

GUANA SCIENCE



ANNUAL REPORT 2014

Annual Report for 2014 Guana Science

James Lazell
The Conservation Agency
6 Swinburne St.,
Jamestown, RI 02835

October of 2014 was the last regular month of terrestrial science work on Guana Island since 1981. I have six separate projects reported on:

1. Introduced Fire Ants and species related both evolutionarily and ecologically, by Tracy Langkilde.
2. Iguanas: the restored population of critically endangered BVI endemic *Cyclura pinguis*, by Krista Mougey and Gad Perry.
3. Sheep ecology and effects on native plants, by TTU Biology Class and Gad Perry and Clint Boal.
4. Data on habitat use of *Anolis* lizards in natural and artificial locations, by Jason Kolbe.
5. Lead metal exposure of lizards and potentially humans too resulting from old batteries, by Adam Duerr.
6. Birds, by Clint Boal. Of note are two species of North American migrant warblers not previously recorded here. I asked Clint if he thought they might be products of an eastward shift in migratory route resulting from global warming, which facilitates migrant departure from farther North – which in North America equals farther East. Clint said he would “not speculate” on that. I will. I think these new records are the result of global warming indeed! But instead of a speculation I will call that a Hypothesis. Hypothesis is the foundation of Science.

We are all sad to see Guana Science come to an end.

Thank you for all the years!



December 19, 2014

Re: 2014 Research Summary Report.

Guana Island – Red Imported Fire Ants, Egg Predation, and Management

Personnel: Christopher Thawley and Gail McCormick

To whom it may concern,

Please find below a summary of the research conducted by the Langkilde Lab on Guana Island in 2014.

Motivation

The Red Imported Fire Ant (*Solenopsis invicta*) is an invasive species of ecological concern. Originally native to South America, it has spread to over 12 countries, including the British Virgin Islands, and is predicted to occupy over 60% of the Earth's land area. Researchers have documented many negative ecological impacts of invasions of *S. invicta*. When Red Imported Fire Ants move into an area, the native arthropod communities are often changed by competition with or direct predation by fire ants on native species. Red Imported Fire Ants also prey on native vertebrates, and some species with particularly vulnerable life stages, such as eggs or juveniles that cannot escape ant attack, may be at particular risk. These ants are aggressive and readily attack humans with venomous stings; medical consequences range from itchy white pustules to anaphylactic shock and potentially even death. Red Imported Fire Ants were first documented on Guana Island in the 1990's, but little is known about how this invasive species may be affecting native ants and vertebrates on the island. Several species, including sea turtles and the endangered Stout Iguana (*Cyclura pinguis*), lay eggs in the island's sandy soils, and these may be at particular risk of fire ant predation.

Our 2014 research had two main aims:

1. To continue monitoring the distribution of the two species of fire ants: *Solenopsis invicta* and *S. geminata*, to track the invasion and determine how these species interact on Guana Island.
2. To conduct preliminary examination of the potential impact of the Red Import Fire Ant on nest survival of the Stout Iguana, to support a model to predict impacts of this invader on the iguana and inform managements decisions.

Research and Results

1. Monitoring the Spread of Invasive Red Imported Fire Ants

Background and Approach

This year, we continued surveys and experimental baiting protocols that have been conducted annually since 2010 to monitor the spread of Red Imported Fire Ants on Guana Island. As in previous years, we walked transects along the flats and beach area as well as all trails on the west side of Guana Island and marked all fire ant mounds by GPS (Fig. 1). Additionally, we experimentally baited for ants in four habitat types: the flats, the forest adjacent to the flats, the sand dunes at the main swimming/tourist

beach (White Bay Beach), and the area immediately surrounding the bar and lounge area at White Bay Beach. The continued collection of this data has allowed us to assess the spread of Red Imported Fire Ants on the island as well as their potential impact on native species and tourists.

Results and Management Implications

Our previous work has determined that numbers of Red Imported Fire Ants increased dramatically from 2010 to 2013. The number and location of fire ant mounds in 2014 were similar to that in 2013, suggesting that their spread may be stabilizing. Fire ants were frequently present at baits in the flats, and persisted in lower numbers near the Tourist Beach. This is likely due to poisoning for fire ants in this area in Spring of 2014. We also found evidence of fire ants on the North “Honeymoon” beach. There is potential for interaction with tourists if fire ants continue persist at these locations.



Figure 1. A researcher checks the side of the road for fire ant mounds. GPS coordinates for all fire ant mounds were taken to continue monitoring of the spread of fire ants on the island.

The mounds we observed, especially in the flats, were small and ants were not particularly active, as revealed in behavioral assays. We suspect this is at least partially due to the dry weather during our research period, which makes the ant retreat deep underground. It is important to note that we did observe native ants foraging in spite of the dry conditions.

2. Potential Predation by Red Imported Fire Ants on Stout Iguana Eggs:

Background

Our research effort focused on studying the potential vulnerability of Stout Iguana (*Cyclura pinguis*) nests to fire ant predation. Invasive fire ants are known to cause nest mortality in many reptiles, including lizards, turtles, and snakes. Underground fire ant foraging tunnels may intersect with reptile nests, and fire ants are capable of chewing through many eggshells as well as attacking hatchlings when they are pipping out of eggs or tunneling to the surface. Stout Iguanas on Anegada may nest at depths of 38-90 cm, but soils this deep are not available on Guana Island. Little is known about the structure of Stout Iguana nests on Guana, but on Caribbean islands with soils similar to Guana, iguanas in the genus *Cyclura* nest at depths from 22-67 cm. More research into this will illuminate the potential for fire ants to impact these nests.

Approach:

We have found fire ants at North Beach, near the primary known Stout Iguana nesting site, both in 2013 and this year. Fire ants appear to be expanding into this area, however densities of fire ants

remained too low to conduct appropriate trials. Stout Iguanas generally nest well above the tideline in sandy areas with moderate vegetation. As a result, we chose to examine potential nest predation in three accessible areas with similar habitat and fire ant populations: on the dunes of Tourist Beach above the high tide line, on the landward side of vegetation/forest bordering Tourist Beach, and in the forested area of the Flats surrounding a recently discovered Stout Iguana nest site.

We used a technique we have successfully employed to assess fire ant predation on lizard eggs in the mainland US. This involves inserting hollow acrylic tubes in the ground to nesting depths and constructing an artificial nest against the outer wall of the tube (Fig. 2). A digital camera is then lowered into the tube allowing for daily monitoring of predation on nests without a need for subsequent disturbance. While we could not use iguana eggs to construct artificial nests due to conservation concerns, we placed hotdog slices, an attractive bait to many ant species, along the outer wall of tubes at depths of 30 cm. Hotdog slices were topped with sticky bait cards to trap any ants attending the baits. Each day, we remotely monitored the buried baits for any activity, using our cameras. At the end of our 5-day research period, we unearthed the baits and examined them for signs of ant predation. We then placed hotdog baits above ground to assess whether fire ants were active on the surface in these areas.



Figure 2. a) A juvenile Stout Iguana basks on a tree. b) A researcher digs a hole in the sand in preparation for tube installation. c) An predation-monitoring tube installed in the ground. A fake “nest” of hot dog slices is placed against the outer wall of the transparent tube at approximated nesting depths. A small digital camera is placed inside the tube to monitor the nests for predation by fire ants.

Results and Implications:

We found that fire ant surface activity was high (63% of baits were attacked) along the inner edge of the beach vegetation, where iguanas are thought to nest. This suggests a high likelihood that hatching iguanas would be attacked by foraging fire ants in these locations. We did not, however, find fire ants on any of our buried baits. One set of baits was partially consumed by a beetle larva, and one other set of baits was being consumed by a native thief ant (likely *Solenopsis pygmaea*). Thief ants are known predators of the eggs of other lizards, including anoles, so this likely represents a naturally-occurring predator.

Our research suggests that predation of buried Stout Iguana eggs will be uncommon, likely because the beach sand is too unstable for fire ants to construct structured underground foraging tunnels. However, predation of hatchling iguanas as they emerge from the eggs and dig out of the soil is likely. As

previously documented, fire ants are currently expanding into the main Stout Iguana nesting areas along North Beach. Fire ants on Guana Island seem to be most common in areas with flatter aspects and sandy, deep soils, the same types of habitats preferred by Stout Iguanas for nesting. As such, the likelihood for interactions between fire ants and hatching iguanas may be increasing on the island.

Fire ants are also known to prey on sea turtle eggs and hatchlings. Both Green turtles (*Chelonia mydas*) and Hawksbills (*Eretmochelys imbricata*) are known to nest on North Beach and could be vulnerable to fire ant predation on eggs of nestlings. We recommend monitoring for the presence of fire ants at the nesting sites of these vulnerable species. If egg predation by fire ants is shown to have detrimental effects on the species of concern, carefully managed fire ant reducing treatments, including targeted application of pesticide baits could be considered, especially during sensitive time periods when iguana eggs are hatching. We again warn that no fire ant specific baits exist, and fire ant numbers could actually increase if the abundance of native ants is decreased following the application of baits.

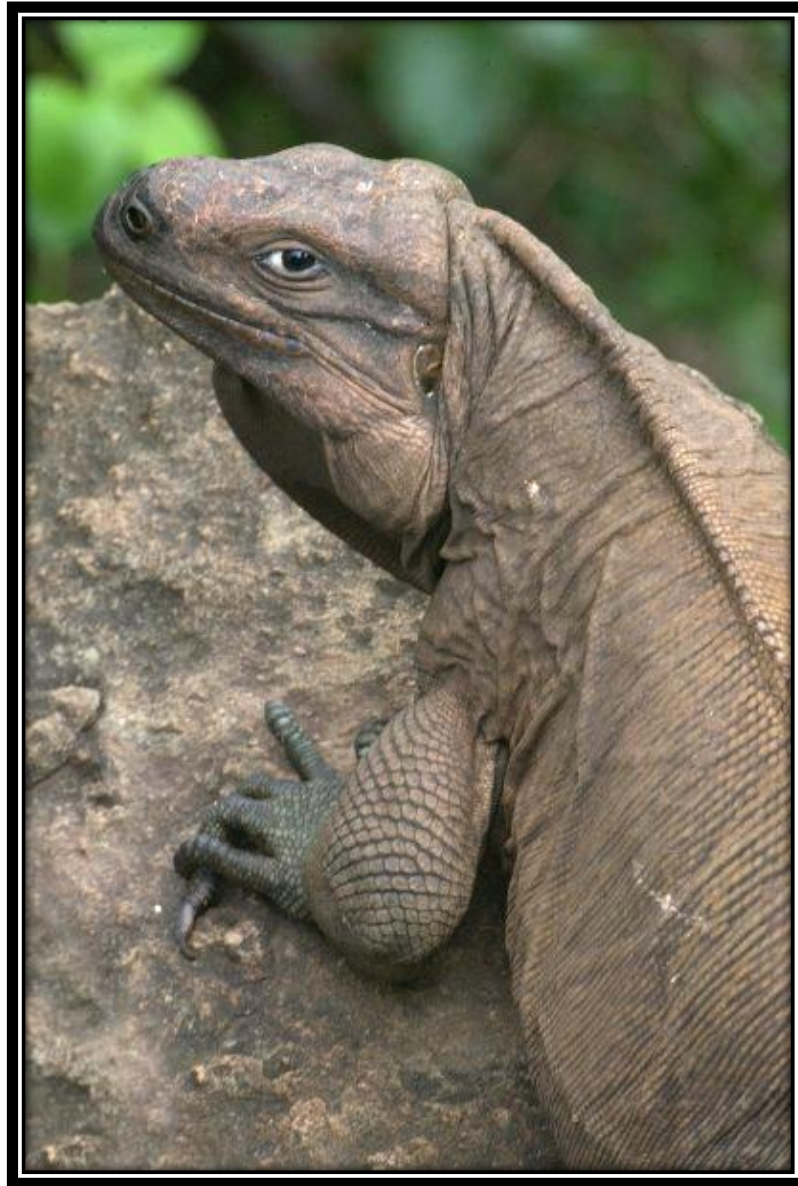
Executive Summary and Management Recommendations

These results have important management implications. While native ant species do not generally attack humans or wildlife, they do provide resistance to invasion by Red Imported Fire Ants, which are a major pest species. Red Imported Fire Ants forage at great distances from their mounds, recruit aggressively to attack prey and defend resources, and can be both a nuisance and serious health concern (in the event of serious or fatal allergic reactions) to humans. We have found continued presence of invasive Red Imported Fire Ants in areas of important human use, including near the main tourist swimming beach and along North Beach, suggesting that future interactions between humans and these invasive ants are likely. Red Imported Fire Ants may also pose a threat to nesting species of conservation concern, including sea turtles and Stout Iguanas.

Eradicating Red Imported Fire Ant mounds via direct applications of pesticides or poisoned baits is possible, but extreme care must be taken to avoid mismanagement of these toxins. Poisons that kill Red Imported Fire Ants can also kill other species. If competitors of fire ants are killed by poisons, then poisoning/baiting could actually promote the spread of these invasive fire ants by releasing them from competition. **Targeted poisoning Red Imported Fire Ants** is essential to prevent removal of native ants. Additionally, **poisoning and baiting should be avoided during dry weather**, when fire ants may not be actively foraging at the surface. Conducting these interventions during the dry season could thus be *worse* for native ants, which provide important ecosystem services on the island.

Careful, targeted use of pesticides in limited, specific areas may help reduce interactions between fire ants and humans and avoid detrimental effects of these invasive predators on island wildlife. Continued monitoring may allow us to determine the most effective management strategies that target times of greatest fire ant activity with the least affect on native species.

Natural History, Ecology, and Conservation of the West Indian Stout
Rock Iguana (*Cyclura pinguis*) on Guana Island, British Virgin Islands



PROGRESS REPORT FOR
SCIENCE MONTH–OCTOBER 2014

Prepared by

Krista Mougey and Gad Perry

Department of Natural Resources Management,
Box 42125, Texas Tech University, Lubbock, TX 79409-2125, USA
krista.mougey@ttu.edu and gad.perry@ttu.edu

CONTENTS

2014 Iguana Research Team:	5
Executive Summary.....	6
Introduction	9
Long-Term Goals of the Cyclura Project.....	12
List of Objectives.....	13
Role of 2014 Data in Meeting Project Objectives	13
Field Methods	15
Progress and Preliminary Findings	20
Objective 1 — Augment and verify findings from previous studies on growth, demographics, habitat use, and spatial ecology of stout iguanas using a more comprehensive dataset	21
Objective 2 — Estimate abundance of the 2014 <i>Cyclura pinguis</i> population on Guana Island.....	30
Objective 3 — Analyze thermal ecology and the thermal dependence of physiological performance in hatchling stout iguanas on both Guana and Anegada	34
Objective 4 — Publication and dissemination of data pertaining to the status of <i>C. pinguis</i> on Guana Island.....	36
The Need for Continued Monitoring and Research.....	37
Literature Cited	39

Appendix 1	41
------------------	----

LIST OF FIGURES

Figure 1—Image plates depicting the differences in elevation, human development, and vegetative associations between Anegada (A, B, & C) and Guana Island (D, E, & F).....	10
Figure 2—Map of the British Virgin Islands showing the current distribution of <i>Cyclura pinguis</i>	11
Figure 3—Map of Guana Island delineating the core research area for the 2003- 2014 <i>Cyclura pinguis</i> Research Project	15
Figure 4—Krista Mougey and Doug Bell processing a small adult iguana captured near the cistern in 2014.....	16
Figure 5—Visual Recapture of Iguana “A85”	17
Figure 6—Map of Guana Island depicting the survey transects used for distance sampling protocols and abundance estimation of the <i>C. pinguis</i> population in non-core regions of the island	19
Figure 7—Field Assistant Josh Cummings measuring perpendicular distance between the trail and an iguana sighting location.....	20
Figure 8—Krista Mougey holding a previously unmarked adult male captured near the orchard nursery	20

Figure 9— A) Depicts the relationship between SVL at initial capture and SVL at recapture for Guana Island hatchlings B) Depicts the relationship between mass at initial capture and mass at recapture for Guana Island hatchlings	22
Figure 10—Field Assistant Doug Bell holding a hatchling (left) and yearling (right) <i>C. pinguis</i> for size comparison.....	23
Figure 11—Graph of the relationship between SVL at initial capture and growth rate in mm/year differentiated by sex illustrating the tapering off of growth in older age classes.....	24
Figure 12—A General view of the nesting location found near the boat storage area of the flat B) Egg shells found on the surface in the nesting area. Egg shells often indicate habitual use of a nesting site, because when the female excavates the nest, she may dig out the egg shells from the previous year.....	25
Figure 13—Hatchling iguana marked with a string bobbin to allow tracking of movements and arboreal habitat use.....	26
Figure 14—Mature adult male utilizing sub-canopy vine mats between the sheep paddock and orchard. This was a habitual use location that was also regularly used as an overnight retreat	27
Figure 15—Map of all iguana sightings and capture localities recorded in 2014.....	28
Figure 16—Density surface of iguana sightings for Guana Island	29

Figure 17—Puerto Rican Racer constricting a marked hatchling on the floor of the clubhouse at 11:00pm on the 24 th of October, 2014.....	31
Figure 18—Map of all iguana sightings recorded outside the core research area in 2014	32
Figure 19—Map of the core research area depicting all iguana capture, resight, and sight record localities for 2014 with relevant landmarks and habitually utilized retreat locations	33
Figure 20—Graphical depiction of endurance capacity for the Anegada and Guana populations, differentiated by thermal treatment group.....	36
Figure 21—Map of locations from the collared female stout iguana showing her nesting location on North Beach and her territory near the hotel complex	38

2014 IGUANA RESEARCH TEAM:

- Krista Mougey
- Gad Perry
- Doug Bell
- Josh Cummings
- Laura Casolino

Additional Assistance From:

- Skip Lazell
- Clint Boal
- The Many Participants of Guana Science Month

EXECUTIVE SUMMARY

The stout iguana is one of the most critically endangered reptiles in the world. Thirty years ago, eight stout iguanas were translocated by Dr. Lazell from Anegada to Guana Island to found a conservation insurance population. While abundance of the Anegada iguana population has shown little recovery, the Guana population has flourished and is now a critical resource for the preservation of the species. Research on the Guana population was implemented to document the recovery process and provide a body of ecological knowledge to aid in the conservation of the stout iguana. We gather information on growth, demographics, survival, habitat use, spatial ecology, and thermal ecology of stout iguanas, monitor population abundance on the island, and contribute to the available scientific knowledge on the species through education, presentations, and publications in the peer-reviewed literature. We also participate in formal management discussions on the species, in which the future of the stout iguana and the strategy for its overall recovery are planned.

In 2014, we focused on completing several ecological studies and performing a rigorous population assessment. Island-wide, we recorded 1,045 iguana sightings. We captured 169 individuals within the core research area. Based on our preliminary analyses, population abundance estimates will likely be slightly higher than those previously reported. The adult sex ratio is near 1:1, and the population age structure is healthy, with ample representation in all age classes. Based on our 12 years of collected size and growth data, we now have a

defensible growth curve for this population that will allow us to estimate age via body-size for both sexes. We have confirmed that detectable levels of hatchling growth occur within the month-long field season and have shown that the iguanas on Guana grow much faster than those headstarted on Anegada and elsewhere.

Iguana density within the core research area has increased, and individuals in all age classes are expressing high site fidelity, meaning they maintain territories in the same locations for multiple years. We have noted an upsurge in tree and canopy use by older, larger, animals and have confirmed nesting at a new location within the island. We have also found a difference in physiological capacity during treadmill tests of wild (Guana) and headstarted (Anegada) individuals, with the latter having lower endurance.

Intense development pressure, non-native predators (primarily cats and dogs), and feral grazing competitors (mostly sheep and goats, but also cattle and donkeys) are all major problems for the Anegada population. On Guana, the density of the stout iguana population near the resort's infrastructures is high. These areas have enhanced resource availability and more open canopy conditions that provide good habitat. Thus, the limited scale of development on the island has resulted in a healthy and stable iguana population near the resort. For the less developed portions of the island, our population assessment highlighted the importance of feral sheep removal. We recorded the lowest number of iguanas in areas with high sheep presence and vegetative damage. The current efforts to cull

feral sheep on the island should be very beneficial to the iguana population, especially in a year where drought has already reduced available resources.

Dissemination of data is a critical part of successful species conservation. The information we have collected over the last 12 years has been the basis for public outreach, minor publications, and scientific presentations. Ms. Mougey's dissertation and the associated manuscripts are now in preparation and will be completed by May of 2016. We expect to generate four or more peer-reviewed journal articles from these data. Ms. Mougey will also be presenting our findings to the Iguana Specialists Group of the IUCN Species Survival Commission in November.

The research we conducted through the Guana Science Program was vital in advancing understanding and conservation of the stout iguana. As conditions change, especially if sheep are eradicated from the island, continued monitoring of population trends will provide valuable information. Furthermore, acquisition of additional data on spatial and reproductive ecology, energy allocation, territoriality, and density-dependant population alterations will be critical to the effective conservation of this population and of the species as a whole.

INTRODUCTION

The stout iguana (*Cyclura pinguis*) is one of nine species of West Indian rock iguana that collectively are recognized as the single most endangered group of lizards in the world. Fossil records indicate that the species was once found throughout the Greater Puerto Rican Bank, but climate change near the end of the Pleistocene, the arrival of Amerindians, and later, colonization by Europeans are all thought to have restricted the distribution of the species (Carey 1975, Pregill and Olson 1981, Bradley and Gerber 2005). Currently, *C. pinguis* survives exclusively in the British Virgin Islands, with the only remaining natural population occurring on Anegada, an island at the northeastern extent of the species' historic range (Perry et al. 2007, Lemm and Alberts 2012).

The first formal study of the ecology and natural history of *C. pinguis* was initiated in 1968. During that study, Carey (1975) concluded that the Anegada iguana population was at the cusp of a catastrophic decline. He recommended immediate action in implementing mammalian control and other conservation measures to protect the remaining population, but his warning went largely unheeded (Carey 1975). Continued precipitous declines of *C. pinguis* were noted in the 1980s and 1990s, with an estimated 80% reduction in iguana numbers (Mitchell 1999, Bradley and Gerber 2005, Lemm and Alberts 2012).

In the late 1980s, it was estimated that there were less than 200 *C. pinguis* left in the wild (Alberts 2000, Bradley and Gerber 2005). Their single island distribution and the unabated threats within their remaining native habitat

generated major conservation concerns that prompted the translocation of several iguanas to found an artificially established “insurance” population. Between 1984 and 1986, eight adult *C. pinguis* (five females and three males) were translocated from Anegada to Guana Island (Figure 1). Despite concerns that *C. pinguis* would not thrive in such disparate habitat (Mitchell 2000), the population underwent rapid growth, and offspring have been observed every year since 1987 (Goodyear and Lazell 1994).

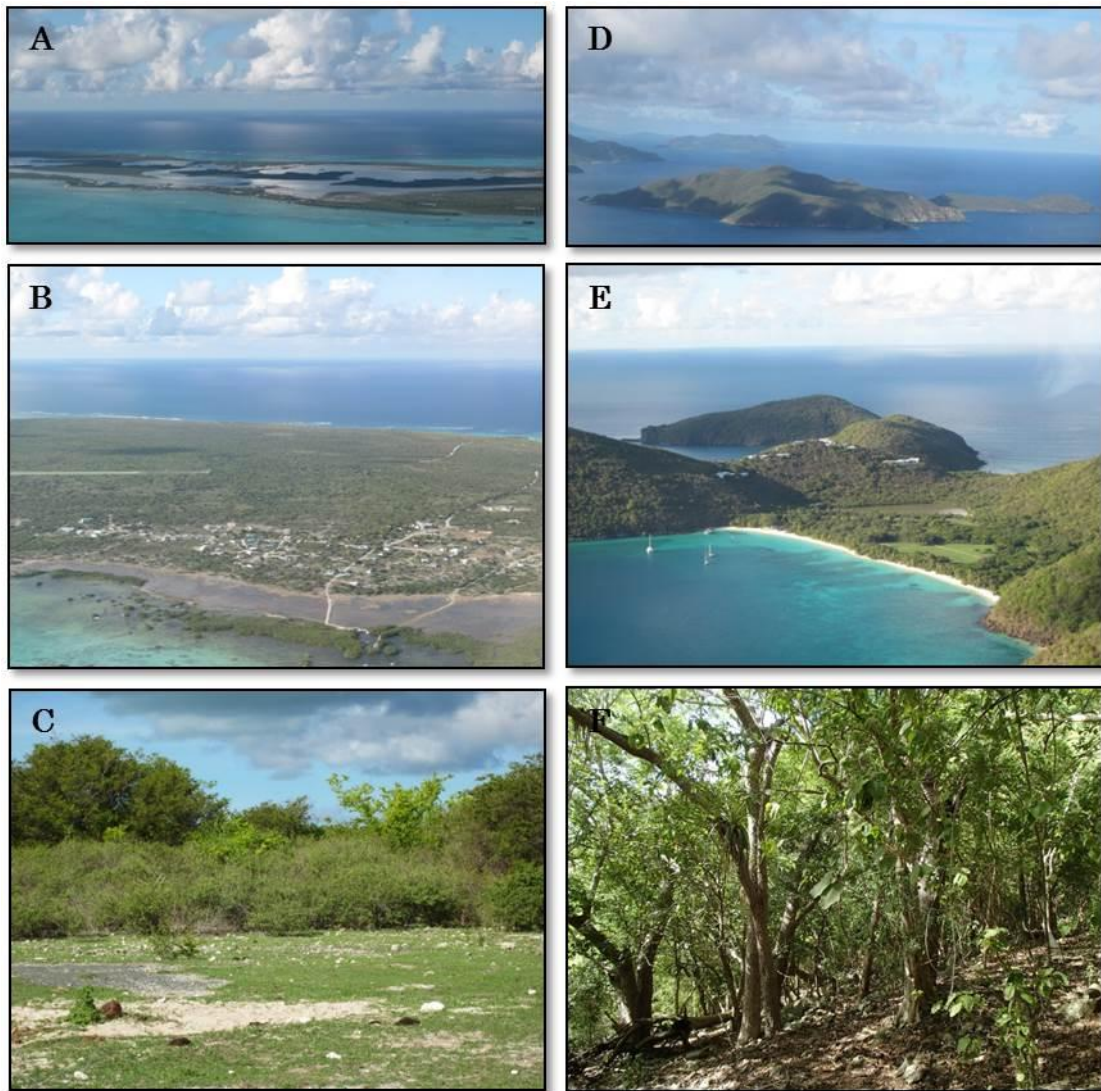


Figure 1—Image plates depicting the differences in elevation, human development, and vegetative associations between Anegada (A, B, & C) and Guana Island (D, E, & F)

Descendents of Guana's original eight iguanas have been used to found additional *C. pinguis* insurance populations on Necker, Norman, and Little Thatch, and in more recent years, animals from Necker have been introduced to Moskito and Eustatia (Figure 2). Based on current estimates, there are now more iguanas



Figure 2—Map of the British Virgin Islands showing the current distribution of *Cyclura pinguis*

in these insurance populations than there are remaining on Anegada. However, due to political controversy surrounding these *ex-situ* conservation efforts, the rock iguanas on islands other than Anegada have traditionally not been considered as legitimate components of the population in terms of conservation planning or management. But, it appears that this situation may be changing. A recent

proposal by members of the Iguana Specialists Group of the IUCN recommended the potential inclusion of these insurance populations into the species recovery plan.

The stout iguana has become an iconic flagship species for the protection of xeric habitat and native wildlife in the Caribbean. The negative relationships between human developments, introduced mammalian species, and *C. pinguis* survival are well established. Yet, despite scientific understanding of both the importance of the species and the causes of its imperilment, the problems on Anegada remain largely unabated. As the Anegada iguana population continues to decline, well established and robust insurance populations, like the one on Guana, will become an irreplaceable and crucial resource. For the long term conservation of the species, it is imperative that we understand the differences in populations between islands, compile information and natural history data from Guana's less disturbed, lower stress environment, and ensure the health and stability of the Guana *C. pinguis* population. Thus, analyses of iguana data collected as a part of the Guana Science Program, continued monitoring efforts, and additional study on the Guana population are all warranted.

LONG-TERM GOALS OF THE CYCLURA PROJECT

Effective conservation requires a detailed understanding of the species. The conservation efforts implemented to save *C. pinguis*, specifically the artificial establishment of insurance populations of the species on multiple islands, provide unique avenues of research into the natural history, ecology, and physiological

plasticity of the species. Although a small body of research does exist for *C. pinguis*, very little is focused on the potential differences between the source and insurance populations, or the importance of these insurance populations to the long-term survival of the species. The goal of this project was to capitalize on these unique conditions in order to provide a body of ecological, spatial, physiological, and methodological knowledge to aid in the conservation of *C. pinguis* and of West Indian cyclurid iguanas, in general.

List of Objectives

1. Augment and verify findings from previous studies on growth, demographics, habitat use, and spatial ecology of stout iguanas using a more comprehensive dataset
2. Estimate abundance of the 2014 *Cyclura pinguis* population on Guana Island
3. Analyze thermal ecology and the thermal dependence of physiological performance in hatchling stout iguanas on both Guana and Anegada
4. Publish and disseminate data pertaining to the status of *C. pinguis* on Guana Island

Role of 2014 Data in Meeting Project Objectives

One of the foremost challenges in working with endangered species is collecting enough data to produce robust statistical calculations and defensible population models. This is especially true of studies conducted under strict time

constraints. As 2014 was the last official year of the Science Month program, we placed intense effort on rounding out existing datasets and finishing ecological projects pertaining to the Guana iguana population. The 2014 data augmented information on within year growth, between year growth, demographic composition, spatial ecology, and thermal-based performance giving us large enough samples sizes to generate sound findings from our multi-year datasets.

Given the potential for insurance populations to be considered in the upcoming revision of the species recovery plan, it was also crucial to perform a population assessment on the abundance and density of cyclurids on Guana. Most of our capture and research efforts have been focused on the developed portion of the island. Several attempts have been made to assess the cyclurid population on Guana, but time and personnel constraints have resulted in population estimates of limited utility. In 2014, we attempted to correct some of these issues and generate more accurate population models. Having been granted bed-nights for additional assistants, we were able to expand our research to outlying regions of the island without compromising data collection in the core research area. Thus, we were able to perform a much more tenable population assessment across the island as a whole. In addition to the work we conducted on Guana, we were also able to make research visits to Necker, Mosquito, and Anegada to enhance our datasets for cross-island comparisons of growth, body condition, population health, and endurance physiology.

FIELD METHODS

Beginning in October of 2003, a long-term marking and monitoring effort was undertaken in areas around the Guana Resort to elucidate the ecology and natural history of the *C. pinguis* population. During October of each successive year (2003-2014), animals near the resort complex (an area hereinafter referred to as the core research area) (Figure 3) were captured, measured, and PIT tagged to allow for subsequent studies on the demographic, morphometric, and life history characteristics of this population (Perry et al. 2007).



Figure 3—Map of Guana Island delineating the core research area for the 2003-2014 *Cyclura pinguis* Research Project

Within the Core Research Area— Iguanas were captured using a combination of hand-capture, noosing, live-trapping, and netting techniques. The position of the iguana at first sighting was recorded using both GPS coordinates and general site descriptors. We noted the time of day and whether the lizard was in full shade, partial shade, or full sun. Upon capture, we immediately took a cloacal temperature and a reading of ambient temperature to the nearest 0.1°C. For each individual, we recorded morphometric measures of snout-vent length (SVL) and tail length (VTL) to the nearest mm, and mass to the nearest 0.1 gram. If the individual was sufficiently mature, we used cloacal probing and relevant physical characteristics to determine sex (Carey 1975, Dellinger and von Hegel 1990). We



Figure 4—Krista Mougey and Doug Bell processing a small adult iguana captured near the cistern in 2014

then scanned each iguana to check for the presence of a Passive Integrated Transponder (PIT) tag to determine if they have been marked in a previous year or capture session. If the animal had been previously marked, the identification code was recorded. If the animal was unmarked, we subcutaneously implanted a unique identification tag in the right dorsal region of the tail base (Alberts 2000). Prior to release, we marked each iguana with a unique paint code (written on both flanks with white-out) to aid in individual identification (Alberts 2000). For a subset of individuals, we collected blood samples from the ventral tail vein. These samples



Figure 5—Visual Recapture of Iguana “A85”

were used by other researchers for projects relating to lead or genetic analyses.

Marked hatchlings were recaptured at the end of the month and re-measured to determine their monthly growth rate. We also recorded visual recaptures of marked animals (Figure 5) and sightings of unmarked animals within the core research area. Based on this capture protocol, we also set up a capture-mark-resight framework to estimate the iguana

population within the core. Capture-mark-resight methodologies are a class of incredibly diverse abundance estimation techniques that can be used when animals are marked (or otherwise indentified) and later detected via physical or visual

capture (Lettink and Armstrong 2003). By dividing the month into week long capture sessions and recording all resightings and recaptures of marked animals throughout the month, we were able to mathematically assess the abundance of both the hatchling and post-hatch year *C. pinguis* populations.

Outside the Core Research Area— Very little is known about the *C. pinguis* population outside the core, but it has been suggested that both sheep presence and topographic aspect have heavily influenced iguana density and distribution (Goodyear and Lazell 1994, Anderson et al. 2010, Skipper et al. 2013). Logistically, it is not possible to conduct a rigorous capture and marking study across the entirety of the island with a small field crew and a single month field season, but, especially if density differs between core and non-core areas, it is important to generate an abundance estimate for the non-core region. We used distance sampling techniques modeled after those used by the Blue Iguana Recovery Program (Burton Unpublished Manuscript) to estimate density of post-hatch-year *C. pinguis* in non-core areas.

Distance sampling techniques are a collection of common methodologies used to estimate the density of biological populations through the calculation of a detection function (Buckland et al. 2001). Distance sampling theory addresses the question of how many organisms exist in an area, given the number of organisms detected and the likelihood that detections decrease with increased distance from the observer. They are popular methodologies, as they do not require that organisms be captured or marked, and have been used successfully with other



Figure 6—Map of Guana Island depicting the survey transects used for distance sampling protocols and abundance estimation of the *C. pinguis* population in non-core regions of the island

iguana species. Using the existing trail systems, we established a series of survey transects (Figure 6). Each transect was surveyed four times during peak iguana activity. Every time an iguana was sighted, the time of sighting, air and substrate temperature, GPS location, age/size class of the iguana, and perpendicular distance

of the iguana from the transect were recorded (Figure 7). We used the perpendicular distance measures to calculate a detection function to estimate the proportion of iguanas not detected during the survey process and to correct our abundance and density estimates for the non-core area.



Figure 7—Field Assistant Josh Cummings measuring perpendicular distance between the trail and an iguana sighting location

PROGRESS AND PRELIMINARY FINDINGS

Ms. Mougey is currently in the process of analyzing data and writing her dissertation based on 12 years of data collection from the Guana *C. pinguis*



Figure 8—Krista Mougey holding a previously unmarked adult male captured near the orchard nursery

population. Therefore, we will be providing only preliminary results for some of the sections. Science Month 2014 was a record year for the Cyclura Project. We had 1,045 sight/resight records across the island, including repeat sightings and both marked and unmarked individuals. Within the core area, we made 209 iguana captures representing 169 individual iguanas. As with other years, the majority (84%) of our captures were young-of-the-year. We handled 136 hatchlings, 7 yearlings, 3 sub-adults, and 23 adults

(Figure 8). In 2011, only about 17% of our post-hatch-year animals were recaptures. Starting that year, our efforts to capture and mark post-hatch-year iguanas have been dramatically higher than in earlier years of the study. However, of the 33 post-hatch-year individuals we handled in 2014, only 17 (51.5%) had been marked in a previous year. This indicates that even within the core research area, a substantial proportion of the iguanas still remain unmarked.

Objective 1 — Augment and verify findings from previous studies on growth, demographics, habitat use, and spatial ecology of stout iguanas using a more comprehensive dataset

Growth

In 2014, we added 136 hatchling measures to our size dataset and 34 hatchling data points to our within-year growth dataset. The average body length (SVL) of hatchlings at initial capture (2003-2014) was 104.5 ± 4.92 mm ($n = 567$), but ranged from 78 to 122 mm. Average undamaged tail length was 199.6 ± 8.69 mm ($n = 499$) with a range of 161 to 235 mm, and average mass was 53.9 ± 7.54 g ($n = 554$) with a range of 24 to 82 g. However, both year and ordinal capture date significantly impacted hatchling morphometric measures.

Hatchlings on Guana were both larger and heavier than hatchlings in the Anegada Headstart facility (SVL 102.89 ± 3.86 mm ($n = 35$), with a range of 96 to 112 mm; average undamaged tail length 197.06 ± 8.63 mm ($n = 35$) with a range of 179 to 216 mm; average mass 51.20 ± 7.46 g ($n = 35$) with a range of 39 to 69 g). Average

morphometric measures of Anegada hatchlings varied by year in a similar pattern to that of the Guana population. As the iguanas in the headstart facility age, the differences in average size and mass between the populations increase dramatically. On average, yearlings in the headstart facility are 55 mm shorter in SVL and 180 g lighter in mass than those in Guana's wild population.

Perry et al. (2007) postulated that detectable levels of hatchling growth likely occurred during month-long trapping sessions on Guana Island, but they were unable to assess the rate, as individuals were measured only once during their study. From 2011 to 2014, we collected morphometric measures for hatchlings at initial capture and again for individuals found at least two weeks after initial capture. We then used these capture-recapture morphometrics to assess within-year growth rates (SVL and mass) of the Guana hatchling population (Figure 9).

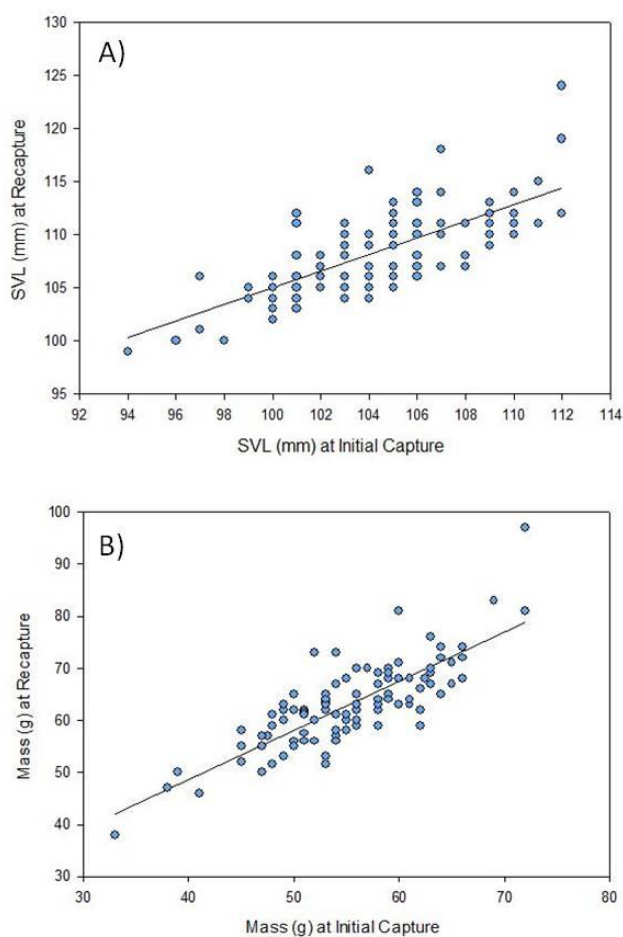


Figure 9— A) Depicts the relationship between SVL at initial capture and SVL at recapture for Guana Island hatchlings B) Depicts the relationship between mass at initial capture and mass at recapture for Guana Island hatchlings

We had usable measurement pairs for 102 individuals, 34 of which were collected in



Figure 10—Field Assistant Doug Bell holding a hatchling (left) and yearling (right) *C. pinguis* for size comparison

2014. When corrected for the elapsed time between initial and subsequent measurements, hatchlings on Guana grew an average of 6 to 7 mm SVL and gained approximately 12 to 13 grams in mass within the month of October.

Our capture records indicate that on average, hatchlings that survive their first year grow to 1.8 times their initial length and gain 5.8 times their initial mass. This gives the yearling age class an average SVL of 192 mm and an average mass of 311 g (Figure 10).

Adult *C. pinguis* are sexually dimorphic, and the males are both significantly larger (446mm versus 417mm SVL) and heavier (4.8kg versus 3.6 kg) than females. Due to the size of our iguana traps, we are less likely to catch the largest members of the male population, so it is quite possible that the degree of dimorphism is actually even higher than our figures indicate. Iguanas continue to grow throughout their lives, but as they age, their rate of growth slows (Figure 11).

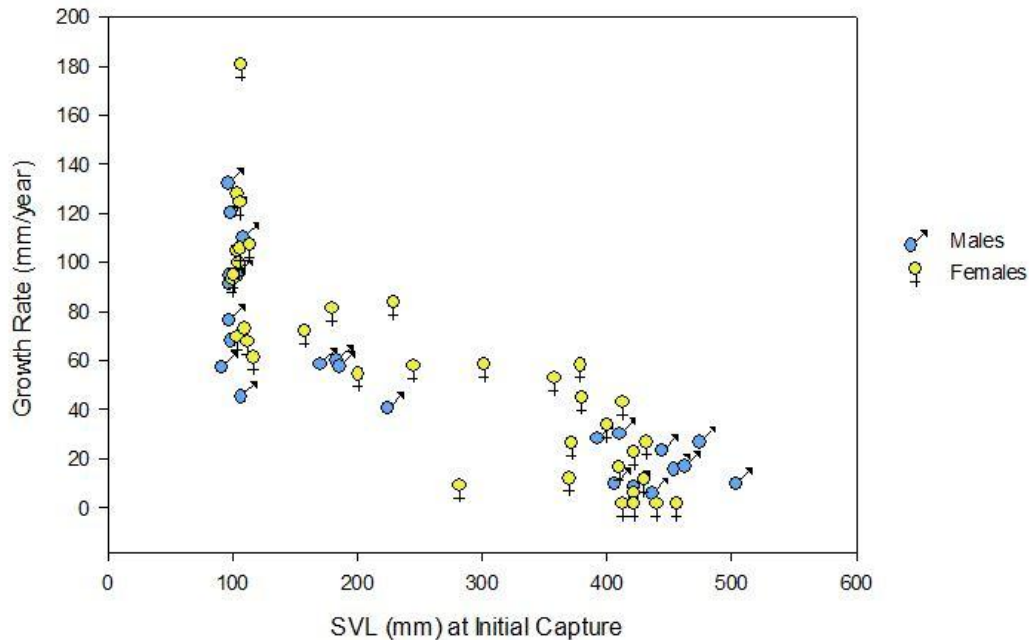


Figure 11—Graph of the relationship between SVL at initial capture and growth rate in mm/year differentiated by sex illustrating the tapering off of growth in older age classes

Demographics

One of the concerns raised about the Anegada iguana population by Carey (1975) and by other more recent studies was that the population appeared to consist primarily of older adults. This aging population demographic and the lack of younger aged animals was a major conservation concern for long term survival. It was also a major contributing factor in the initiation of the Headstart facility. Unlike Anegada, the population age structure on Guana appears to be healthy, with representation in all age classes.

We have captured more females than males; however, our sighting data indicate that this is a product of capture bias rather than sex ratio skewness.

Within the core, we have several males that we can uniquely identify and differentiate through scarring patterns, but that are too large to be reliably trapped with the equipment we have available. Therefore, we have used combination of sighting and capture data gathered between 2011 and 2014 to evaluate adult sex ratio, and it appears to be near 1:1.

Habitat Use

In 2014, we verified continued use of the north beach nesting area and identified a previously unknown nesting location near the boat storage area of the flat (Figure 12). Although density and prevalence of hatchlings near the RO plant,



Figure 12—A General view of the nesting location found near the boat storage area of the flat B) Egg shells found on the surface in the nesting area. Egg shells often indicate habitual use of a nesting site, because when the female excavates the nest, she may dig out the egg shells from the previous year

shop, and dump indicate the likelihood of nearby nests, we have been unable to positively identify individual nest sites in any of these areas. Given the difficulty of locating cryptic cyclurid nesting sites, the most productive way to do so would be to conduct an observational study during the nesting season or to remotely track a number of females with GPS collars during the breeding and nesting seasons.

From our string tracking study in 2011 (Figure 13) (Cheek et al. 2013), we learned a lot about arboreal habitat use in hatchling and yearling age classes, but with few exceptions, adults were rarely seen using upper-story or canopy habitats.

However, within the past few years, the number of records pertaining to the use of arboreal habitat by post-hatch-year iguanas has increased dramatically (Figure 14). This may be due to increased effort and utilization of better search imagery by the researchers or an actual shift in habitat utilization by the iguanas. Increased sheep presence within the core has altered the under- and mid-story vegetation. This, in turn, relates to changes in food availability, alterations in the available thermal habitat, and reductions in visual search obstructions. There may also be density issues at play within the iguana

population. When density is low, only prime habitats need be utilized. As abundance and density increases, competition for prime habitat may be driving a



Figure 13—Hatchling iguana marked with a string bobbin to allow tracking of movements and arboreal habitat use

switch in microhabitat utilization and the changes we are seeing in arboreal habitat use. Most likely, these changes are a result of a combination of these factors.

Spatial Ecology

In 2014, we recorded over 1,000 iguana sightings across the island (Figure 15). Many of these were resightings of individuals, and the majority were locations within the core research area, but we did encounter iguanas in areas where they had not previously been seen. This is likely a result of wider and more intense search



Figure 14—Mature adult male utilizing sub-canopy vine mats between the sheep paddock and orchard. This was a habitual use location that was also regularly used as an overnight retreat

efforts compared to the 2009 and 2010 sampling, although it is probable that range expansions have occurred between the original survey by Goodyear and Lazell (1994) and our current survey effort. Previous research has demonstrated that aspect and the magnitude of sheep-related impacts may influence cyclurid density, and the results of our 2014 iguana surveys outside the core research area support these earlier findings. Iguana density was far higher in the core area, and the portions of the island with the most exposed aspects and most apparent sheep damage had the fewest number of iguana sightings (Figure 16). The current efforts to cull the feral sheep population will likely be highly beneficial to the iguana population, especially in a year where drought has altered resource availability.



Figure 15—Map of all iguana sightings and capture localities recorded in 2014

Within the core area, we see a very high degree of site fidelity by individuals of all age classes, both within year and between years. Although we have recorded a few cases of hatchling dispersal, the majority of hatchlings appear to remain within a small territorial space throughout the month. Differences in detection rates between sedentary and dispersing individuals may influence these findings, but even across years, individuals that we recapture tend to be very near

their original capture localities. But, without a major trapping effort outside the core, it will be impossible to determine if the non-core population is composed of dispersing individuals that have been displaced from core habitat or locally born individuals that have never resided within the core. This kind of trapping effort would also provide important spatial information critical to calculating multi-year survival rates for the population.

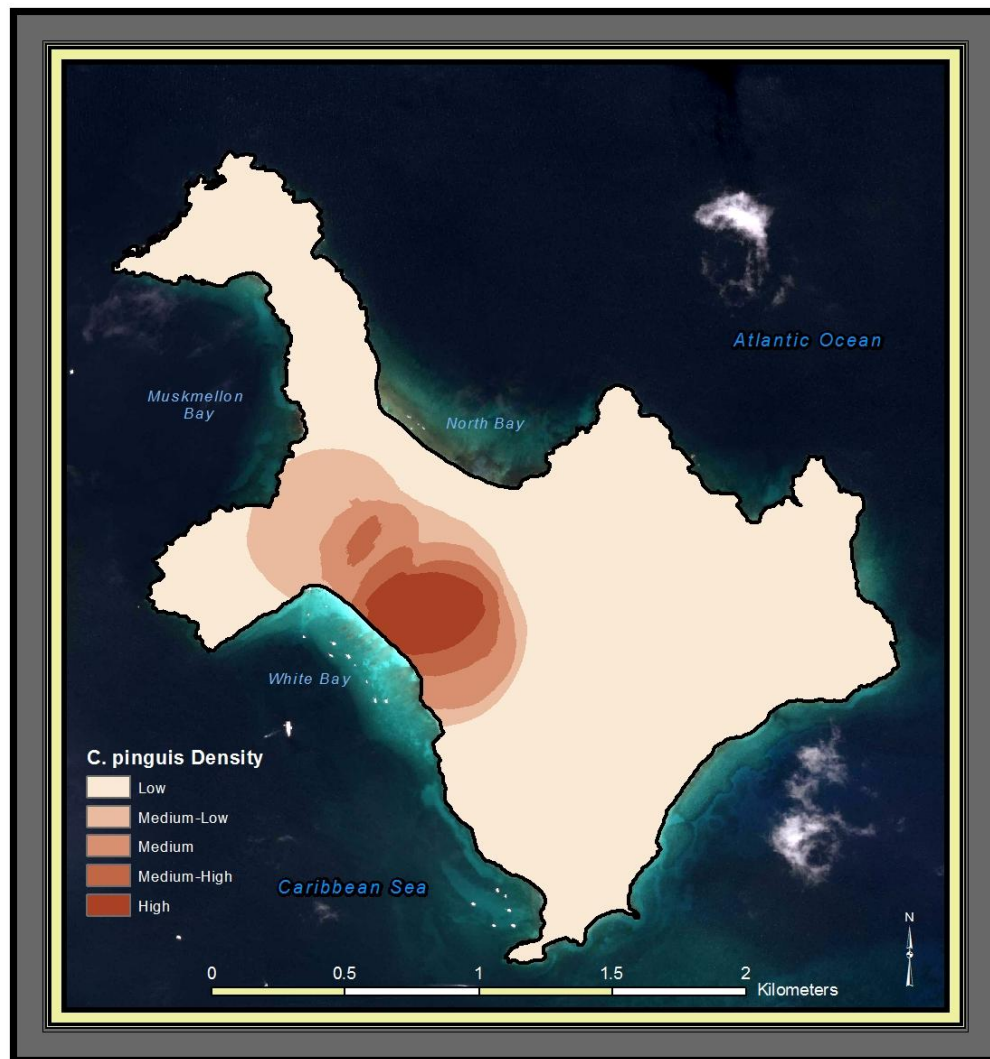


Figure 16—Density surface of iguana sightings for Guana Island

Objective 2 — Estimate abundance of the 2014 *Cyclura pinguis* population on Guana Island

Abundance is one of the most fundamental and sought after parameters in the study of wildlife ecology, as knowledge of population size informs management decisions and wildlife policy (Mills 2007). Accurate estimates of population size are especially important for the successful management and recovery of endangered species. True censuses of wildlife populations are not usually practical or feasible, so numerous field and modeling techniques have been developed to estimate population size and to assess the vital rates influencing population fluctuations (McClintock and White 2009). Two of the most common approaches for estimating population abundances are distance sampling and capture-mark-recapture techniques (Alberts 2000).

The Guana cyclurid population was founded 30 years ago with 8 adult iguanas (Goodyear and Lazell 1994). Since that time, several figures of population abundance have been reported. In 1992, Goodyear and Lazell (1994) estimated the population to be approximately 20 iguanas. Ten years later, Perry and Mitchell (2003) used simple mark-recapture methods to estimate a population of approximately 100 individuals. The most recent estimates of 150-200 iguanas were derived from a mix of occupancy modeling and capture-mark-resight studies conducted between 2008 and 2012 (Bibles unpublished report), but due to low sample sizes and inconsistencies in methodology, the author expressed concerns about the validity of the findings. In 2014, we utilized both distance sampling (in

non-core areas) and capture-mark-resight protocols (within the core) to assess the *C. pinguis* population on Guana Island.

For the purpose of data collection and analyses, we treated the developed and non-developed areas of Guana Island as two separate regions. We also treated hatchling and post-hatch year animals as separate components of the population. October is hatching season for *C. pinguis*, so during the month, an unusually large proportion of the individuals in the population are young-of-the-year. Post-hatch-year-iguanas are large enough that they have no real predators, so during the duration of the study, abundance, rather than survival, is the parameter of primary interest. Conversely, hatchlings are highly susceptible to predation (Figure 17). Also, hatching events occur throughout the month, which can result in the addition



Figure 17—Puerto Rican Racer constricting a marked hatchling on the floor of the clubhouse at 11:00pm on the 24th of October, 2014

of individuals to the existing population, so both abundance and survival are key parameters for this age class. Therefore, population estimates for post-hatch-year iguanas and population estimates and survival for hatchlings must be calculated separately.

We walked over 46 km of survey transects in 2014 and recorded 154 iguana sightings outside the core area. Of these sightings, 110 were part of the distance sampling protocol, and the other 44 were sightings recorded on portions of trails not included in the survey routes or at times outside the sampling interval (Figure 18). We included both visual and auditory cued sighting, and our results



Figure 18—Map of all iguana sightings recorded outside the core research area in 2014

show a clear detection curve with the probability of detection lessening as distance between observer and iguana increased.

Within the core, we captured 136 hatchlings. We recorded 12 sight records of unmarked hatchlings and 467 visual recaptures of previously marked

individuals. We also captured 33 post-hatch-year individuals. We had only 51 visual recaptures of post-hatch-year individuals (14 adults, 4 sub-adults, and 33 yearlings). We did, however, record 305 sight records of unmarked individuals. The majority of these (61%) were unmarked adult sightings, with yearlings being the



Figure 19—Map of the core research area depicting all iguana capture, resight, and sight record localities for 2014 with relevant landmarks and habitually utilized retreat locations

next most commonly sighted unmarked age class. These numbers are quite high, but based on cluster mapping analysis and distinguishing physical markers, we know that some of these sightings were repeat observations of the same individuals (Figure 19).

The population models for the distance sampling and capture-mark-recapture data are not yet complete. But, given the number of captures and sightings, we estimate that our population figures will be slightly higher and more robust than those calculated by Bibles (2012).

Objective 3 — Analyze thermal ecology and the thermal dependence of physiological performance in hatchling stout iguanas on both Guana and Anegada

Compared to other regions of the world, the West Indies has a relatively stable thermal environment. Even so, as a result of the vegetative, microhabitat, and topographic differences between islands, there are dissimilarities in the thermal habitats experienced by cyclurid populations. Therefore, our objectives were to determine the preferred and field-active body temperatures of the Guana hatchling population and to assess the influence of body temperature on both sprint performance and endurance. Additionally, we sought to compare our results to published thermal data from the Anegada population and to evaluate the differences in thermal sensitivity and locomotor performance between head-started and wild hatchlings on the two islands.

Much of the work on hatchling thermal ecology and physiological performance had been completed in previous years (See Appendix 1 – Poster Presentation entitled “Thermal Ecology and Locomotor Performance of the Critically Endangered Stout Iguana (*Cyclura pinguis*)). We determined that sprint performance and endurance capacity exhibited thermal sensitivity. Hatchlings were able to sprint faster and run longer when their body temperatures were between 38° and 40°C, which conformed to both their preferred body temperature range in laboratory testing, and their average field-active body temperatures on Guana and Anegada.

Our initial analysis of physiological performance indicated that endurance capacity in the Guana hatchlings was probably higher than that in the Anegada Headstart population. We also found that Anegada hatchlings had higher endurance capacity than Anegada yearlings, but our Anegada hatchling sample size was quite low. In 2012, the year we began testing on Anegada, we tested 17 yearlings, but there were only 5 healthy hatchlings on the date of our visit. In 2013, their hatchling numbers were higher, and we were able to run 10 additional trials, but as we were testing across two different temperature classes, our sample size for the age class was still too low to accurately compare to the Guana population. Guana’s yearlings were too large to run on our portable treadmill, so we needed more hatchling data to accurately compare our findings. Therefore, the 2014 trip to Anegada was critical to our physiology dataset. We were able to double our hatchling sample, with 16 additional trials. Analyses now give clear evidence that

Guana’s hatchlings have higher endurance capacity than hatchlings from the Headstart facility (Figure 20).

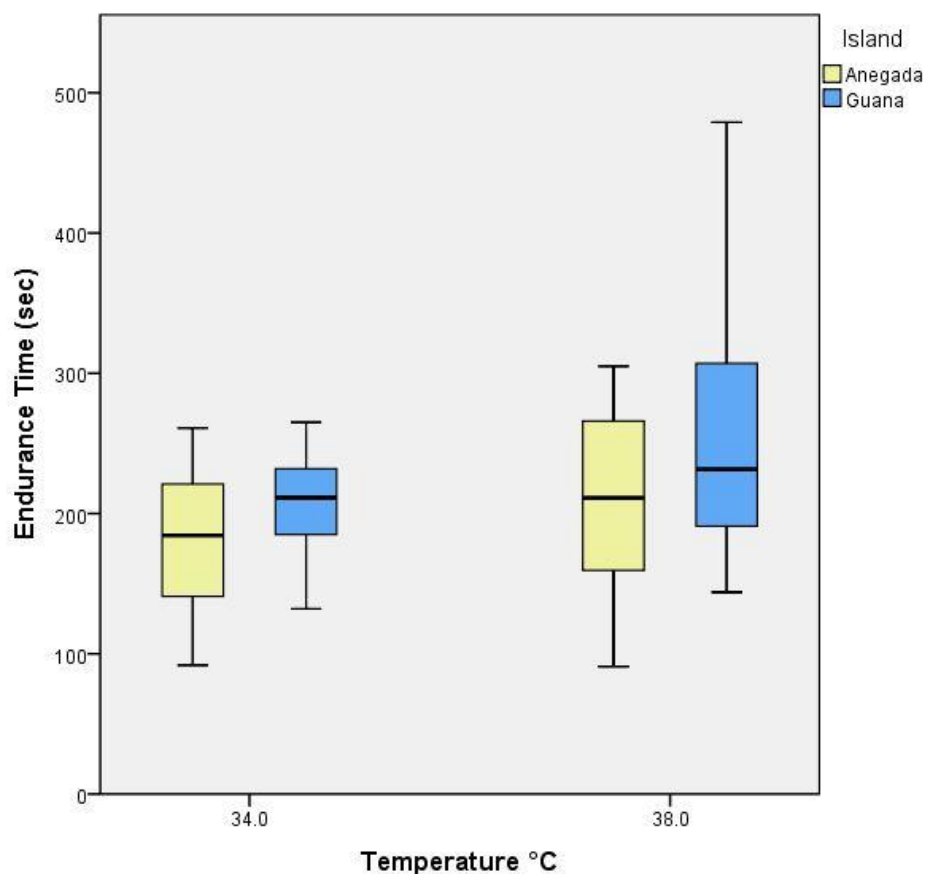


Figure 20—Graphical depiction of endurance capacity for the Anegada and Guana populations, differentiated by thermal treatment group

Objective 4 — Publication and dissemination of data pertaining to the status of *C. pinguis* on Guana Island

Ms. Mougey is currently in the process of finishing the analyses and writing her Ph.D. dissertation based on the 2003 to 2014 iguana data. Although she is including data from before she personally began research, the focus will be on

data she collected between 2011 and 2014. Ms. Mougey is scheduled to complete her dissertation and graduate in May of 2016. Each of the three major chapters of her dissertation will generate at least one paper for publication in the peer reviewed literature, and we will begin the submission process for these papers prior to her graduation.

In addition to the dissertation and resulting publications, she will also be attending several professional conferences and giving presentations based on our findings. At the 2014 Joint Meeting of Ichthyologists and Herpetologists, Ms. Mougey gave a poster presentation on the thermal ecology and locomotor performance of the stout iguana. This was a juried poster competition, and based on her presentation, she won the Henri Siebert Award for the category of Physiology and Morphology. In November of 2015, Ms. Mougey will be attending the Iguana Specialists Group (ISG) Meeting in Florida and giving an oral presentation on the status of the stout iguana population of Guana Island.

THE NEED FOR CONTINUED MONITORING AND RESEARCH

Since 2011, our knowledge base on the natural history and ecology of Guana's *C. pinguis* population has expanded dramatically. However, there is still a great deal of work that needs to be done, with regard to both long-term monitoring and to the acquisition of additional data on crucial seasonal and life history events. In 2011, we GPS collared a single female to test the efficacy of the technique for remotely monitoring iguana movements on Guana Island (Figure 21). The data we

collected from this female is currently the only information we have on nesting movements, timing of reproduction, or year-round homerange fidelity for this population. Further analysis of factors such as spatial ecology, reproductive ecology, energy allocation, foraging, diet, territoriality, and density-dependant population alterations are critical to effective conservation of this population and of the species as a whole. Although the

Science Month program has ended, it is our hope that we may continue this valuable research in the years to come.



Figure 21—Map of locations from the collared female stout iguana showing her nesting location on North Beach and her territory near the hotel complex

LITERATURE CITED

- Alberts, A. C. 2000. West Indian Iguanas: Status Survey and Conservation Action Plan IUCN/SSC West Indian Iguana Specialist Group. Gland, Switzerland.
- Anderson, W. M., G. E. Sorenson, J. D. Lloyd-Strovas, R. J. Arroyo, J. A. Sosa, S. J. Wulff, B. D. Bibles, C. W. Boal, and G. Perry. 2010. Distribution and Habitat Use by the Critically Endangered Stout Iguana (*Cyclura Pinguis*) on Guana Island, British Virgin Islands. *Reptiles and Amphibians* 17:158-165.
- Bradley, K. A., and G. P. Gerber. 2005. Conservation of the Anegada Iguana (*Cyclura Pinguis*). *Iguana* 12:78-85.
- 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 Univeristy Press, Oxford.
- Carey, W. M. 1975. The Rock Iguana, *Cyclura Pinguis*, on Anegada, British Virgin Islands, with Notes on *Cyclura Ricordi* and *Cyclura Cornuta* on Hispaniola. University of Florida.
- Cheek, C. A., S. Hlavaty, K. Mougey, R. N. Perkins, M. A. Peyton, C. N. Ryan, J. C. Zavaleta, C. W. Boal, and G. Perry. 2013. Vertical Structure Use by the Stout Iguana (*Cyclura Pinguis*) on Guana Island, British Virgin Islands. *Reptiles and Amphibians* 20:112-118.
- Dellinger, T., and G. von Hegel. 1990. Sex Identification through Cloacal Probing in Juvenile Marine Iguanas (*Amblyrhynchus Cristatus*). *Journal of Herpetology* 24:424-426.
- Goodyear, N. C., and J. Lazell. 1994. Status of a Relocated Population of Endangered Iguana *Pinguis* on Guana Island, British Virgin Islands. *Restoration Ecology* 2:43-50.
- Lemm, J. M., and A. Alberts. 2012. *Cyclura: Natural History, Husbandry, and Conservation of West Indian Rock Iguanas*. 1st edition. Elsevier/Academic Press, London; Waltham, MA.
- Lettink, M., and D. P. Armstrong. 2003. An Introduction to Using Mark-Recapture Analysis for Monitoring Threatened Species. Department of Conservation Technical Series A 28:5-32.

- McClintock, B. T., and G. C. White. 2009. A Less Field-Intensive Robust Design for Estimating Demographic Parameters with Mark-Resight Data. *Ecology* 90:313-320.
- Mills, L. S. 2007. Conservation of Wildlife Populations: Demography, Genetics, and Management. Blackwell Publishing, Malden, MA.
- Mitchell, N. C. 1999. Effect of Introduced Ungulates on Density, Dietary Preferences, Home Range, and Physical Condition of the Iguana (*Cyclura Pinguis*) on Anegada. *Herpetologica* 55:7-17.
- Mitchell, N. C. 2000. Anegada Iguana. Pages 22-27 *in* Endangered Animals: A Reference Guide to Conflicting Issues. Greenwood Press, London.
- Perry, G., J. Lazell, K. Levering, and N. Mitchell. 2007. Body Size and Timing of Reproduction in the Highly Endangered Stout Iguana, *Cyclura Pinguis*, in the British Virgin Islands. *Caribbean Journal of Science* 43:155-159.
- Perry, G., and N. Mitchell. 2003. Guana and Necker Island Population Assessments 2002. *Iguana* 10:49.
- Pregill, G. K., and S. L. Olson. 1981. Zoogeography of West Indian Vertebrates in Relation to Pleistocene Climatic Cycles. *Annual Review of Ecology and Systematics* 12:75-98.
- Skipper, B., B. Grisham, M. Kalyvaki, K. McGaughey, K. Mougey, L. Navarrete, R. Rondeau, C. Boal, and G. Perry. 2013. Non-Overlapping Distributions of Feral Sheep (*Ovis Aries*) and Stout Iguanas (*Cyclura Pinguis*) on Guana Island, British Virgin Islands. *Reptiles and Amphibians* 20:7-15.



Thermal Ecology and Locomotor Performance of the Critically Endangered Stout Iguana (*Cyclura pinguis*)

Krista Mougey and Gad Perry
Texas Tech University, Lubbock, TX

Introduction

An organism's ability to respond to thermal heterogeneity is defined by the interaction of two continua: 1) the organism's thermoregulatory ability, which measures the extent to which an organism can regulate its body temperature using physiological, morphological, and/or behavioral mechanisms; and 2) the organism's thermal sensitivity, which measures the extent to which body temperature influences metrics of organismal performance. Given that the cellular and enzymatic reaction rates that drive physiological processes are temperature dependent and are often optimized within a narrow temperature range, body temperature (T_b) is one of the most important factors influencing performance, behavior, and fitness of ectotherms.

Herein we describe the results of a thermal ecophysiology study on an artificially established insurance population of critically endangered stout iguanas (*Cyclura pinguis*) translocated from Anegada to Guana Island, British Virgin Islands. Compared to other regions of the world, the West Indies has a relatively stable thermal environment with average diurnal and seasonal temperature variations of only about 5° C. Even so, as a result of the vegetative, microhabitat, and topographic differences between islands, there are dissimilarities in the thermal environments experienced by cyclurid populations. Therefore, our objectives were to determine the preferred and field active body temperatures of the Guana hatchling population and to assess the influence of body temperature on both sprint performance and endurance. Additionally, we sought to compare our results to published thermal data from the Anegada population and to evaluate the differences in thermal sensitivity and locomotor performance between head-started and wild hatchlings on the two islands.



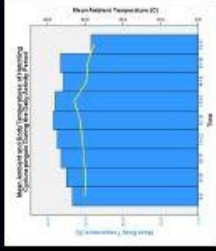
Materials and Methods

- Field Active Body Temperature was assessed during daily activity periods using spot-sampling techniques from hatchling *C. pinguis* captures. Ambient and cloacal body temperatures were taken using a quick-reading Ruke thermometer.
- Preferred Set-Point Temperature Range was assessed using a 1.6 x 0.5 m chamber with a thermal gradient ranging from approximately 28° to 42° C. Initial body temperature was recorded, and the hatchling was released into the gradient and allowed to thermoregulate. Trial duration was one hour, and body temperatures were recorded at 30 minute intervals.
- Thermal Dependence of Sprint Performance was assessed using a 2.5 meter long sprint track equipped with 8 photocell sensors at quarter meter intervals. Hatchlings were assigned to one of four thermal treatment groups (28, 34, 38, or 42°C) and placed in a thermostatically regulated incubator for a minimum of 30 minutes prior to trial commencement. After recording cloacal body temperature, each individual was finger-chased down the length of the track, and the trial was classed as "good" or "poor". Hatchlings were raced 3 times each with a minimum 30 minute rest/incubation period between trials.
- Thermal Dependence of Endurance Capacity was assessed using a low velocity motorized treadmill designed to maintain a belt speed of 0.7 km/h at 0% slope. On Guana, hatchlings were assigned to one of four thermal treatment groups and placed in the incubator for a minimum of 30 minutes prior to the trial. On Anegada, incubation was not possible, and individuals were run near-ambient at 34 or 38°C. After recording body temperature, each individual was placed on the belt and encouraged to maintain consistent locomotion via finger brush techniques. Individuals were run to exhaustion (as indicated by failure to right).
- Statistical Analyses were performed using SPSS version 21 with $\alpha = 0.05$.



Results

- Field Active Body Temperature: Hatchling *C. pinguis* were found active between 8:00 and 17:00, with peak activity occurring during the active period ranging from 26.4 to 37.4 °C, and field active temperatures of the hatchlings ($N = 191$) ranged from 27.6 to 43.0 °C with an average of 36.8 ± 2.7 °C.
- Preferred Set-Point Temperature Range: Hatchlings placed into the thermal gradient for set-point preference trials ($N = 12$) had initial body temperatures between 28.8 and 31.7 °C (average 30.4 ± 0.95 °C). After 30 minutes average body temperature had risen to 30.16 ± 0.68 °C. Average body temperature readings reached 30.49 ± 0.25 °C at 60 minutes, and were statistically indistinguishable from the 30 minute thermal measures ($t_{(11)} = 1.440$, $p = 0.277$).
- Thermal Dependence of Sprint Performance:
 - Maximal sprint performance trials were conducted on 51 hatchlings from Guana Island, but 7 individuals had to be culled due to poor performance across all three test replications ($N = 44$; $N_{28} = 13$, $N_{34} = 8$, $N_{38} = 16$, $N_{42} = 7$).
 - Initial burst-speed on the first 50 cm of track was highly variable across thermal treatment groups, but was significantly correlated with tail length ($F_{(1,44)} = 4.25$; $p = 0.046$); however neither thermal treatment group ($F_{(1,44)} = 1.16$; $p = 0.32$) nor tail length ($F_{(1,44)} = 0.66$; $p = 0.42$) were significant predictors of maximal 50 cm split-speed across the track.
 - Maximum 50 cm split-speeds ranged from 1.97 to 9.43 m/sec (average 3.29 ± 1.17).
 - Average speed across the track as a whole (sprint duration) was significantly correlated with both thermal treatment group ($F_{(1,44)} = 6.99$; $p = 0.012$) and tail length ($F_{(1,44)} = 4.38$; $p = 0.043$).
- Post-hoc analyses using Fisher's LSD indicate that significant differences exist between the sprint speeds of individuals run at 28 °C and 38 °C, but that all other pair-wise comparisons are statistically indistinguishable.
- Neither snout-vent-length nor hind-limb-length were significant predictors of maximal sprint performance in any model.

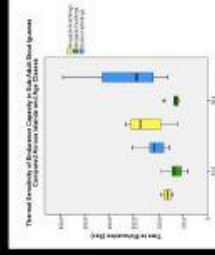
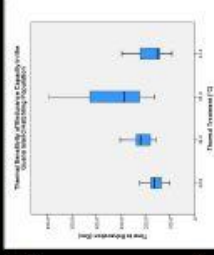


Thermal Dependence of Endurance Capacity

- Endurance capacity trials were successfully conducted on 67 ($N_{28} = 10$, $N_{34} = 11$, $N_{38} = 37$, $N_{42} = 9$) wild hatchlings from Guana Island and on 15 ($N_{28} = 3$, $N_{34} = 12$) captive hatchlings and 20 ($N_{28} = 14$, $N_{34} = 6$) captive yearlings from the Anegada headstart facility. The wild iguanas on Guana Island exhibit higher growth rates than the captive reared animals, and yearlings on Guana were too large to be tested on our low velocity treadmill.
- Two measures of endurance capacity were analyzed:
 - Time to Exhaustion (T_1), no longer able to keep pace with the belt
 - Time to Exhaustion (T_2), lack of righting response
- T_1 (average 219 ± 92 seconds) and T_2 (average 263 ± 119 seconds) were statistically distinct ($t_{(90)} = 7.934$; $p < 0.001$) within the Guana population with the T_2 duration being an average of 44 seconds longer.



- Body temperature was a significant predictor of endurance capacity in Guana population ($F_{(1,48)} = 5.82$; $p = 0.019$), and performance was maximized near 38.0 °C.
- Post-hoc analyses using Fisher's LSD indicate that significant differences exist between the endurance capacity of individuals run at 38 °C and those of all other thermal treatment groups. All other thermal pair-wise comparisons were statistically indistinguishable.
- Within the Anegada population, age class ($F_{(1,44)} = 20.024$; $p < 0.001$) was the only significant predictor of endurance capacity (body temperature $F_{(1,44)} = 3.484$; $p = 0.071$), a likely result of limited sample size.
- Hatchlings from the headstart facility displayed higher endurance capacity than yearlings (average 236 ± 72 seconds and 177 ± 27 seconds respectively).
- Guana's wild ($N = 48$ hatchlings) and Anegada's headstart ($N = 15$ hatchlings; $N = 20$ yearlings) iguanas showed statistically disparate endurance capacities. In cross-island comparisons of the 34°C and 38°C thermal treatment groups, age class ($F_{(1,44)} = 4.005$; $p = 0.049$), Island ($F_{(1,44)} = 5.942$; $p = 0.017$), and body temperature ($F_{(1,44)} = 10.714$; $p = 0.002$) were all significant predictors of endurance capacity.



Discussion

Evaluating the thermal sensitivities of key physiological processes and ecologically meaningful performance indices can provide information on thermoregulatory strategies, ecophysiological performance costs, and other thermally-linked fitness considerations influencing individuals in natural populations. Research into these topics is especially important for the effective conservation of species such as *C. pinguis*, a rock iguana that has been introduced to several disparate Caribbean islands as an ex-situ conservation method. Field active (26.7 to 43.0 °C) and preferred temperatures (30.5 °C) recorded for Guana's cyclurids were similar to the thermal values published for the Anegada source population (30.6 to 42.2 °C and 38.0 °C respectively); however, approximately 4% of the Guana body temperature records were below the Anegada minimum.

In *C. pinguis* both sprint performance and endurance capacity exhibited thermal sensitivity, with maximal performance occurring near the set-point range as would be expected based on thermal ecological theory. The only morphometric measure significantly correlated with either metric of locomotor performance was tail length; however, age class likely displays some degree of collinearity with both mass and snout-vent-length. The statistical differences seen in the endurance capacity and thermal sensitivity of iguanas across islands warrants further study into the differences between populations of this critically endangered species.



Acknowledgements

We thank Texas Tech University and The Conservation Agency for providing financial support through a grant from The Falconwood Foundation. We thank the staff of Guana Island for technical assistance, Little Thatch Resort and Houston Safari Club for equipment funds, and Douglas Bell along with many other volunteers for their dedication in the field. Photographs courtesy of Krista Mougey and Douglas Bell.

Assessing Feral Sheep Effects on Plant Composition and Structure in a Caribbean Tropical Dry Forest

Rachel Granberg^{1,*}, Lori E. Brown¹, Jessica L. East¹, Alexis L. Garcia¹, Alixandra J. Godar¹, Maria F. Mejia¹, Clint W. Boal², and Gad Perry¹

¹Department of Natural Resources Management, Texas Tech University, Lubbock, Texas 79409 ² U.S. Geological Survey, Texas Cooperative Fish and Wildlife Research Unit, Department of Natural Resources Management, Texas Tech University, Lubbock, Texas 79409 * Corresponding author - rachelmgranberg@gmail.com

Abstract - Invasive mammalian herbivores impact ecosystems worldwide, often with poorly documented and understood effects. To determine the effect of feral sheep on vegetation structure and composition in the British Virgin Islands, we conducted stratified random vegetation surveys and compared vegetation between paired ungulate exclosures and control plots. Our results indicate that plant communities <0.5 m in height were significantly reduced by sheep density and by abiotic factors. This was supported by data from experimental exclosures. Island-wide, plants of all age classes were also negatively impacted by sheep density and select abiotic factors, but this was not supported by exclosure plots. Our results suggest that sheep herbivory negatively impacts young plants, potentially affecting plant recruitment, forest age structure, and endangered native wildlife species.

Introduction

Invasion by non-native species is a significant threat to biodiversity, particularly in island ecosystems (Millennium Ecosystem Assessment 2005). Caribbean islands are a biodiversity hotspot (Myers et al. 2000) and the tropical dry forests that occur there are

vulnerable, specialized, and highly endangered (Gerhardt 1993, Janzen 1988, Sanchez-Azofeifa et al. 2005). The fragility of the Caribbean Islands is attributed to soil erosion, hurricanes, and human disturbances (Vieira and Scariot 2006), all of which contribute to alterations in wildlife populations and species extinctions (Lugo et al. 1981). Invasive species are abundant and broadly distributed (e.g., Powell et al. 2011) throughout the Caribbean and are common contributors to species declines.

We examined possible relationships between feral sheep density, abiotic factors, and vegetation community structure and composition on Guana Island, British Virgin Islands. Sheep were introduced to Guana in the 1950's, eventually roaming freely and successfully colonizing the island in the absence of predators (Lazell 2005). Period control activities have at times limited the area inhabited by them and reduced their numbers. Feral sheep appear to influence understory vegetation and distribution of the critically endangered Stout Iguana (*Cyclura pinguis*) on Guana (Skipper et al. 2013). Here we focus on the vegetation that iguanas and other organisms feed on, and test the hypothesis that increased sheep density will negatively affect the native vegetation.

Methods

We conducted island-wide surveys of vegetation understory, overstory, and resident sheep pellet groups (as a proxy of sheep density) on Guana Island ($N = 84$). We also conducted vegetation surveys at paired 16-year sheep exclosure pens and adjacent control plots ($N = 10$). Using these data, we assessed vegetation response to differing levels of sheep density and a suite of abiotic variables.

Field-Site Description

Guana Island is a ~300 hectare privately-owned island in the British Virgin Islands located on the eastern edge of the Greater Antilles Islands (Fig. 1). Rainfall is unimodal, averaging 921 mm annually, with peak precipitation occurring September through November. This pattern creates a distinct dry season and vegetation is adapted to prolonged periods of drought (Lazell 2005). The island is largely (> 90%) covered in subtropical dry forest with an elevation of 264 m (Lazell 1996). Guana is composed of Upper Cretaceous igneous extrusive volcanic rock over 70 million years old (Lazell 2005). Soils are shallow, loamy inceptisols and are considered highly erodible (Rankin 2002). Guana has a steep topographic gradient, with severe ridges, narrow drainages, and a salt pond occupying much of island's flat central area. Extensive research on the island provides an unusually developed understanding of the biota (Lazell 2005).

Vegetation Surveys

Vegetation and fecal pellet count surveys were conducted in October 2014. We subdivided Guana Island into four quadrants representing different levels of perceived sheep density across the island based on Skipper et al. (2013; Fig. 2). Because of terrain constraints, we used the established trail system as a guide for site locations surveying 20-22 randomly-selected locations within each quadrant with a minimum of 10 m between sites. At each site, we established a 20 m² inventory plot, the center line positioned 2 m away from and parallel to the trail. The inventory survey consisted of a 2 m x 10 m belt transect and ten gap 2 m intercepts surveys conducted at each plot, perpendicular to the established belt transect line at each meter mark. All plants rooted in the plot were identified to one of five pre-determined functional groups: vine, epiphyte, succulent, armed woody plant, and unarmed woody plant.

For belt transects, we categorized all plants by functional group (see above) and height class (<0.5 m, 0.5 m – 1.0 m, 1.0 m – 1.5 m, 1.5 m – 2.0 m, and >2.0 m) to calculate stem density and approximate forest size structure (a proxy for age). Percent canopy cover

was visually estimated to the nearest 10% at every meter along the 10 m belt transect line to generate average canopy cover for each transect. In addition to vegetation data, the following site information was collected: slope, aspect and position in the landscape (bottom, lower third, middle third, upper third, ridge) at origin of each transect line, as well as litter depth (measured to the nearest tenth of a centimeter using calipers) at 2-3 equidistant locations along the transect.

Plants <0.5 m were additionally surveyed using a modified gap intercept method (Herrick et al. 2005) to estimate foliar cover of the youngest, most at-risk to herbivory plants. At 10 points along the belt transect line, an observer placed a 2 m line perpendicular to the original transect. The observer measured the distance each plant, identified to functional group, intersected the 2 m line to the nearest cm. Ground cover was also measured to the nearest cm (bare soil, litter, rock, or other) for each gap intercept survey.

Fecal pellet surveys were used to create an index of the relative sheep density across the island. We conducted pellet group searches along the established trail system throughout Guana. For quantification purposes, we considered pellets >5 cm from another group or pellet as a separate group. Using ArcGIS, we created a raster surface that summed pellet groups contained within a 100 m² pixel resolution over the entire island.

Sheep Exclosure Plots

We surveyed 5 pairs of fenced sheep exclosure plots and adjacent control plots. Each plot was 10 m² and pairs consisted of identical topographical covariates (slope, aspect, position in the landscape). Gap intercept and belt transect, outlined above, were conducted at the center of each exclosure and control plot, parallel to the nearest trail. The exclosure plots had been fenced for approximately 16 years. Although sheep have been unable to enter these areas, we assumed insects, birds, and iguanas had not been impeded from accessing the plots.

Analysis

We used Principle Component Analysis (prcomp) in Program R to reduce dimensionality of our data. In accordance with the Kaiser criterion, components with an eigenvalue of > 1.0 were retained for further analysis with any factor loading of a coefficient $> |0.3|$ considered important (Tabachnick and Fidell 2001). We then used MANCOVA to evaluate differences in response of principal components to covariates in the dataset (James and McCulloch 1990), including sheep index, aspect, slope, position within the landscape, and average litter depth.

With belt transect analysis, we used data for unarmed woody plants, armed woody plants, succulents, and vines, each grouped by height categories to approximate age class. For gap transect analysis, we evaluated the presence of total vegetation, canopy cover, unarmed woody plants, armed woody plants, succulents, vines, succulents, epiphytes and ground cover (litter, rock, bare soil, or other). We compared the sheep exclosure plots to respective control plots using one-sided paired t-tests, testing the hypothesis that a greater density of vegetation will be found within the exclosures.

Results

Gap Intercept

We performed a principle component analysis (PCA) with the gap transect data, keeping 4 of the 11 components for further analysis for a cumulative explanatory power of 68.2%. (Table 1). Following the PCA, we ran a MANCOVA ($N = 84$) with the retained components. The assumption of normality was supported based upon histogram plots and Q-Q plots of the 4 components retained. We rejected homogeneity of variance-covariance based

on a Levene's test ($P = 0.005$). However, because our sample sizes were roughly equal among treatments, we continued with a MANCOVA (Tabachnik and Fidell 2001).

For the MANCOVA, we evaluated differences in our 4 principle components based on our index of sheep density, aspect, slope, position within the landscape, average litter depth and their interactions. Greater sheep density index ($p < 0.001$), higher position in the landscape ($P < 0.001$), exposed aspects ($P < 0.001$), and shallow average litter depth ($p < 0.001$) were all correlated with reduced vegetation. The interactions between sheep index and position in the landscape ($P < 0.001$), position in the landscape and aspect ($P < 0.001$), position in the landscape and average litter depth ($P < 0.001$), aspect and average litter depth ($P = 0.002$), aspect and slope ($P = 0.002$), average litter depth and slope ($P = 0.04$), and sheep index, position in the landscape, and slope ($P = 0.03$) were all significant.

Belt Transect

With the belt transect data, we conducted another PCA and retained components meeting the Kaiser criterion (eigenvalue > 1.0), giving us 6 of 16 components (Table 2) with a cumulative 62.9% of the total variance explained. After performing a PCA with the belt transect data, we conducted a MANCOVA ($N = 84$) to evaluate variables that affect observed differences in vegetation. We assessed independence, normality, and homogeneity of variance-covariance. The assumption of normality was satisfactory based upon histogram plots and Q-Q plots of the 6 components retained. We rejected homogeneity of variance-covariance based on a Levene's test ($P = 0.005$). However, because our sample sizes were roughly equal among treatments, we continued with a MANCOVA (Tabachnik and Fidell 2001).

We evaluated differences in the 6 retained principle components based on sheep index, aspect, slope, average litter depth, and position within the landscape and their

interactions. We observed significantly less vegetation in exposed aspects ($P = 0.04$), with the interaction of exposed aspects and steep slopes ($P = 0.02$), and with the interaction of exposed aspects and greater sheep density ($P = 0.02$). We also observed a trend for the interaction between sheep index and average litter depth ($P = 0.09$).

Sheep Exclosure Plots

We found significant differences in density of vegetation <0.5 m and no differences in density of belt transect vegetation ($N = 5$) between the exclosure treatment and control plots. With gap intersect, we found significantly lower density of overall vegetation ($P = 0.03$) in control plots versus plots excluded from sheep grazing. We also found a strong trend ($P = 0.06$) for lower density of unarmed woody plants in control plots. We did not find significant differences in litter depth or density of vines or succulents. For belt transect, we did not find any significant differences in plant density or composition (armed and unarmed woody plants, vines, and succulents) between control and exclosure plots.

Discussion

Biotic Factors

Our study found that higher sheep density was correlated with reduced amount of vegetation present, most notably at low heights easily accessible to sheep. Gap intercept, which assessed vegetation <0.5 m, was responsive to our biotic variable of sheep index for island-wide surveys as well as our paired exclosure/control surveys. Belt transect, measuring plants of all height classes, was not responsive to the index of sheep density alone in island-wide or exclosure/control surveys. However, when considering the interaction of aspect and sheep index, results for belt transect were significant for the island-wide surveys, indicating that sheep may decrease vegetation to a greater degree on more exposed aspects.

Although there are multiple pathways for herbivory to occur, many of the plant species present in the Caribbean are more sensitive to mammalian herbivory because they have evolved in the absence of these organisms (Bowen and Van Vuren 1997, Coley and Barone 1996). Several studies found that feral sheep have a strong preference for consuming endemic island species over mainland plant species (Bowen and Van Vuren 1997, Van Vuren and Coblentz 1987). Furthermore, rates of mammalian herbivory are up to 25 times higher on young, more nutritious plants compared to mature leaves (Coley and Barone 1996), which has the potential to alter plant community composition and age structure through reduced vigor or induced mortality.

In addition to the loss of vegetation biomass due to mammalian herbivores, indirect effects of non-native herbivory are plentiful. Concentrated presence of ungulates, like feral sheep along trail systems, leads to increasing soil erosion (Cumming and Cumming 2003, Sharrow 2007), soil compaction (Basset et al. 2005, Heckel et al. 2010), nutrient availability reduction (Bradshaw 1969), and reduction in seed germination and plant recruitment (Basset et al. 2005). The effects may be synergistic as well; a reduction in litter accumulation accompanies reduction in vegetation (Sharrow 2007), increasing erosion and reducing soil nutrient availability (Bradshaw 1969).

Belt transects conducted in the exclosure/control plots did not demonstrate differences in the amount of vegetation present based on functional group, indicating that older age classes of vegetation may not be as heavily influenced by sheep herbivory as the youngest classes. Given the slow rate of growth and recruitment in tropical dry forests (Murphy and Lugo 1986), longer-term and larger-scale data sets may be necessary to establish relationships between older age classes of plants and sheep herbivory.

Abiotic Factors

Overarching limiting factors of light availability, soil nutrition, and soil moisture affect tropical dry forests differently than the more often-studied tropical moist forests (Ceccon et al. 2006). Light availability, which is a major component in plant recruitment within tropical moist forests, does not have as distinct effects in tropical dry forests due to more open and shorter canopy structure (Brienen et al. 2009, Lebrija-Trejos et al. 2010, Swaine et al. 1990, Wright 2002). The effect of nutrient availability is not extensively studied in the tropical dry forest system; however, there is evidence to suggest that nutrition is a not limiting factor for growth (Murphy and Lugo 1986; but see Marrs et al. 1991). High variability in rainfall amount, length of drought, and precipitation-dependent nutrient availability means that tropical dry forests experience more environmental stress than their mesic counterparts in terms of available soil moisture (Bradshaw 1969, Ceccon et al. 2006). For these reasons, this research focused on easily-measured abiotic covariates that influenced available soil moisture.

With the exception of slope, the youngest age class of plants responded to all abiotic covariates included in analysis as well as interactions between all abiotic covariates. Plants assessed in belt transect surveys responded to abiotic variables, but not to the extent that smaller, younger plants responded. Lack of significance of sheep index and reduced number of significant abiotic covariates with older plants may suggest ontogenetic shifts in growth-limiting factors and causes of plant mortality (Alvarex-Clare and Kitajima 2009, Auspburger 1983) or reflect a lack of ecological relevance for belt transect with respect to older age classes. Although 2 m wide belt transects are common, the transect length (10 m) was short relative to other forestry studies. Widening or lengthening transects may have altered the final results for older age classes; however, research has demonstrated that a shift factors limiting recruitment in common in forest systems (Clark et al. 1999).

Conclusions

We observed a reduction in short vegetation in areas where sheep were most abundant. Smaller vegetation may be more sensitive to herbivory whereas older, taller vegetation may have more efficient defenses against herbivory and may be out of reach for sheep. Our results indicate that herbivory is negatively impacting the youngest vegetation on Guana Island, with potential negative effects on seedling recruitment, forest age structure, and native wildlife if sheep presence is allowed to continue. Although longer-term and larger-scale exclusion studies may develop a greater understanding of the extent of feral ungulate impact and time to forest recovery on Guana Island, these results justify a call for immediate removal of feral sheep.

Acknowledgements

We thank M. Vanlandeghem and R. Rondeau for providing critical insight on various aspects of the project. Funding for this project was provided by The Conservation Agency through a grant from the Falconwood Foundation.

Literature Cited

- Alvarex-Clare, S. and K. Kitajima. 2009. Susceptibility of tree seedlings to biotic and abiotic hazards in the understory of a moist tropical forest in Panama. *Biotropica* 41:47–56.
- Atkinson, I.S. 1989. Introduced animals and extinctions. Pp. 54–69, *In* D. Western and M.C. Pearl (Eds.). *Conservation for the twenty-first century*. Oxford University Press, New York, NY.
- Augspurger, C.K. 1983. Offspring recruitment around tropical trees: Changes in cohort distance with time. *Oikos* 40:189–196.
- Baber, D.W. and B.E. Coblenz. 1986. Density, home range, habitat use and reproduction in feral pigs on Santa Catalina Island. *Journal of Mammalogy* 67:512–525.
- Basset, I.E., R.C. Simcock, and N.D. Mitchell. 2005. Consequences of soil compaction for seedling establishment: Implications for natural regeneration and restoration. *Austral Ecology* 30:827–833.
- Boege, K. and R.J. Marquis. 2005. Facing herbivory as you grow up: The ontogeny of resistance in plants. *Trends in Ecology and Evolution* 20:441–448.
- Bowen, L. and D. Van Vuren. 1997. Insular endemic plants lack defenses against herbivores. *Conservation Biology* 11:1249–1254.
- Bradshaw, A. 1969. An ecologist's viewpoint. Pp. 415–427, *In* I.H. Rorinson (Ed.). *Ecological Aspects of the Mineral Nutrition of Plants*. Blackwell Publishing, Oxford, England.
- Brienen, R.J., P.A. Zuidema, and M. Martinez-Ramos. 2009. Attaining the canopy in dry and moist forests: Strong differences in tree growth trajectories reflect variation in growing conditions. *Oecologia* 163:485–496.
- Bullock, D. and S. North. 1985. Round Island in 1982. *Oryx* 18:36–41.

- Cabin, R.J., S.G. Weller, D.H. Lorence, T.W. Flynn, A.K. Sakai, D. Sandquist, and L.J. Hadway. 2000. Effects of long-term ungulate exclusion and recent alien species control on the preservation and restoration of a Hawaiian tropical dry forest. *Conservation Biology* 14:439–453.
- Carlquist, S. 1974. *Island biology*. Columbia University Press, New York, NY. 656 pp.
- Ceccon, E., S. Sanchez, and J. Campo. 2004. Tree seedling dynamics in two abandoned tropical dry forests of differing successional status in Yucatan, Mexico: A field experiment with N and P fertilization. *Plant Ecology* 170:277–285.
- Ceccon, E., P. Huante, and E. Rincon. 2006. Abiotic factors influencing tropical dry forests regeneration. *Brazilian Archives of Biology and Technology* 49:305–312.
- Clark, J.S., B. Beckage, P. Camill. 1999. Interpreting recruitment limitation in forests. *The American Journal of Botany* 86:1–16.
- Coley, P.D. and J.A. Barone. 1996. Herbivory and plant defenses in tropical forests. *Annual Review of Ecology and Systematics* 27:305–335.
- Comita, L.S., M. Uriarte, J. Thompson, I. Jonckheere, D.D. Canham, and J.K. Zimmerman. 2009. Abiotic and biotic drivers of seedling survival in a hurricane-impacted tropical forest. *Journal of Ecology* 97:1346–1359.
- Cronk, Q.B.C. 1986. The decline of the St. Helena ebony *Trichetiopsis melanoxylon*. *Biological Conservation* 35:159–172.
- Cumming, D.H.M. and G.R. Cumming. 2003. Ungulate community structure and ecological processes: Body size, hoof area and trampling in African savannas. *Oecologia* 134:560–568.
- Dilks, P.J. and P.R. Wilson. 1979. Feral Sheep and cattle and royal albatrosses on Campbell Island; population trends and habitat changes. *New Zealand Journal of Zoology* 6:127–139.

- Dupuy, J.M. and R.L. Chazdon. 2008. Interacting effects of canopy gap, understory vegetation, and leaf litter on seedling recruitment and composition in tropical secondary forests. *Forest Ecology and Management* 255:3716–3725.
- Gerhardt, K. 1993. Tree seedling development in tropical dry abandoned pasture and secondary forest in Costa Rica. *Journal of Vegetation Science* 4:95–102.
- Heckel, C.D., N.A. Bourg, W.J. McShea, and S. Kalisz. 2010. Nonconsumptive effects of a generalist ungulate herbivore drive decline of unpalatable herbs. *Ecology* 91:319–326.
- Herrick, J.E., J.W. Van Zee, K.M. Haystad, L.M. Burkett, and W.G. Whitford. 2005. Monitoring manual for grassland, shrubland, and savanna ecosystems, Volume II. The University of Arizona Press, Tucson, AZ. 201 pp.
- Imbert, D. and J. Portecop. 2008. Hurricane disturbance and forest resilience: Assessing structural vs. functional changes in a Caribbean dry forest. *Forest Ecology and Management* 255:3494–3501.
- James, F.C. and C.E. McCulloch. 1990. Multivariate analysis in ecology and systematics: Panacea or Pandora's Box? *Annual Review of Ecology and Systematics* 21:129–166
- Janzen, D.H. 1988. Tropical dry forests, the most endangered major tropical ecosystem. Pp. 130–137, *In* E. O. Wilson (Ed.). *Biodiversity*. National Academy Press, Washington, D.C.
- Krauss, U. 2010. Invasive Alien Species (IAS) awareness baseline survey, Saint Lucia, 2010. Ministry of Agriculture, Lands, Forestry and Fisheries, Castries, St. Lucia., Forestry Department, St. Lucia. 57 pp.
- Lazell, J. 1996. Guana Island: A Natural History Guide. The Conservation Agency Occasional Paper 1:1–20.
- Lazell, J. 2005. *Island Fact and Theory in Nature*. University of California Press. Berkeley, CA. 402 pp.

- Lebrija-Trejos, E., E.A. Perez-Garcia, J.A. Meave, F. Bongers, and L. Poorter. 2010. Functional traits and environmental filtering drive community assembly in a species-rich tropical system. *Ecology* 91:286–398.
- Long, J. 2003. Introduced mammals of the world: Their history, distribution, and influence. CSIRO Publishing. Collingwood, Victoria, Australia. 590 pp.
- Lugo, A.E., R. Schmidt, and S. Brown. 1981. Tropical forests in the Caribbean. *Ambio* 10:318–324.
- Marrs, R.H., J. Thompson, D. Scott, and J. Proctor. 1991. Nitrogen mineralization and nitrification in Terre Firme Forest and savanna soils on Ilha de Maraca, Roraima, Brazil. *Journal of Tropical Ecology* 7:123–137.
- Millennium Ecosystem Assessment. 2005. Ecosystems and Human Well-being: Biodiversity Synthesis. World Resources Institute, Washington, D. C. 100 pp.
- McIntosh, P.D. and R.B. Allen. 1998. Effect of exclosure on soils, biomass, plant nutrients, and vegetation, on unfertilized steeplands, Upper Waitaki District, South Island, New Zealand. *New Zealand Journal of Ecology* 22:209–217.
- Murphy, P.G. and A.E. Lugo. 1986. Ecology of Tropical Dry Forest. *Annual Review of Ecology and Systematics* 17:67–88.
- Murphy, P.G., A.E. Lugo, A.J. Murphy, and D.C. Nepstad. 1995. The dry forests of Puerto Rico's South coast. *Tropical Forests: Management and Ecology. Ecological Studies* 112:178–209.
- Myers, N., Mittermeier, R.A., Mittermeier, C.G., A.B. da Fonseca, G., Kent, J. 2000. Biodiversity hotspots for conservation priorities. *Nature* 403:853–858.
- Powell, R., R.W. Henderson, M.C. Farmer, M. Breuil, A.C. Echternacht, G. van Buurt, C.M. Romagosa, and G. Perry. 2011. Introduced amphibians and reptiles in the greater Caribbean: Patterns and conservation implications. Pp. 63-143 *In: Conservation of*

- Caribbean Island Herpetofaunas Volume 1. Edited by Adrian Hailey, Byron S. Wilson, and Julia A. Horrocks.
- Rankin, D. 2002. Geology of St. John, U.S. Virgin Islands. US Geological Survey Professional Paper 1631:1–30.
- Sanchez-Azofeifa, G.A., M. Kalacska, M. Quesada, J.C. Calvo-Alvarado, J.M. Nassar, and J.P. Rodriguez. 2005. Need for integrated research for a sustainable future in tropical dry forests. *Conservation Biology* 19:285–286.
- Sayer, E.J. 2005. Using experimental manipulation to assess the roles of leaf litter in the functioning of forest ecosystems. *Biological Review* 80:1–31.
- Sharrow, S.H. 2007. Soil compaction by grazing livestock in silvopastures as evidenced by changes in soil physical properties. *Agroforestry Systems* 71:215–223.
- Skipper, B., B. Grisham, M. Kalyvaki, K. McGaughey, K. Mougey, L. Navarette, R. Rondeau, C. Boal, and G. Perry. 2013. Non-overlapping distributions of feral sheep (*Ovis aries*) and stout iguanas (*Cychura pinguis*) on Guana Island, British Virgin Islands. *ICRF Reptiles and Amphibians* 20:7–15.
- Swaine, M.D., D. Lieberman and J.B. Hall. 1990. Structure and dynamics of a tropical dry forest in Ghana. *Vegetation* 88:31–51.
- Tabachnick, B.G. and L.S. Fidell. 2001. Using multivariate statistics, 4th edition. Allyn and Bacon, Needham Heights, MA. 966 pp.
- Van Vuren, D. 1992. Eradication of feral goats and Sheep from island ecosystems. Pp. 377–381, *In* J.E. Borrecco and R.E. Marsh (Eds.). *Proceedings of the Fifteenth Vertebrate Pest Conference*, University of California, Davis, CA.
- Van Vuren, D. and B.E. Coblenz. 1987. Some ecological effects of feral Sheep on Santa Cruz island, California, USA. *Biological Conservation* 41:253–268.

- Vieira, D.L. and A. Scariot. 2006. Principles of natural regeneration of tropical dry forests for restoration. *Restoration Ecology* 14:11–20.
- Vitousek, P.M. 1988. Diversity and biological invasions of oceanic islands. Pp 130–137, *In* Biodiversity, E.O. Wilson (Ed.) National Academy Press, Washington D.C.
- Walter, H.S. and G.A. Levin. 2008. Feral Sheep on Socorro Island: Facilitators of alien plant colonization ecosystem. *Diversity and Distributions* 14:422–431.
- Weller, S.G., R.J. Cabin, D.H. Lorence, S. Perlman, K. Wood, T. Flynn, and A.K. Sakai. 2011. Alien plant invasions, introduced ungulates, and alternative states in a mesic forest in Hawaii. *Restoration Ecology* 19:671–680.
- Wright, S.J. 2002. Plant diversity in tropical forests: A review of mechanisms of species coexistence. *Oecologia* 130:1–14.

Figures



Fig. 1. Guana Island is part of the British Virgin Islands, north of Tortola, on the eastern edge of the Greater Antilles Islands.

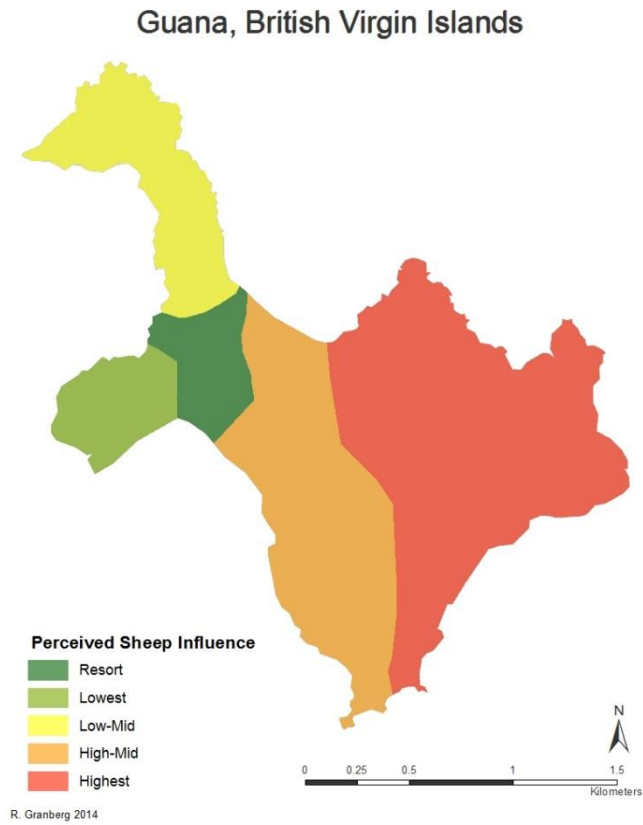


Fig. 2. Perceived sheep density on Guana Island, based on current literature and unpublished data. See text for details.

Tables

Component loadings Variables	Principal component axes			
	1	2	3	4
Functional Group	-0.44	0.01	-0.19	0.05
Total Vegetative Cover	-0.51	0.01	-0.14	-0.10
Canopy Cover	0.31	0.17	-0.34	0.03
Litter	-0.05	0.69	0.12	-0.12
Rock	0.19	-0.57	-0.23	-0.20
Bare Soil	-0.24	-0.37	0.21	0.45
Unarmed Woody Plants	-0.08	0.01	-0.58	-0.44
Armed Woody Plants	-0.05	-0.17	0.07	-0.46
Succulents	-0.35	-0.06	0.31	-0.33
Vines	-0.20	0.07	-0.53	0.46
Epiphytes	-0.43	0.02	-0.02	-0.06
Variance explained by components	1 0.27	2 0.18	3 0.13	4 0.10
Cumulative variance explained (%)	1 0.27	2 0.45	3 0.58	4 0.68

Table 1. Gap intercept (plants <0.5 m) variable loadings for each retained component and variance explained by each component. We considered loading of a coefficient > |0.3| significant; these are highlighted in bold.

Component loadings Variables	Principal component axes					
	1	2	3	4	5	6
Unarmed Woody Plants <0.5 m	-0.27	-0.04	0.33	-0.13	0.14	-0.15
Armed Woody Plants <0.5 m	-0.44	0.23	0.15	0.13	0.15	0.20
Succulent Plants <0.5 m	0.04	0.49	0.01	-0.15	0.12	0.28
Vines <0.5 m	-0.22	0.01	-0.56	-0.10	0.20	-0.03
Unarmed Woody Plants 0.5-1.0 m	-0.32	-0.17	0.27	-0.30	-0.11	-0.27
Armed Woody Plants 0.5-1.0 m	-0.34	0.11	-0.42	0.18	0.12	-0.03
Succulent Plants 0.5-1.0 m	0.04	0.56	0.08	-0.22	0.13	-0.05
Vines 0.5-1.0 m	-0.16	-0.03	-0.44	-0.37	0.03	-0.21
Unarmed Woody Plants 1.0-1.5 m	-0.39	0.04	0.02	0.00	-0.28	-0.15
Armed Woody Plants 1.0-1.5 m	-0.27	0.10	0.15	0.13	-0.42	0.08
Succulent Plants 1.0-1.5 m	0.07	0.33	-0.02	-0.37	-0.25	-0.07
Vines 1.0-1.5 m	-0.13	-0.28	0.11	-0.58	0.07	0.07
Unarmed Woody Plants >2 m	-0.38	-0.04	0.15	0.28	0.36	-0.07
Armed Woody Plants >2 m	-0.22	0.06	-0.12	0.01	-0.50	0.46
Succulent Plants >2 m	0.01	0.34	0.17	-0.01	0.21	-0.23
Vines >2 m	-0.07	-0.18	0.09	-0.22	0.33	0.65
Variance explained by components	1	2	3	4	5	6
	0.15	0.13	0.11	0.09	0.08	0.07
Cumulative variance explained (%)	1	2	3	4	5	6
	0.15	0.28	0.39	0.48	0.56	0.63

Table 2. Belt transect variable loadings for each retained component and variance explained by each component. We considered loading of a coefficient $> |0.3|$ significant; these are highlighted in bold.

Jason Kolbe
University of Rhode Island
October 1, 2015

In October 2014 on Guana, I completed data collection for habitat use of *Anolis* (anoles) lizards in natural and human-modified areas. These data helped complete a manuscript entitled “City slickers: poor performance does not deter *Anolis* lizards from using artificial substrates,” which I’m currently revising for resubmission to *Functional Ecology*. In the paper, we provided evidence to reject the habitat constraint hypothesis – the idea that organisms should use habitats in which they perform better and avoid those habitats which lead to poor performance – for anoles in human-modified areas. Despite poor locomotor performance (running slow, slipping, and falling) on smooth, vertical substrates, anoles commonly used these substrates when available. We also found smaller anoles with relatively longer limbs perform better on smooth, vertical surfaces, such as posts and walls, in human-modified areas. Thus, natural selection is predicted to favor lizards with these traits.

In another project, I followed up on a 2012 Guana project that was recently published in the *Journal of Herpetology* that tested clinging performance on substrates varying in diameter and roughness. In 2014, I attempted to isolate how anoles use their claws to interlock on rough surfaces versus their toepads to adhere to smooth surfaces. This is important for anoles living in human-modified areas where they most often use artificial surfaces (buildings, fences, posts, walls) where their claws are less effective, thus relying primarily on their toepads when moving on smooth, vertical surfaces.

Lead exposure in lizards of the British Virgin Islands

Adam E. Duerr

Division of Forestry and Natural Resources, West Virginia University, PO Box 6125, Morgantown, WV 26506-6125, adam.duerr@mail.wvu.edu, 757.903.5461

Long-term use and local disposal of lead-acid batteries has created a possible sources of this toxic heavy metal to fauna of the British Virgin Islands. Lead-acid batteries have been discarded apparently for decades on islands in the Virgin Island chain and accumulate at dumpsites. Over time, discarded batteries degrade, acid drains and leaches into soil, and lead plates become exposed to the elements. Historically, lead has been used as an additive to paint and gasoline. Although lead paint and leaded gasoline are no longer available, these sources may contribute to the lead levels in the BVI environment.

Effects of lead exposure or lead toxicosis include nervous and organ system disorders, inhibition of hemoglobin synthesis, digestive system stasis, decreased bone mineralization, reduced productivity, and altered behavior in birds (Buerger et al. 1986, Cade 2007, Carpenter et al. 2003, Kenntner et al. 2007, Pokras and Kneeland 2009). In lizards, lead exposure reduced survival, depending upon exposure level (Holem et al. 2006, Salice et al. 2009), but behavioral impairment was equivocal (Holem et al. 2006, Salice et al. 2009).

During preliminary work conducted in 2012, I found what appeared to be elevated levels of lead in blood of Anole lizards (*Anolis* spp.) on Guana Island. These findings raised concerns about potential lead exposure in other species, specifically predators of *Anolis* spp. and the endangered Stout Iguana (*Cyclura pinguis*).

Availability of portable and reliable instruments (Lead Care II, Magellan Diagnostics, North Billerica, MA, USA) allowed for on-site testing of small blood samples (50 µL). This instrument was designed to analyze human blood and has not been verified for herpetological use. Results from the Lead Care II unit tend to be biased low in birds (Boschetto and Pierce 2012). Results from reptiles have to be considered preliminary until supported by testing by an accredited lab. The unit has a detection range of 3.3 – 65 µg/dL. Levels > 5 µg/dL are a concern in humans.

There are no established benchmarks for lead exposure in herpetofauna. Camus et al., (1998) interpreted blood lead levels of 0.2 mg/L (20 µg/dL) as suspicion of lead poisoning and levels of 0.5 mg/L (50 µg/dL) as diagnostic of lead poisoning in American Alligators (*Alligator mississippiensis*). Although I am using these values in this preliminary analysis, it is important to note that there are no data to support levels of acute or chronic toxicity in iguanas. However, lead is not a biologically useful element and lead exposure at all levels is considered poisoning with potential chronic effects.

The objectives of this study are to determine lead exposure in select herpetofauna of the BVI. Specifically, I will test blood lead levels in two species of Anoles (*A. cristatellus* and *A. stratulus*), in Stout Iguanas and in racer snakes (*Alsophis pororicensis*) on Guana Island. I will also test blood lead levels in Anoles and Stout Iguanas from other BVI islands to determine the potential spatial extent of lead exposure in BVI herpetofauna.

METHODS

Lead exposure could differ among areas with potentially different lead contaminants; therefore, I sampled from three types of areas. The first was in the close vicinity (<10m) of exposed lead batteries. The second was in areas where paint was used or stored (often housing areas). The third was in areas without evidence of lead contamination (often native vegetation).

I collected anoles from each of the three types of areas on Guana Island. On other islands, I combined the housing and lead-battery areas. Using lead exposure levels from anoles collected in 2012, I estimated that sample sizes of 11 *A. cristatellus* and 16 *A. stratulus* were needed to determine if the proportion of lizards exposed to lead differed between two sites. I also attempted to collect an equal number of male and female of each species of anole. I applied the sample size of 16 estimated for *A. stratulus* to determine the number of snakes to sample from each type of area.

I used noose poles to capture anoles (*Anolis stratulus* and *A. cristatellus*) and captured snakes by hand. After capture, I placed each anole into a single plastic bag filled with ambient air and transported them to a lab for analysis. I used sharpened scissors to decapitate each anole or snake and used a capillary tube or micropipette to collect 50 µl blood samples.

Stout iguanas were captured as part of other studies in the BVI (K. Mougey and G. Perry, Texas Tech University). Blood samples were collected from caudal veins of iguanas using hypodermic needles, with blood samples transferred immediately to EDTA treated vacutainers for storage.

I also collected blood samples from seven Pearly-eyed Thrashers (*Margarops fuscatus*) that were euthanized to remove problem birds from the orchard or from mist nets used to band birds (C. Boal) where thrashers were observed killing other birds. I collected blood samples directly from hearts of euthanized thrashers.

I used a LeadCare II (Magellan Diagnostics, North Billerica, Massachusetts, USA) portable lead analyzer to determine blood lead levels for all samples collected in 2012-2013 and for all snakes and iguanas. These samples were analyzed on Guana Island. In 2014, I stored 50 µl blood samples from anoles on treated filter paper and tested the remainder of samples using the LeadCare II unit.

I used an ANOVA framework (general linear models) to determine how lead levels varied among species, area types, and islands. I compared lead levels among area types for anoles, snakes, and iguanas, and among islands for anoles.

PRELIMINARY RESULTS and DISCUSSION

I collected blood samples from a total of 172 *Anolis cristatellus* and 76 *A. stratulus* (Table 1). Of these, I used the LeadCare II unit to analyze 109 and 38 blood samples for *A. cristatellus* and *A. stratulus*, respectively. I used this unit to analyze all of 32 blood samples for *Alsophis pororicensis* and X of Y samples for *C. pinguis*.

Blood lead levels of anoles differed among the three area types (Table 1). For *A. cristatellus*, blood lead levels differed by area type ($F = 11.71$, $P < 0.001$) and island ($F = 3.60$, $P = 0.005$). Lead levels were highest for contaminated sites, lowest for uncontaminated sites, and intermediate for housing sites.

For *A. stratulus*, blood lead levels showed marginal differences by area type on Guana Island ($F = 2.87$, $P = 0.073$) but not between uncontaminated sites on Guana and Virgin Gorda Islands ($F = 0.71$, $P = 0.406$). Lead levels were highest for housing areas, intermediate for contaminated sites, and lowest for uncontaminated sites.

As expected, *A. cristatellus* had higher lead levels in contaminated sites than uncontaminated sites. What was not expected was that lead levels in housing areas were nearly as high as for contaminated sites for *A. cristatellus* and that housing areas had slightly higher lead levels than contaminated sites for *A. stratulus*.

Relatively high blood lead levels in Anoles from contaminated and housing areas on Guana Island suggests that they may be exposed to more lead sources than lead batteries. However, lead batteries may still pose a relatively greater risk to these lizards. On Guana Island, a greater number of anoles with diagnostic levels of lead poisoning were found in contaminated areas than in housing areas (Table 2).

In 2012, I collected blood samples from 7 Pearly-eyed Thrashers. They are omnivores with much greater vagility than anoles. In spite of a diet that includes other animals and fruit, one of seven thrashers showed measureable blood lead levels ($5.9 \mu\text{g/dL}$). These results provided evidence that lead was moving through the greater ecosystem on Guana Island and was not restricted to contaminated sites. Other animals, especially predators that consume anoles, may also be exposed to lead. This observation lead to the collection of samples to test lead levels in blood of snakes and iguanas.

I collected 32 blood samples from *Alsophis portoricensis*, 31 from Guana Island and one from Little Thatch Island (Table 1). There was no difference in blood lead levels between contaminated and uncontaminated sites (both islands combined; $F = 0.03$, $P = 0.869$). As snakes are larger and are predators of anoles, their home ranges are expected to be considerably larger than those of anoles. It is likely that snake exposure to small areas contaminated by lead batteries or other lead sources is much lower than for anoles. Some snakes did show measureable lead levels, but all were below levels considered suspicious of lead poisoning.

I analyzed lead levels in blood of endangered *Cyclura pinguis* from 67 samples of 146 samples collected. Of the 67 samples analyzed, I classified 42 by the area type where they were collected. I found no difference in the lead levels among the three area types on Guana Island ($F = 0.37$, $P = 0.691$). No *C.*

pinguis had blood lead levels diagnostic of lead poisoning, one had lead levels suspicious of lead poisoning. Nine had measurable levels of lead in their blood, with levels between 3.3 and 18 µg/dL. Levels in the remaining 57 samples were below detection limits for the LeadCare II unit.

FUTURE ANALYSES

The Lead Care II unit has a limited range for quantifying blood lead levels. To verify results obtained by the Lead Care II unit, I archived a majority of blood samples from anoles, snakes and iguanas from the BVI. These archived samples include blood spots of ~50 microliters stored on treated filter paper and other whole blood samples. These samples were exported to the US (*C. pinguis* exported via CITES permits obtained by Dr. Perry and a CITES import permits to the IUCN Iguana Specialist Group). Lab-based lead analyses have a lower detection limit of 0.5 µg/dL and such results will better show where exposure to lead occurs. I am collaborating with the Diagnostic Center for Population and Animal Health, Michigan State University, to conduct lab-grade analyses. These will allow us to 1) detect much lower levels of lead exposure, 2) develop a correction factor for the Lead Care II unit for lizard blood, and 3) identify stable isotope signatures to distinguish sources of lead, such as leaded gasoline, lead paint, and car batteries (Komárek et al. 2008). This collaboration will also allow for analysis of a greater number of samples (from iguanas, anoles, and snakes) than current funding permits.

Many hatchling iguanas sampled on Guana Island were too young to have fed much, so lead exposure must primarily have been from a maternal source, transferred via the egg. If funding allows, we would also like to conduct genetic analyses to determine relatedness among iguanas. Genetic analyses may be conducted by collaborators at West Virginia University, Purdue University, or University of Rhode Island. In addition to genetics, we can also assess maternal transfer of lead by analyzing lead content in iguana eggshells because lead replaces calcium at the metabolic level.

LITERATURE CITED

- Boschetto and Pierce. 2012. Accuracy of LeadCare® Kit in Veterinary Wildlife Applications. B.S. thesis. Worcester Polytechnic Institute.
- Buerger, T. T., R. E. Mirarchi, and M. E. Lisano. 1986. Effects of Lead Shot Ingestion on Captive Mourning Dove Survivability and Reproduction. *The Journal of Wildlife Management* 50:1-8.
- Cade T. 2007 Exposure of California condors to lead from spent ammunition. *Journal of Wildlife Management* 71(7):2125-2133
- Camus, A. C., M. M. Mitchell, J. E. Williams, and P. L. H. Jowett. 1998. Elevated Lead Levels in Farmed American Alligators *Alligator mississippiensis* Consuming Nutria *Myocastor coypus* Meat Contaminated by Lead Bullets. *Journal of the World Aquaculture Society* 29:370-376.

- Carpenter, J., O. Pattee, S. Fritts, B. Rattner, S. Wiemeyer, J. Royle, and M. Smith. 2003. Experimental lead poisoning in turkey vultures (*Cathartes aura*). *Journal of Wildlife Diseases* 39:96-104.
- Friend, M., and J. C. Franson, editors. 1999. Chapter 42 Lead. pp 317-337 In *Field Manual of Wildlife Diseases*. Volume Information and Technology Report 1999-001. US Geological Survey, Biological Resources Division, Madison, WI.
- Holem, R. R., W. A. Hopkins, and L. G. Talent. 2006. Effect of Acute Exposure to Malathion and Lead on Sprint Performance of the Western Fence Lizard (*Sceloporus occidentalis*). *Archives of Environmental Contamination and Toxicology* 51:111-116.
- Kenntner N, Crettenand Y, Fünfstück H, Janovsky M, Tataruch F. (2007) Lead poisoning and heavy metal exposure of golden eagles (*Aquila chrysaetos*) from the European Alps. *Journal of Ornithology* 128:173-177
- Pokras, M. A., and M. R. Kneeland. 2009. Understanding lead uptake and effects across species lines: a conservation medicine based approach. in R. T. Watson, M. Fuller, and M. Pokras, editors. *Ingestion of Lead from Spent Ammunition: Implication for Wildlife and Humans- Abstract*. The Peregrine Fund, Boise, Idaho, USA.
- Salice, C. J., J. G. Suski, M. A. Bazar, and L. G. Talent. 2009. Effects of inorganic lead on Western fence lizards (*Sceloporus occidentalis*). *Environmental Pollution* 157:3457-3464.

Table 1. Blood lead levels measured by LeadCare II unit for Anole lizards from the British Virgin Islands.

Species	Island	Area type ¹	N	Blood Pb level	
				Mean	SE
Anolis cristatellus	Frenchman Cay		9	.	.
	Guana Island	Contaminated	16	25.4	6.2
		Housing	12	19.8	12.7
		Uncontaminated	13	0.9	0.6
	Jost Van Dyke Island		7	.	.
	Little Thatch	Uncontaminated	7	0.7	0.7
			11	.	.
	Marina Cay	Contaminated	11	2.5	0.9
	Norman Island		11	.	.
	Peter Island		4	.	.
	Salt Island		10	.	.
	Scrub Island		6	.	.
	Tortolla Island	Contaminated	9	14.6	6.3
		Uncontaminated	9	1.3	0.8
	Virgin Gorda Island	Contaminated	11	7.0	2.0
		Housing	3	5.4	1.8
		Uncontaminated	4	0.9	0.9
A. stratulus	Frenchman Cay		2	.	.
	Guana Island	Contaminated	12	87.5	25.6
		Housing	4	153.4	153.4
		Uncontaminated	19	2.0	2.0
			16	.	.
	Jost Van Dyke Island		4	.	.
	Little Thatch		10	.	.
	Norman Island		3	.	.
	Peter Island		3	.	.
	Virgin Gorda Island	Uncontaminated	2	0	0
		Housing	1	0	.
Alsophis portoricensis	Guana Island	Contaminated	14	0.7	1.9
		Uncontaminated	18	0.9	2.1
	Little Thatch	Uncontaminated	1	0	.
Cyclura pinguis	Guana Island	Contaminated	20	2.5	9.2
		Uncontaminated	11	0.4	1.4
		Housing	11	1.58	3.0
		Unclassified	25	1.7	4.5

¹Lines without a treatment type indicate samples collected but not yet analyzed.

Table 2. Number of animals found with blood lead levels that are diagnostic or suspicious for lead poisoning.

Island	Species	Area type	No. diagnostic levels	No. suspicious levels
Guana	A. stratulus	Contaminated	6	3
		Housing	1	
		Uncontaminated		1
	A. cristatellus	Contaminated	2	6
		Housing	2	
Tortolla	A. cristatellus	Contaminated	1	

ORNITHOLOGICAL RESEARCH AND MONITORING ON GUANA ISLAND, BRITISH VIRGIN ISLANDS: PROJECT REPORT 2014

Dr. Clint W. Boal, *USGS Texas Cooperative Fish and Wildlife Research Unit, Texas Tech University, Lubbock, TX 79409-2120*

INTRODUCTION

The Caribbean is an important region for neotropical migrant landbirds during their annual migration from North America to South America. While considerable research has been conducted on migrant ecology in the western Caribbean, comparatively little has been conducted in the Virgin Islands or east Caribbean (Wiley 2000). Thus, ornithological work on Guana Island makes substantive contributions toward a better understanding of the Virgin Islands as stop-over habitat for migrant birds (McNair et al. 2002, Boal et al. 2006, Boal and Estabrook 2007) and the basic ecology of Caribbean birds (Chipley 1991, Boal et al. 2006).

Components of avian research on Guana Island are 1) mist-netting and banding neotropical songbirds that migrate through the Caribbean region during the autumn migration, and 2) specific studies focusing on species resident to the island. Current projects on resident species are 1) a population demography study of bananaquits, 2) a study of distribution and habitat associations of mangrove cuckoos, and 3) monitoring distribution and relative abundance of resident species across the island. This year was the terminal year for the traditional science month on Guana Island. As such, our research efforts in 2014 focused on maximizing collection of data related to our species-specific studies of Bridled Quail Doves and Mangrove Cuckoos. Mist-netting resident and migrant passerines was not a priority due to 1) 10 years of study on Blackpoll Warblers had been written up and was in publication and 2) data collection on Bananaquits since 2003 had been published or was undergoing data analysis for publication.

RESULTS AND DISCUSSION

Mist-Netting and Migrant Ecology

Avian activity was sedate this year. Although we conducted 100 hours of mist-netting, we only captured 3 blackpoll warblers, 1 red-eyed vireo, 2 bananaquits, 2 black-faced grassquits and 1 common yellowthroat. Based on Lazell (2005) and our data, the common yellowthroat is a new species record for Guana Island. We also captured 3 bananaquits and 3 pearly-eyed thrashers that had been banded in previous years.

Bridled Quail-Dove Population

I conducted distance sampling surveys for bridled quail-doves on every trail on Guana Island. I pooled age classes of all detected birds due to the low number of juveniles, and entered all data into Program MARK. Due to limited area and sample sizes, I did not separate surveys by day or trail. I ran 4 models that were all quite similar in results, with the largest ΔAIC being 0.89. Therefore, I model averaged the four models and arrived at an estimate of 479 bridled quail-doves present on the island. However, the confidence interval was very large, suggesting the estimate may range from 153 to 806. There is also an unassessed, as yet, spatial aspect to this associated with topography and vegetation. For example, few detections were from the Long Man Point and Grand Ghut trails. I suspect vegetation (or lack of understory vegetation) may be an issue. Only further study and perhaps radio telemetry will help elucidate what the pattern of dispersal is across the island. However, I am working on some different ways to assess this.

Mangrove Cuckoo Ecology

The lead field investigator for this project is Tracy S. Estabrook (M.S.). The mangrove cuckoo is one of the least-studied North American birds and baseline population estimates and habitat requirements have been identified as among the most important research needs for the species. Surveys across the island were conducted for the mangrove cuckoo and we are now in the process of analyzing several years of repeat surveys to assess distribution, consistency in occupancy, and estimates of relative density across the island.

Other Observations of Note

Although numbers of birds and general activity appeared to be much reduced in 2014, we did capture a new island record (common yellowthroat) as discussed above. We also had observations of at least 2 yellow-throated warblers (also new species records for the island) that were occupying the woodland by the shop.

PROJECT PRODUCTIVITY

Avian research is of little value if not made available to both the scientific community and the general public. Since engaging in avian research on Guana Island in 2003, I have published 4 papers in peer-reviewed journals. I am currently writing one paper addressing timing and condition of blackpoll warblers arriving on Guana Island. I am also modeling the age- and sex-specific survival of island resident species based on mark-recapture methods. My colleagues and I have also made 5 Guana Island-related presentations at professional meetings, the most recent of which was at the annual meeting of Cooper Ornithological Society in Tucson, Arizona in April 2009.

Recent Publications

Boal, C.W. 2014. Age-ratios and condition of en route migrant blackpoll warblers in the Caribbean. *Wilson Journal of Ornithology* 126:568-574.

Boal, C.W., J.M. Wunderle, Jr., and W.J. Arendt. 2013. Autumn monitoring of resident avifauna on Guana Island, B.V.I. *Ornitologia Neotropical* 24:335-343.

ACKNOWLEDGEMENTS

First and foremost, I thank Dr. James Lazell and Dr. Gad Perry for continuing to facilitate avian research activities on Guana Island. I thank Gloria and Henry Jarecki for providing the opportunity for me and other researchers to conduct our studies on Guana Island. I thank the several people who assisted with ornithological studies on Guana Island during the 2008 season. Specifically, these were Tracy Estabrook and Susan Valentine. Funding for this research was provided by The Conservation Agency through a grant from the Falconwood Foundation and by the U.S. Geological Survey, Texas Cooperative Fish and Wildlife Research Unit.