A TRANSLOCATED POPULATION OF THE ST. CROIX GROUND LIZARD:

ANALYZING ITS DETECTION PROBABILITY AND INVESTIGATING ITS

IMPACTS ON THE LOCAL PREY BASE

A Thesis

by

MICHAEL LOUIS TREGLIA

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

August 2010

Major Subject: Wildlife and Fisheries Sciences

A Translocated Population of the St. Croix Ground Lizard: Analyzing Its Detection

Probability and Investigating Its Impacts on the Local Prey Base

A TRANSLOCATED POPULATION OF THE ST. CROIX GROUND LIZARD:

ANALYZING ITS DETECTION PROBABILITY AND INVESTIGATING ITS

IMPACTS ON THE LOCAL PREY BASE

A Thesis

by

MICHAEL LOUIS TREGLIA

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

Approved by:

Chair of Committee,	Lee A. Fitzgerald
Committee Members,	Gil Rosenthal
	Kirk Winemiller
Head of Department,	Tom Lacher

August 2010

Major Subject: Wildlife and Fisheries Sciences

ABSTRACT

A Translocated Population of the St. Croix Ground Lizard: Analyzing Its Detection

Probability and Investigating Its Impacts on the Local Prey Base.

(August 2010)

Michael Louis Treglia, B.S., Cornell University

Chair of Advisory Committee: Dr. Lee A. Fitzgerald

The St. Croix ground lizard, *Ameiva polops*, is a United States endangered species endemic to St. Croix, U.S. Virgin Islands. It was extirpated from St. Croix Proper by invasive mongooses, and remaining populations are on small, nearby cays. In the summer of 2008, as part of the recovery plan for this species, I worked in a multi-agency effort to translocate a population of *A. polops* to Buck Island Reef National Monument, U.S. Virgin Islands to focus on two main objectives: 1) examine the detection probability of *A. polops* and infer the consequences of it on population estimates; and 2) examine whether *A. polops* may deplete its prey base or alter the arthropod assemblage at the translocation site. We used a soft-release strategy for the translocation, in which 57 lizards were initially contained in a series of eight 10 m x 10 m enclosures in the habitat on Buck Island for monitoring. As part of the initial monitoring I conducted visual surveys through all enclosures, with the known number of lizards, to calculate the detection probability and to demonstrate how many individuals would be estimated using visual encounter surveys of this known population. Adjacent

to enclosures housing *A. polops* were control enclosures, without *A. polops*, which I used to test whether the translocated lizards would impact their prey base over 6 weeks. I found that the detection probability of *A. polops* is very low (<0.25), which causes population sizes to be severely underestimated, even using some mark-resight techniques. My study of *A. polops* on the prey community indicated that the lizards generally had no effect on abundance or diversity of arthropods in general, though they may cause small changes for particular taxa. My results help corroborate other evidence that accuracy of population enumeration techniques needs to be improved in order to adequately understand the status of wildlife populations. Additionally, prey resources do not seem to be limiting *A. polops* in the short-term, and I expect the population will grow, expanding through Buck Island. Future monitoring will be carried out by the National Park Service using robust mark-resight techniques.

DEDICATION

This thesis is dedicated to the world's biodiversity, most of which I will never know, but all of which keeps me curious.

ACKNOWLEDGEMENTS

hearts. My parents are the ones to whom I owe my interest in nature, getting me out hiking from a young age and allowing me to spend many hours in the woods looking for snakes, salamanders, and other critters.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER I

INTRODUCTION

Although the efficacy of herpetofaunal translocations has been controversial (Dodd and Seigel 1991), the portion of documented successes has greatly increased in the past 20 years (Germano and Bishop 2009). As Germano and Bishop (2009) suggest, the life history traits of many reptiles actually tend to make them good candidates for translocations because they generally have high fecundity, especially when compared with similarly sized endotherms (Shine 2005). Thus, if survivorship is high reptiles may proliferate in an area relatively quickly with their already high reproductive rates (Vitt and Price 1982, Warner et al. 2007).

However, a major shortcoming of many translocations is the lack of post-translocation monitoring to determine the success of projects. For example, of 47 reptile translocations carried out between 1991 and 2006, approximately 40% of them had uncertain outcomes (Germano and Bishop 2009). This may not even-reflect the total portion of translocations with uncertain outcomes because of publication biases (i.e., tendency of authors not to publish about translocations with failed or unknown results; Dickersin and Min 1993, Scargle 2000). While it is sometimes difficult to predict the long-term success of translocations with only short term monitoring data (Dodd and Seigel 1991), target population sizes can be set based on the available habitat and resources for particular time intervals (Armstrong and Seddon 2008), as is often done in recovery plans for at-

_____

This thesis follows the style of *Ecological Applications.*

risk species (U.S. Fish and Wildlife Service 1984, 2009, Gates et al. 2010).

Unfortunately, monitoring population sizes of many reptile species has proven to be difficult because they are not very active, especially when compared with endotherms because they have lower energy requirements (Bennett and Ruben 1979, Pough 1980). Thus, not all individuals present are necessarily available for detection at any particular time, and population estimation techniques that employ visual surveys assuming all individuals present are available for detection, at least along a transect line (e.g., distance sampling; Buckland et al. 2001, 2004) may not be accurate (Rodda and Campbell 2002, Smolensky 2008, Smolensky and Fitzgerald in press). Although these techniques have been validated for some lizard species using mark-resight methods (Dickinson and Fa 2000, Kacoliris et al. 2009), even mark-resight estimates can be flawed if there is much heterogeneity in the population or the detection probability is low (Pledger 2000). Preliminary studies can be carried out to estimate the proportion of individuals that will not be available for detection, as demonstrated by Grant and Doherty (2007) for *Phrynosoma mcallii*, the flat-tailed horned lizard, though this is generally resource intense.

When reptile translocations do fail, it is most frequently attributed to homing abilities and movement patterns associated with being transported to a new location (Germano and Bishop 2009). However, habitat quantity and quality are also important to ensure that the translocated individuals can carry out normal life processes (Dodd and Seigel 1991). One particular component of habitat quality that may be important is prey availability, which can limit reproductive success of translocated populations (Vitt and

Price 1982, Olsson and Shine 1997, Warner et al. 2007). To support this, multiple

studies have found that *Anolis* lizards can deplete particular components of their prey

base (Schoener and Spiller 1996, Schoener and Spiller 1999). Additionally, Anderson

(1994) found that a population *Cnemidophorus tigris*, the tiger whiptail lizard, spatially

followed its prey resources through time. However, the effects of lizards on their prey

base may be partially controlled by foraging mode (Huey and Pianka 1981), and it is

unclear whether specific types of lizards may alter the local invertebrate prey

community. If a translocated population of lizards were to rapidly deplete enough of its

prey base quickly, the translocation could fail due to too few food resources for the

amount of lizards.

In this study, I used the translocation of the endangered St. Croix ground lizard,

Ameiva polops, to examine aspects of lizard ecology that may influence success of

translocations, and our measurement of translocation success. I had two specific

objectives:

1) Determine how inactivity of a translocated population of *A. polops*
   impacts its detection probability, and make inference to the effect of this
   factor on population estimates of this, and other species.

2) Investigate whether *A. polops* may impact its invertebrate prey base, both
   in abundance and diversity, using a controlled enclosure experiment, and
   infer whether a translocated population risks depleting its own food
   resources.

Ameiva polops, is a small teiid (max snout-vent length=88 mm; Treglia and Fitzgerald, unpublished data), endemic to St. Croix, U.S. Virgin Islands and the surrounding cays (Henderson and Powell 2009). It was extirpated from St. Croix Proper by introduced mongooses, with the last confirmed sighting in Fredriksted in 1968 (Philibosian and Ruibal 1971, Philibosian and Yntema 1976). With the only populations left on two small cays (<10 hectares each) it was given protection under the Endangered Species Act in 1977. The Recovery Plan for *A. polops* (U.S. Fish and Wildlife Service 1984) called for the translocation of a population to the nearby island, Buck Island Reef National Monument following the eradication of mongooses there. Although population was translocated to another small manmade island in 1990 (Knowles 1996), the translocation to Buck Island was still deemed necessary. By 1995 the National park Service successfully eradicated mongooses from Buck Island Reef National Monument (National Park Service 2007), and in 2008 I worked in a collaborative effort with the U.S. National Park Service, U.S. Fish and Wildlife Service, Virgin Islands Department of Planning and Natural Resources- Division of Fish and Wildlife to establish a new population of *A. polops* there, and to carry out the objectives listed above.

CHAPTER II

DETECTABILITY OF AN ENDANGERED LIZARD: AN EMPIRICAL STUDY,

A SIMULATION, AND IMPLICATIONS FOR POPULATION ESTIMATION

INTRODUCTION

Population size is the status quo unit for monitoring wildlife species, as evidenced by most conservation goals and management plans (e.g., U.S. Fish and Wildlife Service 2009). However, accurate estimation of abundance is often hindered by low detectability of focal species (Cook and Jacobson 1979, MacKenzie and Kendall 2002, Mazerolle et al. 2007), making effective monitoring programs difficult to implement. For example, stand-alone visual survey techniques are often ineffective because of researchers' inability to see focal animals (Marsh and Sinclair 1989). For some taxa this is accounted for with complementary techniques such as distance (Buckland et al. 2001, 2004) and aural surveys (e.g., Sepúveda et al. 2006). However, population estimation for other organisms tends to be more difficult because individuals that are present may be unavailable for detection due to overall low levels of activity (e.g., reptiles; Rodda and Campbell 2002, Mazerolle et al. 2007, Smolensky 2008, Smolensky and Fitzgerald in press). Although techniques such as scat and track surveys have been developed to account for this bias (availability bias), feasibility varies across species and location. For example, tracks of small organisms are diminutive and thus difficult to find, particularly in dynamic substrates (e.g., small lizards in sand dunes).

Reptiles are particularly prone to availability bias because of their variable and generally low energy requirements associated with ectothermy (Bennett and Ruben

1979). However, only a few studies have quantified the error in population estimates associated with availability bias. Rodda and Campbell (2002) found that distance sampling (Buckland et al. 2001, 2004) underestimated numbers of geckos (multiple species) and brown tree snakes (*Boiga irregularis*) by factors of 34 and 7, respectively, when compared with results of a complete census technique, total removal plots (Rodda et al. 2001). Using the same methodology for dunes sagebrush lizards (*Sceloporus arenicolus*), Smolensky and Fitzgerald (in press) found that distance sampling underestimated population sizes by a factor 6.5. The reason for this large discrepancy between distance sampling and total removal plots is that distance sampling, while accounting for decreasing visibility of animals with increasing distance from a transect line, has no algorithm for counting individuals that are simply not available for detection. Total removal plots, however, count all individuals within an enclosed area of habitat, regardless of whether they are active. Similarly, double observer methods, originally developed for counting large mammals in aerial surveys (Cook and Jacobson 1979), take into account observer biases and can be employed in herpetological studies to increase the probability of seeing the individuals present (Mazerolle et al. 2007). However, these techniques also rely on individuals being available for detection and have no way to estimate the number of individuals inactive or hidden from view.

In contrast to the studies by Rodda and Campbell (2002) and Smolensky and Fitzgerald (in press), both Dickinson and Fa (2000) and Kacoliris et al. (2009) validated distance sampling for *Cnemidophorus vanzoi* (St. Lucia whiptail lizards) and *Liolaemus multimaculatus* (sand dune lizards), respectively, but using mark-resight techniques.

However, if there is significant heterogeneity in detection probability, or if it is very low overall, mark-resight methods may also underestimate population sizes (Pledger 2000). Grant and Doherty (2007) successfully used distance sampling to estimate the abundance of a short-horned horned lizard population (*Phrynosoma mcalli*) by incorporating an estimate of the portion a population that would be available for detection in their surveys that was derived from radio-tracking data. This method was effective for their study, though it is generally resource intense and impractical. Additionally, this correction procedure would need to be carried out for every survey period, as detectability should be assumed to vary with space and time unless otherwise indicated (MacKenzie and Kendall 2002).

In this study I examined the population level activity of the St. Croix ground lizard, *Ameiva polops*, and made inference to impacts of availability bias on population estimates conducted using strip-transect searches and a Lincoln-Petersen mark-resight technique. *Ameiva polops* is a small teiid lizard (max snout-vent length: 88 mm; Treglia and Fitzgerald, unpublished data) endemic to St. Croix, U.S. Virgin Islands, that has been severely threatened by invasive mongooses. Multiple population estimates have been conducted for the species during the past two decades and although variable, have indicated alarmingly low numbers in each of the three extant populations (McNair 2003,

McNair and Lombard 2004, McNair and Mackay 2005; Table 2.1). In 2008, as part of a multi-agency effort to expand its range, and in accordance with the recovery plan for *A. polops* (U.S. Fish and Wildlife Service 1984), I translocated a population to Buck Island Reef National Monument. I used a soft-release strategy (Scott and Carpenter 1987), temporarily constraining the lizards to a small portion of the available habitat and later releasing them to the surrounding area. This not only facilitated habituation, but also enabled intense short-term monitoring of the translocated population. I took advantage of this post-translocation monitoring of the confined individuals to the assess availability bias due to individuals being inactive during visual encounter surveys during the general activity period of the species. I also simulated mark-resight surveys with varying levels of detectability to estimate at what detection probabilities Lincoln-Petersen estimates may be ineffective. The results of this research have obvious implications for monitoring populations of *A. polops*, and for evaluating the status of difficult to detect species in a more general context.

Table 2.1 Summary of population estimates of *Ameiva polops* from 1967. In the multi-day habitat and strip-transect searches, observers counted lizards multiple times per day for multiple days, and used the maximum count of lizards to extrapolate for the entire island.

Location	Year	Method	Number	Source
Protestant Cay	1967	Unknown	200	Philibosian and Ruibal (1971)
Green Cay	1967	Unknown	300	Philibosian and Ruibal (1971)
Green Cay	1980-1981	Mark-resight (analysis unknown)	300-4300	U.S. Fish and Wildlife Service (1984)
Green Cay	1987	Mark-resight	431*	Meier et al. (1993)
Protestant Cay	2002	Multi-day habitat patch searches	30	McNair (2003)
Green Cay	2002	Multi-day strip transect searches	183	McNair and Lombard (2004)
Ruth Cay	2003	Multi-day strip transect searches	60	McNair and Mackay (2005)

*Note: 431 was the average of 3 estimates, 420,462, and 421 within an eight day period.

METHODS

Translocation

From 29 April–10 May, 2008 our research team translocated 57 adult *A. polops* from Green Cay National Wildlife Refuge to the northwestern beach forest habitat at Buck Island Reef National Monument, St. Croix, U.S. Virgin Islands. All individuals were captured by noosing, and the following data were recorded: sex; snout-vent length; total tail length; regenerated tail length; and mass. The lizards were marked with toe-clipping for permanent, individually-identifying marks (Dodd 1993, Borges-Landáez and Shine 2003). To make them more easily identifiable from a distance, unique combinations of colored glass beads were sutured to the dorsum of lizards' tails using Ethicon suture thread, similar to a technique described by Fisher and Muth (1989) that used surgical steel wire.

All individuals were transported to Buck Island by 1700hrs Atlantic Standard Time each day, and placed into one of eight 10 x 10 m open-top enclosures (Figure 2.1). The sex ratio was 4 females to 3 males for each enclosure, except for one with 4 males and 4 females. I contained all individuals in the enclosures until 10 July 2008, when I removed enclosure walls to allow them into the surrounding habitat.

The translocation habitat was similar to that of the source population. This site had been recommended as a potential translocation site in multiple papers and reports (Meier et al. 1990, National Park Service 2007) and was used for a previous translocation that failed due to the presence of mongooses at the time (Philibosian and Ruibal 1971, Philibosian and Yntema 1976). Generally the vegetation was composed of mature trees creating a canopy that allows dappled sunlight to reach the ground and creates substantial leaf litter and other decaying organic matter to support an invertebrate prey base. Additionally there was a variety of herbaceous and shrubby vegetation, as well as dead woody debris for *A. polops* to use as refugia.

The research team constructed four enclosures made of galvanized tin roofing material and four made of Duraflash vinyl flashing. The bottom of all walls were buried a minimum of 15 cm to prevent *A. polops* from burrowing under, leaving at least 46 cm above ground (walls of the tin roofing-based enclosures extended to 71 cm tall). My observations and previous accounts of *A. polops* (Meier et al. 1993) indicated that they could not climb very well, and watching them in the enclosures gave me confidence that they could not scale the walls. As an extra precaution, I also installed longitudinally cut half-sections of PVC pipe as a lip on the shorter, Duraflash enclosures.
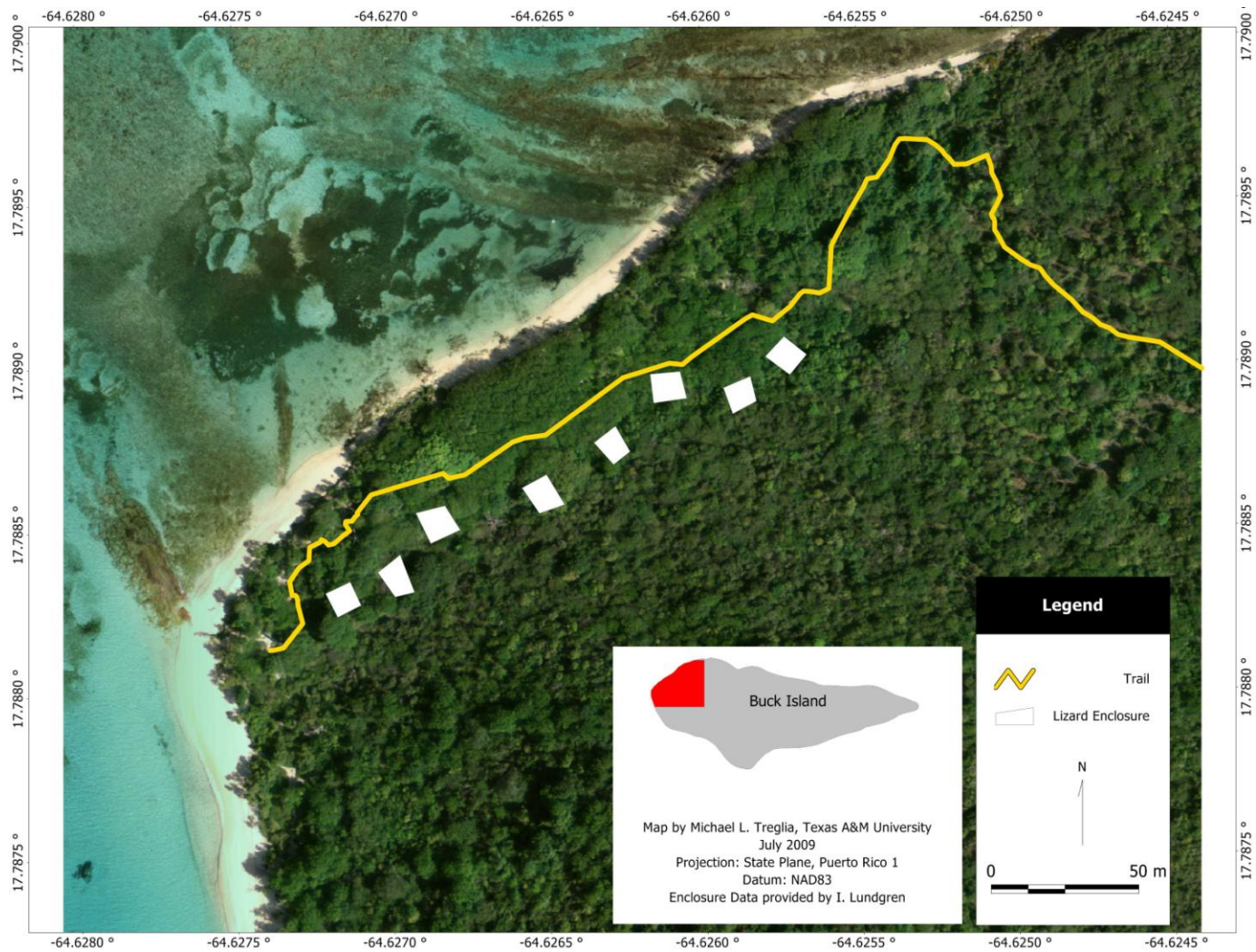
Fig. 2.1. Map of lizard enclosures on Buck Island, US Virgin Islands (Datum: WGS 84; Projection: State Plane - Puerto Rico and State Plane 1; created by Michael Treglia, 2009).

Visual Surveys of Translocated Lizards

After all eight enclosures were stocked with lizards, from 12 May to 2 July 2008 (for 7 weeks), I conducted visual surveys to count and identify the translocated individuals. Survey methodology was designed to be analogous to a timed strip-transect search through the enclosed habitat, and allowed me to calculate how many lizards would be estimated using these visual encounter surveys if they were being used for population enumeration. I conducted all the surveys, and alternated the starting enclosure and the direction walked around enclosures on every complete survey. I spent 10 minutes at each enclosure and used eight-power binoculars to aid in sighting and identifying individuals. I conducted all surveys between 1030 and 1430 hrs AST, corresponding to the majority of the lizards' activity period, and within the time frame of activity found by Meier et al. (1993). I did not conduct any surveys during rain.

Recapture of Translocated Individuals

At the end of the study period, from 3 to 9 July, I installed pitfall traps in the enclosures to recapture individuals that had never been seen in visual surveys. Pitfall traps were 7.6 liter plastic flowerpots with 28 cm sides and 23 cm in diameter, and were placed equidistant from each other. I used 28 cm x 28 cm, corrugated cardboard coverboards to provide shade for the pitfalls and create a refugium. I opened the traps from 0900 until 1600hrs AST each day for six days, for 245 total trap-hours. After realizing initially low trap success, from 5 to 10 July I also noosed as many individuals as possible. I computed size-adjusted mass as a body condition index (Dickinson and Fa 2000) for comparison with original capture data. Because of low sample size (11

individuals), I used a paired two-way Wilcoxon signed-ranks test using R (R

Development Core Team 2005) to determine whether changes in body condition were

significant ($\alpha=0.05$).

Analyses of Population Level Surface Activity

I analyzed the resight probability of *A. polops* as a proxy for population level

surface activity, with the assumption that during visual surveys I detected all individuals

that were active. This is reasonable given the small enclosures (100 m^2), the amount of

time spent at each enclosure, and my use of auditory cues (e.g., rustling of leaf litter) and

binoculars to find and identify individuals.

I used closed capture models (Otis et al. 1978) in Program MARK (White and

Burnham 1999) to estimate the average resight probability. I set the initial detection

probability to 1, as all individuals were "encountered" for the first time upon

translocation. During the monitoring period, some individuals lost their visual marks,

which hindered my ability to identify them. To incorporate sightings of unidentified

individuals into the resight probability, I randomly assigned those to individuals initially

placed in the respective enclosures, constrained to lizards that I previously saw to avoid

incorporating lizards that had possibly died or escaped.

During visual encounter surveys, I identified only 20 individuals in the

enclosures, which was surprising considering 57 individuals were translocated.

Throughout the study, only one lizard was known to escape. The escapee was seen

outside the enclosure and was thus excluded from analyses. I calculated two main sets of

models for resight probability: one for all lizards in the enclosures, and one for only

individuals that I saw during surveys or otherwise confirmed to be present in the

enclosures. Within each of these models, I also estimated the detection probability for all

individuals combined, and separating out males and females to determine whether they

were different. The goal of these models was to calculate the average probability of

seeing the translocated *A. polops* during typical activity times and monitoring periods

(Meier et al. 1993, Mackay 2007).

Simulations of Population Estimates

To simulate the effects of varying levels of detectability on population estimates

based on mark-resight data, I constructed a simulation model. For each detectability

value from 0 to 1 in increments of 0.025, I simulated 1000 two-survey mark-resight

estimates of hypothetical populations with 100 individuals. Individuals had the same

probability of being "observed" in the simulation on both the first and second surveys. I

then used a Chapman modified Lincoln-Petersen calculation (Pollock et al. 1990), as in

the following equation:

$$N_{est} = \left[\frac{(n_1 + 1)(n_2 + 1)}{(m_2 + 1)} \right] - 1.$$

In this equation, the estimated population size (N_{est}) is based on the number of

individuals seen and marked on the first survey (n_1), the total number of individuals seen

on the second survey (n_2), and the number of individuals seen on the second survey that

had been marked on the first (m_2). This estimator and variations of it have been

incorporated into myriad wildlife studies (e.g., Hein and Andelt 1995, Swann et al. 2002,

King et al. 2006), and thus it is appropriate to test the effects of detectability on this

method. I used the Chapman modified Lincoln-Petersen calculation because even when there are no recaptures, it can compute an estimate unlike the original formula, which will try to divide by 0 (see Pollock et al. 1990 for the equations). I carried out the simulations using R (R Development Core Team 2005), and have provided the code in Appendix A.

RESULTS

Visual Surveys of Translocated Lizards

I conducted 26 surveys through all 8 enclosures consecutively (34.67 person-hours of search effort), resulting in 137 observations of *Ameiva polops*. Interestingly, despite small enclosure size and a density of 700 lizards/ha in each enclosure, I identified only 20 of the 57 translocated individuals. Although there were 35 observations for which I could not positively identify individuals, many of these sightings appeared to be of the 20 otherwise confirmed based on visual characteristics, photographs, and notes from initially processing lizards. Of the 20 identified, 9 were females and 11 were males (Table 2.2). On average I detected only 5.3 (1 SD = 2.24) individuals per survey, with a range of 2−10, out of the total 57 lizards translocated into the enclosures.

Table 2.2 Number of *Ameiva polops* identified in enclosures during visual surveys on Buck Island after all 57 individuals were translocated to Buck Island until enclosures were removed.

	Overall	Enclosure 1	2	3	4	5	6	7	8
Known Individuals (♀:♂)	32:25	4:4	4:3	4:3	4:3	4:3	4:3	4:3	4:3
Proportion of Total Seen	20/57 (35%)	4/8	1/7	3/7	0/7	2/7	5/7	0/7	5/7
Proportion of Females Seen	9/32 (28%)	1/4	1/4	2/4	0/4	0/4	3/4	0/4	2/4
Proportion of Males Seen	11/25 (44%)	3/4	0/3	1/3	0/3	2/3	2/3	0/3	3/3

Recapture of Translocated *Ameiva polops* and Body Condition

There was low capture success in the pitfall traps; only two individuals were trapped (0.002 animals/trap-hour). However, one of the individuals trapped had never been seen during the entire study period, and the other had not been observed since the beginning of the study. I captured eleven other individuals by noosing, of which nearly all had an increase in body condition. One lizard escaped and was observed outside the enclosure. There is no evidence of other escapes. I patrolled the enclosure area daily during the entire study period and did not see any other individuals outside the enclosure.

The *A. polops* that were recaptured had significantly better body condition than when originally captured on Green Cay (V=51, $P < 0.05$). Ten of the 11 individuals recaptured had increased in weight, with a mean increase in size-adjusted mass ($\pm$ 1 SD) of 0.017 ± 0.015.

Population Level Surface Activity

When only individuals seen during surveys and captured in pitfall traps were included in closed capture models, the probability of individuals being surface active (resight probability; ± 1 SE) for males and females combined was 0.251 ± 0.019. The model separating females and males produced nearly the same resight probability for both groups (Table 2.3), and model fit for these two was nearly identical based on ranking of Akaike's Information Criterion corrected for overdispersion (AIC$_c$; Burnham and Anderson 2002). When all individuals that may have been present (56 individuals) were included in the models, the resight probability for males and females combined was 0.094 ± 0.008; for males and females it was 0.122 ± 0.013 and 0.073 ± 0.009, respectively. The model assuming homogeneity in the population had considerably worse fit, with the difference in AIC$_c$ between models being 7.7 (Burnham and Anderson 2002). Based on all of these models, the range of probability that individuals were surface active during standard survey times for all individuals was 0.094 to 0.251. For females the range was 0.073 to 0.261, and for males it was 0.122 to 0.244.

Table 2.3 Resight probability as a proxy for probability of surface activity of the translocated *Ameiva polops* monitored in enclosures at Buck Island Reef National Monument. Models are grouped by the individuals considered, and ranked by lowest AIC_c.

Individuals In Models	Model	Resight Probability* (± SE)		AIC_c	ΔAIC_c
		Males	Females		
Individuals Confirmed Present[†]	Homogenous	0.251 ± 0.018		553.61	0.000
	Grouped by Sex	0.244 ± 0.024	0.261 ± 0.028	555.42	1.813
All Translocated Individuals[‡]	Grouped by Sex	0.122 ± 0.013	0.073 ± 0.009	631.90	0.000
	Homogenous	0.094 ± 0.008		639.60	7.705

* Detection probability as a proxy for probability of individuals being surface active; separated by males and females where applicable.
[†] Individuals confirmed present includes all individuals observed during surveys as well as another that was not seen, but captured in a pitfall trap. This individual captured escaped from hand before sex as determined, and was arbitrarily grouped with females to be included in the model. This individual decreased the resight probability for females by 0.005.
[‡] The model for all translocated individuals excludes one individual that had escaped because it was not available for detection in the enclosures.
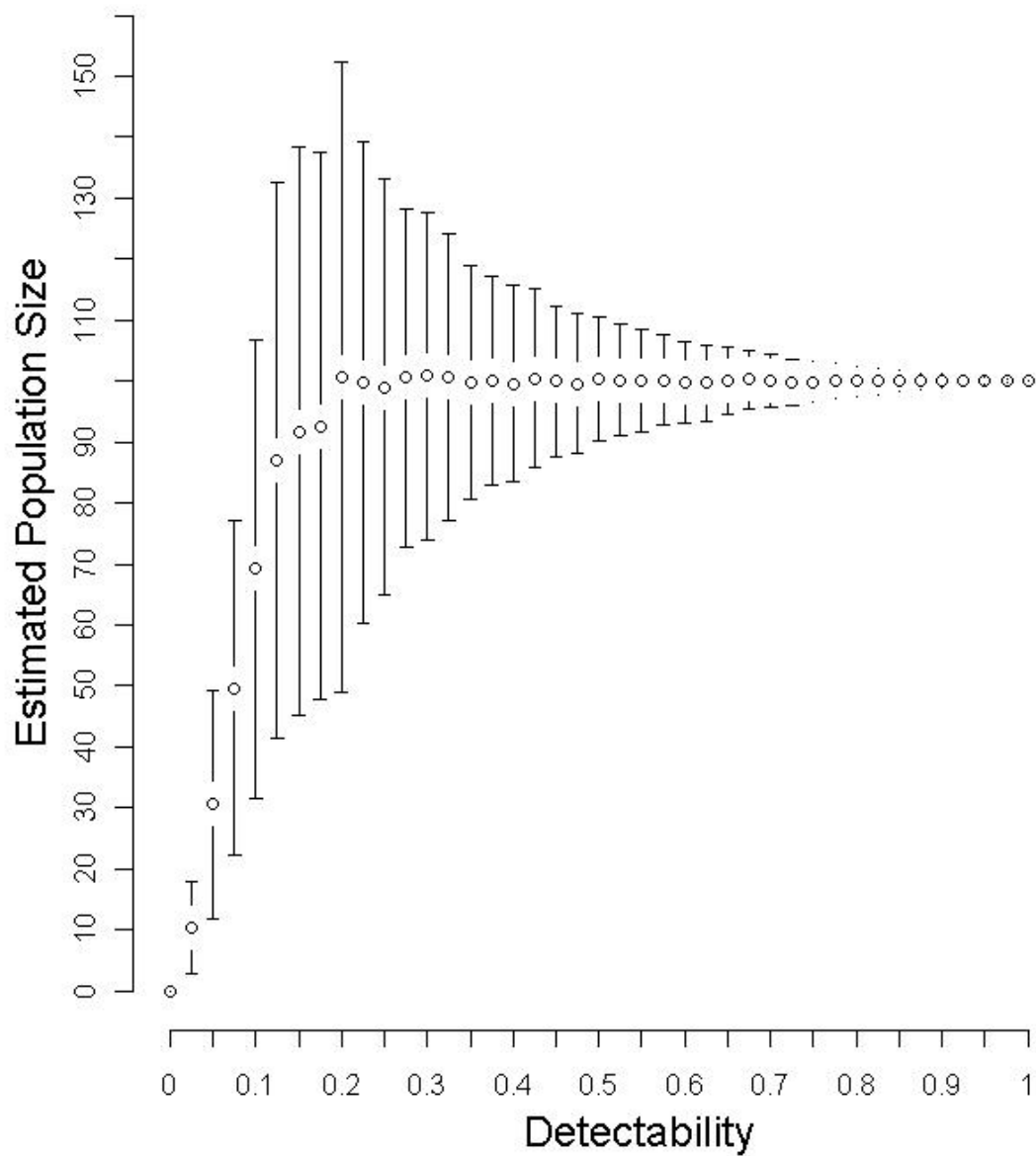
Fig. 2.2 Chapman modified Lincoln-Petersen estimates of hypothetical populations of 100 individuals, simulated 1000 times for each level of detectability from 0 to 1 in increments of 0.025. Means are open circles and error bars represent 1 standard deviation to illustrate the variance in the 1000 simulations.

Simulations

Simulations of Chapman modified Lincoln-Petersen estimates for a hypothetical population of 100 individuals reveled that this mark-resight method was generally not accurate until detection probability reaches 0.2. The standard deviation of estimates was greater than 30% of the actual estimate until the detection probability reached approximately 0.3 (Figure 2.2). When the detection probability was 0.5, population estimates greatly increased in accuracy, with a standard deviation of 10.1% of the actual population size.

DISCUSSION

The most striking result from my surveys was how few lizards were seen despite the containment of seven individuals in small ($100m^2$) enclosures. Of the 56 individuals that may have been present (excluding the one confirmed escapee), I saw only five on average, and never more than 10. Even when considering only the 20 lizards that I could confirm were present through the study, I typically saw only one-fourth of those, and never saw more than one-half on an individual survey. Thus, this experimental demonstration of detectability illustrates that simple visual encounter surveys, using only transects or other non-mark-resight methods, will yield severe underestimates of the population size. My simulations of mark-resight estimates with populations of low detectability were inaccurate, as I expected, but even simulations with detectability values as high as 0.3 were often accurate inaccurate as well, with standard deviations least 30% of the actual simulated population size. The standard deviation of the

simulated estimates did not come within 10% of the actual population size until detectability was set at 0.5.

My empirical data and simulations, together, revealed that even in this controlled situation, it is difficult to determine an accurate and precise detection probability. Even with the range of detection probabilities resulting from this study, it is difficult to speculate what may be the actual detection probability for *A. polops*, and whether there is an actual difference between detection probabilities for males and females. The pitfall traps were largely unsuccessful, and did not even capture individuals that I routinely observed on surveys. It goes without saying that noosing may also be limited, because individuals need to be detectable in order to be noosed. However, the two individuals that were captured in pitfalls provide clear evidence that some *A. polops* that were never seen were in fact present throughout the entire survey period.

Lizards Not Seen During Surveys

Although some translocated lizards could have died during the study period, I was unable to calculate mortality from recaptures in the enclosures. Potential sources of lizard mortality could be stress from the actual translocation, predation, and natural causes such as senescence. However, no mortality from predation or any other source was documented during the study. As in most macroteiids, our field work on *A. polops* (unpublished data) indicates that adults in particular are fairly robust to handling. Measures were taken to reduce handling time and thermal stress, and although some individuals may have experienced some other stress of their new environment, all the lizards behaved completely normally when they were placed in the enclosures.

Potential predators included crabs, which may predate lizards in their burrows, and two bird species. I was unable to estimate crab abundance in enclosures and infer how that may have affected the translocated lizards due to logistic constraints, although lizards observed in enclosures were typically seen throughout the enclosure period, such that there was not an obvious gradual loss to predation. With regard to potential bird predation, pearly-eyed thrashers (*Margarops fuscatus*) and American kestrels (*Falco sparverius*) both inhabit the area. I never observed pearly-eyed thrashers attacking *A. polops*, although both I and a National Park Service employee observed two separate incidents of a kestrel landing on a lizard. In both cases, the lizards escaped and were among the individuals seen during future surveys. Thus, while it is possible that some individuals died in the enclosures, mortality probably did not account for many of the individuals that were not seen.

Another source of potential loss of lizards from enclosures was escape. I only observed one individual outside of an enclosure multiple times. I immediately sealed the one small hole where the lizard escaped, and I continued to see other lizards in that enclosure throughout the remainder of the monitoring period. It is possible that other individuals escaped from enclosures, although if many did, I likely would have seen at least some of them during my time spent between and around enclosures. If lizard escapes were a factor in low detection probability, then we would infer that *A. polops* in this habitat was more detectable than the results indicated. Following, it would stand to reason that escaped lizards were detectable in the areas surrounding the enclosures. However, as mentioned, no other lizards were ever seen outside the enclosures. I was in

the enclosure area daily throughout the entire study period and would have seen escaped lizards.

Sex-Based Detectability Differences

Although the closed-capture model that included only individuals seen or trapped in enclosures estimated nearly the same resight probabilities for males and females, the model that included all translocated individuals estimated the two parameters to be further apart. Females may have simply been less detectable than males due to behavioral differences. Previous work corroborates that males are more detectable than females. For example, males of *Ameiva plei* tend to have larger home ranges than females (Censky 1995). Additionally Censky (1995) found that a small portion of males do most of the mating, and non-mating males have even larger home ranges, which may make lizards confined to an enclosure more active. Males of *Tupinambis rufescens* and *T. merianae* are significantly more active than females, especially during breeding season and this behavioral difference makes them more detectable to hunters and their dogs (Fitzgerald et al. 1991, Fitzgerald 1994). These potential sex-based differences in detectability can have important implications for estimating population sizes, and understanding the long-term viability of populations.

Conclusion

Based on my review of the literature, this appears to be the first study to measure detectability of a study organism in a controlled set of enclosures with a known initial population. Overall, my study reaffirmed what others have found about problems of detectability in population estimation of herpetofauna (Rodda and Campbell 2002,

Smolensky and Fitzgerald in press). However, population estimation techniques that assume all individuals present are detectable are still used for monitoring lizard populations (Kacoliris et al. 2009) and other herpetofauna (Mazerolle et al. 2007). These methods can work reasonably well for some taxa (e.g., crocodilians; Subalusky et al. 2009), and when additional studies are conducted to assess surface activity levels of the focal species (Grant and Doherty 2007). Additionally, models that predict activity of organisms by incorporating other variables can enhance estimates obtained from these methods (Freilich et al. 2005, Nussear and Tracy 2007), but a major assumption, that organisms will have the same detection probability with spatial and temporal variability, is not confirmed (MacKenzie and Kendall 2002).

Thus it is necessary to use methods in which detection probabilities are taken into account while simultaneously estimating population sizes. This may be accomplished using robust design techniques (Kendall et al. 1995), which also allow for long-term monitoring of survivorship and other parameters. However, this is resource-intense, and requires long-term mark-recapture projects. Other techniques are cheaper and equally effective, but are rarely used. For example Heckel and Roughgarden (1979) described a technique for estimating population sizes of *Anolis* lizards in which individuals are marked with paint from a distance using squirt guns over a three-day period. Each day a different color is used, and the number of new individuals marked and the number seen with previous days' colors are recorded. Heckel and Roughgarden (1979) used a contingency table design to estimate population sizes from these data, but we may also employ maximum likelihood estimation techniques, available in a number of software

packages (e.g., Program MARK). When large numbers of individuals are not surface active at a given time, as was the case with *A. polops*, more days of paint-marking would add power are and improve population estimates. Paint-resight methodology has largely been restricted to work with *Anolis* lizards (e.g., Diaz et al. 2005, Hite et al. 2008), but we suggest it may have great application to other lizards and we are implementing this methodology into the monitoring program for *A. polops*.

Overall, there is great value to understanding population dynamics of both rare and common species through long-term monitoring. We can use this information to benefit conservation of endangered species, and also track potential nuisance species. Although the population estimation literature is constantly expanding, many of the newer techniques stand little chance of improving our ability to estimate populations of species that are difficult to detect. Although detectability "is what it is", and can not be changed through analysis of visual encounter data, we can use efficient marking techniques for multi-day surveys to mark relatively large numbers of individuals in the population, and analyze the resulting data with modern numerical methods to obtain more accurate population estimates than can be achieved with visual encounter surveys.

CHAPTER III

IMPACTS OF A TRANSLOCATED ACTIVE FORAGING LIZARD ON ITS

INVERTEBRATE PREY BASE

INTRODUCTION

Limitation of lizard abundance by prey availability has been historically assumed in the literature (Stamps and Tanaka 1981), and has been suggested for a variety of other organisms (Sih et al. 1985). In support of this, multiple studies have demonstrated that lizards can negatively impact particular components of their invertebrate prey base. For example, using an enclosure experiment, Lewis (1989) found that *Ameiva exsul* (Puerto Rican ground lizard) negatively impacted land snails, and Schoener and Spiller (1996) found that introduced *Anolis* lizards on islands can reduce spider diversity and abundance. Additionally, Anderson (1994) found that the foraging behavior of *Cnemidophorus tigris*, (tiger whiptail lizard) followed invertebrate prey availability.

However, other studies have indicated that some lizards were not limited by and did not impact their invertebrate prey base. For example, Schoener and Spiller (1999), found that introduced and natural populations of *Anolis sagrei* did not significantly alter abundances of large (>4 mm) aerial arthropods, and actually had a positive influence on small aerial arthropods. In looking at the distribution of a translocated population of *Cnemidophorus vanzoi* (St. Lucia ground lizards) on Praslin Island, St. Lucia, Dickinson et al. (2001) found that individuals were distributed independently of their prey base, suggesting that another factor influenced their distribution.

It may be posited that foraging mode may drive impacts of lizards on the local prey base. Huey and Pianka (1981) found that active foragers tend to eat more clumped, unpredictably distributed, and sedentary prey when compared with sit-and-wait foragers. Following this, Lewis (1989) inferred that active foragers in particular may easily deplete components of their prey base. However, this has not been well documented in the literature, and in many studies only taxon-specific effects of lizards on their invertebrate prey base have been examined and detected.

In this study I used a controlled enclosure experiment with a translocated population of *Ameiva polops* (the St. Croix Ground Lizard) to determine whether the presence of this species may impact the invertebrate assemblage at the translocation site. *Ameiva polops* is a federally endangered species of the U.S.A., endemic to St. Croix, U.S. Virgin Islands (Henderson and Powell 2009). This species is believed to be extirpated from St. Croix proper by invasive mongooses (Henderson and Powell 2009), with the last sighting in 1968 (Philibosian and Ruibal 1971, Philibosian and Yntema 1976). The only natural populations of *A. polops* persist on two small satellite islands (<10 ha each), Green Cay National Wildlife Refuge and Protestant Cay (McNair 2003, McNair and Lombard 2004) and another population was introduced to a small, dredge-spoil island, Ruth Island, in 1990 (Knowles 1996, McNair and Mackay 2005). The Endangered Species Recovery Plan for *A. polops* (U.S. Fish and Wildlife Service 1984) called for a translocation of the species to Buck Island Reef National Monument, a larger island (72 ha) approximately 2.5 km from St. Croix, following the eradication of mongooses there. Thus, by the mid 1990s mongooses had been eradicated from Buck

Island (National Park Service 2007), and in 2008 a multiagency effort was undertaken to translocate individuals there from Green Cay National Wildlife Refuge. On Buck Island I sampled invertebrates from enclosures with *A. polops* and from a set of paired control enclosures without *A. polops* over the course of 6 weeks. My *a priori* hypothesis was that the presence of *A. polops* would have negative impacts on richness of invertebrate taxa, and cause decreases in abundance of individual taxa in the enclosures.

METHODS

Translocation

From 29 April through 9 May 2008 each of 8 enclosures on Buck Island was stocked with 7 adult (> 50 mm snout-vent-length) *A. polops* (4 females and 3 males), except for one, with 4 females and 4 males. The enclosures were all in the northeastern beach forest habitat of Buck Island, open-top, 10 m x 10 m square, and contained natural, pre-existing substrates and vegetation (Figure 3.1). The habitat was composed of mature trees that creating a canopy allowing dappled sunlight to reach the ground, with rich topsoil and abundant leaf litter. The density of *A. polops* in these enclosures was approximately 700 individuals per hectare, nearly the same as reported for a similarly sized insular species in the same family, *Cnemidophorus arubensis* (Aruban whiptail

lizard; Schall 1974). Although this density was much higher than previously documented for *A. polops*, its population sizes have not been well studied. Because of low detection probability, some previous estimates may have underestimated population densities (Chapter II).

Enclosure walls were a minimum of 46 cm tall, with the bottom edge buried 15 cm below ground level to prevent *A. polops* from burrowing out, and to minimize migration and of potential prey items. Like most macroteiids, *Ameiva polops* are poor climbers (pers. obs., Meier et al. 1993) and individuals could not climb out. Every enclosure that contained *A. polops* (hereafter referred to as "lizard enclosures") was adjacent to a paired, control enclosure. Control enclosures were identical to the lizard enclosures, subject to the same environmental conditions, and in the same habitat, but did not house *A. polops*. Thus, I was able to test for effects of *A. polops* on the local arthropod assemblage while accounting for natural trends and enclosure effects. After the sixth week, control enclosures were removed, though *A. polops* were contained in lizard enclosures for two more weeks of monitoring before being released into the surrounding habitat
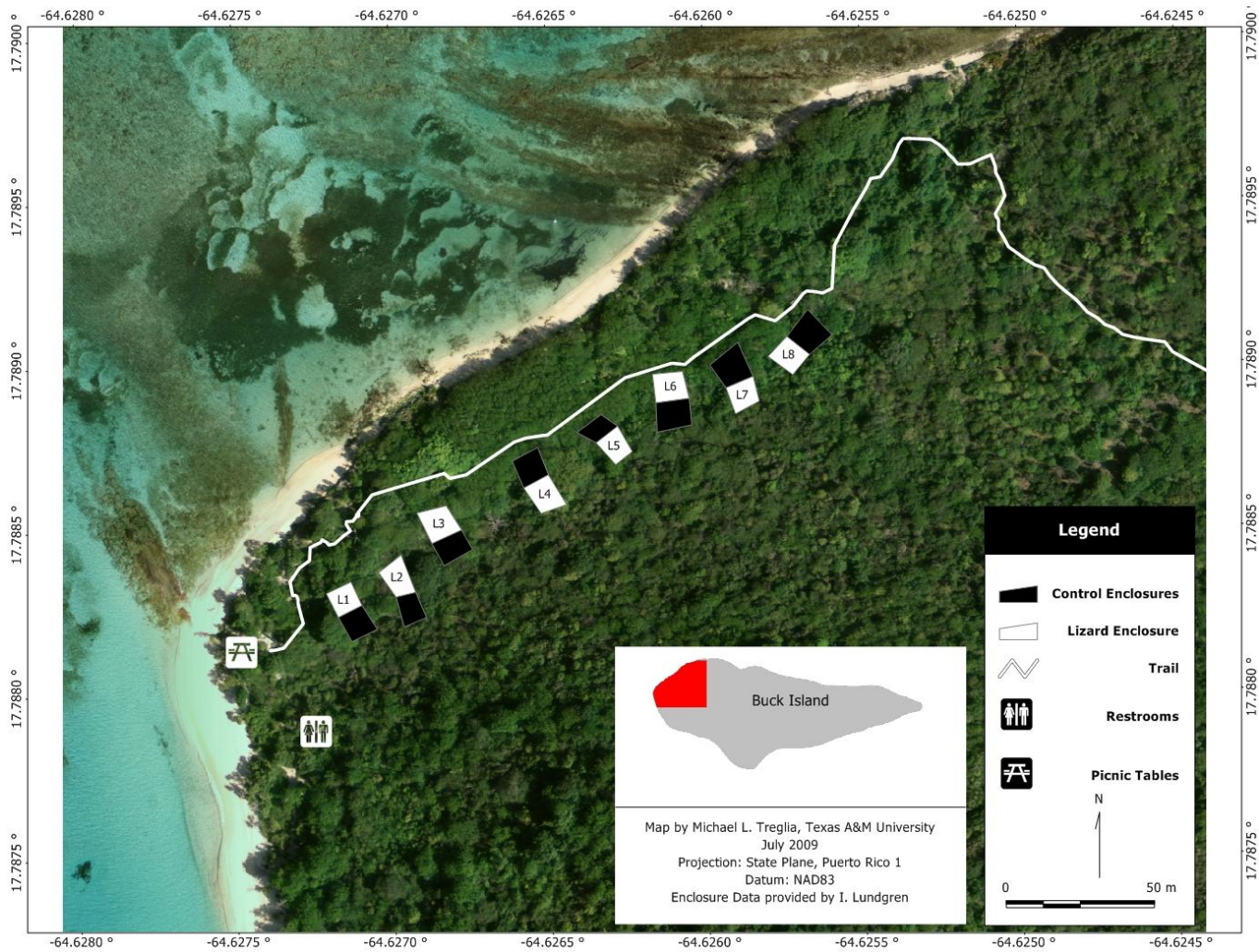
Fig. 3.1 Map of control and lizard enclosures on Buck Island; enclosures 4 and 7 were excluded from the analyses because no *Ameiva polops* were seen in them following the translocation.

Invertebrate Sampling

From 14 May to 18 June 2008, after all lizards had been translocated, I used 8 live-capture pitfall traps randomly placed in each lizard and control enclosure to sample invertebrates. My pitfall trap design (Figure 3.2) consisted of a 473 mL plastic cup (11 cm tall and 9.3 cm in mouth diameter) with a 266 mL cup with the bottom cut off to serve as a baffle (similar to that of Weeks and McIntyre 1997). The entire trap was nested within another 473 mL cup for easy removal of captured organisms, and the entire apparatus was dug in to be flush with the soil or slightly lower. I also installed cut-to-fit covers made of expanded steel mesh with diagonal openings of 2 cm x 0.6 cm to prevent lizards and large crabs from entering the trap and perhaps eating other captured organisms.
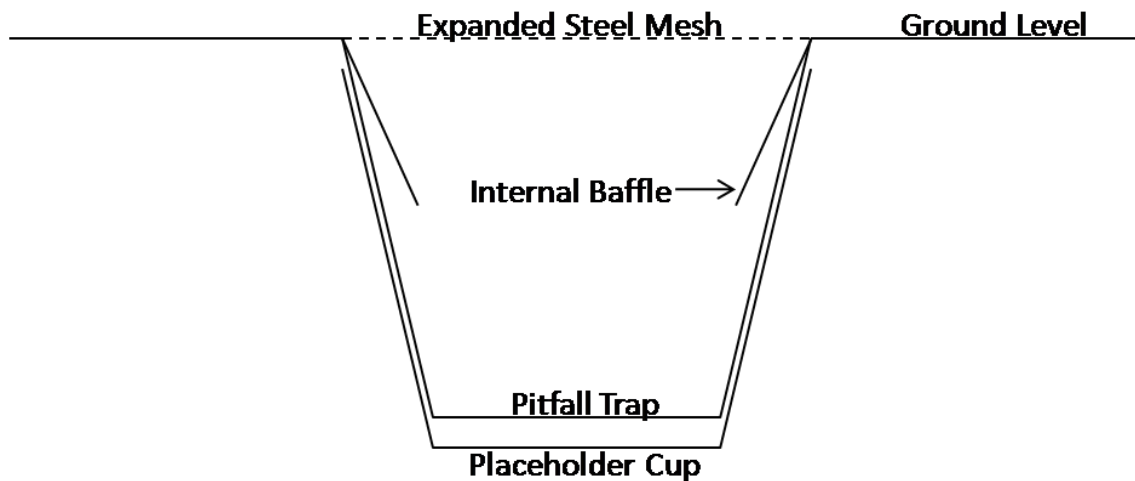
Fig. 3.2 Design of pitfall trap for capturing invertebrates in control and lizard enclosures; similar to that of Weeks and McIntyre (1997).

I opened all pitfall traps for 48 hours per week. At the end of each trapping period, collected samples were preserved in 70% ethanol. During all other times, I closed the traps in place to prevent other organisms from entering. I counted and identified all samples from the first and sixth weeks. I identified all individuals to order except for larvae (grouped together as "larvae"), and ants (which were identified to family [Formicidae] because of their disproportionate abundance).

Analyses

During monitoring of the translocated *A. polops*, there were two enclosures in which no *A. polops* were detected. Although this probably was the result of a low detection probability (Chapter II), it is possible that some individuals may have died or escaped. Therefore, I omitted those pairs of enclosures from the analyses. Additionally, although the lizard and control enclosures were spatially paired, multivariate ordinations of the invertebrates sampled from each enclosure indicate that the lizard enclosures were generally not similar to their paired control enclosure with respect to the invertebrate community at the onset of the experiment. Thus, I used non-paired analyses. Because of low sample size, I used two-way Wilcoxon signed-ranks tests to determine whether changes in individual invertebrate taxa were significantly different between control and lizard enclosures ($\alpha = 0.05$). To analyze changes in abundance, I used proportional changes to account for initial differences between enclosures ([(week 1-week 6)/week 1]). I multiplied the proportion by -1 so that decreases would be indicated as negative. For taxa not present in the first week but present in the sixth, I used the raw abundance increase value. Because increases were rare, this did not affect ranking in statistical

analyses. To assess whether *A. polops* impacted the richness of invertebrate taxa in

enclosures, I used a two-way Wilcoxon signed ranks test to compare the proportional

change in number of taxa present in the control versus lizard enclosures. Where means

are presented they are displayed ± 1 standard deviation. To account for experimentwise

error rate I used a Bonferroni correction (15 comparisons) for all significance values.

To better understand changes of the entire invertebrate community, I ordinated

enclosures from the first and sixth weeks based on the invertebrate samples collected

from them using nonmetric multidimensional scaling (NMDS; Kruskal 1964).

Nonmetric multidimensional scaling is a non-parametric ordination technique that uses

iterations of plotting taxa and sampling sites (enclosures) and randomly moving them

around to obtain a fewer-dimensional, (user-defined) representation of data (McCune

and Grace 2002). More similar sites are plotted closer together, reflecting the chosen

distance or similarity measure. To eliminate influence of rare taxa that may have been

detected or present only by slim chance, I omitted taxa that only occurred once in a

single enclosure. I used a Bray-Curtis similarity metric (Bray and Curtis 1957) with

square-root transformed data to dampen the impact of extremely abundant taxa. The

Bray-Curtis similarity metric is a measure of percentage similarity of taxa between

sampling units (each enclosure in the weeks analyzed here). This metric does not give

any weight to taxa that are mutually absent between two enclosures, and incorporates

quantitative data about each taxon, not just presence/absence. I selected the final

ordination based on lowest-stress with fewest dimensions to reduce the possibility of

drawing false inferences (McCune and Grace 2002). I carried out all analyses in R (R

Development Core Team 2005) and used the package "vegan" for the NMDS (Oksanen et al. 2009).

RESULTS

Overall I collected 5046 individual invertebrate specimens in pitfall traps, comprising 21 orders. Of these, 4637 (and 20 orders) were in the 6 sets of enclosures that I included in statistical analyses. Total abundance decreased in all enclosures from the first week to the sixth, though the change was not significantly different between control and lizard enclosures (W=12, Bonferroni-corrected P=1.00). There were no statistically significant relationships between treatment and proportional changes in abundance for individual taxa (Table 3.1). *Ameiva polops* also did not have an impact on the richness of invertebrate orders in enclosures (W=14.5, P=1.00), though number of taxa present decreased in all enclosures (mean proportional change in number of taxa = -0.2690 ± 0.273).

The two-dimensional NMDS (stress = 12.45, Figure 3.3) revealed some differences in invertebrate community changes between lizard and control enclosures not clear in univariate analyses. Overall, there is a clear shift of all enclosures with respect to the invertebrate community, from negative to positive along NMDS axis 1. Axis 2 also depicts small differences in changes between control and lizard enclosures by the sixth week, with most control enclosures on the negative side of axis 2, and lizard enclosures being neutral or slightly positive.

Table 3.1. Mean proportional changes (± 1 SD) of each taxon in control and lizard enclosures from week 1 to week 6, with test-statistic (W) and Bonferroni-corrected P-value, comparing differences in control and lizard enclosures. Analyses were not carried out for taxa of < 10 individuals.

	Total Sample	Mean Change	Control Enclosures No. of Enclosures with Gain	No. of Enclosures with Loss	No. of Enclosures with No Change	Mean Change	Lizard Enclosures No. of Enclosures with Gain	No. of Enclosures with Loss	No. of Enclosures with No Change	W	P
Total	4637	-0.80 ± 0.20	0	6	0	-0.77 ± 0.15	0	6	0	12	1.00
Formicidae	2120	-0.88 ± 0.17	0	6	0	-0.71 ± 0.28	0	6	0	8	1.00
Acari	1642	-0.25 ± 1.29	1	4	1	-0.36 ± 0.80	2	4	0	10	1.00
Diptera	380	-1.00 ± 0.01	0	6	0	-1.00 ± 0.00	0	6	0	24	1.00
Isopoda	125	-0.67 ± 0.82	1	5	0	-0.42 ± 0.78	1	4	1	8	1.00
Larva	81	-0.80 ± 0.40	0	5	1	-0.50 ± 0.55	0	3	3	13.5	1.00
Thysanoptera	49	7.17 ± 6.15	5	0	1	0.00 ± 0.63	1	1	4	33	0.24
Collembola	49	0.33 ± 1.97	2	3	1	0.50 ± 2.07	2	3	1	17.5	1.00
Aranae	39	0.21 ± 1.40	1	3	2	0.42 ± 1.79	1	2	3	17	1.00
Hymenoptera	35	-0.72 ± 0.44	0	5	1	-0.07 ± 0.65	1	2	3	7.5	1.00
Coleoptera	30	1.83 ± 3.66	3	1	2	0.08 ± 0.49	1	1	4	23	1.00
Psocoptera	20	-0.83 ± 0.41	0	5	1	0.33 ± 1.03	2	1	3	5	0.43
Orthoptera	20	-0.17 ± 0.41	0	1	5	-0.13 ± 1.58	1	4	1	24	1.00
Blattodea	15	0.33 ± 1.37	1	1	4	-0.50 ± 0.84	1	4	1	26	1.00
Hemiptera	7	-0.17 ± 0.41	0	1	5	0.00 ± 0.89	2	2	2	NA	
Pseudoscorpiones	6	-0.33 ± 0.52	0	2	4	-0.17 ± 0.75	1	2	3	NA	
Isoptera	6	-0.17 ± 0.75	1	2	3	-0.17 ± 0.41	0	1	5	NA	
Scolopendromorpha	5	-0.17 ± 0.41	0	1	5	0.33 ± 0.52	2	0	4	NA	
Neuroptera	3	0.17 ± 0.41	1	0	5	0.00 ± 0.00	0	0	6	NA	
Decapoda	2	0.00 ± 0.00	0	0	6	-0.17 ± 0.41	0	1	5	NA	
Lepidoptera	1	0.00 ± 0.00	0	0	6	-0.17 ± 0.41	0	1	5	NA	
Thysanura	1	0.00 ± 0.00	0	0	6	0.17 ± 0.41	1	0	5	NA	
Dermaptera	1	0.00 ± 0.00	0	0	6	-0.17 ± 0.41	0	1	5	NA	
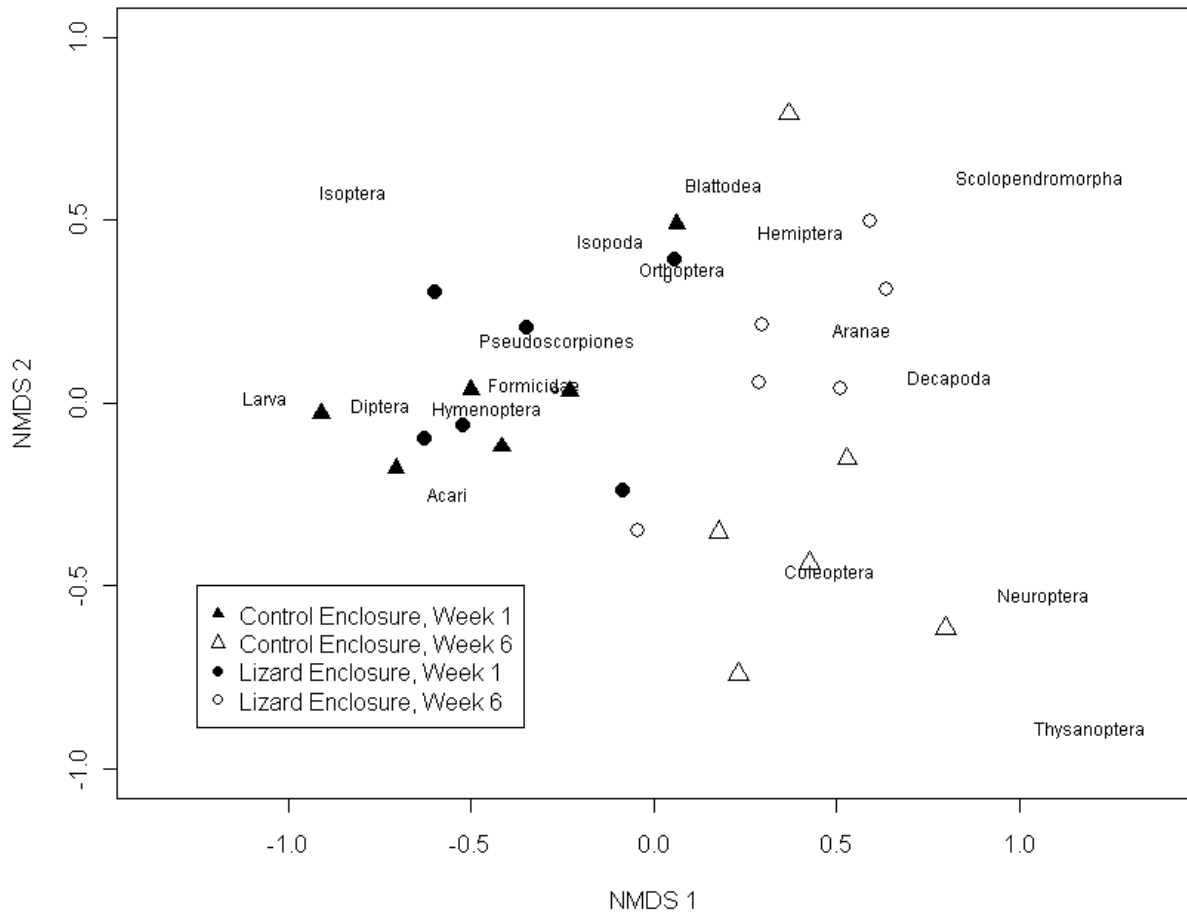
Fig. 3.3 Two dimensional nonmetric multidimensional scaling of enclosures based on the invertebrate community collected from them in the first and sixth weeks with square-root transformed data and a Bray-Curtis similarity metric; stress = 12.45. Although NMDS is inherently without axes, the ordination is displayed here oriented along principal component axes 1 and 2.

DISCUSSION

Overall, *A. polops* did not have a significant impact on the arthropod abundance within the enclosures on Buck Island over the six week study period. However, based on the NMDS the control and lizard enclosures, initially fairly similar, developed some small differences over the six-week period. Exploration with NMDS revealed that although abundance of Thysanopterans was the greatest difference between control and lizard enclosures, the same pattern was present in the ordination even when that group was removed from this analysis. A potential confounding factor was the small overall sample of many taxa, with 8 of 22 groups of arthropods categorized having less than 10 individuals; thus, slight changes may have influenced the ordination, even though extremely rare taxa that occurred only once in a single enclosure were removed from this analysis.

It is possible that artifacts of my sampling strategy or the relatively short study period influenced the results. Because *A. polops* searches for food by rooting through the leaf litter, the lizards may encounter dense patches of less mobile prey that are not well represented in pitfall traps. Although another strategy, such as leaf-litter sampling, may have been more effective to get certain taxa, that also would have its own bias, and it is impossible to know if any sampling methodology would be better or worse overall. With a longer sampling interval, stronger effects of *A. polops* on the arthropod assemblage may have been detected, but extending the period lizards were held in enclosures was not an option for this study.

My univariate results generally agreed with those of Lewis (1989) who found that *Ameiva exsul*, a larger species closely related to *A. polops*, did not impact ground-dwelling arthropods (grouped much broader than presented here), even after 5 months. However, my results contrasted with a number of studies showing that *Anolis* lizards did have major impacts on their prey base (Pacala and Roughgarden 1984, Schoener and Spiller 1996, Schoener and Spiller 1999). This may be reflective of very distinctive foraging behaviors of *Anolis* and *Ameiva*. Most teiids and all *Ameiva* species are active foragers (Vitt et al. 2000) and encounter a wide variety of prey in their habitat during wide-ranging forays (e.g., Lewis 1989, Vitt et al. 2000). In contrast, lizards in the genus *Anolis* are classic sit-and-wait predators that ambush mobile prey that are visible from perch sites (Moermond 1979), and their impacts should be focused on particular taxa. The data here support that notion, as even the most pronounced changes for particular taxa (Thysanoptera and Psocoptera) were not significant.

Small differences that I did observe between control and lizard enclosures may be attributed to natural outbreaks associated with seasonal or environmental changes. For example Levings and Windsor (1984) found that variations in leaf litter moisture influenced arthropod abundance on Barro Colorado Island, Panama. Although all enclosures were in the same habitat and within 100 meters of each other, my small sample size of enclosures and random chance may have caused some taxa to become more prominent in certain sets of enclosures. Additionally, it is possible that *A. polops* consumed certain taxa at finer taxonomic and spatial scales than I examined here, and caused trophic effects that I was unable to interpret from my analyses.

Although *A. polops* did not appear to deplete arthropods in the enclosures, multiple long-term studies of lizards have demonstrated that population sizes fluctuate with environmental variables and invertebrate abundance (Whitford and Creusere 1977, Dunham 1981). Additionally, Anderson (1994) found that foraging patterns of *Cnemidophorus tigris* followed the spatial distribution of local prey resources. When prey resources are less dense, lizards may decrease their foraging efficiency, resulting in greater risks of predation and reduced reproductive output (Vitt and Price 1982, Warner et al. 2007), potentially impacting their population sizes. Although Dickinson et al. (2001) found the distribution of *Cnemidophorus vanzoi* to be independent of arthropod distribution after two years at a translocation site, this may have been during a prey-rich time, or the result of other microhabitat preferences.

In conclusion, I did not find direct impacts of *A. polops* on arthropod abundance in the enclosures, and there is no evidence the lizards' population size would become limited by prey availability provided that conditions do not change. In a follow-up visit approximately 1 year after the experiment, the population appeared to be doing well. Our research team observed gravid adult females that we confirmed had hatched on Buck Island (based on lack of toe-clips), as well as numerous other adults, juveniles, and neonates. Although environmental stochasticity can cause major fluctuations in the population size (Fitzgerald 1994), the population has persisted thus far and appears to be increasing, demonstrating the potential for long-term success with the current resources.

CHAPTER IV

CONCLUSION

Translocation can be an important strategy to re-establish at-risk species in former parts of their range, or to expand their range to new refuge sites (Griffith et al. 1989). Although the feasibility of translocations has been questioned for herpetofauna (Dodd and Seigel 1991), efforts during the last 20 years have been more successful than in the past, particularly when the explicit purpose of the translocations was for species conservation (as opposed to reducing human-wildlife conflict or research; Germano and Bishop 2009). However, of 38 reptile translocations recorded in the literature between 1991 and 2006 for conservation purposes, the confirmed success rate was only nearly 40%, with approximately 45% translocations having uncertain outcomes and 15% actually failing (Germano and Bishop 2009).

Translocation research has pointed out a crucial need for improved monitoring of translocated populations. Although long-term success (the establishment of a self-sustaining population) is impossible to document without long-term data (Griffith et al. 1989, Dodd and Seigel 1991), immediate increases in population size can be used as preliminary indicators of success and to inform management practices (e.g., U.S. Fish and Wildlife Service 1984, 2009, Gates et al. 2010).

Efficient and accurate estimation of reptile population numbers has proven to be difficult. Availability biases make raw visual encounter techniques inaccurate because they generally include the assumption that at a minimum, animals along a transect line are detectable (Rodda and Campbell 2002, Smolensky and Fitzgerald in press). Some

studies have validated distance sampling for select species (Dickinson and Fa 2000, Kacoliris et al. 2009) using mark-resight techniques, and correction factors can be created based on other field work. However, these validations and corrections require extra resources that often make them infeasible, and must be repeated across time and space, as detectability should be assumed to vary across both parameters (MacKenzie and Kendall 2002).

My results reinforce the findings of Rodda and Campbell (2002) and Smolensky and Fitzgerald (in press), illustrating that, at least for some lizard species, low activity levels make visual encounter surveys impractical for actually estimating their abundance. Furthermore, I showed that when Lincoln-Petersen techniques (Pollock et al. 1990) are applied, this low detectability can result in underestimates, and high variance in estimates of population sizes. To improve population estimates of *A. polops*, I recommend using a multiday mark-resight design, similar to that suggested by Heckel and Roughgarden (1979). In this technique, observers search intensely for individuals over multiple days, and mark all individuals seen by squirting them with paint from a distance, using a different color each day and recording the number seen that had been marked on each of the previous days (based on paint marks). This allows more individuals to be marked and re-sighted for increased accuracy. Then, maximum likelihood techniques can be used to derive actual population estimates (Otis et al. 1978, Pledger 2000, White 2008).

In evaluating the potential for long-term success of wildlife translocations, an important aspect to consider is food resource availability at the translocation site. There

has been little research designed to investigate whether food limitation is a potential problem for translocated lizards (Germano and Bishop 2009). Previous work demonstrated that lizard populations can track prey availability in space and time (Anderson 1994), and also can impact particular components of their prey base (Schoener and Spiller 1996, Schoener and Spiller 1999). However, whether or not lizards may quickly deplete local prey resources had not been explicitly studied, but may be an important consideration for translocations.

My results show that, for a translocated population of the active foraging lizard, *A. polops*, there were no measurable impacts on the abundance or diversity of the invertebrate prey base that could be interpreted as food-limiting over a six week period. Considering the large size (72 ha) of the translocation site, and the ecology of the species in this habitat, it seems unlikely that the lizards would ever become prey-limited at the translocation site. The small differences in abundance changes between lizard and control enclosures were not significant. It is possible that finer spatial and taxonomic-scale effects were taking place, which I was unable to detect, or that impacts of *A. polops* needed more time to become pronounced. Potentially, as the population of *A. polops* expands across Buck Island, the population size will track abundance of the invertebrate prey base, as seen with other lizard species (Anderson 1994). However, it seems unlikely that *A. polops* will be constrained by resources in the intermediate term, and the population will likely grow quickly.

One factor that may have influenced the results of my study, particularly for my investigation of impacts of *A. polops* on the invertebrate prey base, is that after lizards

were released in the enclosures I could not be certain how many survived through the entire 6-week period. Thus, I was unable to control for population density at all times. I was only able to analyze the effects of *A. polops* as present vs. absent, and I eliminated two sets of enclosures for which I could not confirm continued presence or absence of *A. polops* at the end of the study period. It is possible that *A. polops* has a larger effect at higher population densities than I was able to measure, though their presence alone did not seem to be important to the arthropod assemblage. Furthermore, the body condition of lizards observed at the end of the experiment was greater than when they were initially translocated from Green Cay, where they are native.

In conclusion, studies conducted to date indicate that reptile translocations have great potential for success. Potential prey limitations, at least for small, active-foraging lizards, may not greatly influence long-term persistence, though sharp drops in resources may lead to decreased reproductive rates of females (Vitt and Price 1982). It is important to establish monitoring protocols for populations of translocated animals so that success and failures can be documented. Future monitoring by the National Park Service at Buck Island will incorporate mark-resight techniques, as suggested in this thesis, to periodically estimate the population size. Monitoring will also include other surveys to monitor the expansion of *A. polops* across Buck Island. The population appears to be growing already, and in a follow-up trip to Buck Island in June, 2009, I observed many juveniles and adults, including gravid females that had hatched there. On short walks through the habitat I saw as many as 13 individuals, and considering the low detectability for this species, it is likely that there are many more present. Thus, the

preliminary outlook for this population is very positive, and I expect the population to

continue growing and expanding throughout the habitat.

LITERATURE CITED

Anderson, R. A. 1994. Functional and population responses of the lizard *Cnemidophorus tigris* to environmental fluctuations. American Zoologist 34:409-421.

Armstrong, D. P., and P. J. Seddon. 2008. Directions in reintroduction biology. Trends in Ecology & Evolution 23:20-25.

Bennett, A. F., and J. A. Ruben. 1979. Endothermy and activity in vertebrates. Science 206:649-654.

Borges-Landáez, P. A., and R. Shine. 2003. Influence of toe-clipping on running speed in *Eulamprus quoyii*, an Australian Scincid lizard. Journal of Herpetology 37:592-595.

Bray, J. R., and J. T. Curtis. 1957. An ordination of the upland forest communities of southern Wisconsin. Ecological Monographs 27:325-349.

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 University Press, New York.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2004. Advanced Distance Sampling: Estimating Abundance of Biological Populations. Oxford University Press, New York.

Burnham, K. P., and D. R. Anderson. 2002. Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach. 2nd edition. Springer, New York.

Censky, E. J. 1995. Mating strategy and reproductive success in the Teiid lizard, *Ameiva plei*. Behaviour 132:529-557.

Cook, R. D., and J. O. Jacobson. 1979. A design for estimating visibility bias in aerial surveys. Biometrics 35:735-742.

Diaz, P. M., H. Heinz, J. S. Parmerlee, Jr., and R. Powell. 2005. Population densities and structural habitats of *Anolis* lizards on St. Eustatius, Netherlands Antilles. Caribbean Journal of Science 41:296-306.

Dickersin, K., and Y.-I. Min. 1993. Publication bias: the problem that won't go away. Annals of the New York Academy of Sciences 703:135-148.

Dickinson, H. C., and J. E. Fa. 2000. Abundance, demographics and body condition of a translocated population of St. Lucia whiptail lizards (*Cnemidophorus vanzoi*). Journal of Zoology 251:187-197.

Dickinson, H. C., J. E. Fa, and S. M. Lenton. 2001. Microhabitat use by a translocated population of St. Lucia whiptail lizards (*Cnemidophorus vanzoi*). Animal Conservation 4:143-156.

Dodd, C. K., Jr. 1993. The effects of toeclipping on sprint performance of the lizard *Cnemidophorus sexlineatus*. Journal of Herpetology 27:209-213.

Dodd, C. K., Jr., and R. A. Seigel. 1991. Relocation, repatriation, and translocation of amphibians and reptiles: are they conservation strategies that work? Herpetologica 47:336-350.

Dunham, A. E. 1981. Populations in a fluctuating environment: the comparative population ecology of iguanid lizards *Sceloporus merriami* and *Urosaurus ornatus*. University of Michigan Museum of Zoology Miscellaneous Publication 158.

Fisher, M., and A. Muth. 1989. A technique for permanently marking lizards. Herpetological Review 20:45-46.

Fitzgerald, L. A. 1994. The interplay between life history and environmental stochasticity: implications for the management of exploited lizard populations. American Zoologist 34:371-381.

Fitzgerald, L. A., J. M. Chani, and O. E. Donadío. 1991. *Tupinambis* lizards in Argentina: implementing management of a traditionally exploited resource. Pages 306-316 *in* J. Robinson and K. Redford, editors. Neotropical Wildlife: Use and Conservation. University of Chicago Press, Chicago.

Freilich, J. E., R. J. Camp, J. J. Duda, and A. E. Karl. 2005. Problems with sampling desert tortoises: a simulation analysis based on field data. Journal of Wildlife Management 69:45-56.

Gates, C. C., C. H. Freese, P. J. P. Cogan, and M. Kotzman. 2010. American Bison: Status Survey and Conservation Guidelines 2010. IUCN, Gland, Switzerland.

Germano, J. M., and P. J. Bishop. 2009. Suitability of amphibians and reptiles for translocation. Conservation Biology 23:7-15.

Grant, T. J., and P. F. Doherty. 2007. Monitoring of the flat-tailed horned lizard with methods incorporating detection probability. Journal of Wildlife Management 71:1050-1056.

Griffith, B., J. M. Scott, J. W. Carpenter, and C. Reed. 1989. Translocation as a species conservation tool: status and strategy. Science 245:477-480.

Heckel, D. G., and J. Roughgarden. 1979. A technique for estimating the size of lizard populations. Ecology 60:996-975.

Hein, E. W., and W. F. Andelt. 1995. Estimating coyote density from mark-resight surveys. The Journal of Wildlife Management 59:164-169.

Henderson, R. W., and R. Powell. 2009. Natural History of West Indian Reptiles and Amphibians. University Press of Florida, Gainesville, FL.

Hite, J. L., C. A. R. Gómez, S. C. Larimer, A. M. Díaz-Lamiero, and R. Powell. 2008. Anoles of St. Vincent (Squamata: Polychrotidae): population densities and structural habitat use. Caribbean Journal of Science 44:102-115.

Huey, R. B., and E. R. Pianka. 1981. Ecological consequences of foraging mode. Ecology 62:991-999.

Kacoliris, F. P., I. Berkunsky, and J. D. Williams. 2009. Methods for assessing population size in sand dune lizards (*Liolaemus multimaculatus*). Herpetologica 65:219-226.

Kendall, W. L., K. H. Pollock, and C. Brownie. 1995. A likelihood-based approach to capture-recapture estimation of demographic parameters under the robust design. Biometrics 51:293-308.

King, R. B., A. Queral-Regil, and K. M. Stanford. 2006. Population size and recovery criteria of the threatened Lake Erie watersnake: integrating multiple methods of population estimation. Herpetological Monographs 20:83-104.

Knowles, W. C. 1996. Conservation of the St. Croix Ground Lizard, *Ameiva polops*. Final report. Endangered species project, Study IIB, Job ES 2-1, Division of Fish and Wildlife, United States Virgin Islands.

Kruskal, J. B. 1964. Nonmetric multidimensional scaling: a numerical method. Psychometrika 29:115-129.

Levings, S. C., and D. M. Windsor. 1984. Litter moisture content as a determinant of litter arthropod distribution and abundance during the dry season on Barro Colorado Island, Panama. Biotropica 16:125-131.

Lewis, A. R. 1989. Diet selection and depression of prey abundance by an intensively foraging lizard. Journal of Herpetology 23:164-170.

Mackay, A. 2007. Conservation and Management of the St. Croix Ground Lizard (*Ameiva polops*) on Green Cay National Wildlife Refuge. University of Maryland, Eastern Shore, Princess Anne, MD.

MacKenzie, D. I., and W. L. Kendall. 2002. How should detection probability be incorporated into estimates of relative abundance? Ecology 83:2387-2393.

Marsh, H., and D. F. Sinclair. 1989. Correcting for visibility bias in strip transect aerial surveys of aquatic fauna. The Journal of Wildlife Management 53:1017-1024.

Mazerolle, M. J., L. L. Bailey, W. L. Kendall, J. Andrew Royle, S. J. Converse, and J. D. Nichols. 2007. Making great leaps forward: accounting for detectability in herpetological field studies. Journal of Herpetology 41:672-689.

McCune, B., and J. B. Grace. 2002. Analysis of Ecological Communities. MjM Software Design, Glenenden Beach, OR.

McNair, D. B. 2003. Population estimate, habitat associations, and conservation of the St. Croix Ground Lizard *Ameiva polops* at Protestant Cay, United States Virgin Islands. Caribbean Journal of Science 39:94-99.

McNair, D. B., and C. D. Lombard. 2004. Population estimates, habitat associations, and management of *Ameiva polops* (Cope) at Green Cay, United States Virgin Islands. Caribbean Journal of Science 40:353-361.

McNair, D. B., and A. Mackay. 2005. Population estimates and management of *Ameiva polops* (Cope) at Ruth Island, United States Virgin Islands. Caribbean Journal of Science 41:352-357.

Meier, A. J., R. E. Noble, and S. L. Rathbun. 1993. Population status and notes on the biology and behavior of the St. Croix ground lizard on Green Cay (St. Croix, U.S. Virgin Islands). Caribbean Journal of Science 29:147-152.

Meier, A. J., R. E. Noble, and P. J. Zwank. 1990. Criteria for the introduction of the St. Croix ground lizard. New York State Museum Bulletin 471:154-156.

Moermond, T. C. 1979. The influence of habitat structure on *Anolis* foraging behavior. Behaviour 70:147-167.

National Park Service. 2007. Capture and Translocation of Endangered Endemic St. Croix Ground Lizard, *Ameiva polops* to Buck Island Reef National Monument, St. Croix, U.S. Virgin Islands. PMIS # 3995, Buck Island Reef National Monument, St. Croix, U.S. Virgin Islands.

Nussear, K. E., and C. R. Tracy. 2007. Can modeling improve estimation of desert tortoise population densities? Ecological Applications 17:579-586.

Oksanen, J., R. Kindt, P. Legendre, B. O'Hara, G. L. Simpson, P. Solymos, M. H. H. Stevens, and H. Wagner. 2009. vegan: Community Ecology Package. R package version 1.15-4. http://CRAN.R-project.org/package=vegan.

Olsson, M., and R. Shine. 1997. The limits to reproductive output: offspring size versus number in the sand lizard (*Lacerta agilis*). The American Naturalist 149:179-188.

Otis, D. L., K. P. Burnham, G. C. White, and D. R. Anderson. 1978. Statistical inference from capture data on closed animal populations. Wildlife Monographs 62:3-135.

Pacala, S., and J. Roughgarden. 1984. Control of arthropod abundance by *Anolis* lizards on St. Eustatius (Neth. Antilles). Oecologia 64:160-162.

Philibosian, R., and R. Ruibal. 1971. Conservation of the lizard *Ameiva polops* in the Virgin Islands. Herpetologica 27:450-454.

Philibosian, R., and J. A. Yntema. 1976. Records and status of some reptiles and amphibians in the Virgin Islands. I. 1968-1975. Herpetologica 32:81-85.

Pledger, S. 2000. Unified maximum likelihood estimates for closed capture-recapture models using mixtures. Biometrics 56:434-442.

Pollock, K. H., J. D. Nichols, C. Brownie, and J. E. Hines. 1990. Statistical inference for capture-recapture experiments. Wildlife Monographs 107:3-97.

Pough, F. H. 1980. The advantages of ectothermy for tetrapods. The American Naturalist 115:92.

R Development Core Team. 2005. R: A Languange and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Rodda, G. H., and E. W. Campbell. 2002. Distance sampling of forest snakes and lizards. Herpetological Review 33:271-274.

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

Scargle, J. D. 2000. Publication bias: the "file-drawer" problem in statistical inference. Journal of Scientific Exploration 14:91-106.

Schall, J. J. 1974. Population structure of the Aruban whiptail lizard, *Cnemidophorus arubensis*, in varied habitats. Herpetologica 30:38-44.

Schoener, T. W., and D. A. Spiller. 1996. Devastation of prey diversity by experimentally introduced predators in the field. Nature 381:691-694.

Schoener, T. W., and D. A. Spiller. 1999. Indirect effects in an experimentally staged invasion by a major predator. American Naturalist 153:347-358.

Scott, J. M., and J. W. Carpenter. 1987. Release of captive-reared or translocated endangered birds: what do we need to know? The Auk 104:544-545.

Sepúveda, M., F. Hernández, D. G. Hewitt, W. P. Kuvlesky, G. Waggerman, and R. L. Bingham. 2006. Evaluation of auditory counts for estimating breeding populations of white-winged doves. Journal of Wildlife Management 70:1393-1402.

Shine, R. 2005. Life-history evolution in reptiles. Annual Review of Ecology, Evolution, and Systematics 36:23-46.

Sih, A., P. Crowley, M. McPeek, J. Petranka, and K. Strohmeier. 1985. Predation, competition, and prey communities: a review of field experiments. Annual Review of Ecology & Systematics 16:269-311.

Smolensky, N. L. 2008. Population enumeration and the effects of oil and gas development on dune-dwelling lizards. Master's Thesis. Texas A&M University, College Station, TX.

Smolensky, N. L., and L. A. Fitzgerald. in press. Distance sampling underestimates population densities of dune-dwelling lizards. Journal of Herpetology.

Stamps, J., and S. Tanaka. 1981. The influence of food and water on growth rates in a tropical lizard (*Anolis aeneus*). Ecology 62:33-40.

Subalusky, A. L., L. L. Smith, and L. A. Fitzgerald. 2009. Detection of American alligators in isolated, seasonal wetlands. Applied Herpetology 6:199-210.

Swann, D. E., R. C. Averill-Murray, and C. R. Schwalbe. 2002. Distance sampling for Sonoran Desert tortoises. The Journal of Wildlife Management 66:969-975.

U.S. Fish and Wildlife Service. 1984. St. Croix Ground Lizard Recovery Plan. U.S. Fish and Wildlife Service, Atlanta, GA.

U.S. Fish and Wildlife Service. 2009. Revised Recovery Plan for the Laysan Duck (*Anas laysanensis*). U.S. Fish and Wildlife Service, Portland, OR.

U.S. Fish and Wildlife Service. 2009. Revised Recovery Plan for the Laysan Duck (*Anas laysanensis*).

Vitt, L. J., and H. J. Price. 1982. Ecological and evolutionary determinants of relative clutch mass in lizards. Herpetologica 38:237-255.

Vitt, L. J., S. S. Sartorius, T. C. S. Avila-Pires, M. C. Espósito, and D. B. Miles. 2000. Niche segregation among sympatric Amazonian Teiid lizards. Oecologia 122:410-420.

Warner, D. A., M. B. Lovern, and R. Shine. 2007. Maternal nutrition affects reproductive output and sex allocation in a lizard with environmental sex determination. Proceedings of the Royal Society B: Biological Sciences 274:883-890.

Weeks, R. D., and N. E. McIntyre. 1997. A comparison of live versus kill pitfall trapping techniques using various killing agents. Entomologia Experimentalis Et Applicata 82:267-273.

White, G. 2008. Closed population estimation models and their extensions in Program MARK. Environmental and Ecological Statistics 15:89-99.

White, G. C., and K. P. Burnham. 1999. Program MARK: Survival estimation from populations of marked animals. Bird Study 46 Supplement:120-138.

Whitford, W. G., and F. M. Creusere. 1977. Seasonal and yearly fluctuations in Chihuahuan desert lizard communities. Herpetologica 33:54-65.

APPENDIX A

R-Code for simulation of estimates of 100 individuals with varying levels of detection probability using a modified Lincoln-Petersen calculation (Pollock et al. 1990). Simulation was carried out 1000 times for each detection value evaluated. Detectability was varied from 0 to 1 in increments of 0.025.

```
#The function to run the simulation
detectability<-function(x){
l<-matrix(nrow=1000,ncol=1)
any.na <- any(is.na(l))
while (any.na) {
        X <- matrix(rbinom(200, 1, prob = x), nrow = 100)
n1<-sum(X[,1])
n2<-sum(X[,2])
rs<-rowSums(X)
m2<-length(rs[rs==2])
nT<-(((n1+1)*(n2+1)/(m2+1))-1)
        first.na <- which(is.na(l))[1]
        l[first.na] <- nT
        any.na <- any(is.na(l))
}
l
}
```

```
#Create a vector from 0 to 1, at intervals of 0.025
detect<-seq(0,1,0.025)
```

```
#Run function for simulation for each vector entry; simulation for each detectability
parameter is #added to the data as a new column
detect.a<-sapply(detect, detectability)
```

VITA

Name: Michael Louis Treglia

Address: 2258 TAMU, College Station, TX 77843-2258

Email Address: mlt35@tamu.edu

Education: B.S., Natural Resources, Cornell University, 2007
                 M.S., Wildlife & Fisheries Sciences, Texas A&M University, 2010

Publications:

Treglia, M.L., O. Ramirez. 2009. *Nerodia rhombifer*. Geographic distribution.

Herpetological Review 40(2):238.

Treglia, M.L. 2008. Kraig Adler: A lifetime promoting herpetology. Iguana. 15(4):234–

237.

Treglia, M.L., A.J. Muensch, R. Powell, and J.S. Parmerlee, Jr. 2008. An invasive anole:

Anolis sagrei on St. Vincent and its potential impact on native anoles. *Caribbean*

Journal of Science 44(2):251–256.

Henderson, R.W., M.L. Treglia, and S.D. Powell. 2007. *Corallus cookii*. Foraging.

Herpetological Review 38(4):466

Powell, S. D., M. L. Treglia, R. W. Henderson, and R. Powell. 2007. Treeboas in the

West Indies: responses of *Corallus cookii* and *C. grenadensis* to disturbed habitats.

In: R. W. Henderson and R. Powell (eds.). *Biology of the Boas and Pythons*. Eagle

Mountain Publ., Eagle Mountain, Utah.

Treglia, M.L. 2006. An annotated checklist of the amphibians and reptiles of St. Vincent,

West Indies. *Iguana* 13: 252–263.