. Four of these six were mentioned by Oustalet (1895, cited by Lemke 1986) but no details about them have been published other than Lemke's (1986) clarifications. In 1932 a total of 14 bats were collected on Rota and formed the basis of the subspecies description by Yamashina (1943). In June 1984, Lemke (1986) collected two females at Guano Cave on Aguiguan, and these specimens are housed at the American Museum of Natural History (AMNH 256514 and 256515). These and the two specimens we collected are the only samples of this subspecies in museum collections in the United States.

Published data on morphometrics of this subspecies are apparently limited to perhaps two bats mentioned by Yamashina (1943) and the two bats collected by Lemke (1986). The forearm lengths and body masses we measured on bats released after capture provide an improved estimate of the range of body sizes known for *Emballonura semicaudata rotensis* based on both the minima and maxima of forearm lengths of adult males and adult females (Table VI-3). It is important to note that the subspecies designations for *E. semicaudata* follow geographic distributions, but also have been morphologically defined on the basis of body size and qualitative features of skull morphology in small numbers of specimens, with a linear series from smallest to largest defined as *E. s. semicaudata-E. s. palauensis-E. s. rotensis-E. s. sulcata* (Koopman 1997, Helgen and Flannery 2002). Our measurements of forearm lengths show that the Aguiguan population has overlap in body size with each of these other three subspecies, making size alone a poor criterion for subspecies definitions.

We updated summaries of museum specimen records for all subspecies of *Emballonura semicaudata* previously provided in part by Lemke (1986), Koopman (1997), and Helgen and Flannery (2002). This new summary is not exhaustive, but suggests that at least 386 specimens from wide areas of the species range are housed in the world's museums (Table VI-5). Subspecies designations made in the past (see reviews in Koopman 1997, Helgen and Flannery 2002) were based on far fewer specimens than are now known to be available, and did not include any genetic analyses. A comprehensive systematic review and new morphometric and genetic analyses would be desirable to improve our understanding of the degree of differentiation among these taxa. The current ranking of *Emballonura semicaudata rotensis* as a Category 3 candidate for listing under the U.S. Endangered Species Act is based largely on its systematic status as a subspecies of a more widely ranging species (U.S. Fish and Wildlife Service 2007).

Wing Biopsies for Genetics Studies

We obtained wing biopsies from all 12 adult bats captured during this study. These are now archived at -80° C at the USGS laboratory at Fort Collins with plans for molecular genetics studies in 2009 by USGS staff and cooperators at the Rocky Mountain Center for Conservation Genetics and Systematics. Analysis of these samples will focus on understanding the current level of genetic diversity in the isolated population on Aguiguan, as well as assessing the overall level of differentiation of this population in comparison with another, previously analyzed population of *Emballonura semicaudata semicaudata* on Fiji (Colgan and Soheili 2008). The latter analysis also might help ascertain if a subspecies designation is appropriate or if genetic divisions of sheath-tailed bats in the Mariana Islands are even deeper. To our knowledge the only genetics research involving Pacific sheath-tailed bats was the comparative analysis of *E. semicaudata* from Fiji with three other full species of *Emballonura* and a fifth emballonurid species, *Mosia nigrescens,* from the southwestern Pacific (Colgan and Soheili 2008). This study examined segments of the mitochondrial DNA genome in *E. semicaudata* from 12 bats sampled at three locations on Fiji. Objectives of the study were aimed at understanding biogeography and evolutionary processes as exemplified by the regional bat fauna rather than at estimating the genetic diversity or depth of divergences within *E. semicaudata*. However, the development of primers and data on gene sequences deposited in GenBank by Colgan and Soheili (2008) provides basic information that will expedite a preliminary analysis of variation in mtDNA of Pacific sheath-tailed bats on Aguiguan by USGS biologists. Results of the mtDNA studies may be expanded to other segments of the genome depending on preliminary findings.

Cave Temperatures and Humidities

We deployed and recovered 58 functional temperature data loggers at 31 stations: 3 stations in camp, and 28 stations at 7 caves (Tables VI-6 and VI-7). Duplicate readings were obtained at 27 of the 31 stations, with 4 other stations based on single logger records. Agreement on temperature readings between the loggers at each station was good. The distribution of differences between 5,828 hourly readings of paired loggers placed at caves was: 0.0°C (58.9 % of readings), 0.5°C(30.4 %), 1.0°C(10.4 %), 1.5°C(0.2 %), > 1.5 °C (0.0 %).

Temperatures at camp during the recording period averaged about 27 °C, fluctuating over a 10 °C range from a minimum of 22 °C to a maximum of 32.8 °C (Table VI-6). Temperatures at stations in all caves were much less variable, ranging from no variation to at most 3 °C, with mean temperatures at all stations in caves ranging narrowly from 26 °C to 27 °C. During the period when all recording stations were operating simultaneously (Table VI-7), minimum and maximum temperatures at all stations within caves varied by 0 to 2 °C; temperatures within most caves at depths of 20 m or more and heights of 1-3 m above cave floors were essentially constant, although the inner chamber at Guano Cave fluctuated by about 1 °C during this period. Slightly greater fluctuations were recorded over longer periods. Variation with height within caves was also minor in the three caves where this was measured (Guano Cave at two locations, Dangkolo Cave, and Fault Line Cave 1): means at various heights were within 1 °C or less, and differed by only fractions of degrees over a height range of 6 m directly below the roosting bats at Guano Cave. Maximum temperatures in all caves over the full recording periods also spanned a narrow range, from 26 °C to 28 °C (Table VI-6).

Overall there were no obvious major differences in thermal regimes of the caves we sampled regardless of history of occupancy by bats. Given that the manufacturer's specifications for these data loggers are ± 1 °C , the differences in mean temperatures we observed when computational averages were rounded to the nearest °C were at most 1 °C, with rounded means at all stations within caves either 26°C or 27 °C (Tables VI-6 and VI-7). Stations within two caves that had no history of occupancy by bats (Dangkolo Cave and Cave 68) averaged 26 °C and did not reach the maxima recorded in Guano Cave (most stations in Guano Cave averaged 27 °C), the only cave we sampled that was used consistently by bats on every visit from 1984 through 2008 (see Section III in this administrative report). Maxima at stations in Guano cave reached 27 °C to 28 °C. Crevice Cave, used by a very small number of bats since discovered as a roost in 1995, also averaged 27 °C and had maxima of 27 °C to 28 °C (Tables VI-6 and VI-7). Caves with inconsistent histories of use by bats varied from means of 26 °C (Fault Line Cave 1, New Cave 1) to 27 °C (Pillar Cave). Perhaps the slightly warmer caves are preferred, but it seems doubtful to us that a 1 °C difference between rounded means is biologically meaningful given this limited sampling effort to characterize the thermal environments of these caves. This is even more evident considering the low variability in temperatures in this region and the thermal characteristics of other roosts used by other emballonurids (see below).

Although additional sampling of cave temperatures at greater levels of accuracy and using a more systematic series of sampling stations might reveal some subtle differences in thermal environments among caves, subtle differences in cave temperature patterns are unlikely to limit their use as roosts by these bats. Most caves we observed (see section III of this report) do not appear to have major internal complexities that might create strong heterogeneity in internal microclimates. Cave temperatures anywhere in the world typically reflect the mean annual surface air temperatures of a region (e.g. Dwyer 1971). There are no long-term temperature data readily available for Aguiguan or neighboring islands, but the mean annual temperatures at three weather stations on Guam are 26.2 °C, 26.9 °C, and 27.7 °C (National Oceanic and Atmospheric Administration 2008), very consistent with the temperatures at the stations we sampled in caves on Aguiguan. Mean annual maxima and minima at these three weather stations on Guam varied between 22.6 °C and 30.7 °C, suggesting that even if caves were more complex the ranges of air temperatures that might occur in trapped internal air masses would be small. We did not observe any major air movement within the 41 caves we investigated in 2008 to indicate that highly complex thermal patterns might be found in these caves.

Furthermore, it is thought that the thermal environments of caves used by those species of bats that do not regularly enter torpor are less important than those of caves used by more heterothermic bats at higher latitudes (Dwyer 1971). We believe that Pacific sheath-tailed bats do not enter torpor readily and normally maintain a resting body temperature that is high enough to allow alertness and quick flight under the thermal conditions prevailing in the caves on Aguiguan. Maintenance of homeothermy under normal environmental conditions is also typical for other tropical emballonurids (Bonaccorso 1998). These bats always seemed wary, readily flew within the roost when we approached from outside, did not cluster, and did not become torpid when we placed them in cloth sacks. (Indeed, it is possible that some of the stunning and the death we observed in the first bats we captured may have been due to heat stress from struggling while held in gloves in the high humidities on Aguiguan.) The lack of regular use of torpor has been demonstrated experimentally in some (but not all) other species of emballonurids elsewhere in the world (e.g. Genoud and others 1990, Genoud and Bonaccorso 1986). Temperatures of roosts used by two of these other species of emballonurids that do not normally enter torpor also have been measured (although measurements were over shorter periods than some on Aguiguan). Two tree roosts of *Saccopteryx bilineata* (an 8.2 g emballonurid) in Costa Rica fluctuated less than 1 °C daily and had mean temperatures of 26.1 °C and 26.5 °C; temperatures at a third roost ranged 26.4 °C to 27.5 °C at noon (Genoud and Bonaccorso 1986). Temperatures in roosts used by the 5 g emballonurid *Peropteryx macrotis* in caves in Venezuela averaged 27.8 ± 1 °C (Genoud and others 1990). Temperatures in 6 caves used by a third species of small (5.3 g) emballonurid (*Balantiopteryx plicata*) in Mexico averaged 26.7 ± 3.1 (SD) °C (Avila-Flores and Medellin 2004). The similarity in temperatures of roosts used by these other emballonurids with those we measured in the Aguiguan caves is noteworthy, and is also suggestive that the thermal characteristics of caves on Aguiguan do not limit their use by Pacific sheath-tailed bats.

Relative humidity in caves was high. We recorded relative humidity at about 1.5 m height above the cave floor in five caves. Three were identical at 92 %, whereas New Cave 1 and Dangkolo Cave were slightly more humid at 96 %. The latter cave has no history of bat occupancy and was the only cave we entered that had occasional dripping water. Relative humidities taken at various times and locations outside of caves were generally lower, ranging from 74% to 92% (Table VI-8). The role of humidity in use of caves by bats on Aguiguan, if any, remains to be studied. We did not measure humidity in a large number of caves and given the lack of access to pools of freshwater for drinking, roost environments that minimize evaporative water loss may be of importance to these bats. However, given the uniformly high humidities in the caves that we measured it seems unlikely that variability in humidity among caves will be great enough to be a factor limiting sheath-tailed bat populations on Aguiguan. The humidities we observed were also within the ranges in caves utilized as roosts by many other bats in the tropics (e.g. 70-98 % in 12 species summarized by McNab 1969)

Acknowledgments

We thank E. Masga, F. Muna, and J. Omar for shepherding us to various locations around Aguiguan and facilitating all aspects of this work, including captures of bats at Guano Cave. F. Bonaccorso helped with captures of bats at Guano Cave and M. Gorresen helped with captures at East Black Noddy Cave. F. Bonaccorso, M. Gorresen, and G. Wiles contributed to many useful discussions in the field. P. Cryan and G. Wiles provided comments on the manuscript. The research we report on herein was sponsored by the U.S. Fish and Wildlife Service (USFWS), with C. Kessler of the Honolulu office coordinating and guiding the project. Bat captures and collecting took place under authority of Scientific Research Permit 01048-08 issued by the Commonwealth of the Northern Mariana Islands. Helpful information on specimens in Japan was provided by Dr. T. Yamasaki, Yamashina Institute for Ornithology, Chiba, Japan. Any use of trade, product, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the U.S. Government.

References Cited

Anthony, E.L.P., 1988, Age determination in bats, *in* Kunz, T.H., ed., Ecological and behavioral methods for the study of bats: Washington, D.C., Smithsonian Institution Press, p. 47–58.

Avila-Flores, R., and Medelin, R., 2004, Ecological, taxonomic, and physiological correlates of cave use by Mexican bats: Journal of Mammalogy, vol. 85, p. 675-687.

Bonaccorso, F. J., 1998, Bats of Papua New Guinea: Washington, D.C., Conservation International, 489 p.

Brown, P., 1976, Vocal communication in the pallid bat, Antrozous pallidus: Zeitschrift für Tierpsychologie, vol. 41, p. 34-54.

Bruner, P. L., and Pratt, H. D., 1979, Notes on the status and natural history of Micronesian bats: ʻElepaio, vol. 40, p.1-4.

Colgan, D. J., and Soheili, S., 2008, Evolutionary lineages in *Emballonura* and *Mosia* bats (Mammalia: Microchiroptera) from the Southwestern Pacific: Pacific Science, vol. 62, p. 219-232.

Dobson, G. E., 1878, Catalogue of the Chiroptera in the British Museum: London, British Museum (Natural History), 567 p.

Dwyer, P. D., 1971, Temperature regulation and cave-dwelling in bats: an evolutionary perspective: Mammalia, vol. 35, p. 424-455.

Flannery, T., 1995, Mammals of the South-West Pacific and Moluccan Islands: Ithaca, New York, Cornell University Press, 464 p.

Genoud, M., and Bonaccorso, F. J., 1986, Temperature regulation, rate of metabolism, and roost temperature in the greater white-lined bat Saccopteryx bilineata (Emballonuridae): Physiological Zoology, vol. 59, p. 49-54.

Genoud, M., Bonaccorso, F. J., and Arends, A., 1990, Rate of metabolism and temperature regulation in two small tropical insectivorous bats (*Peropteryx macrotis* and *Natalus tumidirostris*): Comparative Biochemistry and Physiology Part A, vol. 97, p. 229-234.

Helgen, K. M., and Flannery, T. F., 2002, Distribution of the endangered Pacific sheath-tailed bat *Emballonura semicaudata*: Australian Mammalogy, vol. 24, 209-212.

Johnson, D. H., 1962, IV. A. Rodents and other Micronesian mammals collected, *in* Storer, T. I., ed., Pacific island rat ecology: Honolulu, Hawaii, Bernice P. Bishop Museum Bulletin, vol. 225, p. 21-38.

Koopman, K. F., 1997, The subspecies of *Emballonura semicaudata* (Chiroptera: Emballonuridae): Journal of Mammalogy, vol. 78, p. 358-360.

Kunz, T. H., ed., 1988, Ecological and behavioral methods for the study of bats: Washington, D.C., Smithsonian Institution Press.

Kunz, T.H., Tidemann, C. R., and Richards, G. C., 1996, Small volant mammals, *in* Wilson, D. E., Cole, F. R., Nichols, J. S., Rudran, R., and Foster, M. S., eds., Measuring and monitoring biological diversity—Standard methods for mammals: Washington, D.C., Smithsonian Institution Press, p. 122–146.

Lemke, T. O. 1986. Distribution and status of the sheath-tailed bat (*Emballonura semicaudata*) in the Mariana Islands: Journal of Mammalogy, vol. 67, p. 743-746.

McNab, B. K., 1969, The economics of temperature regulation in Neotropical bats: Comparative Biochemistry and Physiology, vol. 31, p. 227-268.

Neubaum, M. A., Douglas, M. R., Douglas, M. E., and O'Shea, T. J., 2007, Molecular ecology of the big brown bat (*Eptesicus fuscus*): genetic and natural history variation in a hybrid zone: Journal of Mammalogy, vol. 88, p. 1230–1238.

National Oceanic and Atmospheric Administration, 2008, Monthly station normals of temperature, precipitation, and heating and cooling degree days 1971-2000, 91 Pacific Islands: Asheville, North Carolina, National Climatic Data Center, Climatography of the United States No. 81, 11 p.

Orr, R. T., 1954, The natural history of the pallid bat, *Antrozous pallidus* (Le Conte): Proceedings of the California Academy of Sciences, vol. 28, p. 165-264.

O'Shea, T. J., and Vaughan, T. A., 1977, Nocturnal and seasonal activities of the pallid bat, *Antrozous pallidus*: Journal of Mammalogy, vol. 58, p.269-284.

Oustalet, M. E., 1895, Les mammiféres et les oiseaux des Iles Mariannes. Nouvelles Archives du Museum d'Histoire Naturelle Paris Série 3, vol. 7, p.141-228. (Cited by Lemke 1986).

Sanborn, C. C., 1949, Notes on the Caroline sheath-tailed bat (*Emballonura sulcata* Miller): Natural History Miscellanea The Chicago Academy of Sciences, vol. 48, p. 1-2.

Tate, G. H. H., 1934, Bats from the Pacific Islands, including a new fruit bat from Guam : American Museum Novitates, vol. 713, p. 1-3.

Tate, G.H.H., and Archbold, R., 1939, Results of the Archbold Expeditions. No. 23, a revision of the genus *Emballonura* (Chiroptera): American Museum Novitates, vol. 1035, p. 1-14.

U.S. Fish and Wildlife Service, 2007, 50 CFR Part 17; Endangered and Threatened Wildlife and Plants; Review of Native Species That Are Candidates for Listing as Endangered or Threatened; Annual Notice of Findings on Resubmitted Petitions; Annual Description of Progress on Listing Actions; Proposed Rule. Federal Register (December 6, 2007)72 (234): 69033-69106.

Vaughan, T. A., 1959, Functional morphology of three bats: Eumops, Myotis, Macrotus: University of Kansas Publications, Musuem of Natural History, vol. 12, 153 p.

Wiles, G. J., Engbring, J., and Otobed, D., 1997, Abundance, biology, and human exploitation of bats in the Palau Islands: Journal of Zoology (London), vol. 241, p. 203-227.

Worthington Wilmer, J., and Barratt, E., 1996, A non-lethal method of tissue sampling for genetic studies of chiropterans: Bat Research News, vol. 37, p. 1–3.

Yamashina, M. Y., 1943, On a new subspecies of *Emballonura semicaudata* from the Mariana Islands: Bulletin of the Biogeographical Society of Japan, vol. 13, p. 99-100. (In Japanese, cited by Lemke 1986).

Yamashina, M. Y., 1932, New species of bat from the Japanese South Pacific mandated islands: Transactions of the Natural History Society of Formosa, vol. 22, p. 240-241. (In Japanese, cited by Lemke 1986).

Table VI-1. Efforts made to capture Pacific sheath-tailed bats on Aguiguan in June and July 2008, with a summary of results and observations. Locations are in Universal Transverse Mercator (UTM), Zone 55 P, WGS 84 datum; elevations and estimated location errors are available on request.

Date	Location	Number and Lengths of Nets Deployed, Other Methods	Time	Results
26 June	Vicinity of Fault Line Cave 1 (=Cave 94, across gaps in boulders and cliffs, 344804 E, 1643215 N.	1-6 m, 1-9 m; bottoms 2.5-3.0 m above ground	18:00-22:00	In the dusk from 1846 to 1913 we watched 14 bats fly to within 1-2 m of the net and turn to the side, and 8 fly directly to net and then up and over the top. No captures.
28 June	Slope about 40 m below Guano Cave (= Cave 2) 343039E, 1642089N (EPE 8, elevation 100 m); mouth of Guano Cave	2-12 m nets in forest on slope below cave; 1-2.6 m net at mouth of Guano Cave. Intent to capture bats returning rather than bats emerging	18:30-22:00	No bats captured. Some bats returned to cave but likely entered above the net,
29 June	Vicinity of Fault Line Cave 1 (=Cave 94), across gaps in boulders and cliffs, 344804 E, 1643215 N; in forest on terrace ca. 100 m inland from Cave 94	1-9 m across gap, 1-18 m in forest. Insect sweep net as bats dispersed at dusk	18:00-22:00	Watched two bats circle go over or fly parallel to 9 m net within 1 m. Captured one male in sweep net. Male appeared "stunned" during 20 mins of handling. Put it on a cliff wall about 2.5 m above ground, where it remained for 30 mins, then flew off after disturbed by light from headlamp.
30 June	Slope about 40 m below Guano Cave (= Cave 2), 343039E, 1642089N (EPE 8, elevation 100 m)	1-12 m in forest on slope below cave	18:15-20:15	One adult female captured in mist net. Held in hand for 10 mins while measuring, photographing. Bat died.
1 July	Forest on terrace above East Black Noddy Cave (= Cave 76), 343939 E, 1642819 N	1-12 m, 1-18 m in "v" under canopy, in open flat section of younger forest. Bottoms 1.5 m above ground	18:00-22:15	No bats seen approaching the nets at dusk. Only 2 bats seen under canopy, both 1840-1850. One heard at 1907h, no further bat activity noticed.
1 July	East Black Noddy Cave (=Cave 76), 344004E, 1642923N (EPE 8 m, elevation 30m)	1-9 m across mouth of the smaller of the two caves. Bottom 1 m above ground	19:50-22:30	Bats were observed exiting the cave, and circled near the rim of the cliff, then dispersed. A bat was captured exiting the cave, after mass exodus, but escaped from net. A second bat, a pregnant female,

				was captured. It appeared stunned after handling for 13 mins, was placed on a rock wall but did not fly off until 20 mins later.
2-3 July	Forest on terrace below Fault Line Cave 1 area, 344767 E, 1643240 N	1-9 m, 3-12 m, 1-18 m. Bottoms 2 m above ground	18:00-00:15	Bats were observed foraging in the area, and on multiple occasions we watched them fly within 1 m of a net and turn. No bats captured.
3-4 July	Forest on terrace below Fault Line Cave 1 area, 344767 E, 1643240 N	2-9 m, 3-12 m, 2-18 m. Bottoms 2 m above ground.	18:00-00:15	Bats were observed foraging in the area, and on multiple occasions we watched them fly within 1 m of a net and turn. No bats captured.
5 July	East Black Noddy Cave (=Cave 76), 344004E, 1642923N (EPE 8, elevation 30m), small opening	1-9 m across mouth of the smaller of the two caves. Bottom 1 m above ground	20:30-22:30	Bats observed exiting cave. Audible calls heard from bats outside of cave, after exiting. These calls were similar to audible coaxing calls made by other species of bats elsewhere (e.g., *Antrozous pallidus*). Two bats were captured at the same time, with one capture of a bat entering the cave and the other exiting.
6 July	Vicinity of Cave 94, in boulders and cliffs above fault line, 344804 E, 1643215 N.	Insect sweep net	18:00-19:10	A minimum of 17 bats dispersed past and overhead, but none was caught in the insect sweep net. No captures.
10 July	Crevice Cave (= Cave 17), 0343541 E, 1641526 N	1-2.6 m net across opening during day, sweep net inside	Day	Two bats present, evaded capture.
11 July	Vicinity of Cave 94, in boulders and cliffs above fault line, 344804 E, 1643215 N	1- 6 m net 2-5 m above rocks; insect sweep net	18:00-20:00	Multiple bats dispersed past and overhead at dusk, one male caught in the insect sweep net. Placed in bag, held for 30 min and handled for 10 min additional, flew readily on release
12 July	Guano Cave (= Cave 2), 343031 E, 1642084 N	2-2.6 m nets on 3 m poles across opening to second chamber that bats use as a refuge when disturbed during the day. Disturbance was shining light until 1-2 bats fled the main chamber, repeated at 15-30 min intervals	14:00-16:30	Five bats (including one attached young) captured in three episodes. All immediately placed in cloth bags and held for 10-30 mins, female with young for 2 hrs. None was stunned or unable to fly off readily on release.
13 July	Vicinity of fault Line Cave 1	1- 6 m net 2-5 m above rocks;	18:00-20:00	Multiple bats dispersed past and overhead at dusk.

	(=Cave 94), in boulders and cliffs above fault line, 344804 E, 1643215 N	sweep net		One male caught in mist net, stunned, was sacrificed as voucher specimen. Second male caught in the insect sweep net as it veered away from the mist net. Placed in bag, held and handled over 30 min period, flew readily on release

Table VI-2 . Records of sheath-tailed bats captured at Aguiguan, June-July 2008. See Table VI-1 for details on locations and methods of capture, Table R-1 for information about reproductive status. Abbreviations: A = adult, F= female, M= male, ND = not determined, Y = young.

Field ID	Date	General Location	Age & Sex	Forearm, Body Mass	Samples Taken	Comments
STB-1	29 June	Vicinity of Cave 94	A M	43.5 mm, ND	Wing biopsy	No visible ectoparasites. Held 21 mins without placing in bag, appeared stunned afterwards. Placed on cliff face where it remained still for 30 mins before flying off.
STB-2	30 June	Vicinity of Cave 2	A F	45.0 mm, 7.6 g	Wing biopsy	No visible ectoparasites. Held in hand 10 mins while measuring, photographing, without placing in bag. Bat died. Voucher specimen, U.S. Geological Survey Colection, Museum of Southwestern Biology, study skin and skeleton (catalog no. MSB125656), organs (heart, kidney, liver, spleen, lungs, and muscle; catalog no. NK104278) in 95% ethanol.
STB-3	1 July	Cave 76	A F	45.3 mm, 7.7 g	Wing biopsy,	No visible ectoparasites. Bat appeared stunned after handling for 13 mins without holding in bag, was placed on a rock wall but did not fly off until 20 mins later.
STB-4	5 July	Cave 76	A F	46.1 mm, 6.5 g	Wing biopsy, 1 guano pellet	No visible ectoparasites.
STB-5	5 July	Cave 76	A F	45.5 mm, 8.0 g	Wing biopsy, 3 guano pellets	No visible ectoparasites.
STB-6	11 July	Vicinity of Cave 94	A M	45.9 mm, 5.8 g	Wing biopsy	No visible ectoparasites. Held in bag about 30 mins, handled 10 mins thereafter. Flew readily on release.
STB-7	12 July	Cave 2	A F	45.3 mm, 6.5 g	Wing biopsy	No visible ectoparasites. Held in cloth bag for ca. 30 mins, handled 10 mins, placed back in bag 90 min. Flew readily on release.
STB-8	12 July	Cave 2	Y F	[24.4 mm, 2.3 g]	None	Attached nursing young of STB-7.

STB-9	12 July	Cave 2	A F	43.8 mm, 5.8 g	Wing biopsy	No visible ectoparasites. Held in cloth bag about 15 mins, 5 mins handling. Flew readily on release.
STB-10	12 July	Cave 2	A F	47.0 mm, 7.0 g	Wing biopsy	No visible ectoparasites. Held in cloth bag about 25 mins, 5 mins handling. Flew readily on release.
STB-11	12 July	Cave 2	A F	46.4 mm, 5.8 g	Wing biopsy	No visible ectoparasites. Held for 10 mins in cloth bag, about 5 mins for handling. Flew readily on release.
STB-12	13 July	Vicinity of Cave 94	A M	46.0 mm, 5.7 g	Wing biopsy	No visible ectoparasites. Stunned after 10 mins of handling. Voucher specimen, U.S. Geological Survey Colection, Museum of Southwestern Biology, study skin and skeleton (catalog no. MSB125657), organs (heart, kidney, liver, spleen, lungs, and muscle; catalog no. NK104279) in 95% ethanol
STB-13	13 July	Vicinity of Cave 94	A M	45.5 mm, 5.1 g	Wing biopsy	No visible ectoparasites. Kept in cloth bag for about 25 mins, handled about 5 mins. Flew readily on release.

Table VI-3. Published data on forearm lengths and body mass of *Emballonura semicaudata* from throughout the species distribution in comparison with measurements of bats from Aguiguan. Measurements given are ranges or individual measurements. NR = not reported. Numbers in parentheses are sample sizes if different from sample description. Subspecies names and distribution follow Koopman (1997).

Sample	**Body Mass (range in g)**	**Forearm Length (range in mm)**	**Source**
E. semicaudata rotensis: 8 adult females, Aguiguan, 2008	5.8-8.0	43.8-47.0	This study
E. semicaudata rotensis: 4 adult males, Aguiguan, 2008	5.1-5.8 (3)	43.5-46.0 (4)	This study
E. semicaudata rotensis: 6 adult females, Aguiguan, 2003	NR	44.5-47.3	Esselstyn (unpublished)
E. semicaudata rotensis: 2 adult females, Aguiguan, 1984	5-7	44-45	Lemke (1986)
E. semicaudata rotensis: 1 male, 1 female	NR	45, 45	Yamashina (1943); T. Yamasaki (pers. commun.)
E. semicaudata semicaudata: 3 adult males, Fiji	5.5-5.5	44.4-45.5	Flannery (1995)
E. semicaudata semicaudata : 5 adult females, Fiji	6.5-8.0	45.3-47.9	Flannery (1995)
E. semicaudata semicaudata: type specimen	NR	41.0	Tate and Archbold (1939)
E. semicaudata palauensis: 4 males, 7 females	NR	39-44.5	Tate and Archbold (1939)
E. semicaudata palauensis: 2 females	NR	42-43	Johnson (1962)
Emballonura semicaudata sulcata: 8 females, 1 male, Pohnpei	NR	48.5-52.5	Sanborn (1949)
Emballonura semicaudata sulcata: 4 adult males, Pohnpei and Chuuk	4, 4 (2)	46-49.5	Tate (1934), Tate and Archbold (1939), Bruner and Pratt (1979)
Emballonura semicaudata sulcata: 4 adult females, Pohnpei and Chuuk	4-6 (3)	43-54	Bruner and Pratt (1979)
Emballonura semicaudata sulcata: 3 adult males, Pohnpei	7.0-7.5	49-51	Johnson (1962)
E. semicaudata sulcata: 11 adults, sex or location unspecified	NR	48.4-52.5	Literature summary in Lemke (1986)

Table **VI-4**. Observations of bats dispersing in the Fault Line area in the vicinity of Cave 66 and Cave 94, during June and July 2008. Observer initials in parentheses (EM = Elvin Masga, GJW= Gary J. Wiles, PMG = P. Marcos Gorresen, GC = Greg Camacho).

Location	**Date**	**Time 1st bat seen**	**Summary of observations**
Near Cave 66 344840, 1643270	24 June	18:49	18:49-19:13 (TJO): 17 bats observed flying through the forest below canopy, most uphill but 3 noted coming downhill. Heard ultrasonic calls on bat detector in isolation 39 times, and multiple times while bats observed flying back and forth in foraging beats (including feeding "buzzes" or pursuit calls). On at least 5 instances bats were feeding rather than flying directly through the area. Audible vocalizations (social calls) were heard on 3 occasions. Most feeding seemed to be at least 3 m above ground and over the fault line fissure. 19:14- 19:45 (TJO): 1 bat seen at 1935, heard echolocation calls on bat detector in isolation 9 times, 1 seen foraging 19:38-19:42 in headlight beam over an elliptical foraging beat with accompanying feeding buzzes on bat detector. Distinctive audible vocalizations heard on 3 occasions.
Near Cave 94, near 344804 E, 1643215 N	24 June	--	18:50-19:00 (EM): 47 bats observed flying uphill below canopy at this station
Near Cave 94, near 344804 E, 1643215 N	26 June	18:41	18;41-19:13 (GJW): Counted 25-30 bats (PMG nearby saw 43) coming up hillside from lower terrace to the northwest. None seen emerging from caves. Most traveled uphill, all under canopy, early bats easily visible in fading light. Some fly 2-3 feet above ground, some stop to circle and forage briefly, continue uphill. Collared kingfishers still active. PMG counts highest at 1900-1910. GC reported seeing a few split off and fly east low over the top of the canopy. 18:46-19:13 (TJO): 27 bats observed flying uphill below canopy, did not use bat detector.
Near Cave 94, near 344804 E, 1643215 N	29 June	18:37	18:37-18:42 (TJO): 2 bats dispersed uphill below canopy, 1 made several audible calls. Observation period limited because of bat capture.
Near Cave 94, near 344804 E, 1643215 N	6 July	18:50	18:50-19:08 (TJO): 12 bats observed flying uphill below canopy, 6 ultrasonic calls on bat detector, heard > 6 audible calls. Two fed together in the same elliptical foraging beat for two minutes 18:57-18:59; another two feeding separately at 19:02-19:03, feeding buzzes on bat detector. Light drizzle may have affected activity.
Near Cave 94, near 344804 E, 1643215 N	11 July	18:31	18:31-18:53 (TJO, EWV): 18 bats observed flying uphill below canopy, 10 ultrasonic calls on bat detector, heard > 30 audible calls. One bat foraged in an elliptical beat below the canopy and 4-8 m above ground for 1-2 mins. Observation period limited because of bat capture at 18:53.
Near Cave 94, near 344804 E, 1643215 N	13 July	18:36	18:36-18:53 (TJO): 7 bats observed flying uphill below canopy, did not use bat detector. Observation period limited because of bat capture at 18:53.

Table VI-5. Summary of records of standard museum specimens (study skins and skulls, or fluid preserved whole bats) of *Emballonura semicaudata*. This summary is based primarily on published records available to us through 2008 and queries of the online database maintained by the Global Biodiversity Information Facility (GBIF). It is not exhaustive, but probably includes most of the specimens housed in museum collections. Slight overlap may occur among sources.

Collecting Locality	**Number of Specimens**	**Museum Collection**	**Source of Information**
Aguiguan	2	American Museum of Natural History	Lemke (1986)
Aguiguan	2	Museum of Southwestern Biology	This study
American Samoa	1	Museum of Vertebrate Zoology, University of California Berkeley	GBIF
American Samoa	24	Museum of Vertebrate Zoology, University of California Berkeley	Helgen and Flannery (2002)
American Samoa	2	Bernice Pauahi Bishop Museum	GBIF
American Samoa	31	United States National Museum of Natural History	GBIF
Fiji	47	Harvard University Museum of Comparative Zoology	GBIF
Fiji	1	Los Angeles County Museum	GBIF
Fiji	9	United States National Museum of Natural History	GBIF
Fiji	2	Australian Museum, Sydney	Helgen and Flannery (2002)
Fiji	8	Australian Museum, Sydney	Colgan and Soheili (2008)
Fiji	15	Institut vor Taxonomie, Amsterdam	Helgen and Flannery (2002)
Fiji	1	Academy of Natural Sciences, Philadelphia	Helgen and Flannery (2002)
Guam	6	Museum National d'Histoire Naturelle, Paris	Lemke (1986)
Palau	68	United States National Museum of Natural History	GBIF
Palau	16	Yamashina Institute for Ornithology, Chiba, Japan	Yamashina (1932) cited by Lemke (1986); T. Yamasaki (pers.

			commun.)
Pohnpei	1	Florida Museum of Natural History	GBIF
Pohnpei	2	Bernice Pauahi Bishop Museum	GBIF
Pohnpei	7	California Academy of Sciences	GBIF
Pohnpei	21	Field Museum of Natural History	GBIF
Pohnpei	5	United States National Museum of Natural History	GBIF
Rota, CNMI	14	Yamashina Institute for Ornithology, Chiba, Japan	Yamashina (1943), T. Yamasaki (pers. commun.)
Samoa	7	Harvard University Museum of Comparative Zoology	GBIF
Samoa	16	Bernice Pauahi Bishop Museum	GBIF
Samoa	1	Bell Museum of Natural History	Helgen and Flannery (2002)
Tonga	2	Burke Museum, University of Washington	GBIF
Tonga	3	Los Angeles County Museum	GBIF
Tonga	7	United States National Museum of Natural History	GBIF
Tonga	1	Western Australia Museum	Helgen and Flannery (2002)
Truk	3	Louisiana State University Museum of Natural Science	GBIF
Truk and Pohnpei	32	Yamashina Institute for Ornithology, Chiba, Japan	Yamashina (1943) cited by Lemke (1986); T. Yamasaki (pers. commun.)
Vanuatu	1	Harvard University Museum of Comparative Zoology	GBIF; see also Helgen and Flannery (2002)
Vanuatu	1	British Museum of Natural History	Dobson (1878) cited in Koopman (1997)
Unspecified	19	Museum fur Naturkunde, Berlin	GBIF
Unspecified	1	Swedish Museum of Natural History	GBIF
Unspecified	1	Los Angeles County Museum	GBIF
Pohnpei	unspecified	American Museum of Natural History	Koopman (1997)

Unspecified	6	American Museum of Natural History	Griffiths and others 1991

Table VI-6. Summary statistics for temperature data (°C) from data logger sampling stations on Aguiguan, 2008.

Station	Period	N Hourly Readings	Computational Mean (95% CI)	Rounded Mean	Range	Min	Max
Camp							
Camp 1 m above ground	6/26, 14:30 to 7/13, 13:30	408	26.8 (26.55, 26.99)	27	9.7	22.3	32
Camp 2 m above ground	6/27, 14:30 to 7/13, 13:30	408	27.0 (26.77, 27.25)	27	10.0	22	32
Camp 3 m above ground	6/26, 14:30 to 7/13, 13:30	408	27.1 (26.90, 27.37)	27	10.5	22.3	32.8
Guano Cave							
Guano Cave entrance, 2 m high	6/30, 19:00 to 7/11 18:00	264	27.05 (27.01, 27.09)	27	1.3	26.3	27.5
Guano Cave 10 m depth, 2 m high	7/7, 17:00 to 7/11 16:00	96	26.58 (26.55, 26.60	27	0.5	26.5	27.0
Guano Cave inner chamber below bats, 1 m high	6/25, 15:00 to 7/12, 14:00	408	26.75 (26.71, 26.79)	27	2.5	25.3	27.8
Guano Cave inner chamber below bats, 2 m high	6/25, 15:00 to 7/12, 14:00	408	26.84 (26.79, 26.88)	27	3.0	25	28
Guano Cave inner chamber below bats, 6 m high	6/25, 15:00 to 7/12, 14:00	408	26.48 (26.44, 26.52)	26	2.5	25	27.5
Guano cave outer chamber 2 m high	7/7, 18:00 to 7/11, 17:00	96	26.26 (26.17, 26.34)	26	1.8	25.3	27
Guano cave outer chamber 3 m high	7/7, 18:00 to 7/11, 17:00	96	26.18 (26.10, 26.27	26	2	25	27
Guano cave outer chamber 4 m high	7/7, 17:00 to 7/11, 16:00	96	26.59 (26.53, 26.66)	27	1.5	26	27.5
Guano cave outer chamber 5 m high	7/7, 17:00 to 7/11, 16:00	96	27.09 (27.03, 27.15)	27	1	26.5	27.5
Guano cave outer chamber 6 m high	7/7, 18:00 to 7/11, 17:00	96	27.16 (27.12, 27.19)	27	0.75	26.75	27.5
Crevice Cave							
Crevice Cave at entrance, 2 m high	6/27, 18:00 to 7/9, 17:00	288	26.86 (26.81, 26.91	27	2.5	25.5	28
Crevice Cave at 3 m depth, 3 m high	6/27, 18:00 to 7/9, 17:00	288	26.60 (26.57, 26.63)	27	1.5	25.75	27.25
Crevice Cave at 5 m depth, 4 m	6/27, 18:00 to 7/9, 17:00	288	27.12 (27.10, 27.14)	27	0.75	26.75	27.5

high							
Fault Line Cave 1							
Fault Line Cave at entrance, 2 m high	6/30, 17:00 to 7/13, 16:00	312	26.11 (26.05, 26.18)	26	2.5	25	27.5
Fault Line Cave 1, 8 m depth, 1 m high	6/30, 17:00 to 7/13, 16:00	312	26.05 (25.98, 26.11)	26	3.0	24.5	27.5
Fault Line Cave 1, 8 m depth, 2 m high	6/30, 17:00 to 7/13, 16:00	312	26.20 (26.14, 26.26)	26	2.75	24.5	27.25
Fault Line Cave 1, 8 m depth, 3 m high	6/30, 17:00 to 7/13, 16:00	312	26.45 (26.39, 26.51)	26	2.5	25	27.5
New Cave 1							
New Cave 1, 10 m deep, 6 m above lower chamber	7/5, 18:00 to 7/9, 17:00	96	25.80 (25.75, 25.85)	26	1	25.25	26.25
Pillar Cave							
Pillar Cave at entrance, 2 m high	7/7, 19:00 to 7/11, 18:00	96	27.32 (27.23, 27.40)	27	1.5	26.5	28
Pillar Cave, 35 m deep, 2 m high	7/7, 2:000 to 7/11, 19:00	96	27	27	0	27	27
Dangkolo Cave							
Dangkolo Cave entrance 2 m high	6/27, 17:00 to 7/9, 16:00	288	27.14 (27.09, 27.18)	27	1.75	26.25	28
Dangkolo Cave 10 m depth, 1 m high	6/27, 17:00 to 7/9, 16:00	288	26.22 (26.21, 26.23)	26	0.5	26	26.5
Dangkolo Cave 20 m depth, 1 m high	6/27, 17:00 to 7/9, 16:00	288	26	26	0	26	26
Dangkolo Cave 45 m depth, 1 m high	6/27, 17:00 to 7/9, 16:00	288	26.5	26	0	26	26
Dangkolo Cave 45 m depth, 2 m high	6/27, 17:00 to 7/9, 16:00	288	26.5	26	0.25	26.25	26.5
Dangkolo Cave 45 m depth, 3 m high	6/27, 17:00 to 7/9, 16:00	288	26.5	26	0	26.5	26.5
Cave 68							
Cave 68, rear (uphill) chamber, 5 m above ground, 3 m below opening	7/3, 17:00 to 7/13, 16:00	240	25.72 (25.66, 25.78)	26	2	24.75	26.75

Draft

Cave 68, 18 m depth from lower entrance, 2 m high	7/3, 17:00 to 7/13, 16:00	240	26.23 (26.18, 26.27)	26	1.25	25.5	26.75

Table VI-7. Summary statistics for temperature data (°C) from data logger sampling stations on Aguiguan, 2008, during the period from 7 July (19:00 h) to 10 July (15:00 h) when data were recorded simultaneously at all locations . For each station *N* = 68 hourly readings.

Station	**Computational Mean (95% CI)**	**Rounded Mean**	**Range**	**Minimum**	**Maximum**
Camp 1 m above ground	25.5 (25.14, 25.83)	26	6	23	29
Camp 2 m above ground	25.6 (25.26, 25.99)	26	6.5	23	29.5
Camp 3 m above ground	25.8 (25.41, 26.17)	26	7	23.3	30.3
Guano Cave entrance	26.8 (26.74, 26.90)	27	1.3	26.3	27.5
Guano Cave 10 m depth, 2 m high	26.60 (26.56, 26.63)	27	0.5	26.5	27.0
Guano Cave inner chamber below bats 1 m high	26.57 (26.49, 26.65)	27	1.3	25.8	27.0
Guano Cave inner chamber below bats, 2 m high	26.63 (26.55, 26.70)	27	1	26	27
Guano Cave inner chamber below bats, 6 m high	26.25 (26.18, 26.32)	26	1	25.8	26.8
Guano cave outer chamber 2 m high	26.24 (26.14, 26.35)	26	1.7	25.3	27
Guano cave outer chamber 3 m high	26.17 (26.06, 26.27)	26	2	25	27
Guano cave outer chamber 4 m high	26.58 (26.51, 26.66)	27	1	26	27
Guano cave outer chamber 5 m high	27.09 (27.01, 27.17)	27	1	26.5	27.5
Guano cave outer chamber 6 m high	27.19 (27.14, 27.23)	27	0.75	26.75	27.5
Crevice Cave					
Crevice Cave at entrance, 2 m high	26.68 (26.54, 26.82)	27	2	25.75	27.75
Crevice cave at 3 m depth, 3 m high	26.43 (26.32, 26.54)	26	2	25.75	27.75
Crevice Cave at 5 m depth, 4 m high	27.01 (26.95, 27.06)	27	1.25	26.75	28
Fault Line Cave 1					
Fault Line Cave at entrance, 2 m high	25.78 (25.68, 25.88)	26	1.5	25	26.5
Fault Line Cave 1, 8 m depth, 1 m high	25.89 (25.77, 26.01)	26	1.5	25	26.5
Fault Line Cave 1, 8 m	25.99 (25.89, 26.09)	26	1.5	25	26.5

depth, 2 m high					
Fault Line Cave 1, 8 m depth, 3 m high	26.29 (26.20, 26.38)	26	1.5	25.5	27
New Cave 1					
New Cave 1, 10 m deep, 6 m above lower chamber	25.71 (25.64, 25.78)	26	1	25.25	26.25
Pillar Cave					
Pillar Cave at entrance, 2 m high	27.35 (27.23, 27.46)	27	1.5	26.5	28
Pillar Cave, 35 m deep, 2 m high	27	27	0	27	27
Dangkolo Cave					
Dangkolo Cave entrance 2 m high	26.74 (26.66, 26.83)	27	1.5	26	27.5
Dangkolo Cave 10 m depth, 1 m high	26.09 (26.06, 26.12)	26	0.5	25.75	26.25
Dangkolo Cave 20 m depth, 1 m high	26	26	0	26	26
Dangkolo Cave 45 m depth, 1 m high	26.5	26	0	26.5	26.5
Dangkolo Cave 45 m depth, 2 m high	26.45 (26.42, 26.47)	26	0.25	26.25	26.5
Dangkolo Cave 45 m depth, 3 m high	26.5	26	0	26.5	26.5
Cave 68					
Cave 68 rear (uphill) chamber, 5 m above ground, 3 m below opening	25.65 (25.57, 25.73)	26	1.25	25	26.25
Cave 68, 18 m depth from lower entrance, 2 m high	26.16 (26.11, 26.22)	26	0.75	25.75	26.5

Table VI-8. Relative humidity at caves and other locations on Aguiguan, 2008.

Date	**Time**	**Location**	**Relative Humidity**
5 July	17:00	New Cave 1, 10 m depth	96%
6 July	13:45	Fault Line Cave 1, 18 m depth	92%
7 July	15:15	Guano Cave, 10 m depth	92%
7 July	18:20	Pillar Cave at 35 m 1 m high	92%
10 July	16:00	Dangkolo Cave at 40 m depth	96 %
6 July	13:45	Outside mouth of Fault Line Cave 1	88%
7 July	15:30	Outside mouth of Guano Cave	84%
7 July	18:25	Outside mouth of Pillar Cave	92% (drizzling)
4 July	11:15	Camp	74%
4 July	11:18	Camp	75%
5 July	09:40	Camp	81%
5 July	12:00	Camp	78%

Section VII. Reproduction of Pacific Sheath-Tailed Bats (*Emballonura semicaudata rotensis*) on Aguiguan, Commonwealth of the Northern Mariana Islands

Thomas J. O'Shea and Ernest W. Valdez

ABSTRACT

There is very little information available about reproduction in Pacific sheath-tailed bats or other species of the genus *Emballonura*. Basic information about ecological aspects of reproduction is important for understanding the population dynamics of rare mammals. We found that 7 of 8 adult females we captured at two caves on Aguiguan in June and July 2008 were reproductive (5 were pregnant and 2 were lactating). A pregnant female was reported in the literature in June 1984, but none of six adult female bats examined in September of 2003 by others were pregnant. We also observed 11 pups at roosts in June and July 2008, but captured no volant young of the year. These observations suggest that Pacific sheath-tailed bats on Aguiguan may have a diffuse seasonality in reproduction, timing the period of late gestation, lactation, and maturation of young to coincide with the late June-early November rainy season. We observed one large embryo in a female dissected in June 2008, as was also observed in a female dissected in June 1984, and each of the pups we observed in caves were singletons. These observations suggest a litter size of one. Although additional sampling is needed to fully understand patterns of reproduction in these bats on Aguiguan, findings are consistent with the scant literature on other species of *Emballonura* and with the population dynamics of many other species of bats. If reproduction occurs annually and litter size is one, then the capacity for population growth in Pacific sheath-tailed bats will be very limited.

INTRODUCTION

Basic information on ecological aspects of reproduction is essential to understanding the population dynamics of rare species of mammals. Little information is available in the literature on reproduction in Pacific sheath-tailed bats in the Mariana Islands or elsewhere. Lemke (1986) reported that one of two females captured at Guano Cave on Aguiguan on 22 June 1984 was pregnant with a single fetus. On 17 September 2003 six females were captured at Guano Cave, palpated, and released by Jake Esselstyn (personal communication) and others. None of the bats handled by Esselstyn was palpably pregnant. We are unaware of any other records describing reproduction in this species in the Mariana Islands or anywhere else in its range. Our objectives in this section are to highlight and synthesize the limited findings pertinent to this topic based on our investigations on Aguiguan in 2008 and related information.

MATERIALS AND METHODS

We employed standard field methods for assessing reproductive condition in bats. Females were palpated to determine pregnancy, the condition of the teats was assessed to determine if bats were in lactation, and males were examined to determine if testes or cauda epididymides were engorged (Racey 1988). Age classes (adult or large volant juvenile) were assigned based on the degree of closure of the phalangeal epiphyses (Anthony 1988). Reproductive condition was also assessed by examination of internal organs of two bats saved as voucher specimens. We also report on the reproductive condition of six bats examined by Jake Esselstyn and others at Guano Cave in 2003 following similar techniques. Observations of young bats in caves were made using the night vision equipment described in Section III of this administrative report.

RESULTS AND DISCUSSION

Reproductive status of adult female bats captured at two caves on Aguiguan from late June to mid-July 2008 show that most (7 of 8, or 87%) were in active reproductive condition (5 pregnant and 2 lactating). These and other records suggest a hypothesis that birthing in Pacific sheath-tailed bats on Aguiguan is diffusely seasonal and timed to coincide with the rainy season. General ecosystem productivity (including small insects) is likely to be higher in the rainy season than in the dry months. The general seasonal pattern in the region is that almost four times as much rainfall occurs in the wet season than in the dry season (Lander 2004). The lowest monthly rainfall occurs in December through May. Rainfall increases in June, is highest in July through October, and then decreases in November (Lander 2004). In other tropical areas of the world, insectivorous bats time their reproduction to coincide with rainy season productivity and are non-reproductive during dry seasons; this is thought to be due to increases in food abundance as a result of seasonably predictable rainfall (*e.g*. Bernard and Cumming 1997; Fleming and others 1972), and accompanying higher insect abundance (McWilliam 1987; Bradbury and Vehrencamp 1976).

Although sampling efforts have been very limited, several lines of evidence support a hypothesis of diffuse seasonal reproduction in Pacific sheath-trailed bats on Aguiguan. Esselstyn (pers. comm.) and others captured six adult females at Guano Cave later in the rainy season on 17 September 2003. None of these was pregnant. Using the night viewing device, we observed a pup on 23 and 27 June in the roost at Crevice Cave that also held 2-3 adults; 10 smaller bats were each observed roosting in very close association with 10 single larger adults within the colony at Guano Cave on 25 June. We assume these smaller bats at Guano Cave each were juveniles roosting with their mothers. Volant juvenile bats can be readily distinguished from adults based on epiphyseal closure and ossification for several months after birth (Anthony 1988). All of the 12 Pacific sheath-tailed bats we caught in flight at three different locations in June and July were adults (Table VII-1); the absence of readily discernible young bats in our samples suggests that reproduction was limited over the preceding few months of the late dry season. The pregnant females we handled all seemed to be in fairly advanced pregnancy based on abdominal distension, and this was supported by the presence of a large fetus (crown-rump length of 23 mm) in the museum voucher specimen taken on 30 June (see Section VI of this administrative report). The fetus observed by Lemke (1986) on 22 June 1984 was advanced but slightly smaller at 19 mm in crown-rump length. The other female he examined, however, was non-reproductive. In addition to possible seasonality in birthing, our observations of roosting pups, the single attached young caught with its mother at Guano Cave (Table VII-1, see also Table VI-2, Section VI of this administrative report), and the dissections of two females by us and Lemke (1986) all suggest a litter size of one. Very little information is available on litter size in other species in the genus *Emballonura*, but litter sizes of one seem consistent with our observations of Pacific sheath-tailed bats on Aguiguan. A litter size of one has been reported for *Emballonura tiavato* and *Emballonura atrata* on Madagascar, where reproduction may be seasonal but available data are also limited (Goodman and others 2006). Only single embryos have been reported thus far in the few samples of *Emballonura beccarii* and *Emballonura dianae* that have been taken on Papua New Guinea (Bonaccorso 1998), and single embryos have been reported for Emballonura monticola (Nowak 1999). A seasonal birthing period coinciding with the rainy season is known for other emballonurids, but the pattern in Pacific sheath-tailed bats on Aguiguan suggested by our data could be more complex and will require additional sampling to verify. For example, some Neotropical emballonurids also give birth to singletons once annually in synchrony with rainy seasons, whereas other species of emballonurids may have more than one birthing period each year (Bradbury and Vehrencamp 1976).

The single male that we dissected had no swelling of the cauda epididymides and small testes (measuring 1 x 3 mm) that were withdrawn into the inguinal canal, indicating that it was not in mating condition. The other three males we captured showed no external evidence of distended testes or epididymides. Perhaps mating occurs earlier in the year during the dry season. This would be compatible with mating systems of other emballonurids which include defending access to feeding areas (e.g. Bradbury and Emmons 1974, Bradbury and Vehrencamp 1976), if food is a more limited resource in the dry season.

Although bats are a diverse group with about 1,200 species worldwide and can show a concomitant variability in life history traits, most have limited potential for population growth based on reproduction alone (and thus require high adult survival to prevent population declines). Sexual maturity in bats is usually not reached until one year of age or older, birth typically occurs once annually (but some species of tropical bats in aseasonal environments may reproduce year-round), and litter size is small but can vary among species and habitats from one to four, with most producing one or two young at parturition (see review of ecological aspects of bat reproduction in Racey and Entwistle 2000). Our observations suggest that Pacific sheath-tailed bats on Aguiguan may be at the low range of reproductive potential for bats if they give birth once annually with litter sizes of one. This low reproductive potential will increase the time required for the population to recover and reach carrying capacity.

Acknowledgments

We thank E. Masga, F. Muna, and J. Omar for shepherding us to various locations around Aguiguan and facilitating all aspects of this work, including captures of bats at Guano Cave. F. Bonaccorso also helped with captures of bats at Guano Cave, and he, M. Gorresen, and G. Wiles contributed many useful discussions in the field. P. Cryan and G. Wiles provided comments on the manuscript. The research we report on herein was sponsored by the U.S. Fish and Wildlife Service (USFWS), with C. Kessler of the Honolulu office coordinating and guiding the project. Bat captures and collecting took place under authority of Scientific Research Permit 01048-08 issued by the Commonwealth of the Northern Mariana Islands. Any use of trade, product, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the U.S. Government.

References Cited

Anthony, E. L. P., 1988, Age determination in bats, *in* Kunz, T. H., ed., Ecological and behavioral methods for the study of bats: Washington, D.C., Smithsonian Institution Press, p. 47–58.

Bernard, R. T. F., and Cumming, G. S., 1997, African bats: evolution of reproductive patterns and delays: Quarterly Review of Biology, vol. 72, p. 253-274.

Bonaccorso, F. J., 1998, Bats of Papua New Guinea: Washington, D.C., Conservation International, 489 p.

Bradbury, J. W., and Emmons, L. H., 1974, Social organization of some Trinidad bats. I. Emballonuridae: Zeitschrift für Tierpsychologie, vol. 36, p. 137-183.

Bradbury, J. W., and Vehrencamp, S. L., 1976, Social organization and foraging in emballonurid bats: Behavioral Ecology and Sociobiology, vol. 1, p. 337-381.

Fleming, T. H., Hooper, E. T., and Wilson, D. E., 1972, Three Central American bat communities: structure, reproductive cycles, and movement patterns: Ecology, vol. 53, p. 555-569.

Goodman, S. M., Cardiff, S. G., Ranivo, J., Russell, A. L., and Yoder, A.D., 2006, A New Species of *Emballonura* (Chiroptera: Emballonuridae) from the Dry Regions of Madagascar: American Museum Novitates, no. 3538, p. 1-24.

Lander, M. A., 2004, A rainfall climatology for Saipan: distribution, return-periods, El Niño, tropical cyclones, and long-term variations: Mangilao, Guam, University of Guam Water and Environmental Research Institute of the Western Pacific, Technical Report 103, 50 p.

Lemke, T. O., 1986, Distribution and status of the sheath-tailed bat (*Emballonura semicaudata*) in the Mariana Islands: Journal of Mammalogy, vol. 67, p. 743-746.

McWilliam, A. N., 1987, The reproductive and social biology of *Coleura afra* in a seasonal environment, *in* Fenton, M. B., Racey, P., and Rayner, J. M. V. , eds., Recent advances in the study of bats: Cambridge, England, Cambridge University Press, p. 324-350.

Nowak, R. M., 1994, Walker's Bats of the World: Baltimore, Maryland, Johns Hopkins University Press. 287 p.

Racey, P. A., 1988, Reproductive assessment in bats, *in* Kunz, T.H., ed., Ecological and behavioral methods for the study of bats: Baltimore, Md., Smithsonian Institution Press, p. 31–46.

Racey, P. A., and Entwistle, A. C., 2000, Life-history and reproductive strategies of bats, *in* Crichton, E. G., and Krutzsch, P. O. H., eds., Reproductive biology of bats: Academic Press, San Diego, CA., p. 363-414

Table VII-1. Summary of reproductive status of bats captured on Aguiguan, June-July 2008.

Field ID	Date	General Location	Age & Sex	Reproductive Condition
STB-1	29 June	Vicinity of Fault Line Cave 1 (Cave 94)	A M	Non-reproductive
STB-2	30 June	Vicinity of Guano Cave (Cave 2)	A F	Pregnant
STB-3	1 July	East Black Noddy Cave (Cave 76)	A F	Pregnant
STB-4	5 July	East Black Noddy Cave (Cave 76)	A F	Lactating
STB-5	5 July	East Black Noddy Cave (Cave 76)	A F	Pregnant
STB-6	11 July	Vicinity of Fault Line Cave 1 (Cave 94)	A M	Non-reproductive
STB-7	12 July	Guano Cave (Cave 2)	A F	Lactating
STB-8	12 July	Guano Cave (Cave 2)	Y F	Small nursing young attached to adult
STB-9	12 July	Guano Cave (Cave 2)	A F	Pregnant
STB-10	12 July	Guano Cave (Cave 2)	A F	Pregnant
STB-11	12 July	Guano Cave (Cave 2)	A F	Non-reproductive
STB-12	13 July	Vicinity of Fault Line Cave 1 (Cave 94)	A M	Non-reproductive
STB-13	13 July	Vicinity of Fault Line Cave 1 (Cave 94)	A M	Non-reproductive

Section VIII. Sampling Guano for Organochlorine Insecticides and Other Contaminants

Thomas J. O’Shea and Ernest W. Valdez

ABSTRACT

Past studies have shown that guano of bats can be used to assess the degree of contamination with organochlorine pesticides and the likelihood that the observed level of exposure has caused bat mortality. The scientific literature on conservation of Pacific sheath-tailed bats speculates that past organochlorine pesticide contamination may have played a role in their decline in some areas. However, Pacific sheath-tailed bats have never been assessed for organochlorine contaminants anywhere in their range. We obtained a small number of guano samples at different depths in an area of accumulation at Guano Cave on Aguiguan using chemically cleaned glass jars with teflon-lined lids. These samples are stored at our laboratory and can be made available for chemical analysis. They may be particularly useful for comparison with samples that may be taken in the future at other islands where these bats have declined, because there is no known history of organochlorine pesticide use on Aguiguan. However, our samples likely include some guano from swiftlets. Attempts to assess the degree of mixing of the two kinds of guano based on microscopic examination will be necessary prior to analysis of these and any comparative samples from other locations.

INTRODUCTION

There has been speculation that exposure to insecticides (particularly the persistent organochlorines) has been a contributing cause of the decline of Pacific sheath-tailed bats in the Mariana Islands and elsewhere (Esselstyn and others 2004, Flannery 1995, Tarburton 2002). The organochlorine insecticide DDT and the organophosphate malathion were applied in Guam, Saipan, and Tinian between the 1940s and 1970s (Baker 1946, Townes 1946, Drahos 1977, Jenkins 1983). Researchers also have speculated that declines in swiftlets and sheath-tailed bats on Guam were linked (Lemke 1986). However, there is no firm chemical or toxicological supporting evidence that the disappearance of swiftlets was a result of pesticide exposure. Concentrations of DDE (a major break-down product of DDT) in swiftlet tissues and swiftlet guano samples from Guam measured in 1981 were much less than those associated with avian mortality or reproductive failure in studies of other species of birds (Grue 1985), and an order of magnitude less than concentrations in bat guano that have been linked to mortality or population declines in other species of insectivorous bats (Clark and others, 1982, 1995; Clark and Shore 2001). Nonetheless there continue to be suspicions that swiftlet and bat declines in years past may have been a result of exposure to DDT or other pesticides (Cruz and others 2008).

Concentrations of contaminants have never been measured in sheath-tailed bat guano or tissues. DDT was used extensively in Palau in the 1940s (Baker 1946) but sheath-tailed bats have been characterized as abundant there in the recent past (Wiles and others 1997). The carbamate and organophosphate insecticides that were also widely used globally are not persistent in tissues or guano of birds or mammals, and exposure of bats or swiftlets to these compounds has not been assessed. However, it has been speculated that these additional classes of insecticides were likely responsible for deaths of bats in the United States in the 1960s (Clark and Shore 2001), so that such past poisoning also seems feasible (although only speculatively so) in the Mariana Islands. It is unknown but seemingly unlikely that there is any history of major organochlorine insecticide use on Aguiguan (most of the plantation activity of the Japanese colonists was prior to the 1940s when other substances may have been used for pest control, including elemental based compounds; recognition and introduction of DDT as an insecticide began in the 1940s (Metcalf 1973). However, count data suggest that the population of sheath-tailed bats on Aguiguan was much reduced by the early 1980s (see Section III of this administrative report).

Determining concentrations of contaminants in samples of guano from sheath-tailed bats on Aguiguan might be instructive, particularly in relation to samples from elsewhere in the CNMI or Guam where sheath-tailed bats have disappeared. Concentrations of organochlorines (particularly DDT, DDE, and dieldrin) in guano that are associated with declines or direct mortality in other species of insectivorous bats have been established (Clark and others 1982) and used to evaluate the likely impacts of exposure on bat populations (e.g., Clark and others 1988, 1995, O'Shea and others 2001). Comparison of concentrations of organochlorines in samples of guano from sheath-tailed bats on Aguiguan with samples from other colonies of sheath-tailed bats that have gone extinct (e.g. on Guam) might also be useful in evaluating the degree of any past threat associated with these substances. Therefore we also sought to use careful protocols to collect a small series of guano samples from below roosts of sheath-tailed bats in Guano Cave on Aguiguan for archival and possible future analyses.

METHODS AND MATERIALS

Samples were collected in Guano Cave directly below the high domed ceiling where the bats roost daily (see Sections III and V of this administrative report), and which does not appear to be heavily used by nesting cave swiftlets. However, mixing of swiftlet guano with sheath-tailed bat guano could have occurred, particularly over the long periods of time during which the guano pile was formed. Microscopic examination of subsamples will be required to quantify the extent of this mixing (see below). Samples were placed in chemically cleaned 120 ml glass jars with teflon-lined lids that were pre-washed following EPA procedures and specifications (Eagle Picher Lot G3255020). The samples were removed from the surface layer, and from depths at 10 and 20 cm, using a stainless steel spoon wrapped in aluminum foil, with the foil changed between each sample. Duplicate samples were taken from areas about 15 cm apart. Original teflon-lined lids were sealed to the jars using adhesive tape.

RESULT AND DISCUSSION

We obtained six samples of guano at three depths below the sheath-tailed bat roosting area in Guano cave (Table VIII-1). The amounts listed are similar to those collected in other studies of contaminants in guano of insectivorous bats (e.g. Clark and others 1995) and sufficient to allow duplicate analyses of subsamples. Given the small numbers of bats utilizing this site over the last few decades (Section III of this administrative report), we suspect the three different depths represent substantial differences in dates of deposition. However, there are no means to calibrate the layers of guano at this cave, and interpretations of differences in findings at the three depths will only be relative to each other. These samples are currently stored at room temperature at the Fort Collins Science Center and can be made available for analysis, particularly if similar material can be obtained at additional locations for comparison. Unlike animal tissues, which must be kept frozen in storage and shipment, bat guano can be stored at room temperature, but samples or subsamples should be dried in a dessicator to constant dry weight prior to any subsampling for chemical analyses (e.g. Clark and others 1982, 1995, O'Shea and others 2001). Future chemical analyses should attempt to include as many persistent contaminants as possible (e.g. metals as well as organochlorines) given the logistic difficulty of obtaining these samples. It would also be useful to examine small subsamples microscopically to verify by degree of mastication that the material taken at various depths below the surface layer are primarily of bat origin rather than from swiftlets (see Section V of this administrative report). We recommend that these samples be analyzed for a range of persistent contaminants in conjunction with samples that may be obtained from caves on Guam and from elsewhere in the CNMI where these bats formerly roosted but no longer occur.

REFERENCE CITED:

Baker, R. H., 1946, Some effects of the war on the wildlife of Micronesia: Transactions of the North American Wildlife Conference, vol. 11, p. 205-213.

Clark, D.R., Jr., and Shore, R.F., 2001, Chiroptera, in Shore, R.F. and Rattner, B.A., eds., Ecotoxicology of Wild Mammals: London, John Wiley & Sons, Ltd., p. 159-214.

Clark, D.R., Jr., Bagley, F.M., and Johnson, W.W., 1988, Northern Alabama colonies of the endangered grey bat *Myotis grisescens* - organochlorine contamination and mortality: Biological Conservation, vol. 43, p. 213-225.

Clark, D.R., Jr., LaVal, R.K., and Tuttle, M.D., 1982, Estimating pesticide burdens of bats from guano analyses: Bulletin of Environmental Contamination and Toxicology, vol. 29, p. 214-220.

Clark, D.R., Jr., Moreno-Valdez, A., and Mora, M.A., 1995, Organochlorine residues in bat guano from nine Mexican caves, 1991: Ecotoxicology, vol. 4, p. 258-265.

Cruz, J B., Kremer, S. R., Martin, G., Williams, L. L., and Camacho, V. A., 2008, Relative abundance and distribution of Mariana swiftlets (Aves: Apodidae) in the Northern Mariana Islands: Pacific Science, vol. 62, p. 233-246.

Drahos, N., 1977, Status of the Vanikoro swiftlet on Guam (*Collocalia vanikorensis* Quoy and Gaimard): Mangilao, Guam , Division of Aquatic and Wildlife Resources.

Esselstyn, J. A., Wiles, G.J., and Amar, A., 2004, Habitat use of the Pacific sheath-tailed bat (*Emballonura semicaudata*) on Aguiguan, Mariana Islands: Acta Chiropterologica, vol. 6, p. 303-308.

Flannery, T. 1995. Mammals of the South-West Pacific & Moluccan Islands: Ithaca, NY., Cornell University Press.

Grue, C. E., 1985, Pesticides and the decline of Guam's native birds: Nature, vol. 316, p. 301.

Helgen, K. M., and Flannery, T. F., 2002, Distribution of the endangered Pacific sheath-tailed bat *Emballonura semicaudata*: Australian Mammalogy, vol. 24, 209-212.

Jenkins, J. M., 1983, The native forest birds of Guam: Ornithological Monographs, vol. 31, p. 1-61.

Lemke, T. O., 1986, Distribution and status of the sheath-tailed bat (*Emballonura semicaudata*) in the Mariana Islands: Journal of Mammalogy, vol. 67, p. 743-746.

Metcalf, R.L., 1973, A century of DDT: Journal of Agricultural and Food Chemistry, vol. 21, p. 511-519.

O’Shea, T.J., Everette, A.L., and Ellison, L.E., 2001, Cyclodiene, DDE, DDT, arsenic and mercury contamination of big brown bats foraging at a Colorado superfund site: Archives of Environmental Contamination and Toxicology, vol. 40, p. 112-120.

Steadman, D. W., 1999, The prehistory of vertebrates, especially birds, on Tinian, Aguiguan, and Rota, Northern Mariana Islands: Micronesica, vol. 31, p. 319-345.

Tarburton, M. K., 2002, Demise of the Polynesian sheath-tailed bat *Emballonura semicaudata* in Samoa: Micronesica, vol. 34, p.105-108.

Townes, H. K., 1946, Entomology. Part I. Non-agricultural plants. Economic Survey of Micronesia, Vol. 12: Honolulu, Hawaii., U.S. Commercial Company

Wiles, G. J., Engbring, J., and Otobed, D., 1997, Abundance, biology, and human exploitation of bats in the Palau Islands: Journal of Zoology (London), vol. 241, p. 203-227.

Table VIII-1. List of samples of guano obtained from Guano Cave for possible future analyses for contaminants.

Sample Jar Serial Number	USGS Sample Id	Depth of Sample	Approximate Sample Mass
A1374270	7-12-08 S-1	Surface layer at location 1	14 g
A1374275	7-12-08 S-2	Surface layer at location 2	12 g
A1374261	7-12-08 10-cm 1	10 cm at location 1	20 g
A1374287	7-12-08 10-cm 2	10 cm at location 2	16 g
A1374256	7-12-08 20-cm 1	20 cm at location 1	24 g
A1374290	7-12-08 20-cm 2	20 cm at location 2	24 g

Section IX. Assessment for Pacific Sheath-Tailed Bats (*Emballonura semicaudata rotensis*) on Tinian, Commonwealth of the Northern Mariana Islands

Thomas J. O'Shea, Ernest W. Valdez, and Frank J. Bonaccorso

ABSTRACT

Pacific sheath-tailed bats are unknown from Tinian except from prehistoric deposits in caves. We used ultrasonic detectors to sample for the presence of echolocating bats in areas of native forest on Mount Lasso and in the Kastiyu Forest on four nights. We did not detect any bats. We did not see any bats in flight during this sampling, and did not receive verbal reports from knowledgeable residents that would indicate their presence. We also found no records of specimens in museums. Native forest habitat was found to be very important in supporting the population of sheath-tailed bats on Aguiguan. However, native vegetation was impacted by Chamorro settlers prior to European contact, and by subsequent introduced ungulate grazing; thereafter most of the native forest on Tinian was cleared for sugar cane plantations by Japanese colonists in the 1920s and 1930s. If sheath-tailed bats survived this loss of foraging habitat, their roosts in caves may have been destroyed by military activities in World War II. Our lack of findings provides further negative evidence that these bats occur on Tinian, but our sampling efforts were very limited. Additional sampling with echolocation detectors across a wider area of forest and searches of caves are needed to further rule out the possibility that a relict population of these bats may remain on Tinian.

INTRODUCTION

Pacific sheath-tailed bats are not known from Tinian other than in prehistoric times. Bones of this species have been found in three caves, crevices or rock shelters in the Carolinas area of Tinian in prehistoric stratigraphic layers, at least one of which has been dated at 2,400-2,200 years before present by radiocarbon analysis (Steadman 1999). In contrast, the contemporary presence of sheath-tailed bats on Tinian has never been documented by biologists, and there are no specimens (other than zooarchaeological material) from Tinian known in museum collections (Lemke 1986, Helgen and Flannery 2002; see also Section VI of this administrative report). We had a limited amount of time available to make an assessment for the presence of Pacific sheath-tailed bats on Tinian. We arrived on Tinian during the day of 15 July 2008 and were scheduled to leave on 19 July. We decided that with 4 days and nights available the most efficient approach to identify the presence of these bats would be to use echolocation detectors in likely foraging habitat each night, rather than search caves in cliffs during the day. Stafford (2003) had previously conducted a geological inventory of 88 caves on Tinian, and did not see roosting bats in them (K. W. Stafford, personal communication 2008). Detection of even a single bat echolocation call would provide evidence to support additional survey effort in the future. We concentrated our use of echolocation detector equipment to native forest on Tinian, based on findings on Aguiguan that foraging and associated echolocation activity of Pacific sheath-tailed bats is limited primarily to this habitat (Esselstyn and others 2004; see also Section IV of this administrative report).

MATERIALS AND METHODS

We chose two widely separated stands of native forest to sample echolocation activity. One was a section of forest on Mount Lasso in the north central interior of Tinian. This area of the forest was also used during herpetological surveys in 2008 (G. Rodda, pers. comm.). The second area we sampled was 10 km to the southeast of the Mount Lasso area on the southeastern part of Tinian in the Kastiyu Forest, as recommended by CNMI Division of Fish and Wildlife staff (E. Masga and T. Castro, pers. comm.). This area was closer to sections of sea cliffs with caves than was the Mount Lasso forest, although a few small caves were also known from the Mount Lasso area.

On 15 July 2008 we activated two fixed echolocation detector stations using some of the equipment that had been used in the Aguiguan study (Section IV of this administrative report). These stations were located on a trail through the forest at Mount Lasso. Location 1 was located on Mount Lasso along the forest edge at a clearing near the trailhead, 352715E, 1663323N, elevation 196 m. Location 2 was under the canopy at an area with limited understory vegetation about 525 m NNW of Location 1, at 352609E and 1663835N, elevation 156 m. Each Anabat sampling station was programmed to sample continuously all night long (see Section IV of this administrative report for more details on the equipment and methods used at sampling stations). On 16 July 2008 we placed the echolocation detector units in the Kastiyu Forest. Location 1 in the Kastiyu Forest was at 355660E and 1653903N, elevation 161 m. The second station at the Kastiyu forest was about 400 m SSE of Location 1, at 355732E and 1653511N, elevation 158 m. Both locations where detector stations were established in the Kastiyu Forest area were below the canopy, about 30 m interior from the forest edge.

In addition to the fixed sampling stations that automatically collected echolocation activity all night long, we also sampled using *ad hoc* walking transects and hand-held Anabat II SD1 CF ultrasonic detectors at Mount Lasso and the Kastiyu Forest on 17 and 18 July 2008. These transects were sampled from dusk until about 2130 h, typically a period of peak detection of foraging bats on Aguiguan (see Section IV of this administrative report). At the Mount Lasso site we walked the interior forest trail on a path about 870 m long, beginning at a point at 352715E, 1663323N and ending at 352287E and 1663969N (travelling through the point where the second fixed station of 15 July was located). Two observers each held separate ultrasonic detectors aimed upward at the surrounding airspace with moderately high sensitivity settings and an audible broadcast setting. Detectors were deployed throughout the entire length of the transect and return. At the Kastiyu Forest area where we sampled there were no interior forest trails. Therefore we walked a route along the immediate forest edge on an overgrown road bed over a distance of 0.5 km with the detectors on continuously. An old fence line separated the forest from the old road bed, which was typically within 10 m of the canopy edge. Every 125 m we crossed the fence and entered into the forest at a perpendicular distance of about 30 m into the forest interior from the transect and stood about 15 m apart with each detector scanning upward across the air space for a 5 min fixed point sample. We walked the entire transect route and returned twice. The coordinates for each end of the transect and the locations for fixed point stations at the Kastiyu Forest are given in Table IX-1.

In addition to echolocation detector sampling we also queried employees of the CNMI Division of Wildlife and Tinian Department of Land and Natural Resources and other people for local knowledge about this bat on Tinian, and kept alert for visual observations of bats in flight when traveling at night by foot or by vehicle.

RESULTS AND DISCUSSION

We did not detect any ultrasounds in our limited sampling with echolocation detectors at two forested areas on Tinian. This is consistent with the lack of any previous historic records of Pacific sheath-tailed bats on Tinian (e.g. Lemke 1986, Helgen and Flannery 2002). We also found no records of museum specimens on Tinian in our search of museum databases (see also Section VI in this administrative report). Additionally, we saw no bats in flight at any time, and there was no apparent local knowledge of the existence of these bats on Tinian based on our limited number of conversations with residents and Tinian Department of Land and Natural Resources and CNMI Division of Fish and Wildlife employees. This is also consistent with the verbal report by Stafford (personal communication) that he saw none of these bats during his geological inventory of 88 caves on Tinian (Stafford 2003). Eleven caves specifically searched for bats in 1984-1985, failed to have evidence of occupancy by bats, although investigators suggested that additional caves should be searched in the Kastiyu area (Wiles and others 1990).

Although Pacific sheath-tailed bats are present in deposits in caves from prehistoric times (Steadman 1999), a long history of habitat degradation by human influences may have negatively impacted their foraging habitat. Tinian has a larger land mass than Aguiguan and has areas with considerable karst topography, cliffs, and caves, but the few remaining “native” forested areas on Tinian are small. As reviewed by Wiles and others (1990), the Chamorro people probably strongly modified native vegetation of Tinian prior to European contact, which was exacerbated by introduction of exotic ungulates thereafter. Thousands of cattle roamed the island and a large population of feral pigs existed between the 1700s and 1900s. During the 1920s and 1930s Japanese colonists cleared most of the island for sugar cane plantations, with very little native forest left standing. According to one estimate (Bowers 1951) as little as 2% of the island may have been left in native forest. It is likely that the extensive development and clearing, military activity, and combat operations on Tinian during and after World War II also impacted caves as roosting habitat and forests as foraging habitat. Local knowledge indicated that accessible caves were used as strongholds by soldiers, and were subject to grenade explosions and flamethrower operations during the invasion by the United States. If Pacific sheath-tailed bats were present on Tinian after loss of forest habitat to agriculture prior to World War II, it is likely that populations were severely reduced or eliminated during the war and perhaps thereafter (see Lemke 1986, and Section III of this administrative report for a history of investigations on sheath-tailed bat distribution in the Mariana Islands). If these bats currently exist on Tinian despite the absence of any historic records or contemporary evidence, then it is likely that they do so in a very small remnant population that will take considerable effort to discover. Such efforts will require additional use of ultrasonic detectors and intensive searches of caves for roosting bats. Searches of many of these caves will require specialists with technical climbing and caving skills. Additional effort at surveying forests using echolocation detectors should be devoted to a larger area of forest in the Kastiyu and Carolinas area of southeastern Tinian. This region has a significant amount of karst geology and associated caves (Stafford 2003) as well as native limestone forest, some of which may have never been cleared for agriculture because of the karst substrate. If a relict population of Pacific sheath-tailed bats exists on Tinian it would most likely be in this general location.

Similarly, the literature also reports possible sightings of small numbers of these bats on Anatahan and the east island of Maug in the late 1970s and early 1980s (Lemke 1986, 1987). These two northern islands should be surveyed more intensively for sheath-tailed bats using echolocation detectors as well as searches of caves (other subspecies of Pacific sheath-tailed bats will roost in hollow trees and overhangs in other island groups) to rule out the possibility of an established population on an island other than Aguiguan. Discovery of a second population in the Marianas Islands would help bolster the prospects for survival of this subspecies. Consideration of other management options such as translocation from Aguiguan to other locations could be deferred until further assessments on other islands are completed. Although there are 1,200 species of bats worldwide, with many species of conservation concern, with one exception translocation of insectivorous bat populations for conservation has to our knowledge never been successfully attempted. The exception involved the unusual case of the short-tailed bat (*Mystacina tuberculata*) in New Zealand. This translocation required an intermediate step of captive breeding, and has been too recent to judge its ultimate success (New Zealand Department of Conservation, 2008).

ACKNOWLEDGMENTS

We thank E. Masga, Ton Castro, and other CNMI government and mayor's office employees on Tinian for their guidance in the Field. G. Rodda suggested the location on Mt. Lasso, and the Kastiyu Forest granted after-hours access to their lands. This section of the report was improved by comments on an earlier draft by P. Cryan and G. Wiles. Any use of trade, product, or firm names in this publication is for descriptive purposes only and does not imply endorsement by the U.S. Government.

REFERENCES CITED

Bowers, W.N., 1951, Problems of resettlement on Saipan, Tinian and Rota, Mariana Islands: Ann Arbor, Michigan, Ph.D. Dissertation, University of Michigan, 492 p.

Esselstyn, J. A., Wiles, G.J., and Amar, A., 2004, Habitat use of the Pacific sheath-tailed bat (*Emballonura semicaudata*) on Aguiguan, Mariana Islands: Acta Chiropterologica, vol. 6, p. 303-308.

Helgen, K. M., and Flannery, T. F., 2002, Distribution of the endangered Pacific sheath-tailed bat *Emballonura semicaudata*: Australian Mammalogy, vol. 24, 209-212.

Lemke, T. O., 1986, Distribution and status of the sheath-tailed bat (*Emballonura semicaudata*) in the Mariana Islands: Journal of Mammalogy, vol. 67, p. 743-746.

Lemke, T. O., 1987, Sheath-tailed bats rediscovered in the Mariana Islands: Bats Magazine, vol. 5, p. 5-7.

New Zealand Department of Conservation, 2008, Short-tailed bat translocation,http://www.doc.govt.nz/ conservation/ native-animals/bats/short-tailed-bat/ docs-work/short-tailed-bat-translocation/, accessed 23 December 2008.

Stafford, K. W., 2003, Structural controls on megaporosity in eogenetic carbonate rocks: Tinian, CNMI. M.S. thesis, Mississippi State University, Mississippi State, Mississippi. 340 p.

Steadman, D. W. 1999. The prehistory of vertebrates, especially birds, on Tinian, Aguiguan, and Rota, Northern Mariana Islands. Micronesica 31:319-345.

Wiles, G. J., Amerson, A. B., and Beck, R. E., Jr., 1990, The mammals of Tinian, Mariana Islands: Micronesica, vol. 23, p. 167-180.

Table IX-1. Coordinates (UTM, Datum: WGS 84; Zone 55P) for start point, end point, and fixed point stations using the Anabat detectors in Kastiyu Forest on 18 July 2008.

Location	**Easting Coordinate**	**Northing Coordinate**
Transect beginning	355635E	1654000N
Forest interior station 1	355660E	1653903N
Forest interior station 2	355662E	1653767N
Forest interior station 3	355692E	1653629N
Forest interior station 4 and transect end	355732E	1653511N

Section X. Considerations for Future Management, Monitoring, and Research

Thomas J. O'Shea, Ernest W. Valdez, and Gary J. Wiles

A number of considerations for future activities stem from the findings of this assessment. These are best characterized as activities related to management for conservation, monitoring, and research.

Considerations for Management for Conservation

Increasing the extent of native limestone forest, decreasing existing stands of invasive plants, and eliminating or avoiding actions that would reduce the amount of native limestone forest on Aguiguan. Results presented in Section IV of this administrative report show that Pacific sheath-tailed bats on Aguiguan forage primarily in native limestone forest, particularly in stands of taller stature. The importance of this habitat may also be reflected by some elements of their diet as reported in Section V of this administrative report. Browsing by feral goats has limited the regeneration of native forest and has likely altered tree species composition and diversity by favoring unpalatable species. Control or elimination of goats could favor forest regeneration. If such control takes place, foraging activity of sheath-tailed bats should be monitored to ensure that a developing understory provides favorable feeding habitat compared with the existing open conditions maintained by goats. A program of managing extensive areas of non-native vegetation to encourage its replacement by native forest could enhance the amount of foraging habitat available to Pacific sheath-tailed bats on Aguiguan. Similarly, new military activities that could be destructive to the remnant forest should be avoided.

Limiting disturbance and access to caves used by roosting bats. Although many caves exist on Aguiguan, only a few are used as roosts by Pacific sheath-tailed bats, despite seeming similarities among caves in structure and conditions of temperature and humidity (see Sections III and VI of this administrative report). Only two caves are known to regularly have 50 or more bats. These caves have histories of use by bats that indicate occupancy by sheath-tailed bats on every visit since first discovered by researchers. They are also occupied by endangered Mariana swiftlets. Under current levels of visitation of Aguiguan by people, activities that might disturb bats or swiftlets at caves seem minimal. However, any increase in visitation by people or increases in other activities that could disturb these colonies could have strong negative effects. Disturbance is well known to have long-lasting negative effects on other species of bats around the world that rely on caves for roosts, and a variety of techniques for protecting caves used by bats have been developed.

Considerations for Monitoring

Monitoring numbers of bats utilizing key caves on Aguiguan. Monitoring numbers of bats at caves will provide an index of population status (increasing, stable, or declining) over time, and will allow measurement of responses to habitat change (e.g. from typhoons or habitat management). Use of caves by bats could be monitored using emergence counts or internal counts with night vision equipment where possible. Counts should be made on a regular basis using a standard protocol. Caves with a history of use and past series of counts identified in Section III of this administrative report can be focal points, but new colonies that may be discovered with time could also be added. Monitoring should include some initial efforts to further assess daily and seasonal variability in counts, as well as to quantify possible observer effects. Results from variability assessments can be used to guide scheduling and levels of effort (e.g. seasons and numbers of replicate counts at a site) for monitoring.

Monitoring of the use of foraging habitat on Aguiguan using echolocation detectors and site occupancy models. Replication of sampling efforts described in Section IV of this report using echolocation detectors and site occupancy models could be carried out periodically to assess changes in the amount of activity and in habitat use by foraging sheath-tailed bats. Monitoring can detect changes in habitat use over time, changes in response to catastrophic typhoons, and changes in response to management intended to improve the population status of these bats.

Considerations for Research

Searching the more inaccessible areas on Aguiguan for the presence of additional colonies of Pacific sheath-tailed bats that may occupy caves requiring technical climbing and caving skills to reach. Pacific sheath-tailed bats on Aguiguan currently appear to roost only in caves, and occupy just a small fraction of known caves (Section III of this administrative report). Some of our observations of foraging activity and of bats dispersing early at dusk (Sections IV and VI) leave open the possibility that one or more undiscovered colonies of sheath-tailed bats may still exist on Aguiguan. If so, these are likely to be in caves that are in inaccessible sections of cliffs or caves with deep shaft-like entrances that will require technical skills at climbing and caving to search properly. Future searches for roosts should include a team of bat biologists combined with persons with good technical climbing and caving skills and appropriate safety training, dedicated exclusively to searching less accessible caves for colonies of sheath-tailed bats.

Increasing the foundation of ecological knowledge of this species pertinent to its conservation and management. The biology of Pacific sheath-tailed bats anywhere in their range is poorly known. Although we made a few new observations on the biology of these bats during the course of this assessment, determination of potentially important patterns in their ecology will require additional intensive sampling. As examples, if reproduction or foraging habitat use follows a seasonal pattern, then management needs may differ by season; understanding of vital parameters such as timing of reproduction and litter size could influence expectations for time needed for population recovery; possibilities for complexity in social behavior such as foraging territoriality (known in other emballonurid bats) could limit population density given the restricted area of native forest currently available. A focused multi-year program of research on the natural history, ecology, and biology of Pacific sheath-tailed bats on Aguiguan would provide much information of potential use for management. Development of capture techniques, analysis of cave deposits for past histories of use by these bats and other animals, analysis of fecal material for past exposure to contaminants and more in-depth assessment of diet, and further sampling to estimate genetic diversity are other areas of research that could improve biological knowledge about this population pertinent to its conservation and management.

Conducting a modern analysis of the taxonomic status of Emballonura semicaudata and its subspecies using combined quantitative morphometric and molecular genetic approaches. The management of the Aguiguan population as a subspecies of a more widely distributed Pacific sheath-tailed bat is based on a limited taxonomic understanding. The current systematic status of the subspecies in the Mariana Islands rests largely on qualitative features of skull morphology described 65 years ago after examination of a limited number of specimens. The subspecies designation was also based on size, which our measurements of bats from Aguiguan (Section VI of this administrative report) suggest may no longer be a valid criterion. Review of subspecific distinctions in 1997 was also limited in scope. There are a large number of specimens in existence in various museum collections that could be used in a systematic reassessment, and there are also a number of molecular genetic approaches that are now routinely used in the study of bat taxonomy. Modern techniques employing a variety of morphometric and molecular genetic analyses should be applied to a study of the systematics of this species throughout its range, including the Mariana Islands and American Samoa (where a different putative subspecies of *E. semicaudata* is also a Category 3 candidate for listing under the U.S. Endangered Species Act). New sampling using wing biopsy techniques like those we applied on Aguiguan may be necessary at some locations.

Further assessing for the occurrence of Pacific sheath-tailed bats on Tinian and other islands in the Commonwealth of the Northern Mariana Islands. Our negative findings on Tinian (Section IX of this administrative report) were based on very limited sampling and should be expanded to more conclusively rule out the possibility that Tinian may still be occupied by a remnant population of Pacific sheath-tailed bats. Similarly, the literature also reports possible sightings of these bats on Anatahan and the east island of Maug in the late 1970s and early 1980s. These two northern islands should be surveyed more intensively for sheath-tailed bats using echolocation detectors to rule out the possibility of an established population on an island other than Aguiguan. Discovery of a second population in the Mariana Islands would help bolster the prospects for survival of this subspecies. Consideration of other management options such as translocation from Aguiguan to other locations could be deferred until further assessments on other islands are completed. Although there are 1,200 species of bats worldwide, with many species of conservation concern, to our knowledge translocation of an insectivorous bat population for conservation has been attempted only once. This involved the short-tailed bat (*Mystacina tuberculata*) in New Zealand and required an intermediate step of captive breeding. This translocation has been too recent to judge its ultimate success.

Author Affiliations and Addresses

Frank J. Bonaccorso
United States Geological Survey, Pacific Island Ecosystems Research Center, Kīlauea Field Station, Hawaiʻi National Park, HI 96718, USA

Jacob A. Esselstyn
Affiliation while conducting field work (prior to 2008): Division of Fish and Wildlife, P.O. Box 1397, Rota, Commonwealth of the Northern Mariana Islands 96951, USA.
Current address: Natural History Museum and Biodiversity Research Center & Department of Ecology and Evolutionary Biology, University of Kansas, Lawrence, Kansas 66045, USA.

P. Marcos Gorresen
Hawaiʻi Cooperative Studies Unit (PACRC, UH Hilo), Pacific Island Ecosystems Research Center, Kīlauea Field Station, Hawaiʻi National Park, HI 96718, USA

Thomas J. O'Shea
United States Geological Survey, Fort Collins Science Center, 2150 Centre Avenue, Building C, Fort Collins, Colorado, 80526-8118, USA

Corrina A. Pinzari
United States Geological Survey, Pacific Island Ecosystems Research Center, Kīlauea Field Station, Hawaiʻi National Park, HI 96718, USA

Ernest W. Valdez
United States Geological Survey, Fort Collins Science Center, Department of Biology, MSC03 2020, 1 University of New Mexico, Albuquerque, New Mexico 87131-0001, USA

Gary J. Wiles
Affiliation while conducting field work prior to 2008: Division of Aquatic and Wildlife Resources, 192 Dairy Road, Mangilao, Guam 96913, USA.
Current address: Washington Department of Fish and Wildlife, 600 Capitol Way North, Olympia, Washington 98501-1091 USA

David J. Worthington
Affiliation while conducting field work (prior to 2008): Division of Fish and Wildlife, P.O. Box 1397, Rota, Commonwealth of the Northern Mariana Islands 96951, USA.
Current address: Capitol Reef National Park, Torrey, Utah 84775, USA