CONSERVATION OF THE ENDANGERED SAN SALVADOR ROCK IGUANAS (*CYCLURA RILEYI RILEYI*): POPULATION ESTIMATION, INVASIVE SPECIES CONTROL, TRANSLOCATION, AND HEADSTARTING

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Abstract.—This study describes our efforts to identify and ameliorate threats to San Salvador Rock Iguanas (*Cyclura rileyi rileyi*). Repeated surveys during 1994–2013 of the six small cays occupied suggest a fairly stable population of 429–646 individuals, but with impaired carrying capacity due to habitat deterioration from storm damage and invasive species. Cactus Moth (*Cactoblastis cactorum*) larvae dramatically reduced the biomass of *Opuntia stricta* cacti on several cays inhabited by iguanas. Black Rat (*Rattus rattus*) presence was positively associated with cay size, and a greater proportion of inland-lake cays were occupied compared to offshore cays. We eradicated rats in 2000 and Australian Pines (*Casuarina equisetifolia*) in 2005 on Low Cay, but detected no subsequent increase in iguanas. Of two translocation efforts, the illegal transfer of iguanas to the San Salvador Club Med Resort in 2000 failed for unknown reasons, and the sanctioned establishment of iguanas on Cut Cay in 2005 experienced reproduction failure, possibly because of rat predation. Iguanas at Club Med (altered environment) but not those on Cut Cay (natural vegetation) increased dramatically in body size compared to their source population, suggesting that nutrition of these herbivores affects body size more than low population density and reduced social interactions. We established an iguana headstarting facility at the Gerace Research Centre to augment the wild population (by release of captive-raised iguanas), enhance conservation education, and promote formal habitat protection within a proposed national park. Collectively, these findings provide useful details for monitoring and managing this taxon in the future.

Key Words.—Bahamas; *Cactoblastis cactorum*; *Casuarina equisetifolia*; conservation education; eradication; population surveys; *Rattus rattus*

INTRODUCTION

Among the West Indian Rock Iguanas, the Central Bahamian Rock Iguanas, *Cyclura rileyi*, remain one of the most threatened species, with two of the three recognized subspecies deemed Critically Endangered (*C. rileyi cristata*, *C. rileyi rileyi*) and the third (*C. rileyi nuchalis*) as Endangered by the IUCN Red List of Threatened Species (IUCN. 2014. The IUCN Red List of Threatened Species. Available from http://www.iucnredlist.org [Accessed 30 September 2014]). Although each subspecies formerly occupied large islands, today these mostly herbivorous lizards are confined to small, remote, uninhabited cays of three island groups (Hayes et al. 2004). At present, Sandy Cay or White Cay Rock Iguanas (*C. rileyi cristata*) are restricted to Sandy (White) Cay in the southern Exumas, where an estimated 1000+ individuals persist (Hayes et al. 2004, this volume).

Acklins Rock Iguanas (*Cyclura rileyi nuchalis*) exist naturally on just two cays in the Acklins Bight of the Crooked/Acklins Island group, but a third population has been introduced to a cay in the central Exumas (Iverson et al. this volume). Although as many as 10,000 individuals of this taxon may remain, the current distribution represents a tiny fraction (0.2%) of its former range (Hayes et al. 2004). San Salvador Rock Iguanas (*Cyclura rileyi rileyi*) are largely confined to four tiny offshore cays and two islets within the hypersaline lakes of San Salvador Island, though a small and possibly non-sustaining population persists on the main island itself, where non-native predators and vehicles threaten persistence (Hayes et al. 2004 and further described here). With fewer than 600 individuals remaining, this taxon similarly occupies a mere fraction (0.2%) of its former range. Morphometric analyses suggest that these three taxa warrant status as distinct

subspecies and independent management units (Carter and Hayes 2004; Hayes and Carter 2005), but Malone et al. (2000) found no mtDNA sequence differences among the three forms. Nevertheless, some authorities consider the three taxa to be distinct species (Powell and Henderson 2011).

Lamentably, San Salvador Rock Iguanas continue to decline. In recent decades, a number of populations have become extirpated (Barn Cay in the 1970s; High Cay in the 1980s; Gaulin Cay in 1999). The identified threats include illicit smuggling, invasive species, disease, and catastrophic storms that have damaged both soil and vegetation (e.g., Hurricanes Erin in 1995, Bertha and Lili in 1996, Floyd in 1999, Frances in 2004, Irene in 2011, and Sandy in 2012).

This report summarizes our work on San Salvador Rock Iguanas from 1994 through 2014, and provides an update of the early activities summarized only briefly by Carter and Hayes (2004) and Hayes et al. (2004). It encompasses three major efforts. First, to better understand potential threats and assess management efforts, we repeatedly surveyed the population size and age structure of each iguana population using both mark-resighting and transect surveys. Repeated surveys on one cay with marked iguanas provided useful information on the factors related to detection of iguanas during surveys, including the sex and age class of iguanas, and the time of day that surveys were conducted. Second, we undertook several mitigation measures intended to benefit the lizards, including invasive species control and population translocation. In doing so, we evaluated the success of these efforts and considered hypotheses regarding the factors associated with presence of Black Rats (*Rattus rattus*) and the relative influence of nutrition versus population density on iguana body size. Third, we initiated a headstarting program implemented in part to boost the wild population of iguanas by raising hatchlings in captivity and eventually releasing them at a size less vulnerable to predators and other risks (Alberts et al. 2004; Alberts 2007; Escobar et al. 2010). We conclude by discussing some of the opportunities and obstacles associated with the conservation of this threatened Bahamian endemic.

MATERIALS AND METHODS

Study site.—The island of San Salvador occupies its own bank on the eastern side of the Bahamas archipelago. Although a relatively small island (163 km^2) composed of limestone with karst features, it nevertheless supports diverse habitats, including coastal rock, sand strand and Sea Oats (*Uniola paniculata*), coastal coppice, blacklands coppice, and mangrove communities (Smith 1993). Numerous small islets occur within the extensive system of mostly hypersaline lakes in the island's interior, and eleven small cays occur offshore. Iguana
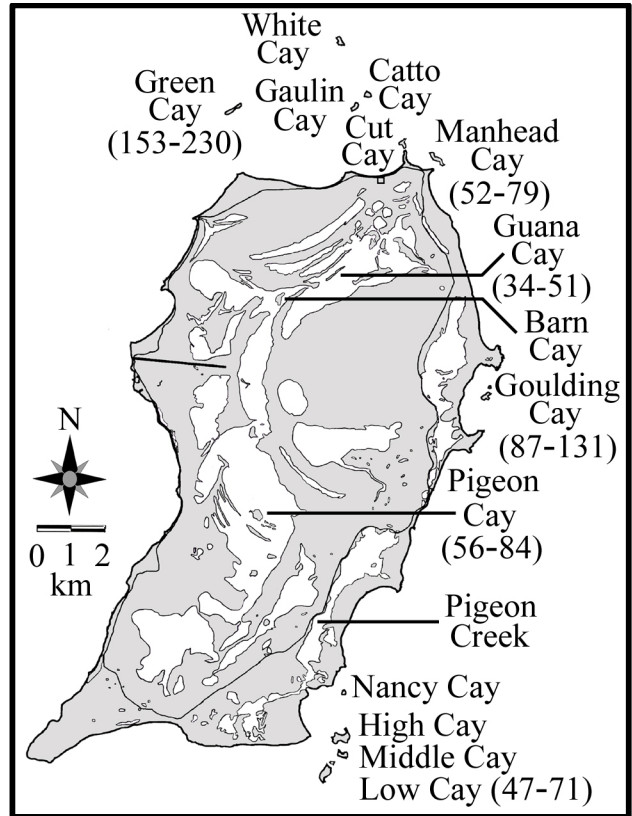
FIGURE 1. San Salvador Island, The Bahamas, with its offshore and inland lake cays. Estimated population size of San Salvador Rock Iguanas (*Cyclura rileyi rileyi*) is indicated for those cays that currently host natural populations. Populations have become extirpated on Barn, Gaulin, and High Cays. A very small population occurs on the main island. A translocated population (14 adults) was established on Cut Cay in 2005, but has been non-viable, presumably due to presence of Black Rats (*Rattus rattus*).

populations occur primarily on six of the cays (Fig. 1). Some of these cays also host substantial seabird colonies and several species compete with the iguanas for burrows (Hayes 2003; Hayes et al. 2004; Trimm and Hayes 2005). Five additional native reptile species co-occur with the iguanas, including one small snake, the San Salvador Threadsnake (*Epictia columbi*, Hayes et al. 2004; Hillbrand et al. 2011).

Iguana body size and sex.—We captured iguanas by noose or by hand to measure a number of morphological variables, including mass (nearest gram weighed by Pesola spring scale) and snout-vent length (SVL, to nearest millimeter). We determined the sex of iguanas by probing of cloacal pouches. We marked the lizards individually using colored glass beads sutured to the nuchal crest (Hayes et al. 2000). During summer 1999 on Green Cay, we also painted an alphanumeric code on both sides of the animal using white enamel paint for short-term identification (Cyril 2001). Methods and analyses of morphological data appear elsewhere (Carter

and Hayes 2004; Hayes et al. 2004, 2012); these summarize sexual differences, variation in body size and scalation among the populations, effects of habitat change on body condition, the relationship between body size and reproductive parameters, and ecological factors associated with toe and tail injuries. Here, we compare body size between translocated iguanas and other individuals from the source cay.

Iguana population surveys.—We conducted two sets of iguana surveys. The first set consisted of entire-cay visual-encounter surveys (Hayes et al. 1995; Hayes and Carter 2000) of each of the six iguana populations on the offshore and inland-lake cays using a standardized route and method. Individual populations were surveyed from three to 14 times during the period 1994 to 2013, with dates (month and year) provided in Table 1. Two to three researchers (always including WKH) spread out and walked slowly through all accessible habitat on each cay, and gently prodded the vegetation with bamboo fishing rods or sticks to flush iguanas. The route walked varied from cay to cay, depending on cay size and habitat structure, but followed the habitat structure rather than lines or strips, and was standardized for each cay. We recorded all iguanas detected by sight or by sound (as they scampered noisily into the vegetation) and the size of all iguanas seen sufficiently well. We paid careful attention not to double-count iguanas, which was made easier by the small size of the cays, the relatively small number of iguanas, limited vegetation structure to impede observation of iguana movements, and communication among observers. On Green Cay, where double-counting was expected to be most problematic due to high population density, most of the adult iguanas were marked, and the unmarked individuals including smaller numbers of juveniles and subadults, were spatially dispersed and easy to keep track of. We visually categorized iguanas into four size classes based on approximate snout-vent length: juveniles (< 12 cm), subadults (12–19 cm), small adults (20–27 cm), and large adults (> 28 cm). These size classes were fairly easy to distinguish except at the boundary of adjacent categories, where some error was inevitable. The observers came to a collective agreement on categorizing size before and during the survey; most iguanas were observed by more than one researcher during the surveys. With the exception of Low Cay, which has mostly large adults, we eventually discontinued distinguishing between the two adult size classes. We conducted surveys between mid-morning and mid-afternoon (1000–1500) during dry weather. Surveys were generally completed within 1 h.

To verify and better interpret our survey results, we conducted a second set of intensive surveys of a single iguana population. One of us (SC) independently (without helpers) conducted 37 surveys of iguanas on Green Cay, where numerous marked individuals were present, from 23 June to 22 July 1999. Using an approach similar to the first set, these surveys encompassed 80% of the cay while avoiding a dense seabird colony that had settled on the eastern end that year. Surveys were done during the morning (0800–1030), midday (1300–1430), and evening (1700–1930) hours, and required 30–75 minutes to complete. Usually two or three surveys were conducted per day, typically separated by 6 h or more, but always by a minimum of 3 h to maintain independence between successive surveys. The sequence of approach for surveying three demarcated regions of the cay was randomized to help reduce iguana detection of the oncoming investigator. During each survey, all iguanas seen were identified as marked or unmarked, and the size class was recorded when possible. To evaluate how iguana activity levels and detectability varied with gender and time of day, we examined the proportion of marked iguanas resighted during each survey using a 2 x 3 (sex x time of day) repeated-measures analysis of variance (ANOVA, Green and Salkind 2005) via SPSS (SPSS v13.0 for Windows, Statistical Package for the Social Sciences, Inc., Chicago, Illinois, USA, 2004), with alpha = 0.05. We treated sex as a within-subjects factor because both sexes were surveyed at the same time (i.e., to avoid pseudo-replication, surveys were used as the appropriate unit of replication for degrees-of-freedom), and time of day as a between-subjects factor. We followed this with Least Significant Difference multiple comparisons. Proportions were arcsine-transformed prior to analysis, and parametric assumptions were met. We computed effect sizes as partial eta-squared (η^2), interpreted as approximate percent of variance explained by an effect or interaction, with values of ~ 0.01, ~ 0.06 and ≥ 0.14 loosely regarded as small, moderate, and large effects, respectively (Cohen 1988). For each survey, a population estimate was also derived based on the Lincoln-Petersen equation (Hayes et al. 1995; Hayes and Carter 2000). The number of marked iguanas increased from 62 to 76 between the first and last surveys, and the number used in the equation was adjusted each day that new iguanas were captured and marked. A one-way ANOVA (with assumptions met) was used to compare Lincoln-Petersen population estimates for the three times of day. To compare the proportions of individuals of each age class that were observed during surveys, we used a 3 x 3 (age class x time of day) ANOVA that treated age class (juveniles, subadults, adults) as a within-subjects factor (because all three age classes were surveyed at the same time) and time of day as a between-subjects factor. Proportions were rank-transformed prior to analysis (with assumptions met) to avoid problems associated with proportions summing to 1.0. Finally, we computed coefficients of variation (CV = SD x 100/mean) to compare the precision of iguana counts for each age class, the count of all iguanas, and for the Lincoln-Petersen estimates.

Previous mark-resighting surveys of *C. rileyi* subspecies on six cays (including four of those studied here; Hayes et al. 2004), and repeated surveys of *C. rileyi nuchalis* (Thornton 2000) and *C. rileyi rileyi* (see results of this study) within single populations, indicated detectability in the range of 0.25 to 0.5, with a mean of 0.33 for 37 surveys on Green Cay (reported here). Thus, for each survey in the first set, which encompassed all of the inhabited cays, we divided the total number of iguanas detected by 0.33 and by 0.5 to derive a range for each population estimate. All population estimates of each population (excluding autumn surveys with a large pulse of neonate iguanas) were then averaged to yield a final estimate, which was similarly expressed as a range.

Invasive species.—We continually monitored each cay harboring iguanas to identify any invasive species present. Numerous non-native species exist on the main island, including many plant species, Red Imported Fire Ants (*Solenopsis invicta*), and feral companion animals (*Canis familiaris*, *Felis catus*) and farm animals (*Bos taurus*, *Sus* sp.; Smith 1993; Deyrup 1994; Smith 2010; William Hayes pers. obs.). However, we focused on the three invasive species that have become established on the offshore and inland lake cays: Australian Pines (*Casuarina equisetifolia*), Cactus Moths (*Cactoblastis cactorum*), and Black Rats (*Rattus rattus*).

TABLE 1. Population estimates for San Salvador Rock Iguanas (*Cyclura rileyi rileyi*), expressed as a range, and age class percentages (within parentheses) of juveniles, subadults, small adults, and large adults, respectively. Population estimates based on standardized visual encounter survey densities divided by detectability of 0.33 and 0.5 (Thornton 2000; Hayes et al. 2004; Table 2 of this study) to provide a range. Brackets indicate that the percentages of small and large adults were combined for some surveys. Mean estimate = mean of all surveys excluding those conducted in October and November.

Date	Goulding Cay	Green Cay	Guana Cay	Low Cay	Manhead Cay	Pigeon Cay
May, July 1994	60–90	158–237	16–24	50–75	56–84	
June 1998	116–174 (5.26.61.7)	130–195 (6.16.60.18)	30–45 (20.27.33.20)	42–63 (0.0.7.93)	32–48 (14.43.43.0)	70–105 (3.6.77.14)
June 1999			56–84 (4.11.61.25)			38–57 (0.11.84.5)
October 1999		142–213 (1.11.[87])		96–144 (46.6.4.44)	56–84 (31.23.46.0)	
May 2000				36–54 (0.6.0.94)		
November 2000		192–288 (1.7.[92])		106–159 (26.7.5.63)	36–54 (29.18.53.0)	
June 2001				54–81 (0.0.14.86)		58–87 (0.19.[81])
May 2002				64–96 (0.5.10.86)		
October 2002		160–240 (12.8.[81])		50–75 (0.11.22.67)		
June 2003	86–129 (0.43.[57])			42–63 (0.14.7.79)		
July 2004		200–300 (6.7.[87])		46–69 (0.7.21.71)	70–105 (29.35.35.0)	
February 2005				54–81 (5.0.10.86)		
June 2005		122–183 (3.7.[90])				
June 2006		172–258 (7.7.[86])		54–81 (0.5.14.82)		
June 2007				52–78 (0.20.5.75)		
May 2012		136–204 (3.18.[78])				
June 2013				28–42 (0.0.0.100)		
Mean Estimate	87–131 (*n* = 3)	153–230 (*n* = 6)	34–51 (*n* = 3)	47–71 (*n* = 11)	52–79 (*n* = 3)	56–84 (*n* = 3)
Trend	Stable	Stable	Unclear	Declining?	Unclear	Unclear

We removed all Australian Pines from Low Cay by hand or by machete, without applying chemicals, and monitored their potential reestablishment on subsequent visits to the cay. To measure Prickly Pear Cacti (*Opuntia stricta*) loss due to grazing by Cactus Moth larvae, we recorded the number of cactus pads (cladodes) and maximum height of individual plants (to nearest centimeter) on three representative 2 m² plots on the western portion of Green Cay in August 1994, May 1995, November 1995, June 1998, and June 1999. Browsing by the moth larvae will generally kill smaller *Opuntia* plants and destroy the terminal segments of larger plants, thereby reducing their overall size (Jezorek 2011; Jezorek et al. 2012). Each plot appeared to have numerous plants, but we could not distinguish individuals reliably due to vegetative multiplication (Reyes-Agüero et al. 2006). Considering the small sample size, we limited statistical analysis to computation of effect size as Kendall's *W*, with 0.1, 0.3, and 0.5 corresponding loosely to small, medium, and large effects (Green and Salkind 2005). Compared to statistical significance, effect size is independent of sample size, biologically more meaning-ful, and can be more readily compared among different studies (Nakagawa and Cuthill 2007).

We used collapsible Sherman aluminum live rodent traps (7.5 x 9 x 23 cm; H.P. Sherman, Tallahassee, Florida, USA) baited with several peanuts to assess the presence of rats on all offshore cays and at one location on the main island. Trapping was conducted during various months of the year, but mostly in June and October. Our goal was to identify their presence rather than population density; *R. rattus* can breed seasonally or year-round, and population size and trap success can fluctuate independent of season (Ewer 1971; Tamarin and Malecha 1972; Clark 1980). Traps were set in the evening after darkness and retrieved early in the morning, as the rats are primarily nocturnal. Any rats captured during the 310 trap nights were euthanized by cervical dislocation. We employed stepwise binary logistic regression (Mertler and Vannatta 2004) using SPSS software and $\alpha = 0.05$ to test which factors influenced the confirmed presence of rats among the 12 cays. The predictors included cay size, water body type (inland lake versus ocean, a categorical predictor), and distance from the main island. We used a stepwise model because of the small sample size (12 cays).

We attempted rat eradication on Low Cay in summer 1999 by using brodifacoum rodenticide, but the effort failed when abundant Hermit Crabs (*Clibanarius tricolor*) blocked access to the bait stations. The rodenticide and application over a 20 m grid system are partially described in Day et al. (1998) and Hayes et al. (2004), but we provide more details on the bait stations here. To reduce the risk of incidental poisoning of iguanas and birds, we secured the bait blocks via wire within rectangular plastic bait stations (roughly 20 × 7.5 × 7.5 cm; Fig. 2A).

We repeated the procedure in summer 2000 with a modified bait delivery system, in which the same bait stations were elevated 15–20 cm above the ground on a single PVC stake to limit entry of Hermit Crabs into the stations (Fig. 2B, 2C). Each station had holes drilled through it to insert the stake, and was held in place on the stake by a plastic zip tie secured around the stake and beneath the station.

Translocation.—Two iguana translocations were undertaken. The first, conducted illegally by employees of Club Med in 2000 or earlier, involved the removal of six or more adult iguanas (sex ratio unknown) from Green Cay to the resort grounds on the main island. We gleaned details about the iguanas from interviews of staff at the resort and by observing and capturing iguanas there in October 2002. The second translocation, conducted with permission of the landowner and a permit from the Bahamas Ministry of Agriculture, involved the transfer of 14 adult iguanas (6 ♂♂, 8 ♀♀) from Green Cay to Cut Cay in February 2005. These animals were transported individually, out of direct sunlight, within cloth pillowcases (51 x 91 cm). Prior to the second translocation, we set rat traps on Cut Cay and on the adjacent main island (North Point) to determine whether rats were present. We compared the body sizes of adult iguanas captured at Club Med (October 2002) and those observed at Cut Cay (June 2012) to the large sample captured at Green Cay, the source population (1993–1999). We hypothesized that translocated iguanas on Club Med and Cut Cay would attain a larger body size (length and mass) than those of Green Cay (the source of origin) due to a richer food source and potential supple-mental feeding (at Club Med), less food competition (both locations), and/or fewer social inter-actions that would otherwise constrain time devoted to feeding (both populations; see Carter and Hayes 2004; Hayes et al. this volume; Iverson et al. this volume). We used an independent samples *t*-test (Green and Salkind 2005) to compare the body sizes of adult (≥ 20 cm) male iguanas from Club Med and Green Cay, and a one-sample *t*-test (Green and Salkind 2005) to compare the body size of the one adult female iguana from Club Med to adult females from Green Cay. We had no iguana recaptures from Cut Cay, but their body size relative to Green Cay iguanas could readily be inferred by observation in the field.

RESULTS

Iguana population estimates.—For the repeated surveys on Green Cay in 1999, we conducted separate analyses of marked iguanas and counts of all iguanas. The 2 x 3 (sex x time of day) ANOVA revealed that the mean proportion of marked adult iguanas detected during a given survey was similar for the two sexes ($F_{1,34} = 0.23$, $P = 0.63$, partial $\eta^2 < 0.01$) but varied

FIGURE 2. Bait station (plastic drain pipe roughly 25 × 7.5 × 7.5 cm) for delivering brodifacoum (Weatherblock XT® Rodenticide) to eradicate non-native Black Rats (*Rattus rattus*) on Low Cay, San Salvador, The Bahamas. (A) Three bait blocks secured by wire within the bait station. (B) Failed station placement on ground in 1999, when abundant Hermit Crabs (*Clibanarius tricolor*) swarmed the stations and blocked rat access to the bait. (C) Successful station placement 15–20 cm above ground on a single PVC stake in 2000, which precluded Hermit Crab entry but allowed access by rats. (Photographed by William K. Hayes).

TABLE 2. Percentages (mean ± 1 SE) of marked adult San Salvador Rock Iguanas resighted during 37 surveys on Green Cay between 23 June and 22 July 1999, and corresponding Lincoln-Petersen (L-P) estimates of adult population size. Surveys were undertaken during morning (0800–1030), midday (1300–1430), and evening (1700–1930) hours.

Time of day	n	Males (%)		Females (%)		All iguanas (%)		L-P estimate	
		Mean ± SE	Range	Mean ± SE	Range	Mean ± SE	Range	Mean ± SE	Range
Morning	16	30.6 ± 1.6	22–47	30.7 ± 2.7	11–47	31.2 ± 1.8	19–47	133.6 ± 4.1	114–168
Midday	5	20.0 ± 5.4	6–38	24.1 ± 3.0	16–34	22.8 ± 3.2	17–31	128.9 ± 14.6	105–181
Evening	16	38.7 ± 1.8	25–50	37.3 ± 2.3	21–49	38.6 ± 1.8	24–47	132.9 ± 3.8	112–165

significantly with time of day ($F_{2,34} = 11.01$, $P < 0.001$, partial $\eta^2 = 0.39$). Proportions (= detectability) for both sexes combined averaged 0.31 in the morning ($n = 16$ surveys), 0.23 at midday ($n = 5$), and 0.39 in the evening ($n = 16$; Table 2, reported as percentages). Post-hoc comparisons indicated that proportions for each time of day differed from each of the other times. No interaction existed between sex and time of day ($F_{2,34} = 0.49$, $P = 0.62$, partial $\eta^2 = 0.03$), indicating that relative levels of activity and detectability of males and females were similar during the three time periods. Mean overall detectability for the 37 surveys was 0.33. The one-way ANOVA for Lincoln-Petersen estimates, in contrast, showed no effect for time of day ($F_{2,34} = 0.13$, $P = 0.88$, partial $\eta^2 < 0.01$). Thus, the mean population estimate from surveys was similar for the three times of day (129–134), though individual survey estimates ranged from 105–181, with an overall mean of 133 (Table 2). Assuming that 133 iguanas was the actual adult population size within the study area, the maximum Lincoln-Petersen estimate error for an individual survey

was 18.0% (24 iguanas) below and 36.1% (48 iguanas) above this number.

The 3 x 3 (age class x time of day) ANOVA for proportions of each age class detected during surveys revealed significant differences among the age classes ($F_{2,68} = 213.71$, $P < 0.001$, partial $\eta^2 = 0.86$), with representation greatest for adults (71.3%), less for sub-adults (25.3%), and least for juveniles (3.4%; Table 3). However, no differences existed for time of day ($F_{2,34} = 0.58$, $P = 0.57$, partial $\eta^2 = 0.03$), and there was no interaction between age class and time of day ($F_{4,68} = 0.11$, $P = 0.98$, partial $\eta^2 < 0.01$). Thus, relative levels of activity and detectability were highly similar among all age classes regardless of time of day. We do not know the actual detectability of juveniles and subadults due to insufficient marked individuals, but assuming it is similar to adults, the 71.3% representation for 133 adult iguanas suggests a conservative estimate of 187 iguanas within the study area during June–July 2009.

TABLE 3. Percentages (mean ± 1 SE) of San Salvador Rock Iguanas of three age classes sighted during 37 surveys on Green Cay between 23 June and 22 July 1999. Surveys were undertaken during morning (0800–1030), midday (1300–1430), and evening (1700–1930) hours.

		Juveniles (%)		Subadults (%)		Adults (%)	
Time of day	n	Mean ± SE	Range	Mean ± SE	Range	Mean ± SE	Range
Morning	16	3.1 ± 0.8	0–11	25.8 ± 1.6	16–42	71.1 ± 1.6	58–79
Midday	5	3.9 ± 1.7	0–9	24.2 ± 3.9	14–33	71.9 ± 4.4	62–84
Evening	16	3.6 ± 0.5	0–8	25.1 ± 1.2	16–33	71.3 ± 1.1	65–82

With means of 1.6, 11.6, and 32.7 juvenile, subadult, and adult iguanas detected per survey, respectively, corresponding CV values were 82.7, 34.9, and 26.1. Thus, although representation of age classes was consistent among surveys, the precision for number of animals detected was comparatively low for juveniles in particular. With 48.1 total iguanas detected per survey (including those of undetermined size class), the CV for total iguana counts was 26.1. With a mean Lincoln-Petersen estimate of 133 adult iguanas, the CV for iguana mark-resighting estimates was 13.7, approximately half that of the counts for adults, indicating greater precision. Given the inherent variation in population estimates, these results encouraged us to report the population-wide surveys (below) as a range based on detectability of 0.33–0.5 (see Discussion for further justification).

Surveys of all iguana populations indicated that population size and age structure varied substantially among the cays (Table 1). However, several trends were evident. First, the largest estimates occurred during autumn surveys, when a large pulse of hatchlings typically increased the population. For the one exception, most of the nests were apparently destroyed on Green Cay in September 1999, when Hurricane Floyd swept over much of the low-lying island (hatchlings comprised 1% compared to 46% and 31% on Low and Manhead Cays, respectively, in late October 1999). This hurricane also extirpated the small Gaulin Cay population (Hayes et al. 2004). Low hatchling numbers on Green Cay occurred again in November 2000, but hatchling success was improved in early October 2002 (apparently before nest emergence on Low Cay). Second, some populations consistently experienced more recruitment of juveniles into the population than others. The Low Cay population, for example, was comprised almost exclusively of adults throughout the study (usually > 90% during the summer months), whereas Manhead Cay had much greater representation of

juveniles and subadults (57–64% during the summer months). Third, we saw no detectable increase in recruitment and population growth on Low Cay following eradication of rats in 2000. Finally, those populations sampled most often have appeared to be stable, except that the last survey of Low Cay suggested a possible recent decline. The sum of estimates from all islands (Table 1) provided a total population estimate of 429–646 across our 20-year study.

Because iguanas are encountered only rarely on the main island, the sparse and possibly non-viable population there could not be surveyed. In the 1980s, John Winter (pers. comm.) observed two adults on the eastern side of the island, one on the western shore of Six Pack Pond, and the other on the northeastern shore of Granny Lake. For the past two decades, we and others have occasionally seen up to two adults at the Fortune Hill Ruins, also on the east side of the island. Shortly after Hurricane Floyd in 1999, Eberth Jones (pers. comm.) reported an adult on the northeastern end of the island near North Point. Finally, between 2005 and 2011, we observed three different adults that appeared briefly at the Gerace Research Centre on the northern end of the island. One of the latter was killed by a vehicle on the main highway (Vince Voegeli, pers. comm.).

Invasive species and their control.—Although Australian Pine occurs abundantly on the main island, we found it present on only one offshore cay (Low Cay) and on none of the sampled inland lake cays. In July 2004, we pulled up and cut down ca. 10 saplings on the northeastern portion of Low Cay. Three of these trees were nearly 3 m high. No regrowth occurred through 2013.

We confirmed Cactus Moths in 1994 on Green Cay (Hayes et al. 1995). At that time, substantial Prickly Pear Cacti remained, though some loss might have occurred. During the period 1994–1999, the cactus patches declined precipitously in biomass, with an 86.6 ± (SE) 6.3% loss
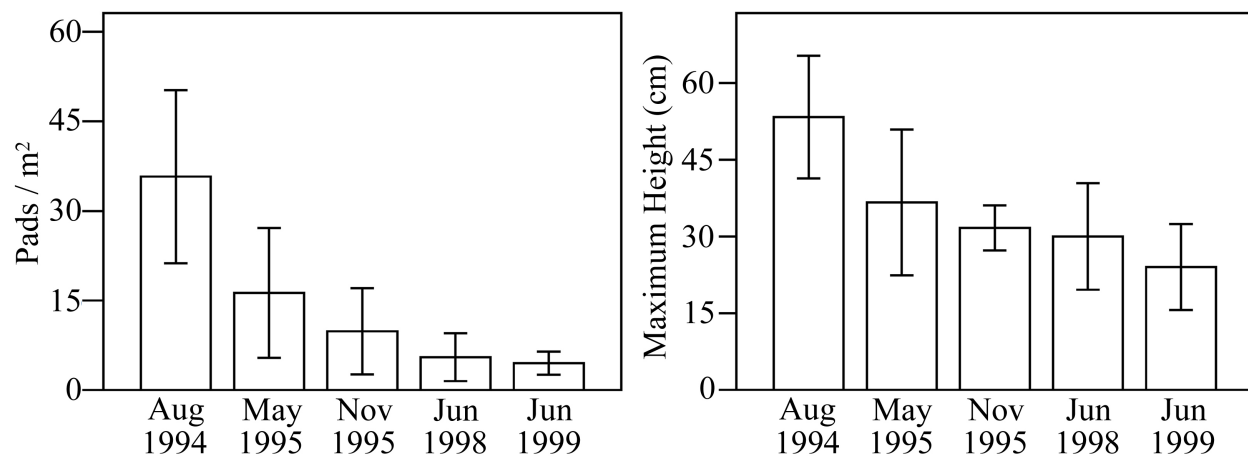
FIGURE 3. Mean (± 1 SE) number of Prickly Pear Cacti (*Opuntia stricta*) pads (cladodes) and maximum height recorded in three 2-m² plots on the west end of Green Cay, San Salvador, The Bahamas, from 1994–1999. The biomass decline resulted from intense herbivory by non-native larvae of Cactus Moths (*Cactoblastis cactorum*), and continued beyond June 1999 (data not obtained).

TABLE 4. Presence of Black Rats (*Rattus rattus*) on offshore cays, inland lake cays, and North Point of San Salvador Island, The Bahamas (see Fig. 1). Distance provided between each cay and the main island of San Salvador, which serves as a reservoir for invasion. Sherman rodent live trap results from 1999–2001. S = sighting; C = carcass; T = trap capture.

| Location	Area (ha)	Distance to island (m)	Water body	Trap nights	Trap captures	Rats confirmed
Catto	1.9	2,050	Ocean	16	0	
Cut	3.0	30	Ocean	40	0	S (2012)
Gaulin	1.6	2,000	Ocean	11	0	
Goulding	2.9	590	Ocean	15	0	
Green	5.1	1,950	Ocean	11	0	
Guana	1.6	180	Lake	4	0	S (before 1994)
High	13.4	580	Ocean	42	0	C (1995)
Low	10.8	1,010	Ocean	14	2	S (1994–2000), T (1999–2000)
Manhead	3.3	280	Ocean	66	0	
Middle	2.7	960	Ocean	15	0	
Nancy	1.0	350	Ocean	10	0	
North Point	—	—	—	22	3	T (2001)
Pigeon Cay	7.8	900	Lake	44	0	S (1995)

(range, 74.7–96.4%) in the number of pads and 49.1% loss ± (SE) 20.0% (range, 16.7–85.0%) in maximum plant height (*n* = 3 plots, Kendall's W = 0.73 and 0.59, respectively; both values represent a large effect size; Fig. 3). Further loss occurred in subsequent years (not quantified), resulting in persistent patches of stunted cacti mostly < 30 cm in height. Large cactus patches experienced dramatic reductions during this time also on Manhead and Gaulin Cays, and on two other cays devoid of iguanas (Catto and Middle Cays). Widespread cactus patches were lost on High Cay and Low Cay prior to this period, where extensive areas of small cactus plants remained as of 2013. We have not attempted to control this moth.

Our trapping efforts for rats yielded positive results at two locations (Low Cay and North Point on the main island adjacent to Cut Cay), and negative results at all other locations (Table 4). We observed up to three rats on Low Cay during daylight hours (Hayes et al. 1995) on virtually every trip to the island through 2000. Rats also were reported on Guana Cay in the inland lake (Don Gerace, pers. comm.); a carcass was found on High Cay in 1995 (John Iverson, pers. comm.; Auffenberg, cited in Hayes et al. 1995, thought rats might have caused extirpation of these iguanas); and we observed a rat on Pigeon Cay in 1995 (Table 4). Stepwise binomial logistic regression yielded a significant model (χ^2 = 10.04, df = 2, *P* = 0.007, Nagelkerke R^2 = 0.76) for confirmed rat presence (last column of Table 4), which predicted the presence of rats on a given cay with 91.7% success (7/7 cays with rats absent; 4/5 cays with rats present). However, neither of the two predictors in the final model attained independent significance (cay size: *P* = 0.17; water body type: *P* = 0.999), due in part to limited statistical power. The odds ratio for cay size (1.85, 95% CI = 0.76–4.51) exceeded 1.0, which suggested that large cays were more likely to have a confirmed rat presence than small cays. Rats were documented on both of the inland lake cays, but only on three of the 10 offshore cays. Distance from the main island was excluded from the final stepwise model.

Our attempt to eradicate the rats on Low Cay in 1999 failed, but we believe we succeeded with the modified bait system in 2000, as described in Hayes et al. (2004). Subsequent trapping effort in 2000 (37 trap-nights) and absence of rat sightings during daylight in the years since suggest the continued absence of rats on Low Cay through 2013.

Prior to translocating iguanas to Cut Cay, we failed to catch rats there despite relatively intense trapping (40 trap nights). This assumed absence of rats contrasted sharply with trap success on North Point on the adjacent portion of the main island (three captures in 22 trap nights; Table 4). However, we sighted a rat on Cut Cay during an iguana survey in May 2012.

Body size and translocation.—In November 2000, we observed fresh iguana tracks on the Club Med grounds during an overnight visit, and were told by employees that they had colored glass beads affixed to their nuchal crests (Hayes et al. 2000, 2004). In October 2002, we observed five adult iguanas and one subadult, confirming that breeding took place, as reported to us by the staff. The resort reportedly removed all stray cats and dogs, and poisoned the rats; we encountered one rat that was obviously incapacitated and dying. We captured three unmarked iguanas (two males, one female), which we recognized beforehand as being exceptionally massive, and the data confirmed that they were, indeed, substantially larger than any iguana measured previously on Green Cay (*n* = 92; Fig. 4). The two males from Club Med (34.8 and 36.0 cm SVL) were significantly larger than the 39 adult males from Green Cay captured between 1993 and 1999 (t_{39} = 5.28, two-tailed *P* < 0.001), and the single female from Club Med (29.5 cm SVL) was significantly larger than the 36 adult females from Green Cay (t_{35} = 16.50, two-tailed *P* < 0.001). Although some supplemental feeding occurred (Janet Storr, pers. comm.), the iguanas we found were skittish, wary of our approach, and mostly ignored the food items we tossed to them to facilitate capture. We
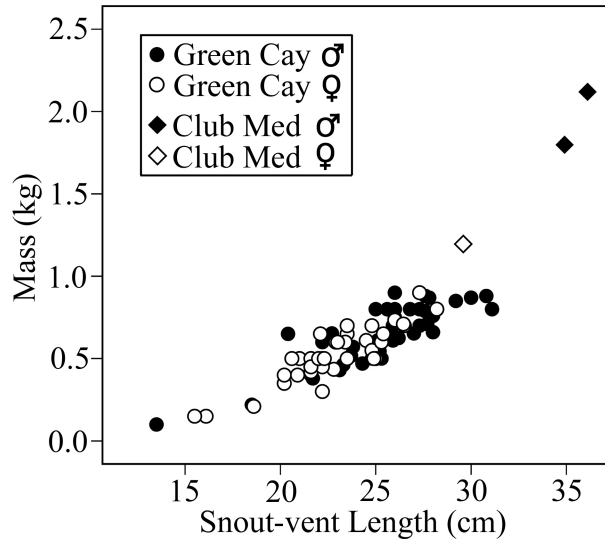
FIGURE 4. Body sizes of male and female San Salvador Rock Iguanas, illustrating the substantially larger size of specimens translocated to the San Salvador Club Med Resort (*n* = 1 ♀, 2 ♂♂), presumably in 2000 or earlier and measured in October 2002, compared to those of their source population on Green Cay (*n* = 92).

obtained permission to return to Club Med again in June 2007, when we managed to locate (but not capture) only two large adult iguanas despite extensive searching. We were told that a few others were still present, but that no more breeding was occurring. The population subsequently dwindled to the point that none remained as of 2012, according to Club Med staff who declined to give us permission to enter the property. At least two iguanas escaped over the fence and were killed by vehicles on the main highway (Janet Storr, pers. comm.).

We translocated adult iguanas (6 ♂♂, 8 ♀♀) from Green Cay to Cut Cay in February 2005. No hatchlings or subadults were observed during visits by us in June 2006, June 2007, January 2012, May 2012, and June 2013. During the May 2012 survey, we sighted 12 of the 14 translocated adult iguanas. The four iguanas for which we had unobstructed views (others were in the shrubs) appeared to be similar in size to those on Green Cay.

DISCUSSION

Our standardized population surveys have provided us with useful data to evaluate conservation threats and management actions for San Salvador Rock Iguanas. Unfortunately, population estimation remains imprecise, as estimates can be affected by numerous environmental and methodological factors. One key concern is the proportion of iguanas detected, which can substantially affect estimates via visual encounter surveys and distance transects (Hayes and Carter 2000; Lovich et al. 2012; Burton and Rivera-Milán 2014). Our data suggest that detectability is similar for males and females, but that time of day can have an influence, with iguanas generally becoming less active at midday, resulting in reduced detections during surveys. Similar patterns of bimodal activity, with higher levels during morning and evening, have been reported for *Cyclura carinata* (Iverson 1979), *Cyclura rileyi nuchalis* (Thornton 2000), *Cyclura cychlura* (Wilcox et al. 1973), and *Cyclura nubila caymanensis* (Carey 1966), however, *Cyclura lewisi* concentrated activity in the midday during the fall and spread out activity over the entire day during the summer (Goodman 2007). Although time-intensive and costly, our data suggest that mark-resighting estimates offer greater precision (coefficient of variation = 13.7) than count data (25.5), and can overcome problems that result from variation in detectability. By comparison, Burton and Rivera-Milán (2014) reported coefficients of 10 to 15 for distance sampling and repeated counts of *C. lewisi* on Grand Cayman. However, the accuracy of mark-resighting becomes limited if there is bias among representation of different age groups, such that separate estimates may be required of each size class. Our data further indicate that juvenile iguanas are more difficult to detect and to survey reliably than other age groups, as evidenced by the relatively high CV values from our surveys (82.7 versus 34.8 for subadults and 26.1 for adults). Collectively, these difficulties prompted us to report population estimates derived from counts as a range of values. Although mean detectability of iguanas on Green Cay was 0.33, we suspect this value was low because of the frequent nature of surveys (up to three per day) and ongoing capture efforts, which probably caused the previously captured iguanas to become more wary. Additional mark-resighting surveys of six *C. rileyi* ssp. populations, which allowed for more time after marking, provided detectability values of up to 0.55 (Hayes et al. 2004). Similar repeated surveys of marked *C. rileyi nuchalis* in the Acklins Bight averaged 0.47 in the morning, 0.24 at midday, and 0.44 during the evening (Thornton 2000). We suggest that detectability of 0.33–0.50 is reasonable for our taxon and location. For comparison, Burton and Rivera-Milán (2014) reported a mean detectability of 0.57 for male and female *C. lewisi* combined. Differences in detectability among studies and populations could result not only from survey methods, but also from population or species differences in activity and habitat use.

Despite the limitations of our surveys, we now have meaningful baseline data, some of which has allowed us to identify trends of potential relevance. We have documented: (1) the negative impact of Hurricane Floyd in 1999 on iguana reproduction on Green Cay and the extirpation of an entire population on Gaulin Cay; (2) obvious differences in age class representation and recruitment among the populations, which presumably relate to food availability, rat presence, and/or disease (Hayes et al. 2004); (3) the absence of a demographic or population response to rat eradication on Low Cay; and

(4) general stability among the populations sampled most frequently, with the possible exception of an apparent recent decline on Low Cay. Some populations have not been surveyed in more than a decade, which needs to be addressed. Although we lack quantitative data, the vegetation on Green Cay, which hosts the largest iguana population, is steadily deteriorating, presumably due to soil loss from catastrophic storms and this merits continued monitoring and further study.

Iguana populations on San Salvador appear to be impacted to some extent by invasive species, particularly the Cactus Moths and Black Rats. Although Australian Pines have the potential to become established and destroy habitat on a number of cays that host iguanas (Hayes et al. this volume), seedlings are easily detected and can be readily managed, as we demonstrated on Low Cay. Unfortunately, large stands exist on the main island (Rodgers 2005; Rodgers and Gamble 2008), which unless controlled will remain a source of more colonizing seedlings in the future.

Our survey of Prickly Pear Cacti stands on Green Cay illustrates the rapid devastation of formerly extensive cactus stands by the larvae of the Cactus Moths. When *Opuntia* cacti are available, the iguanas associate with and feed preferentially on it (Cyril 2001). We suspect that cactus loss has reduced the carrying capacity of iguanas on a number of cays, including Green Cay. Our previous analysis of morphological measures suggested that iguanas prior to (or during) the cactus decline on Green Cay (1993–1995) had comparatively more body mass than after the cactus decline (1998–1999), and this was reinforced by the absence of a similar trend in two populations unaffected by cactus loss (Goulding Cay and Pigeon Cay; Hayes et al. 2004). However, Hurricanes Erin in 1995 and Lili in 1996 may have impacted Green Cay's vegetation to a greater extent than the other cays, offering an alternative explanation. Iguana body condition has been linked to habitat quality in iguanas elsewhere (Pasachnik et al. 2012a, b; Pasachnik 2013). Our iguana population surveys on Green Cay do not suggest an ongoing population decline, but iguanas there have become accustomed to frequent human visitation (and occasional feeding), and detectability may have increased in the past decade (e.g., Hines 2011; Knapp et al. 2013; Smith and Iverson this volume), resulting in greater population estimates. We have not made an effort to control the moth, though new developments suggest feasibility through pheromonal attraction, release of sterile individuals, and fungal, bacterial, parasitoid, or nematode biological control agents (Zimmerman et al. 2004; Hight et al. 2005; Heath et al. 2006).

Accumulating evidence suggests that rats can also impact the iguanas. Populations coexisting with rats, for example, tend to have reduced population density (Hayes et al. 2004) and a greater frequency of tail injuries and tail furcations resulting from rodent bites (Carter and Hayes

2004; Hayes et al. 2012). Unfortunately, our trapping data underscore the difficulty of documenting the presence of rats. When a presumably dense rat population exists, capture rates are fairly high, as occurred on Low Cay and on North Point (14.3% and 13.6% per trap-night, respectively; Table 2), on the main island of San Salvador (16.2–20.5%, Hall and Dougherty 2003), and on Sandy Cay in the Exumas (9.3%, Hayes et al. this volume). Although rats have been documented on High Cay and Pigeon Cay, the populations there appear to be either unstable or of low density, as suggested by an absence of captures despite fairly intensive trapping effort (Table 4). Temporary rat infestations of small islands may be frequent, especially for those having limited water and unfavorable vegetation (Palmer and Pons 2001). The logistic regression results hinted that larger cays and those within the inland lake were more likely to have rat presence confirmed. Distance from the main island appears to have less predictive power, which may result, in part, from the relative strength of water currents that separate the land masses (the northern cays are amidst stronger currents than the southern cays). A similar analysis of rat presence on 14 islets in Spain's Cabrera Archipelago Maritime-Terrestrial National Park determined that plant composition rather than geographic variables (island area, height, storm exposure) best explained the presence of rats (Palmer and Pons 2001).

We expected a vigorous demographic and population response following rat eradication on Low Cay, but this has not happened. Undetected rat recolonization remains a possible explanation. Although the iguana population is not particularly dense on Low Cay, perhaps it has been at carrying capacity, possibly reduced following the massive loss of cactus stands on the cay (a large area of very small cactus plants remains), rendering substantial recruitment unlikely. Nesting habitat abounds, and large numbers of hatchlings have been observed in the autumn, so reproductive capacity is unlikely to be the limiting factor. High levels of territoriality, particularly by adult males, may contribute to the low population density (Pérez-Buitrago et al. 2010), presumably by excluding young animals from suitable habitat, and therefore subjecting them to greater mortality.

The two translocation failures add to our understanding of what it takes to successfully establish a new population. The Club Med Resort provided a lush environment for the herbivorous iguanas, and for a time they appeared to be breeding and successfully established. We do not know what went wrong. We have no information on the exact number or sex ratio of iguanas originally translocated to the property, though we observed six individuals on one visit, captured individuals of both sexes, and observed one young animal. Despite the suitable habitat, some threats still remained, including auto collision, dog attack, secondary poisoning from consuming poisoned rodents (compare Luther et al. 2012), and disease. More surprising, however,

was the complete failure of the Cut Cay translocation in 2005. We expected substantial population growth in the eight subsequent years, similar to that seen for other translocated populations of *Cyclura* (reviewed by Knapp and Hudson 2004; Iverson et al. 2006, this volume). Only one explanation stands out, as the habitat is exceptional, the iguanas on the cay are clearly thriving, and suitable nesting habitat abounds. Despite our initial confidence that rats were absent from the Cay, we sighted one in daylight in 2012. We therefore suspect that rat predation on nests and hatchlings was responsible for the failure. Although iguanas in some areas coexist with rats, we suspect that small iguana populations are much more vulnerable due to the higher ratio of rats to their prey (Hayes et al. 2012). One possibility to help this iguana population to become viable would be to increase iguana density on Cut Cay to the point where nests and eggs would saturate the predatory capacity of rats. Another option, of course, would be to eradicate the rats.

Comparison of body size of translocated iguanas to the source population allowed us to test the hypothesis that translocated iguanas on Club Med and Cut Cay would attain a larger body size (length and mass) due to a richer food source (at Club Med), less food competition (both locations), and/or fewer social interactions that would otherwise constrain time devoted to feeding (both populations), as discussed elsewhere (Carter and Hayes 2004; Hayes et al. this volume; Iverson et al. this volume). The contrast between the translocated populations was unambiguous: iguanas at Club Med became massive compared to those on Green Cay (Fig. 4), whereas those translocated to Cut Cay showed no obvious increase in body size. This finding suggests that nutrition may exert a more profound influence on adult body size of *C. rileyi rileyi* than does population density. Iguanas of both translocated populations were at low density, but those at Club Med had access to the rich, exotic foliage maintained by the grounds crew, and may have benefitted from supplemental feeding by humans, whereas those on Cut Cay persisted on natural vegetation similar to that of the

Figure 5. Exhibition and education enclosure of the San Salvador Iguana Conservation Centre, a headstarting facility partially completed and opened at the Gerace Research Centre in 2012. (Photographed by William K. Hayes).

source cay. Other explanations may also exist; for example, the Club Med location is on the leeward side of the main island, and may offer thermoregulatory advantages for digestive efficiency (Tracy et al. 2005).

At present, no acceptable options for translocation exist to augment the natural population of *C. rileyi rileyi*. Several cays currently uninhabited by iguanas are too small to support a sizeable population (e.g., Catto, Gaulin, Middle, Nancy), and others of questionable quality in the inland lake have ownership issues and, very likely, rats. High Cay would clearly be the best translocation site due to its large size and superb habitat, but the owners have firmly declined permission for such an undertaking. Translocating iguanas to the main island might work, but only to remote areas, preferably with limited human access, devoid of feral dogs, cats, and hogs. Two locations may be feasible: the peninsula east of Pigeon Creek, and the western shore of Green's Bay, where Goulding Cay is located (Fig. 1). Natural features limit access to those locations, and fencing could be used strategically to secure a predator-free area and to contain the iguanas. Some adult iguanas do live on the main island, but they are widely dispersed and rarely encountered, suggesting a minimally viable population subject to non-native predators and vehicle collisions. Dozens, if not hundreds, of neonates currently die each year on the inhabited cays because the populations are at carrying capacity and support only limited recruitment of young into the adult population. These hatchlings could be translocated to the main island, but they are much less likely to survive than headstarted or adult iguanas (e.g., Pérez-Buitrago and Sabat 2007; Knapp et al. 2010; see also Iverson 2007), yet we are reluctant to remove a large number of adults from the cays. Given these considerations, we have sought to implement a head-starting program for many years.

We opened a small headstarting facility, the San Salvador Iguana Conservation Centre, in May 2012. This facility initially included a large (9 x 6 m), block-walled exhibition and education enclosure designed to house up to 3 adult ♂♂ and 5 adult ♀♀ iguanas that were transferred from Green Cay (Fig. 5). We planned on constructing a number of smaller enclosures later. Due to lack of soil stability and moisture retention associated with root growth (extensive root penetration of the soil can take many months), we were uncertain whether nesting would be successful in the large enclosure the first year, but one young iguana emerged in October 2012. With several setbacks we were unable to house additional iguanas in smaller pens through 2014. Gravid females dug burrows and deposited eggs in the exhibition and education enclosure in 2013 and 2014, but no hatchlings appeared those years.

The future of this facility remains unclear. The original concept sought to achieve three purposes: (1) to enhance awareness of the iguanas and promote environmental

education for island inhabitants and guests of the Gerace Research Centre; (2) to stimulate renewed efforts to protect critical habitats supporting diverse marine and terrestrial wildlife, including a number of threatened and endangered species; and (3) to boost the wild iguana population through release of headstarted individuals.

At this point, we appear to be succeeding with the first two goals. Those who visit the facility express considerable appreciation for seeing the animals. Although we plan to keep the facility small, its expansion with additional smaller cages and animals, a rock pathway with natural vegetation, and additional information signage will enhance the visitor experience. We anticipate that visits to the facility will ease some of the ecotourism pressure on Green Cay. Ecotourism, and supplemental feeding in particular, may have detrimental effects on rock iguanas (Hines 2011; Knapp et al. 2013; Smith and Iverson this volume). Perhaps most important, the Bahamas National Trust has now revived its decade-old proposal to create a new national park that would protect the most critical iguana, seabird, and marine populations on San Salvador, tentatively including Green, Goulding, Manhead, and Pigeon Cays. The island residents appear to support this plan, and a prior education program that focused on the iguanas (Carter et al. 2005) has helped to pave the way. However, to make progress with the third goal, that of augmenting the wild iguana population, we need to finish construction of the smaller pens and stock them with hatchling iguanas taken from the cays. Some of these headstarted iguanas could then be released to create new populations on remote parts of the main island, as described above, released on Cut Cay to supplement the low-density population there, or taken to a cay that might eventually become available for translocation. Others can be retained to form the core of a small breeding group. A wild population could conceivably be managed in anthropogenic habitat, such as Club Med or the Gerace Research Centre, with proper predator control and roadway signs to urge caution. Headstarting has benefited recovery efforts for many endangered iguanas, including the Anegada Iguana (Cyclura pinguis, Bradley and Gerber 2005), Cuban Iguana (Cyclura nubila nubila, Alberts et al. 2004), Grand Cayman Blue Iguana (Cyclura lewisi, Knapp and Hudson 2004), Jamaican Iguana (Cyclura collei, Wilson et al. 2004), Mona Island Iguana (Cyclura stejnegeri, Garcia et al. 2007; Pérez-Buitrago et al. 2008), and Galápagos Land Iguana (Conolophus subcristatus, Cayot et al. 1994). This iguana could benefit as well.

In summary, San Salvador Rock Iguanas face an uncertain future. Our population surveys over the course of two decades give us a tool and a history that will be integral to further conservation and management efforts on behalf of this taxon. Unfortunately, the small and isolated iguana populations have been plagued by catastrophic storms and invasive species, such that there is no tangible hope of population expansion without serious intervention. We see opportunity to benefit the 600 or fewer remaining iguanas through more aggressive efforts at invasive species control, translocation with limited options, and a headstarting program that can not only augment the wild population, but also enhance iguana awareness, conservation education, and the pursuit of formal habitat protection. Management would also benefit from detailed genetic studies at the population and subspecies levels. Inbreeding depression may have contributed, for example, to the extirpation of populations in recent decades (e.g., Barn Cay, High Cay), and some of the extant populations may be prone as well.

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