Department of Environment
Cayman Islands Government

Population assessment of the invasive Green Iguana (*Iguana iguana*) in Grand Cayman

Terrestrial Resources Unit, Department of Environment 17 September 2025

Vaughn Bodden, TayVanis Oyog

Background

The Green Iguana (*Iguana iguana*) is an invasive species in the Cayman Islands that has caused large scale defoliation of native plants, disease introduction to the endemic Grand Cayman Blue Iguana (*Cyclura lewisi*), hybridization with endemic Sister Islands Rock Iguana (*Cyclura nubila caymanensis*), along with other nuisance affects in the built environment iguanas (Moss et al., 2018; Rivera-Milán & Haakonsson, 2020; Conley et al., 2021). The species is abundant and widespread in Grand Cayman but is only in the establishment stage in Little Cayman, and is in the established/ dispersal stage in Cayman Brac (Blackburn et al., 2011). The Grand Cayman population acts as the source for those invasions due to poor biosecurity practices for inter-island cargo. The Cayman Islands Green Iguana Culling Program, starting in October 2018, has removed over 1.6 million green iguanas from Grand Cayman to date; but eradication was assessed as unfeasible for this population. Instead ongoing adaptive management of the culling program is necessary to manage the population at low levels to reduce the negative impacts on the environment at a reasonable ongoing cost (Williams et al., 2009). To inform this adaptive management strategy, the Department of Environment surveys the Green Iguana population on Grand Cayman every year to estimate population size and distribution across the island. Here we present the results from the 2025 survey.

Field methods

This survey is designed to monitor trends in the population size over time, with planned surveys originally occurring every August since 2014. This most recent survey was conducted from 18 August – 02 September 2025. Between 124 – 213 points were surveyed each year with 165 points surveyed in 2025 (Figure 1). Each point is between 200 – 800 m from the nearest point, with points that are closer in distance typically being in off-road habitats. Each point is surveyed for six minutes searching the immediate area on foot and using range finder binoculars to scan the trees at a distance to search for Green Iguanas. A subset of transects (between 50 – 75% each survey) were repeated to increase the number of detections and to estimate repeated count parameters. Point visits and repeats were staggered to different parts of the island throughout the survey period. Observers recorded the distance

to each Green Iguana detection made using range finders or a measuring tool for distances < 10 meters, the age of the iguana (hatchling, born in the survey year; sub-adult, one to three years old; and adults, more than three years old), along with other detection and site covariates.

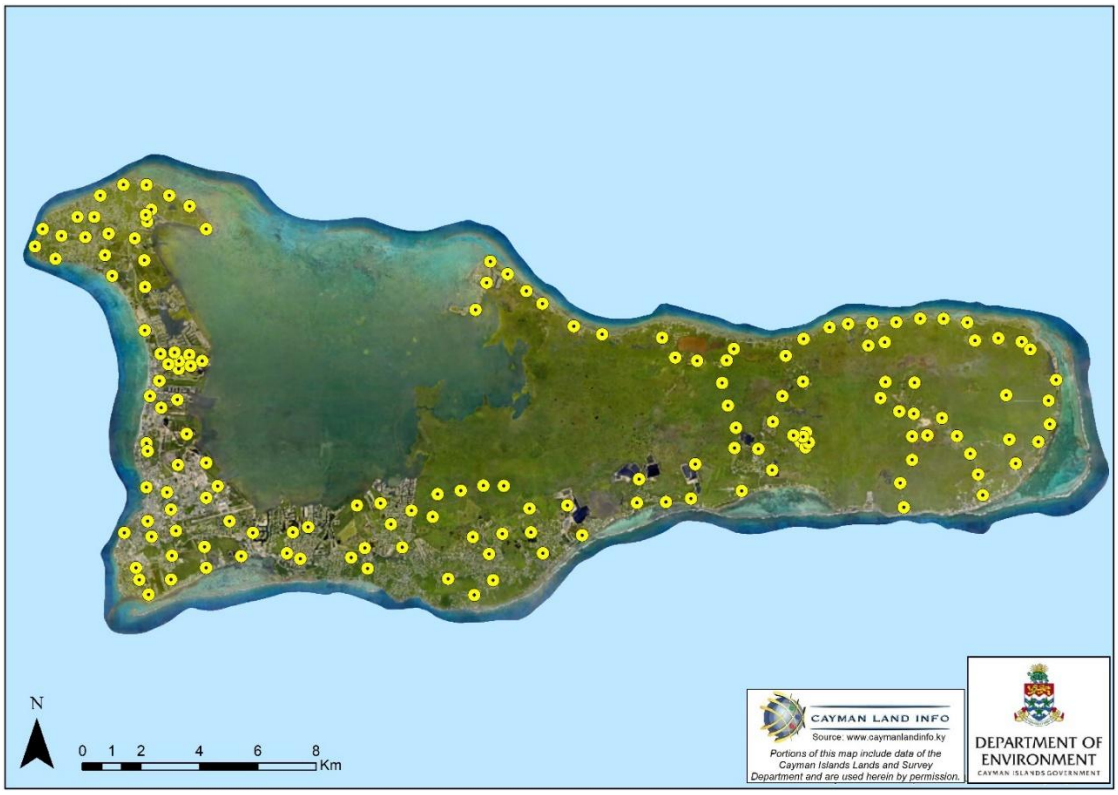

Figure 1. Map of points surveyed in 2025.

Statistical methods overview

Population estimates were made using distance sampling & N-mixture models that account for detection probability (Buckland et al., 2001). Population estimates were made for adults and sub-adults combined and hatchlings separately due to differences in behavior and detection patterns. These two estimates were added together to get the total population estimates with variance propagated through non-parametric bootstrapping. Distribution estimates were made using Integrated Nested Laplace Approximation (INLA) models that account spatial and temporal correlation along with covariate effects (Lingren et al., 2011). All analyses were done in the R programming language using the 'Distance', 'unmarked', and 'R-INLA' libraries (Fiske & Chandler, 2011; Lindgren & Rue, 2015; Miller et al., 2017; R Core Team, 2021). For detailed statistical methods, model diagnostics, and additional parameter estimate tables see the Appendix.

Population size estimates

The estimated number of Green Iguana **adults and sub-adults** in Grand Cayman increased significantly in 2025 when compared to the 2024, and the 2020 - 2022 estimates (Z-test: p-values < 0.05), is similar to the 2023, 2019, and 2014 estimates (z-test: p-values > 0.05), and is significantly lower than the 2015 – 2018 estimates (z-test: p-value < 0.005) (Figure 2; Table 1).

The estimated number of Green Iguana **hatchlings** in Grand Cayman is statistically similar to 2014 – 2017 and 2019 – 2024 estimates (Z-test: P-values > 0.15) and is only significantly lower than the estimate in 2018 (Z-test: p-value < 0.001) (Table 1). This lack of statistical difference between the years highlights the high uncertainty in the hatchling estimates.

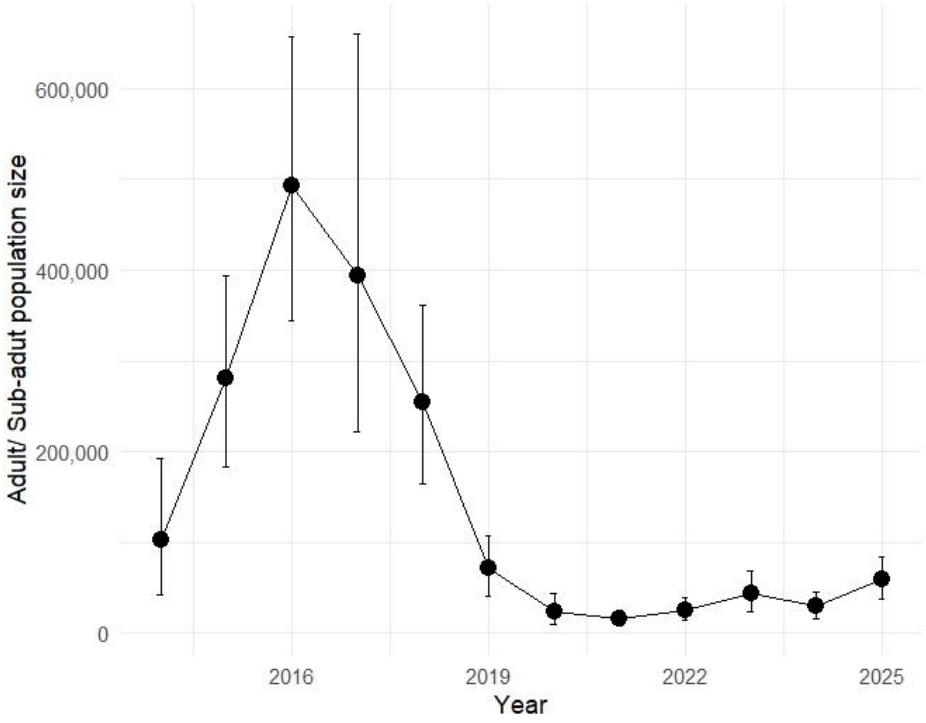

Figure 2. Population size estimates for the **Adult/ Sub-adult** age classes combined 2014 – 2025; points are mean estimates with 95% Confidence interval error bars.

Table 2. Population estimates of adults and sub-adults combined and hatchlings from the Green Iguana survey 2014 - 2025. 95% Confidence Intervals were rounded to the nearest thousand.

	Adults & Sub-adults				Hatchlings		
Year	**Mean**	**95% Confidence Intervals**	**Coefficient of Variation**		**Mean**	**95% Confidence Intervals**	**Coefficient of Variation**
2014	102,311	43,000 – 192,000	0.40		56,281	23,000 – 138,000	0.48
2015	280,363	183,000 – 393,000	0.19		39,822	17,000 – 96,000	0.47
2016	492,579	344,000 – 657,000	0.17		196,476	95,000– 346,000	0.34
2017	394,394	222,000 – 660,000	0.32		172,677	83,000 – 288,000	0.30
2018	254,557	165,000 – 361,000	0.20		385,966	250,000 – 543,000	0.20
2019	71,329	41,000 – 107,000	0.24		152,856	62,000 – 311,000	0.44
2020	24,433	10,000 – 25,000	0.39		15,514	4,000 – 54,000	0.70
2021	16,028	9,000 – 25,000	0.26		34,288	7,000 – 158,000	0.92
2022	25,438	14,000 – 40,000	0.26		53,757	19,000 – 150,000	0.56
2023	44,094	24,000 – 69,000	0.28		134,395	44,000 – 277,000	0.48
2024	29,784	17,000 – 46,000	0.25		147,004	54,000 – 278,000	0.42
2025	58,712	38,000 – 84,000	0.20		84,836	21,000 – 198,000	0.65

Table 3. Total population (adult/sub-adult & hatchlings combined) estimates from the Green Iguana survey 2014 - 2025. 95% Confidence Intervals were rounded to the nearest thousand.

Year	Mean	95% Confidence Intervals	Coefficient of variation
2014	158,822	87,000 – 270,000	0.30
2015	320,710	226,000 – 445,000	0.17
2016	688,122	506,000 – 909,000	0.15
2017	566,333	360,000 – 861,000	0.23
2018	640,893	482,000 – 833,000	0.14
2019	224,303	130,000 – 382,000	0.29
2020	39,895	20,000 – 74,000	0.35
2021	50,224	19,000 – 134,000	0.64
2022	79,394	39,000 – 157,000	0.38
2023	178,537	94,000 – 330,000	0.34
2024	176,384	91,000 – 323,000	0.34
2025	143,689	79,000 – 278,000	0.36

The estimated total number of Green Iguanas (i.e. adults/ sub-adults & hatchlings combined) in 2025 is statistically similar to estimates from 2021 – 2024, 2019, and 2014 (Z-test: p-value > 0.1), is marginally higher than the 2020 estimate (Z-test: p-value = 0.05), and is significantly smaller than 2015 – 2018 estimates (Z-test: p-value < 0.005) (Figure 3; Table 2).

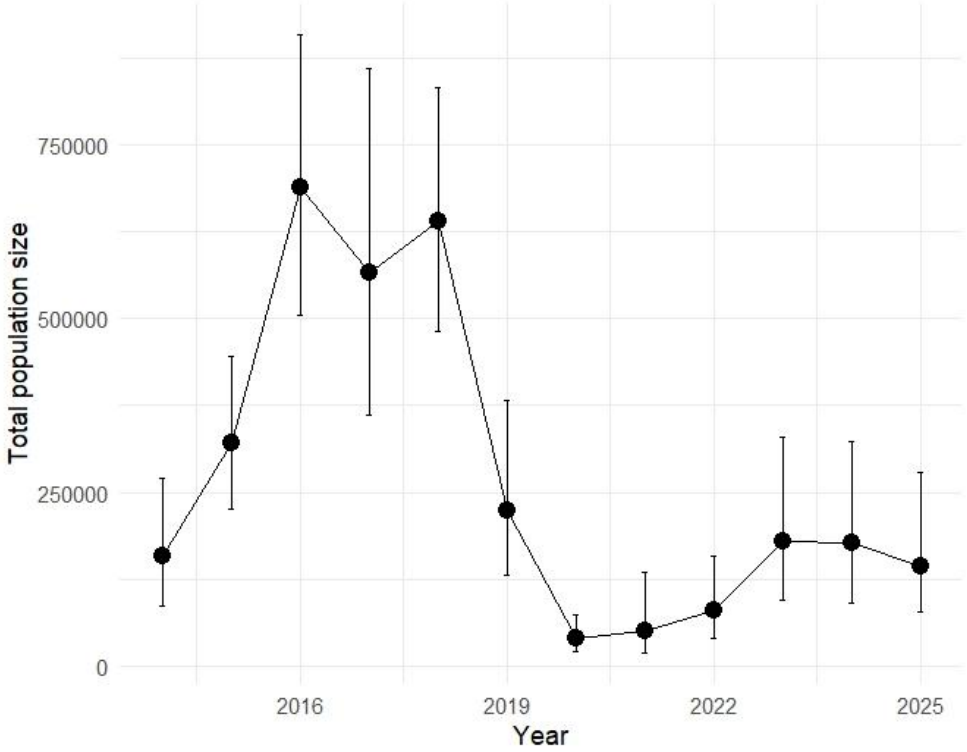

Figure 3. Total population size estimate for Green Iguanas in Grand Cayman 2014 – 2025; points are mean estimates with 95% Confidence interval error bars.

Population distribution estimates

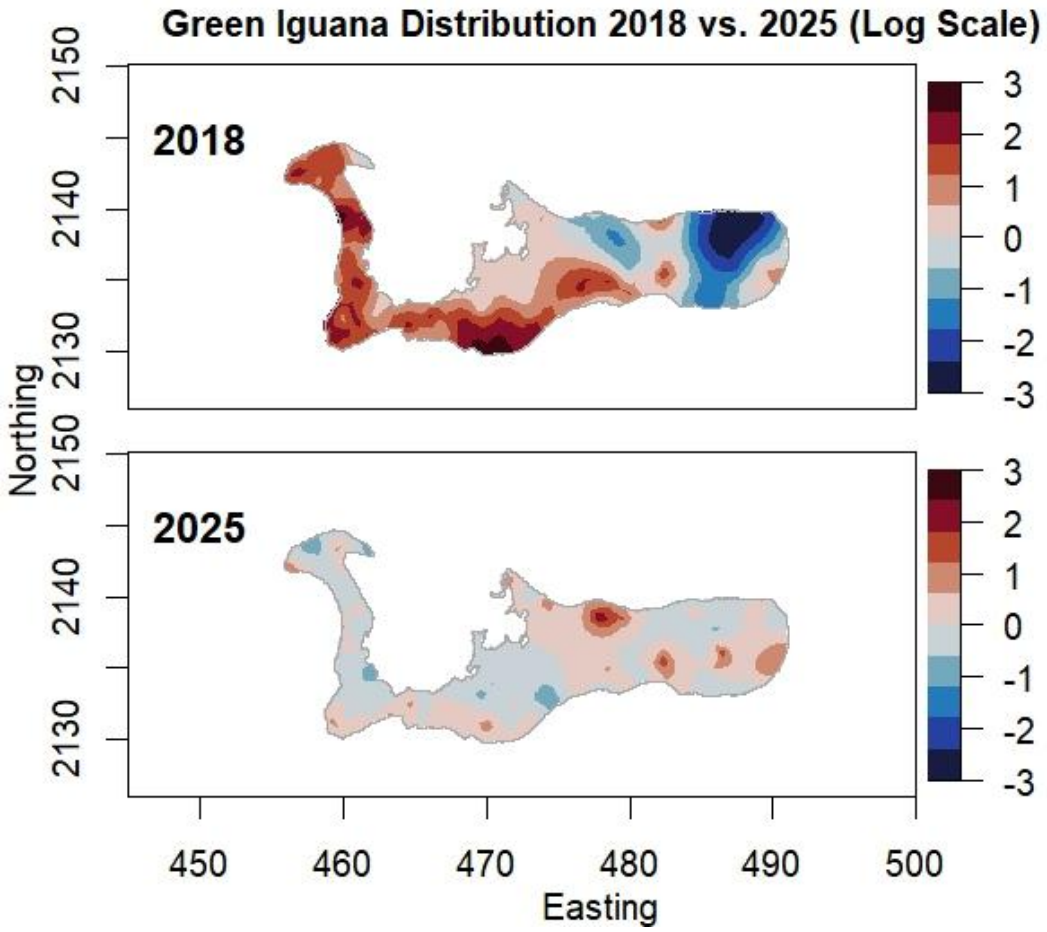

Figure 4. Green Iguana distribution 2018 versus 2025; visualized on log scale, areas in red shades are increasingly abundant while areas in blue shades have little to no iguanas.

The distribution trends overtime shows the large-scale removal of the Green Iguanas from the western half of the island after the start of the island-wide cull (Oct. 2018 start date) and relative persistence of green iguanas in the eastern half of the island where relative abundance is now highest (Figure 4). This is corroborated by the urban covariate association over time (Figure 5). The eastern half of the island is generally less man-modified and supports more primary habitats. The distribution shift and more localized spatial structure in the maps represent the change in spatial correlation estimates 2019 onwards with a much lower spatial range estimate (Figure 6). In 2025, the current Green Iguana population distribution is widespread yet unevenly distributed across the island with major hotspots in the North Side and East End districts, with additional high abundance areas along the south coast in the South Sound to Bodden Town area and smaller isolated patches in West Bay (Figure 4).

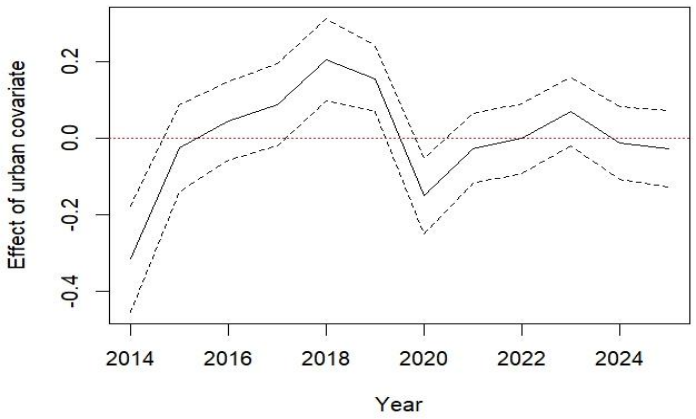

Figure 5. Effect of urban covariate over time (2014 – 2025).

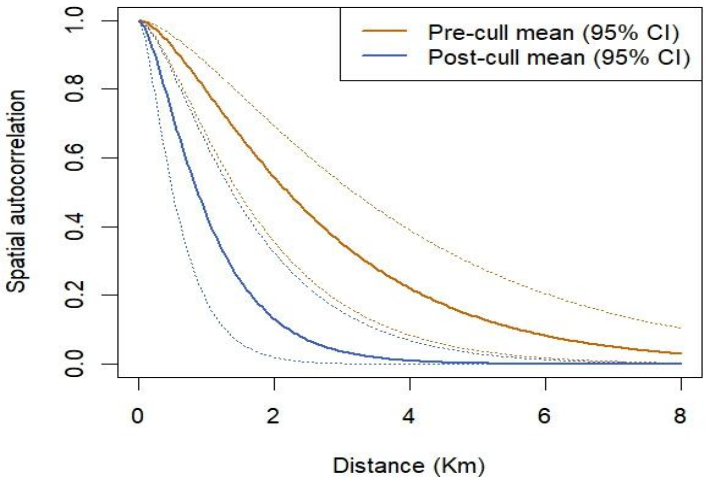

Figure 6. Spatial autocorrelation pre- (2014 – 2018) and post-cull (2019 – 2025).

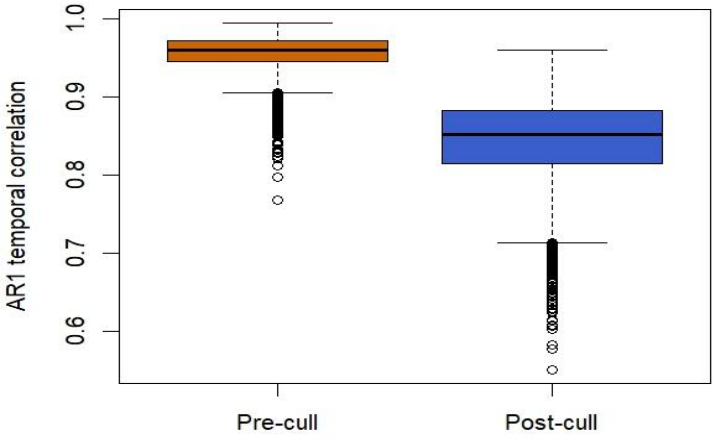

Figure 7. Temporal autocorrelation pre- (2014 – 2018) and post-cull (2019 – 2025).

Conclusions and recommendations

The number of adult & sub-adult green iguanas estimated in 2025 is the highest since the low point in 2020 and continues the trend observed in 2023 of significantly increasing population estimates for the age-classes (Table 3). The hatchling estimates show no statistical difference across most of the time-series which highlights the need adapt the survey design to obtain more precise estimates. The high variance in the hatchling estimate is propagated into the total population estimates which also limits its applicability for year-to-year adaptive management. The most reliable estimates from these models of population size are the adult and sub-adult age class combined estimates; these have the precision to detect year-to-year changes. The ability to detect change is a precondition for the use of these estimates in adaptive management and forecasting (Williams et al., 2009).

The increased incentives placed on the adult age-class in 2022 (increasing price from $5 to $8 - $10 dependent on increasing size categories) so far appear to have not been effective in reducing the overall adult and sub-adult population numbers. However, due to the mismatch in age-class categories between this survey and the culling station, it could be that the incentives were successful in reducing the largest animals, but it is now the young adult (three to five year old) iguanas that are increasing. Indeed, in recent survey years the adults typically detected are younger (three to five years old) in contrast to the pre-cull years when older and larger iguanas (eight years old and above; that have increased incentives) were more common. Aligning the age-class categories for the culling incentives and the survey would increase the usefulness of these estimates for adaptive management of culler incentives. The increased incentive program should be treated as an experiment, and we should evaluate its success or failure within the next two years, five years total since the incentive structure was initiated, and adapt if necessary (Williams et al., 2009). For example, a demographic study of Green Iguanas in Panama by Bock et al. (2016) found that survival in the first two to three years of life (i.e. sub-adult and hatchlings in this study) had the largest influence on demographic trends for the total population. This is the opposite of the age-classes we currently incentive with increased bounties. Green Iguanas in Panama have many more natural predators than they do in Grand Cayman so these results may not be directly applicable but if continued increases are observed in the adult/ sub-adult age class in the next two years it is worth testing this hypothesis.

The current distribution of Green Iguanas highlights the need for increased effort in the less-developed eastern districts where the relative abundance is the highest and where there are the most environmentally sensitive areas, such as the Blue Iguana release sites in the Salina Reserve and Collier's Wilderness Reserve in the East End (Figure 4; Burton & Rivera-Milán, 2014). The current culling-induced distribution is likely being driven by the relatively lower effort per area in the eastern half of the island due to lack of access to interior habitats for cullers and the lower number of registered cullers operating in these districts. The threat to the environmentally sensitive areas is supported by wildlife camera surveys done in the Salina Reserve and Collier's Wilderness Reserve where Green Iguanas are seen within protected area and interacting directly with endemic Blue Iguanas, increasing the risk of disease transmission and hybridization (Moss et al., 2018; Conley et al., 2021).

Control within less-accessible primary habitats is difficult with the current set up of independent cullers and open-cull. But it is worth pushing for higher culler registration and participation in the Eastern districts to test whether increasing the number of independent cullers there will increase overall removal effort enough on the periphery to ultimately keep the Green Iguana population numbers suppressed in these areas over time. For areas that are particularly difficult to reach and/or retrieve iguanas from, such as private quarry areas, protected pond/ wetland areas, and other protected or environmentally sensitive habitats, targeted teams of two to three people with access permissions, necessary equipment, and adequate incentive should be used to prevent these areas from becoming refuge for the Green Iguana population in Grand Cayman.

Population management recommendations:

- Increase culler registration and participation in the Eastern Districts (i.e. East End and North Side). This can be done, for example, via registration drives, targeted advertising, and local cull station. Monitor efforts for participation and effectiveness.
- Increase culler registration and participation overall. Enhanced advertising and reducing barriers to registering and maintaining a green iguana culling license.
- Teams approach for environmentally sensitive areas (e.g. Botanic Park, Protected Ponds) these areas are consistently abundant with Green Iguanas that are difficult to retrieve to turn in for bounty. Incentivize small teams (e.g. 2 – 3 people) or Department of Environment staff to target these areas periodically for control.
- Increase biosecurity at the seaports to prevent further incursions of Green Iguanas into and between the Cayman Islands. Particularly targeting inter-island biosecurity to prevent further supplementing the established and growing populations of Green Iguanas on Cayman Brac and Little Cayman.

Survey design conclusions and recommendations

The current timing of the survey is not ideal for detecting hatchlings as we typically survey within the month of August but the hatchling emergence doesn't start in earnest until the first week of September and is variable dependent on precipitation in any given year. We have kept the August date as it was what has been done in the past but with the relatively low population size and the very sparse hatchling detections each year (only 22 detections this year) there is a need to optimize this survey. In terms of survey date, a shift to the end of or post-hatchling emergence (i.e. November – January) theoretically should result in a higher number of hatchlings available for detection. This time is also after the peak in hatchling returns at the cull station so a survey during this time would represent the number of hatchlings that have survived the initial removal effort and are more likely to progress into sub-adults the following year (i.e. an estimate post-emergence removal peak would be more informative as to what to expect in the sub-adult age class the following year).

Overall effort is low for estimating the availability parameter and there is a need to increase the number of repeated points. The availability parameter is highly influential in the final estimates, often multiplying the distance sampling estimate by 1.5 to 4.5 times higher for adults/ sub-adults and 3 to 15 times higher for hatchlings (Table 2.). However, we only repeat a fraction of the total points once to estimate this parameter, resulting in availability estimates with relatively high standard errors which are propagated into the final estimates. A balance needs to be struck between the number of points surveyed, the number of repeats, and staff capacity to participate in the survey. We recommend two survey teams of two people for 10 – 12 days, repeating 100 - 125 points three to four times. Rising the overall effort from the 200 – 250 points visit range to 300 – 500 point visits.

It is important to isolate the effect of these changes to understand their impact and adjust accordingly before the next official survey period. For example, if the change in survey date to the November – January period results in a large increase in hatchling detections then we may only need to increase overall effort by 25 – 50% instead of increasing it by 100%. Optimizing the survey moving forward is necessary to use the hatchling estimate in adaptive management decisions.

Survey management recommendations:

- Trial a post-hatchling survey period (e.g. late November 2025) with 100 points repeated twice to test for increases in hatchling availability. Implement change in survey date for 2026 if trial is successful.
- Increase survey effort overall from 150 points with 50 – 75% visited twice (i.e. 200 – 250 point visits) to ~100 – 125 survey points surveyed 3 – 4 times (i.e. 300 – 500 point visits) done with two teams of two people. Aim for 10 – 12 days survey duration. Adjust overall increase in effort depending on results from trial for change in survey date.
- Plan survey personnel six months in advance to accommodate the increased effort.

Appendix

Detailed statistical methods

Abundance model:

Abundance was estimated for hatchlings separately from sub-adults and adults due to differences in behavior and detection patterns. For example, hatchlings are typically detected on the ground very close to the transect and are difficult to see at a distance while sub-adults and adults are more often in trees and can be seen at a distance with binoculars. The two separate estimates were added together for total population abundance with variances estimated using non-parametric bootstrapping.

The distance between the line transects and detected green iguanas was used to estimate the detection probability (P_d) and density using distance sampling methods (Buckland et al., 2001). Density was multiplied by the habitable area of 19,600 hectares in Grand Cayman which excluded roadways, buildings, and ponds. Repeated visits to transects were used to estimate availability given detection (P_{da}), or the temporary emigration of animals from the sampling area, using N-mixture models (REF). Availability was derived by dividing P_{da} by P_d to give P_a, we propagated the variance of the ratio of these two parameters using the delta method (Lye & Hirschberg, 2018). The availability estimate was used as a multiplier in the distance sampling model, which adjust the density estimate to account for the temporary emigration of animals during the survey period.

Detection distances were binned in intervals of 0 – 15, 16 – 30, 31 – 45, 45 – 60, 61 – 90, 91 – 120, 121 – 180, 181 – 240, 241 – 340, 341 + meters.

The distance sampling models were fit to each year individually given the changing surveyor teams of varying skill-levels over time and the large-scale intervention in the population in the latter half of the time-series. However, due to the relatively low number of repeated points each year and sparse detections for the hatchling age-class, the N-mixture model included all years together with the year as a covariate allowing for individual estimates each year and random effects on point ID to handle additional variance. Model selection was done using AIC scores and goodness-of-fit test. Differences in population estimate are compared using standard Z-test to test for significance.

Spatial model:

We used the count of iguanas at each point in a Bayesian INLA model to map the distribution of Green Iguanas across Grand Cayman while accounting for spatial x temporal correlation and covariate effects (Miller, 2012). For points that were visited twice we used the visit with the highest count in this model. Age-classes were not separated in this model therefore estimates are for the entire population. Model selection was done using conditional predictive ordinance (CPO) scores which is a leave-one-out procedure that measures the predictive performance of a model (Lindgren & Rue, 2015).

We tested covariate terms for linear, quadratic, and for time-varying effects in the model and included covariate terms based on whether their effects were meaningful (i.e. 95% confidence intervals do not span positive and negative effects in a large way), whether they contributed to the model's explanatory power, and whether they were ecologically relevant for the Green Iguanas distribution.

Between Oct. 2018 (start of the cull) and September 2019 (just after 2019 survey period) there were 874,188 green iguanas removed from the island. To understand the population distribution trends overall and how the removal effort has affected the spatial structure population, we tested the difference in spatial and temporal correlation by explicitly modelling both correlations for the pre- and post-cull periods (i.e. 2014 – 2018 and 2019 – 2025 periods, respectively) in a single model. This model approach was compared to a model without the split in correlation parameters to test for predictive performance, explanatory power, and ecological relevance. The spatial range estimates represent the distance range at which correlation is < 0.1. Temporal correlation was modelled as auto-regressive first-order correlation meaning it compares the distribution year to year to measure average correlation, years further apart can be less correlated.

Covariates for this model were derived from the Department of Environment's landcover layer based on the 2013 (2-meter resolution). The categorical landcover types were summarized into three covariates that describe attributes of each landcover type: moisture, urbanization, vegetation height. The landcover types were assigned values 0 – 5 to correspond with increasing values of each covariate. For example, primary habitats had an urbanization score of 0 and roads/ buildings had a score of 5; or dry shrubland and dry forest had a moisture score of 0 while ponds/ lagoons had a score of 5. In addition, a covariate for distance to coast, distance from roads/ buildings/ and density of roads/ buildings were generated. The 2013 imagery is the most recent imagery classified to landcover use categories by the Department of Environment; this means the association with recently man-modified habitats are not accounted for in this analysis. The raster maps and feature density were sampled as the average from within a 200-meter buffer from each point.

Table 4. Land-use categories and assigned covariate values used in the spatial model.

Land-use category	Vegetation Height	Urbanisation	Moisture
Black candlewood forest	3	0	0
Coastal shrubland	2	1	0
Dry forest > 20 ft	5	0	0
Dry forest < 20 ft	4	0	0
Dry shrubland	3	0	0
Dwarf vegetation and shrubs	2	1	0
Casuarina forest	4	2	1
Man-modified with trees	4	3	0
Man-modified without trees	1	4	0
Freshwater ponds	0	0	5
Saltwater ponds & lagoons	0	0	5
Salt-tolerant succulents	1	0	1
Seasonally flooded forest	4	0	2
Seasonally flooded grasslands	1	1	2
Seasonally flooded mangrove forest	4	0	3
Seasonally flooded mangrove shrubland	3	0	3
Semi-permanent flooded grasslands	1	1	3
Sparsely vegetated rock	0	0	0
Tidally flooded mangrove forest	4	1	4
Tidally flooded mangrove shrubland	3	1	4
Urban structure (e.g. roads, buildings)	0	5	0

Detailed Results

Abundance models goodness-of-fit metrics:

The distance sampling data was fitted with a half-normal detection function to each year individually at 45-90 meters for adults and sub-adults together each year. 2025 was the only year truncated at 90 m due to a relatively wide shoulder on the half-normal detection function. The number of adult and sub-adult detections ranged from 81 to 812 detections each year. In 2025 we detected 145 adults and sub-adults. The distance sampling models for adults and sub-adults vary in the goodness-of-fit over the years with the 2014 – 2018 period having Chi-square P-values between 0.0001 – 0.047 however, the detections functions appear to be fine and the low p-values are likely due to the chi-square test being overly sensitive to the high number of detections in the 2015 – 2018 years which had between 248 and 472 detections. The first year of the survey 2014 had a large spike in the first distance bin and flat detections in the following two bins causing the poor fit. Similarly, 2022 had poor fit (Chi-square: p-value = 0.02), large spike in first distance bin, and a relatively low number of detections. The remaining years 2019 – 2021 and 2023 – 2025 all passed the goodness-of-fit test and had appropriate detection functions (Chi-square: P-values = 0.07 – 0.61).

The distance sampling data for hatchlings were fitted with a half-normal detection function to each year individually with a truncation distance of 45 meters for all years. The number of hatchling detections ranged from 14 to 115 detections over the years, with 2025 having 22 detections. The distance sampling models for hatchling had varying goodness-of-fit metrics over the years with 2014 – 2016, 2018 – 2019, and 2023 – 2025 all having adequate scores (Chi-square: p-value > 0.05) and appropriate detection functions. Years 2017, 2020 – 2022 all failed the goodness-of-fit test (Chi-square: p-values < 0.05) and had detection functions with a large spike in the first distance bin and very low detections in the next two distance bins. The 2020 – 2022 period was after the massive initial removal effort from the culling program when hatchling detections were very sparse.

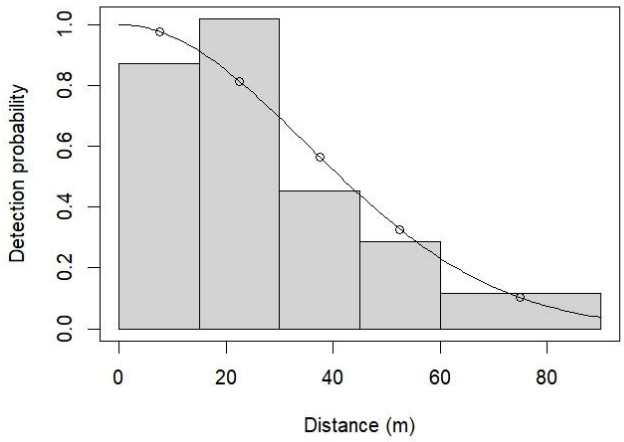

Figure 8. Adult & sub-adult detection function for 2025 survey data.

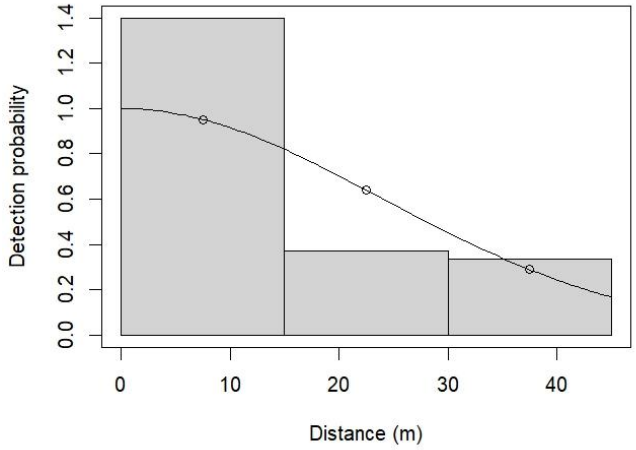

Figure 9. Hatchling detection function for 2025 survey data.

The N-mixture models for adults and sub-adults combined and hatchlings separately were both fitted with data from 649 points that were visited twice to date. We fitted both models with a Poisson distribution, K parameter of 200, random effect of point ID to capture extra-Poisson variation among points over time, and a covariate for year used to derive year-specific detection probability estimates (Pda). The adult and sub-adult model passed the Kologmorov-Smirnov goodness-of-fit test (p-value = 0.11) and the quantile plots indicated a good fit. The hatchling model passed Kologmorov-Smirnov goodness-of-fit test (p-value = 0.63), had no significant dispersion, and an adequate quantile plot.

Note that 2014 and 2015 had no repeated points to estimate Pda and derive availability to be used as a multiplier. Instead, I assigned the N-mixture Pda estimate from 2016 to 2014 and 2015 assuming Pda was similar among those three years.

Detection & availability probabilities:

The detection probability of adult & sub-adult iguanas has ranged between 0.25 – 0.55 over the survey years (Table 5) while availability has been more variable ranging between 0.23 – 0.73. The availability estimates are less certain with larger standard errors due to the relatively low number of points that are visited twice each year which is used to derive the availability estimate. The hatchling detection and availability estimates range between 0.07 – 0.34 over the survey years (Table 5) but both sets of estimates have relatively large standard errors due to the sparse number of detections and low number of repeated points each year.

Table 5. Detection and availability probability estimates from the Sister Islands Rock Iguana surveys on Little Cayman 2014 - 2025.

	Adults & Sub-adults			Hatchlings	
Year	**Detection Probability (P_d)** (S.E.)	**Availability Probability (P_a)** (S.E.)		**Detection Probability (P_d)** (S.E.)	**Availability Probability (P_a)** (S.E.)
2014	0.40 (0.09)	0.23 (0.04)		0.13 (0.04)	0.34 (0.05)
2015	0.37 (0.05)	0.25 (0.03)		0.18 (0.06)	0.24 (0.04)
2016	0.39 (0.03)	0.24 (0.02)		0.21 (0.03)	0.21 (0.02)
2017	0.25 (0.08)	0.38 (0.07)		0.20 (0.04)	0.20 (0.03)
2018	0.55 (0.06)	0.31 (0.08)		0.29 (0.04)	0.14 (0.02)
2019	0.43 (0.06)	0.51 (0.13)		0.22 (0.05)	0.10 (0.06)
2020	0.46 (0.11)	0.35 (0.15)		0.16 (0.06)	0.12 (0.03)
2021	0.36 (0.05)	0.73 (0.17)		0.18 (0.06)	0.39 (0.32)
2022	0.46 (0.06)	0.53 (0.12)		0.12 (0.04)	0.13 (0.03)
2023	0.35 (0.05)	0.34 (0.11)		0.22 (0.07)	0.15 (0.03)
2024	0.47 (0.07)	0.37 (0.10)		0.29 (0.08)	0.11 (0.02)
2025	0.28 (0.04)	0.22 (0.10)		0.32 (0.11)	0.07 (0.02)

Distribution Estimates:

The spatiotemporal model generally underpredicted the actual counts of iguanas (Figure 10). However, some smoothing over of extreme counts is expected when accounting for spatial autocorrelation which would result in under prediction (as well as over prediction for low count points) lowering overall predictive ability. The model with explicit parameterization of the pre- and post-cull period was used as the final model based on the ecological relevance of the estimates for spatial and temporal autocorrelation when using this approach. This more complicated model had similar predictive ability as the simpler model but allowed for inferences on the impact of the island-wide cull that has removed > 1.5 million iguanas to date.

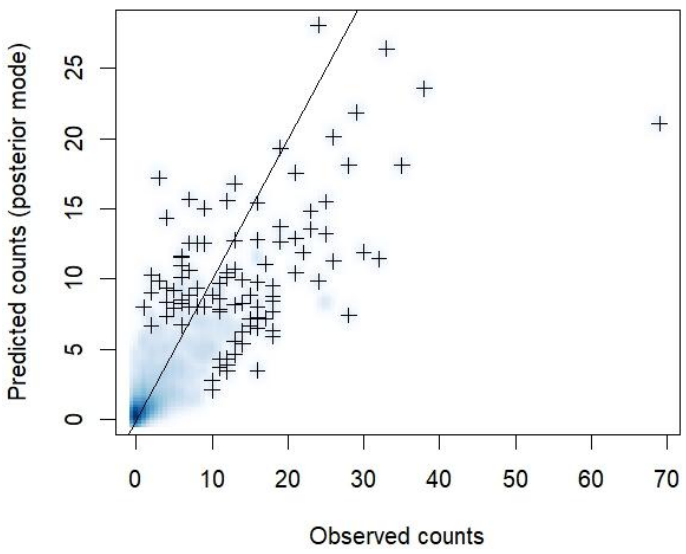

Figure 10. Predicted versus observed counts of iguanas.

The covariate effects indicate a strong association with moisture (coded to represent increasingly wet habitats) that was non-linear indicating a preference for mangrove wetland habitats more so than dry forest and dry shrubland or completely wet habitats (i.e. ponds/ lagoons which are used by green iguanas but only temporarily) (Table 6). The urban linear covariate suggests a strong associated with man-modified habitats overall when compared to undisturbed habitats (Table 6), however, the trend overtime indicates a varying relationship with negative association in 2014 to a strong association to urban environments during the 2015 – 2018 period, and to a negative association immediately after the start of the island-wide cull in 2020. This trend in urban covariate association tracks the population increase in the more urban West Bay to Savannah (western half) of the island and then the large-scale removal of green iguanas from these areas in 2020 (Figure 5). From 2021 onwards there is no effect of urbanization in addition to the baseline linear association (Table 6).

The Green Iguana distribution was significantly correlated at a larger range in the pre-cull years versus the post-cull years (Table 7). The variation within the range was also significantly higher in the pre-cull period (Table 7). This indicates that the population overall shifted from large continuity in its distribution pre-cull to a more fragmented distribution (Figure 11). In addition, the lower standard deviation in the spatial estimate post-cull indicates a more subdued or "flattened" distribution with low, sparse abundance across the landscape compared to the pre-cull period with pronounced hotspot areas with extremely high counts. For example, in the Safe Haven gold course area (~ 90 Hectare) counts of iguanas ranged from 3 – 69 iguanas per point in 2018; whereas in 2025 counts made in the same area ranged from 1 – 2 iguanas per point. Similarly, the pre-cull period showed a high temporal correlation (AR1) year-to-year whereas in the post-cull period this correlation was reduced (Table 7.) indicating the post-cull distribution is less stable (i.e. iguanas were seen less in the same areas and were being seen more in new areas) than the pre-cull distribution (Figure 11).

Table 6. Parameter estimates for the spatiotemporal model 2014 – 2025.

Parameter	Mean	95% Confidence Intervals
Neg. Bin. overdispersion	2.52	2.05 – 3.21
Intercept	-1.45	-1.92 – -1.98
Moisture	0.91	0.60 – 1.22
Moisture ^2	-0.16	-0.25 – -0.06
Urbanisation	0.28	0.15 – 0.41
Precision for urbanisation by year (rw1)	30.55	13.52 – 96.31

Table 7. Parameter estimates for autocorrelation parameters pre- (2014 – 2018) and post-cull (2019 – 2025).

	Pre-cull			Post-cull	
Parameter	**Mean**	**95% Confidence Intervals**		**Mean**	**95% Confidence Intervals**
Spatial range (km)	4.63	3.34 – 7.13		1.79	1.15 – 3.24
Spatial S.D.	1.45	1.17 – 1.89		0.76	0.64 – 0.91
AR1 temporal correlation	0.96	0.91 – 0.98		0.86	0.73 – 0.92

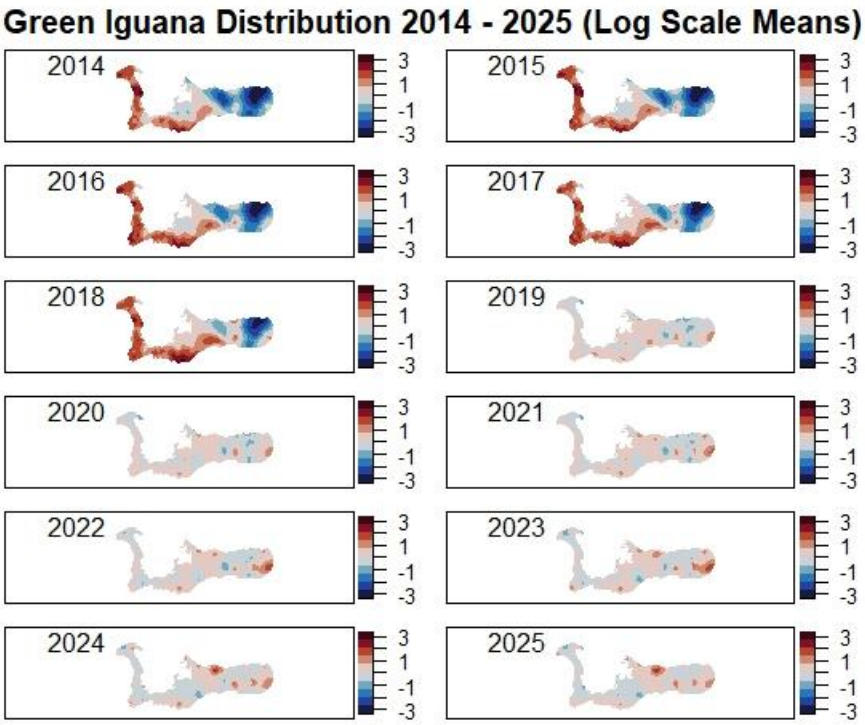

Figure 11. Distribution estimates of Green Iguanas in Grand Cayman 2014 – 2025, log scale means.

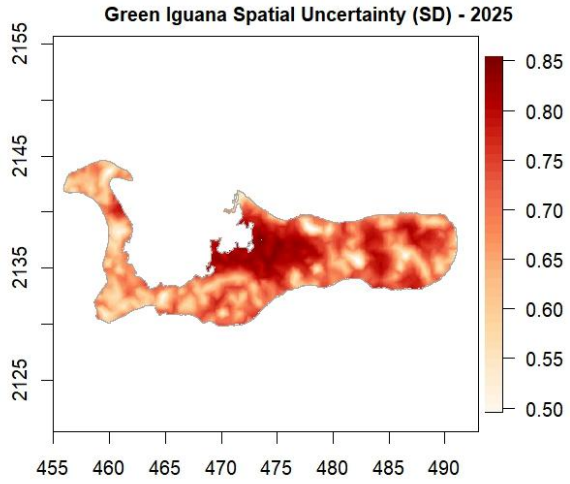

Figure 12. The standard deviation of the log-scale mean spatial estimate for 2025. Areas with higher standard deviation correspond to areas with low survey coverage.

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