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Distribution and Abundance of Endangered Florida Key Deer on Outer Islands

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ABSTRACT Status assessment of endangered Florida Key deer (*Odocoileus virginianus clavium*) is currently limited by a paucity of information regarding population estimates for outer islands, which collectively comprise approximately 70% of potential habitat within the Key deer range. Practical limitations and financial considerations render traditional survey techniques impractical for application on remote outer islands. Our objective was to evaluate the utility of infrared-triggered cameras to estimate Key deer abundance on outer islands. We used digital infrared-triggered cameras and mark–resight methods to estimate Key deer abundance on 20 outer islands. Abundance estimates for primary subpopulations ranged from 15 to 16 for Howe Key, 5 to 10 for Knockemdown complex, and 13 to 17 for Little Pine Key. Other island complexes such as Ramrod Key, Water Key, and Annette complex maintain only small subpopulations (i.e., $\leq$5 individuals) and other previously inhabited island complexes (e.g., Johnson complex and Summerland Key) no longer maintain subpopulations. Key deer abundance was well below estimated carrying capacities on all outer islands, with larger natural populations occurring closest to Big Pine Key. Our results suggest that camera-based surveys offer a practical method to monitor abundance and population trends of Key deer on outer islands. Our study is the first to estimate Key deer abundance in these areas using technically structured model-based methods and provides managers with current and baseline information regarding Key deer subpopulations. (JOURNAL OF WILDLIFE MANAGEMENT 72(2):360–366; 2008)

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KEY WORDS abundance, digital infrared-triggered cameras, distribution, Florida Key deer, *Odocoileus virginianus clavium*, outer islands.

The endangered Florida Key deer (*Odocoileus virginianus clavium*), the smallest subspecies of North American white-tailed deer, is endemic to the Lower Florida Keys, Florida, USA (Lopez et al. 2003). Key deer exhibit a restricted distribution within a highly fragmented landscape consisting of 20–25 small oceanic islands (Fig. 1). Big Pine (2,428 ha) and No Name (459 ha) keys are relatively large islands and comprise the core of the Key deer range (Lopez et al. 2003, Harveson et al. 2006). An estimated 75% of the total population resides on Big Pine and No Name keys, with the remaining deer inhabiting outer islands (islands within the range of Key deer excluding Big Pine and No Name keys) that vary considerably in size and habitat quality (Lopez 2001). Collectively, these islands comprise approximately 70% of available habitat within the Key deer range (Fig. 1). However, previous research on Key deer demographics, density, population structure, and habitat use have focused on Key deer within their core population on Big Pine and No Name keys (e.g., Lopez et al. 2003, 2004; Harveson et al. 2004). Few studies have addressed the abundance and distribution of Key deer inhabiting outer islands (e.g., Klimstra et al. 1974, Silvy 1975) with no recent studies available for status assessment.

Traditional survey techniques used for Key deer on Big Pine and No Name Keys include road-counts, strip-counts, and mark–recapture methods (Silvy 1975, Lopez et al. 2004, Roberts et al. 2006). Practical limitations (i.e., dearth of roads, dense vegetation) and financial considerations render these survey techniques impractical for application on outer islands (Roberts et al. 2006). Camera-based surveys have been used to detect and monitor occupancy of rare mammalian species in remote locations (Bull et al. 1992, Foster and Humphrey 1995, Zielinski and Kucera 1995, Foresman and Pearson 1998) and to estimate abundance of rare or elusive large mammals (Mace et al. 1994, Karanth and Nichols 1998, Sweitzer et al. 2000, Silver et al. 2004, Heilbrun et al. 2006). Camera-based surveys also have been used to survey large mammals in densely vegetated habitats where visual observation is inadequate to facilitate traditional survey techniques (Seydack 1984, Karanth and

Figure 1. Map of the Lower Florida Keys, Florida, USA, within the known distribution of the endangered Florida Key deer, 2006.

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Table 1. Estimated abundance of Key deer subpopulations on outer islands derived from digital infrared-triggered camera data, Lower Florida Keys, Florida, USA, 2005.

Island complex	Area (ha)	Jacobson estimate	Peterson estimate (adjusted)	Minta–Mangal estimate	MNKA[a]	CPUE[b]
Annette complex	222	NC[c]	NC[c]	NC[c]	1	0.05
Cudjoe	1,319	9	8	11	7	0.99
Howe	373	15	16	16	9	1.80
Johnson complex	154	0	0	0	0	0.00
Knockemdown complex	582	5	9	10	3	0.27
Little Pine	381	17	12	13	4	0.67
Newfound Harbor complex	76	36	93	150	6	4.15
Ramrod	374	NC[c]	NC[c]	NC[c]	1	0.35
Sugarloaf (survey 1)	1,399	27	22	69	5	0.59
Sugarloaf (survey 2)	1,399	23	30	25	5	0.64
Summerland	436	0	0	0	0	0.00
Big Torch	626	17	29	NC[c]	4	0.17
Little Torch	305	8	5	NC[c]	3	0.20
Middle Torch	410	3	4	3	3	0.05
Water	92	NC[c]	NC[c]	NC[c]	1	0.10

[a] MNKA = absolute min. no. of individual Key deer known to be alive during survey.
[b] CPUE = catch per unit effort, defined as: (total no. of valid photographic-captures)/(total no. of camera days [i.e., no. of cameras × 20 days]).
[c] NC = estimate was not calculable due to insufficient captures of individually identifiable Key deer during capture or recapture periods.

Nichols 1998, Carbone et al. 2001). Infrared-triggered cameras may provide a practical and accurate method to estimate Key deer abundance and monitor population trends on outer islands (Roberts et al. 2006). Roberts et al. (2006) compared abundance estimates derived from traditional Key deer survey techniques to camera-based surveys on No Name Key and concluded that camera-based surveys were a suitable alternative to traditional survey techniques, particularly in areas where accessibility was an issue. Our objective was to develop a practical method to estimate and monitor the distribution and abundance of Key deer on outer islands and to obtain baseline information regarding Key deer in these areas. Our study is the first to estimate Key deer abundance on outer islands, which collectively represent 70% of suitable habitat within the Key deer's range.

STUDY AREA

Our study area was located in the Lower Florida Keys, Monroe County, Florida (Fig. 1) and consisted of federal, private, and state lands including the National Key Deer Refuge, Great White Heron National Wildlife Refuge, and The Nature Conservancy's Torchwood Hammock Preserve. Geology of the Lower Florida Keys is dominated by 2 Pleistocene formations: Miami limestone (oolite) and older Key Largo limestone, which influence the distribution and availability of fresh water throughout the study area (Hoffmeister and Multer 1968, Halley et al. 1997). Topographic relief is extremely low and most islands are nearly flat with elevations <2 m. Climate was subtropical-marine with mean January temperatures of 21° C, mean July temperatures of 29° C, and 98.8 cm average annual rainfall (National Oceanic and Atmospheric Administration 2006). Vegetation was principally West Indian in origin and was significantly influenced by elevation (Dickson 1955, Long 1974, Folk 1991, Lopez et al. 2004). Vegetative communities near sea level were primarily comprised of black mangrove (*Avicennia germinans*), white mangrove (*Laguncularia racemosa*), red mangrove (*Rhizophora mangle*), and other halophytes. These salt-tolerant communities were successively replaced by buttonwood (*Conocarpus erectus*), hammock, and pineland communities with increasing elevation (Lopez et al. 2004).

We conducted surveys on 14 islands or island complexes within the Lower Florida Keys (Fig. 1). Island complexes are groups of islands close (≤0.29 km) to one another, separated by tidal mudflats or shallow waters that can, in regards to Key deer population dynamics, be considered functionally one island (Harveson et al. 2006). Klimstra et al. (1974) identified 14 island complexes where Key deer dispersal was relatively unrestricted and observed Key deer or their sign (e.g., tracks, pellets, evidence of browsing) on all 14 complexes. Our study occurred on the same 14 island complexes: Annette complex (4 islands), Big Torch Key, Cudjoe Key, Howe Key, Johnson complex (3 islands), Knockemdown complex (3 islands), Little Pine Key, Little Torch Key, Middle Torch Key, Newfound Harbor complex (5 islands), Ramrod Key, Sugarloaf Key, Summerland Key, and Water Key (Fig. 1, Table 1).

METHODS

Data Collection

We conducted camera surveys on 14 outer islands or complexes (Fig. 1) between August and December 2005. We divided Key deer habitat on each island or complex into 50-ha sampling units as suggested by Roberts et al. (2006). We placed digital infrared-triggered cameras (Cuddeback; Non-typical Inc., Park Falls, WI) at a density of one camera per sample unit (50-ha) comparable to average seasonal ranges reported by Silvy (1975) to ensure complete coverage of Key deer habitat within each island or complex. We placed ≥2 cameras on each island complex to account for island complexes comprised of several small islands (<100 ha). Preliminary field tests indicated that placement of camera stations in high-use areas was crucial to acquiring adequate photographic captures of Key deer. Thus, we attempted to centrally locate camera stations within each sampling unit; however, to increase photographic captures, we placed camera stations at freshwater sources, trails, or other areas where we observed Key deer or their sign near sampling unit centers. Furthermore, we baited camera stations once during initial setup with $\leq$3.8 L domestic feed (i.e., sweet feed and cracked corn) to entice exploration by Key deer. We spread bait sparsely over a large area (50–100 m^2) around each camera station and did not re-bait areas after initial setup.

We conducted camera surveys on each island or complex for 20 consecutive 24-hour periods (days). We mounted cameras on trees or constructed platforms approximately 30 cm above ground and set each camera to manufacturer recommended settings for local conditions. We set picture delay to one minute and used video mode to record 10–20 seconds of video after the initial photograph to increase accuracy during identification of individuals and to document neonates and other deer that might not be recorded in initial photographs. Date and time of photographs were automatically recorded on each digital photograph by the camera system. In addition to collecting photographic captures with infrared-triggered cameras, we also conducted intensive searches for presence or absence of Key deer sign (e.g., tracks, pellets, etc.) and areas of high use (e.g., trails, freshwater holes) within each sampling unit to compare current Key deer distribution with historic island occupancies.

Data Analysis

We reviewed digital photographs to identify individual male Key deer by antler configuration. We grouped photographic captures into 4 categories as described by Jacobson et al. (1997): 1) individually identifiable adult males (which served as marked individuals during analyses), 2) adult females, 3) indistinguishable antlered males (e.g., spikes), and 4) subadults. We excluded photographic captures from analysis if we could not conclusively identify the individual. Preliminary field observations indicated that marked individuals (both M and F) lingered at recently baited camera stations $\leq$47 minutes. Thus, we excluded multiple photographs of Key deer within a $\leq$1-hour period to minimize double-counting. We calculated valid captures separately for each camera station and then pooled them for each island or complex. In addition to adult males, we used females with numbered collars from a concurrent study on Cudjoe and Sugarloaf keys as marked females for analysis of sightability between male and female Key deer; however, we treated collared females as unmarked adult females during calculation of subpopulation abundance (see below).

Using the 20-day camera data, we compared 3 frequently used methods to calculate abundance from camera-based survey data for each island or complex. First, we used methods described by Jacobson et al. (1997) to calculate abundance estimates for a 20-day survey period. Second, using a 10-day mark and 10-day resight period, we calculated an abundance estimate with a 2-sample Peterson model (Seber 1982). We used Bailey's binomial model as it is more robust to small sample sizes (Bailey 1952, Seber 1982). We then fit resultant abundance estimates to a Poisson distribution and used simulations to develop corrected population estimates using sightability of known deer on Cudjoe and Sugarloaf keys (70–100%). Finally, we obtained a third abundance estimate using methods developed by Minta and Mangel (1989) using Program NOREMARK (White 1996), which uses the capture frequencies of marked individuals to estimate capture frequencies of nonmarked individuals by means of bootstrap methods. We adjusted the lower bounds of confidence intervals to account for the minimum number known alive on each island complex where applicable, and we rounded all fractions up to the nearest whole number.

We defined total sampling effort as the sum of all 24-hour periods (days) cameras were operated on an island or complex. We defined catch per unit effort as C/X, where C is the total valid photographic captures per survey period and X is the total sampling effort for each island complex. We defined photographic capture rate as the number of valid photographic captures per day summed for all camera stations on an island or complex. We developed sighting histories for all individually identifiable Key deer on all outer islands using each day as a sampling period ($n = 20$). We tested equal sightability among Key deer by day using a zero-truncated Poisson test for equal sightability (Caughley 1977). We derived expected values from observed data, amalgamated expected values $<$1.0 into one category (i.e., $\geq$9 captures) and tested the hypothesis of equal sightability among Key deer by day using a chi-square goodness-of-fit test. We tested sightability between sexes using capture histories for individually identifiable males on all outer islands and collared females on Cudjoe and Sugarloaf keys using Program MARK (White and Burnham 1999). Finally, White and Garrott (1990) noted the precision of abundance estimates can be improved through replication of surveys. Therefore, we conducted 2 surveys on Sugarloaf Key to gain insight into the precision of camera-based estimates of Key deer abundance. Practical limitations (e.g., no. of cameras available), the need to obtain baseline data for all outer islands within one year, and setbacks due to the most active Atlantic hurricane season in recorded history prohibited replication of surveys on all outer islands.

RESULTS

We detected Key deer or their sign on 12 of 14 historically occupied outer islands or complexes. However, we did not detect evidence of Key deer presence on Summerland Key or Johnson complex (Fig. 1). In addition, we documented Key deer presence on Spanish Harbor Key where Key deer were not historically known to occur. We obtained 649 valid photographic captures of Key deer on outer islands and photographically captured 16 of 19 (84%) known individuals (i.e., marked Key deer) by day 10. We photographically captured 70% of known individual Key deer on Sugarloaf Key and 100% of known individuals on Cudjoe Key by day

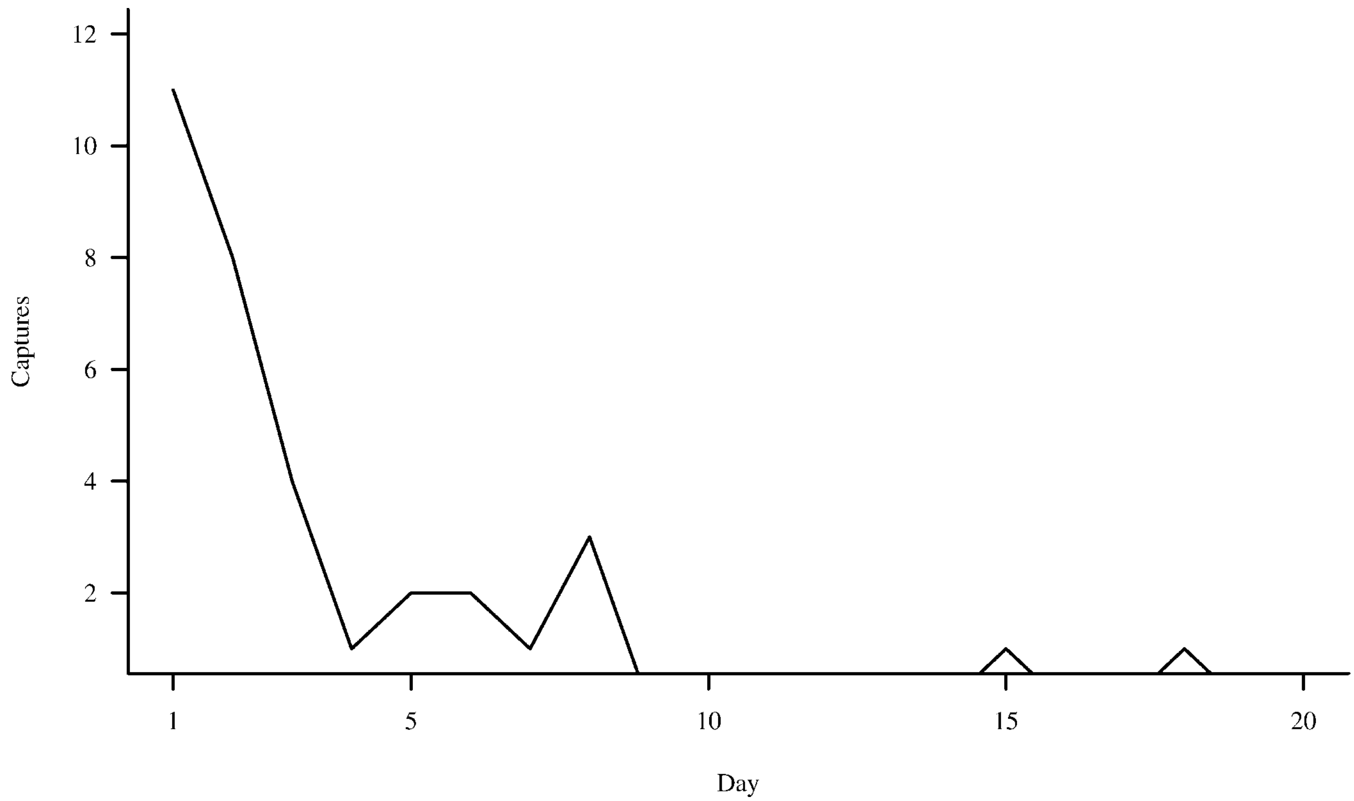

Figure 2. Capture rate for new individual Key deer (*y*-axis) captured by day using digital infrared-triggered cameras on outer islands, Lower Florida Keys, Florida, USA, 2006.

10. Our cumulative capture rate for new marked Key deer reached an asymptote around day 9 with only 2 previously unmarked Key deer being captured after day 10 (Fig. 2). One of these new marked Key deer appearing after day 10 likely immigrated to the Newfound Harbor complex from south Big Pine Key. We regularly captured marked Key deer throughout the 20-day survey period at multiple camera stations. Number of recaptures for marked Key deer during recapture sessions (i.e., last 10 days) ranged from 1 to 7 with a mean of 2.7 for males and 2.4 for females. We found sightability of marked Key deer by day was not equal among all individuals (zero-truncated Poisson tests of equal sightability; $P < 0.001$); however, model selection indicated no significant difference in sighting probabilities by day between sexes (Program MARK; M = 0.244, SE = 0.031; F = 0.224, SE = 0.019).

We found abundance estimates for Key deer subpopulations varied considerably among outer islands (Table 1). Failure to photographically capture mature males on 3 island complexes (i.e., Annette complex, Ramrod Key, and Water Key; Table 1) prohibited calculation of abundance estimates. For these islands or complexes, we considered the minimum number known alive (MNKA) to be a conservative subpopulation estimate. We noted the Peterson-estimator produced negatively biased preadjustment estimates (i.e., <MNKA) on Cudjoe Key due to high sightability of male Key deer. Conversely, we suspect that estimators may have overestimated abundance on Newfound Harbor complex due to violations of closure assumptions, low sightability of males during recapture sessions, an unusually high density of Key deer due to proximity to Big Pine Key, and anthropogenic factors described in more detail below.

We noted that photographic captures were highest during the first 2–4 days of each survey and declined considerably until becoming more stable around 5–6 days (Fig. 3). We excluded the Newfound Harbor complex from cumulative results for several reasons: 1) Newfound Harbor complex is a primary dispersal area for south Big Pine Key, 2) Newfound Harbor has an unusually high density of Key deer due to anthropogenic factors (e.g., camp grounds, ornamental plants), and 3) Newfound Harbor is used as a corridor from Big Pine Key to Little Palm Island where Key deer feed on ornamental plants, use freshwater resources, and are regularly fed by island occupants (Fig. 1).

DISCUSSION

Our results indicate that camera-based surveys provide a practical method to estimate Key deer distribution and abundance on outer islands. Most Key deer inhabiting outer islands were recurrently photographed during the survey periods, suggesting camera density on islands (1 camera/50 ha) was sufficient in obtaining abundance estimates. Although higher camera density might result in capture of all individuals, it may not be practical. Misidentification of individuals, particularly marked individuals, during analysis could potentially bias survey results. However, marked individuals in our study offered distinctly recognizable characteristics that permitted accurate identification. We

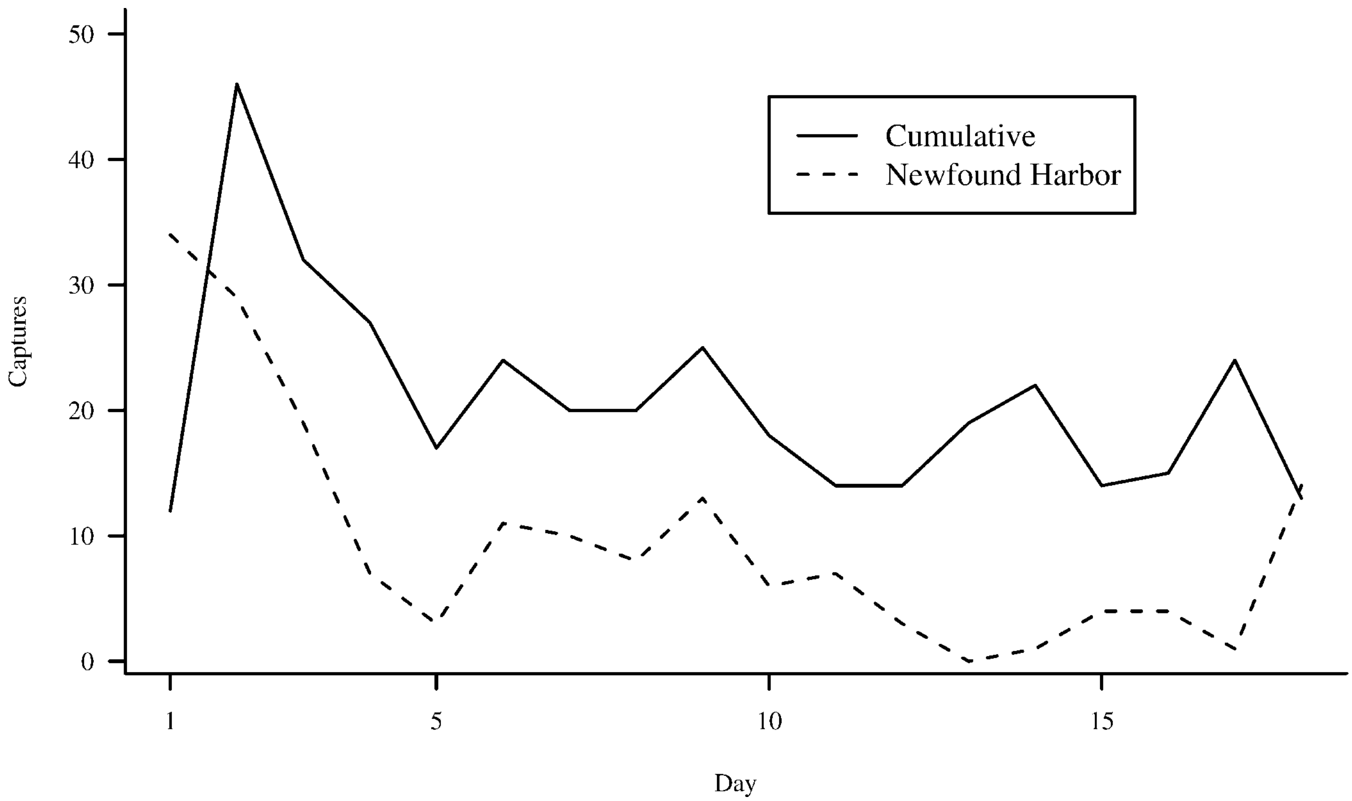

Figure 3. Cumulative capture rate for Key deer on all outer islands (excluding Newfound Harbor complex) and Newfound Harbor complex using digital infrared-triggered cameras, Lower Florida Keys, Florida, USA, 2006.

photographically captured 84% of new marked individuals by day 10 and repeatedly photographed marked Key deer at multiple camera stations throughout survey periods. Our study results were congruent with those observed by Jacobson et al. (1997) who reported 100% and 88% of marked white-tailed deer were photographically captured during a 14-day period and 100% and 82% were captured during the first 10 days of the survey.

Methods to estimate population size from camera-based survey data rely heavily on assumptions of geographic and demographic closure. Using methods presented here it is difficult to formally address closure assumptions; however, capture rates for new deer suggest closure assumptions were met in our study. Additionally, geographic-closure can be reasonably assumed because individual island complexes represent functionally closed island populations and we sampled entire island complexes simultaneously. The short duration of surveys (i.e., 20 days) likely satisfies demographic closure assumptions and also permits the assumption of mark retention for the entirety of the survey period (i.e., given that surveys were not conducted when males might cast antlers or prior to sufficient growth for identification). Use of cameras also allows assumptions of little or no disturbance to natural activities that are commonly associated with other methods to estimate population size (e.g., live-capture studies).

Heterogeneous sightability of age and sex classes could bias abundance estimates based on camera-survey data (Karanth 1995, Jacobson et al. 1997, Koerth et al. 1997). Because it is impossible to test assumptions of equal sightability among survey periods using a 2-sample Peterson model with any statistical power (Krebs 1999), we could not conduct tests of equal sightability by survey-period (i.e., 10 days). Sightability of Key deer by day was heterogeneous among individuals. Although it is difficult to relate sightability by day to sightability by survey period (10 days), model assumptions of homogeneous sightability among all individuals may have been violated. However, mean number of photographic captures for marked individuals during recapture sessions and results of model selection indicate sightability of Key deer was not a function of sex. Equivalent detection probabilities between sexes indicate use of male Key deer as marked individuals in subpopulations was permissible.

We compared 3 frequently applied methods to estimate abundance from camera-based survey data. All 3 estimators produced reasonable results (i.e., compared to sign obs) when model assumptions were not perceptively violated. However, abundance estimators produced spurious results for subpopulations where assumptions of homogeneous sightability were likely violated. Abundance estimates for islands such as Newfound Harbor complex were likely biased high due to low sightability of male Key deer during recapture sessions and violations of closure assumptions. Conversely, on Cudjoe Key, 2 males were captured on as many as 5 and 7 occasions during the 10-day recapture session whereas other marked Key deer on Cudjoe were typically captured on 2–3 occasions during this same time period. Thus, abundance estimates for Cudjoe Key were negatively biased and below the minimum number of Key deer known to inhabit the island. All mark–resight estimators used to develop abundance estimates are subject to inherent biases associated with small samples sizes and sightability of individual animals. Sightability of individual Key deer is an inherent problem when estimating Key deer abundance using camera-based methods. Methods developed by Minta and Mangel (1989) address variation in sightability among individuals within subpopulations and we recommend application of this method when estimating abundance of small populations such as Key deer on outer islands.

Camera-based studies often use baited camera stations to maximize photographic captures (e.g., Mace et al. 1994, Jacobson et al. 1997, Koerth et al. 1997). We suggest use of bait at camera stations may violate model assumptions of homogeneous sightability and can affect the accuracy and precision of abundance estimates (White et al. 1982). A decline in cumulative capture rates (Fig. 3) indicates that baiting affected sightability of Key deer. Bias associated with heterogeneous sightability among all individuals is likely to be exacerbated if camera stations are regularly baited as trap-happy individuals can substantially influence results of abundance estimates. Thus, baiting should be avoided when collecting data for use with methods described by Jacobson et al. (1997). A possible method to minimize effects of baiting using the Jacobson estimator would be to exclude the period where influence of bait was most significant (e.g., days 1–5 in our study). Conversely, the 2-sample Peterson model is not affected by differential sightability between marking and resighting sessions provided that sightability is homogeneous within the resight session and observation in the first session is independent of observation in the second. Cumulative capture rates indicated effects of baiting Key deer dissipated by day 5 allowing increased captures during marking periods (i.e., increased marked individuals) without violating model assumptions.

Finally, our study provides current information regarding the distribution and status of Key deer inhabiting outer islands. Results of camera-based surveys were comparable to intensive searches for Key deer sign within sampling units. However, cameras detected Key deer presence in 2 sampling units where sign surveys did not produce indications of deer presence. Survey data suggests the distribution of Key deer may have constricted since the early 1970s (Klimstra et al. 1974). Absence of Key deer on previously occupied islands such as Big Johnson and Summerland keys may be an indication of declines on outer islands. Big Johnson Key likely serves as a sink or dispersal area for the subpopulation on Little Pine Key (Harveson et al. 2006). Thus, the current absence of Key deer on Big Johnson Key may be indicative of subpopulation declines on Little Pine Key and the same holds true for source populations near Summerland and Ramrod keys. However, meta-populations are dynamic and the absence or low density of Key deer on specific outer islands may be the result of natural fluctuation in meta-population structure. Our study is the first effort to estimate abundance of Key deer subpopulations on outer islands using technically structured model-based methods. Natural Key deer abundance (i.e., excluding translocated animals) was greatest on island complexes closest to Big Pine Key (e.g., Howe Key, Little Pine Key, and Big Torch). Estimates for other island complexes such as Ramrod Key, Water Key, and Annette complex indicate that these islands maintain only small subpopulations (i.e., $\leq$5 individuals) that are likely seasonal or ephemeral in nature. Model-based research by Harveson et al. (2006) concluded that distance from source populations and island size significantly influence Key deer abundance on outer islands, and our results support their conclusions.

MANAGEMENT IMPLICATIONS

Population estimates are fundamental to the management of large mammal populations and the implementation of appropriate conservation strategies, particularly for endangered species such as the Florida Key deer. Reclassification of Key deer from endangered to threatened has recently been proposed as a result of increased Key deer densities within their core range (i.e., Big Pine and No Name keys; Lopez et al. 2004). An important aspect in the proposed reclassification process will include an evaluation of the status of Key deer on outer islands (i.e., abundance estimates). Our results suggest camera-based surveys offer a practical method to monitor abundance and population trends of Key deer on outer islands. Similar sightability among male and female Key deer allow the use of natural marks in obtaining estimates, thereby reducing the temporal and financial costs associated with traditional Key deer survey methods, which require physical capture and direct observation. Though we recommend the use of camera surveys to survey Key deer on outer islands, wildlife managers should be cognizant of inherent biases associated with estimating small subpopulations.

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