AN EVALUATION OF HABITAT SELECTION AND AN ABUNDANCE ESTIMATE FOR THE ENDANGERED KEY LARGO WOODRAT

by

CHRIS WINCHESTER

(Under the direction of Michael T. Mengak and Steven B. Castleberry)

ABSTRACT

The Key Largo woodrat (KLWR; *Neotoma floridana smalli*) is an endangered species endemic to the island of Key Largo. Despite protection of its remaining habitat from development, the population continued to decline over the last 20 years. Information on KLWR habitat preferences required to effectively manage the population is lacking, as is a means to monitor abundance. The objectives of this study were to examine habitat selection by KLWR at two spatial scales, the macro and micro-scales, and identify an efficient sampling design that can be used to monitor this rare population. Results of habitat and nest site selection suggest KLWR is limited by the availability of quality nest habitat and would benefit from the addition of artificial nest substrate and the continued protection of the forested uplands of Key Largo. Stratified random sampling was the most effective design and can be used to monitor the status of KLWR.

INDEX WORDS: Key Largo woodrat, *Neotoma floridana smalli*, macrohabitat selection, microhabitat selection, nest site selection, logistic regression, resource selection function (RSF), adaptive cluster sampling, radiotelemetry, abundance, closed population model, occupancy.

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DEDICATION

To my mom for her love and support.

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INTRODUCTION AND LITERATURE REVIEW

SUMMARY

The Key Largo woodrat (KLWR, *Neotoma floridana smalli*) is a federally endangered subspecies of the eastern woodrat (*Neotoma floridana*) endemic to Key Largo, Florida. KLWR historically ranged throughout the upland forests (hardwood hammocks) of Key Largo but is now restricted to federal and state managed lands on the northern one-third of the island (Barbour and Humphrey 1982). Despite protection of its remaining habitat from development, KLWR continued to decline since its listing in 1984 (McCleery et al. 2006b). The cause of the population decline is unknown, but the quality of existing habitat appears severely compromised. Past disturbances to the hardwood hammocks, such as farming and abandoned development projects, altered the structure and composition of the forested landscape. The degraded forest habitat may be further compromised by fragmentation and proximity to heavily urbanized areas. Fragmentation increased the amount of forest edge, which assisted in the establishment of red imported fire ants (*Solenopsis invicta*; Forys et al. 2002). Fire ants are known to alter habitat use by small mammals (Smith et al. 1990, Killion and Grant 1993, Killion et al. 1995, Holtcamp et al. 1997) and may have reduced the quality of foraging habitat for KLWR. Urban areas offer year round food sources to predators, such as feral cats (*Felis catus*) and raccoons (*Procyon lotor*), potentially increasing their abundances (Hoffman and Gottschanng 1977, Prange et al. 2005) and KLWR predation levels.

Past KLWR studies focused on the differential use of forest patches at varying stages of recovery from disturbances, and were conducted at large spatial scales. Results were conflicting and shed little light on factors restricting KLWR (Hersh 1978, Barbour and Humphrey 1982, Keith and Gaines 2002, Sasso and Gaines 2002, McCleery et al. 2006b; Goodyear, unpublished report). In addition, little information is available on microhabitat selection by KLWR. Since animal populations are limited by factors that occur at multiple spatial scales, a multi-scale approach to observing habitat selection is often most informative (Hilden 1965, Johnson 1980). Given the uncertainty surrounding habitat requirements of KLWR and factors contributing to its decline, it is imperative that additional information on habitat selection be gathered to assess the quality of existing habitat and effectively guide recovery of the population

Nest sites are critical resources for KLWR, and are used as protective cover and refugia during inactive periods (McCleery et al. 2006b; Goodyear, unpublished report). Although KLWR are known to build and maintain stick nests (Goodyear, unpublished report), recent results suggest artificial nest substrate (i.e., rock and debris piles) is a preferred resource for nesting (McCleery et al. 2006b). In addition, recently disturbed areas characterized as young hammock were believed to be preferred nesting habitat (McCleery et al. 2006b). However, artificial substrate occurs frequently in recently disturbed areas, and may be confounded with young hammock. Distinguishing between young hammock and artificial nest substrate as preferred resources for nesting is critical since management recommendations differ greatly for each.

In addition to information on habitat selection, recovery of KLWR would benefit from a monitoring program to gauge the effectiveness of prescribed management. Monitoring the abundance of a rare species, like KLWR, is challenging using traditional sampling designs.

Equal probability designs, such as simple random sampling, often yield imprecise estimates even with large sample sizes. In addition to being rare, there is evidence KLWR may be clustered due to an association with artificial nest substrate (McCleery et al. 2006b). Adaptive cluster sampling (ACS) was developed as means to estimate the abundance of rare and clustered populations with greater precision (Thompson 1990). If successful, the increased efficiency of ACS would allow for a more cost-effective method to monitor KLWR and evaluate effectiveness of prescribed management.

The goals of this research were to examine a variety of factors potentially restricting KLWR, such as predator activity, structure and composition of vegetation, and availability of nest sites, by identifying macro and microhabitat selection. Nest site selection was evaluated to distinguish between the importance of artificial nest substrate and hammock age as nest habitat for KLWR. Collectively, results were used to recommend a management strategy to recover the population and reduce the risk of extinction. In addition, I compared the efficiency of adaptive cluster sampling to stratified random sampling in order to recommend an optimal design to monitor KLWR abundance and evaluate the effectiveness of recovery efforts.

LITERATURE REVIEW

KLWR Ecology and Habitat Use

The Key Largo woodrat (KLWR, *Neotoma floridana smalli*) is the southern-most sub-species of the eastern woodrat (*Neotoma floridana*) and is endemic to Key Largo, Florida. The insular population of KLWR is separated from its nearest conspecific by the southern one-third of the Florida peninsula (Hersh 1981). Although KLWR historically occurred throughout the upland forests of Key Largo, the remaining population is now restricted to the northern one-third of the island due to habitat loss (Barbour and Humphrey 1982). Previous attempts at evaluating population status varied in methodology, but collectively demonstrated a consistent decline over the last 25 years (McCleery et al. 2006b). The population decline occurred despite the protection of its remaining habitat from development and its listing as an endangered species. Recent population estimates and a viability analysis suggest KLWR is at a high risk of extinction within the next ten years (McCleery et al. 2005, McCleery et al. 2006b).

Past research on KLWR habitat selection focused on differential use of forest patches, characterized by age class, in an effort to assess habitat quality and guide recovery of the population (Hersh 1978, Barbour and Humphrey 1982, Keith and Gaines 2002, Sasso and Gaines 2002, McCleery et al. 2006b; Goodyear, unpublished report). Unnatural disturbances, such as farming and land clearing, coupled with excavation for commercial development, created a mosaic of forest conditions throughout the landscape, varying in seral stage (Ross et al., unpublished report). It was hypothesized that the effect of these disturbances, and resulting heterogeneity in patch age, negatively impacted the persistence of KLWR. However, results were conflicting. Early efforts determined KLWR used mature or climax forest more frequently and were negatively impacted by the overall reduction of mature forest (Hersh 1981, Barbour and Humphrey 1982). In a later study, a difference in densities of KLWR was noted between northern and southern portions of Key Largo and was attributed to a preference for mature forests and areas far removed from residential development (Humphrey 1988). However, additional studies found KLWR to use a variety of forest types and did not determine a preference for any particular age class (Keith and Gaines 2002, Sasso and Gaines 2002; Goodyear, unpublished report). More recently, KLWR was found to prefer newly disturbed patches of forest and was absent from mature forest (McCleery et al. 2006b). The discrepancy in results from past research creates uncertainty as to the relevance and predictive ability of using forest age class to identify habitat for KLWR. In order to better guide the recovery of KLWR and identify factors restricting the population, additional information on habitat selection extending beyond the use of different forest age classes is required.

Potential Restricting Factors

KLWR requires protective cover for suitable nest sites and quality foraging habitat within close proximity to the nest in order to minimize predation risk and optimally acquire resources (Orians and Pearson 1979). Due to their moderate size and terrestrial mode of life, *Neotoma* species are believed to be at a high risk to predation and rely heavily on nest sites and cover for protection (Rainey 1956). Changes in forest structure and composition could limit the quality of available nest sites and foraging habitat. Naturally occurring nest sites typically utilized by KLWR include the root systems of standing or wind-blown trees, logs, and solution holes (holes in the limestone substrate created by erosion; Goodyear, unpublished report). Recently, KLWR was found to use artificial substrate for nest sites, such as rock piles and debris from dumping activities, at a high frequency (McCleery et al. 2006b). Although it is common for *Neotoma* species to use anthropogenic structures for nesting, it is unclear if the observed pattern of nest use by KLWR was opportunistic or indicative of a decrease in the quality of naturally occurring nest substrate.

Housing developments, roads, and a waste transfer station, within and around the remaining hardwood hammock, resulted in fragmentation and an increase in anthropogenic food resources for raccoons (*Procyon lotor*) and feral cats (*Felis catus*), potentially increasing their densities (Hoffman and Gottschanng 1977, Prange et al. 2005). Feral cats are well documented predators of small mammals (Baker et al. 2005, Espinosa-Gayosso and Alvarez-Castaneda 2006) and raccoons have been shown to depredate small mammals, including Norway rats (*Rattus* *norvegicus*, Hoffman and Gottschanng 1977). Increased predation levels due to unnaturally high predator densities could be contributing to KLWR population declines. The increase in forest edge resulting from fragmentation has facilitated the establishment of red imported fire ants (*Solenopsis invicta*), which favor disturbed areas lacking canopy cover (Forys et al. 2002). Past studies found small mammals altered patterns of habitat use to avoid fire ants (Smith et al. 1990, Killion and Grant 1993, Killion et al. 1995, Holtcamp et al. 1997). High fire ant abundance may have lowered the quality of foraging habitat for KLWR.

Habitat Selection and the Importance of Scale

Selection is the process by which an animal chooses a resource or habitat (Johnson 1980). The use of a habitat is considered selective if the habitat is exploited disproportionately to its availability (Johnson 1980). Animals select habitats at several spatial scales, and factors influencing selection vary across scales (Hilden 1965, Johnson 1980, Litviatis et al. 1994). In order to gain meaningful insights into habitat selection criteria, careful consideration must be given to scales chosen and factors likely affecting selection at those scales (Manly et al. 2002).

Habitat selection is commonly viewed at 2 spatial scales, the macro and micro-scales (Litviatis et al. 1994). Macro-scale studies examine landscape level processes, which limit the distribution of a population. An examination of micro-habitat selection typically focuses on selection of resources, such as food and cover, required for an individual to survive and reproduce. Considering the multi-scale effects experienced by animals, examining habitat selection at both the micro and macro-scales may offer the greatest insight to animal-habitat relationships (Morris 1984, Snyder and Best 1988).

Previous KLWR studies focused on habitat selection at the macro-scale. An examination of micro-scale habitat selection, selection of resources within the individual foraging range, may provide valuable information on habitat preferences which are obscured at the macro-scale (Jorgensen and Demarais 1999, Castleberry et al. 2002). Sasso and Gaines (2006) evaluated KLWR microhabitat selection by comparing habitat around used and unused trap locations and suggested larger canopy trees and open understory were most important. However, traps may bias observed habitat selection due to imperfect detection and from luring animals into poor quality habitats. Radio-telemetry can be used to estimate habitat selection without the biases of trap response (Douglas 1989). McCleery et al. (2006a) used radio-telemetry to estimate selection of hammock age class by foraging individuals but measured availability at the scale of the study area and did not infer fine-scale selection. Microhabitat selection by KLWR has yet to be evaluated using radio-telemetry and could provide useful information on important habitat and resources.

Identifying habitat selection is critically dependent on accurate and biologically meaningful definitions of availability (Johnson 1980, McClean et al. 1998). Within areas defined as available to an individual, it is typically assumed that resources are equally available. The distribution of resources relative to the individual is not considered. For central place foragers, resources farther from the central place may be less available, as predation risk and energy expended increase with increasing distance from the central place (Orians and Pearson 1979). Due to increased predation risk at far distances, central-place foraging theory predicts individuals will be more selective of resources at greater distances from the central place (Schoener 1979). *Neotoma* species are nocturnal herbivores which exhibit central place foraging behavior (McGinley 1984). Woodrats typically maintain one or more nests that serve as daytime refugia and bases for nightly foraging bouts. Little information is available on KLWR foraging behavior, but individuals are known to maintain small home ranges, incorporating one or several nests (McCleery et al. 2006b). If both availability and selectivity vary with distance from the nest, habitat selection models incorporating distance of foraging locations along with habitat should be more informative (Rosenberg and McKelvey 1999).

Nest Site Selection

Nest sites are critical resources for KLWR and are used as protective cover and refugia during inactive periods (McCleery et al. 2006b; Goodyear, unpublished report). KLWR use a variety of nest types including large free-standing stick nests, natural cavities in the limestone substrate, root systems of large trees, and artificial substrate, such as rock and debris piles (McCleery et al. 2006b; Goodyear, unpublished report). Differential use of nest substrate was noted in different hammock age classes, with free-standing stick nests found less commonly in mature hammock where natural cavities created by large trees were more available (Goodyear, unpublished report). More recently, free-standing stick nests were found to be absent form north Key Largo, with KLWR restricting use to natural and artificial cavities for nest sites (McCleery et al. 2006a). The majority of nests consisted of artificial substrate and occurred more frequently in younger hammock (McCleery et al. 2006b). It was suggested that young hammock was a preferred habitat for nesting (McCleery et al. 2006b). Higher proportional use of artificial nest substrate may be indicative of a lack of quality nest sites naturally available. In addition, artificial substrate is likely to occur in recently disturbed areas, and may, therefore, be confounded with young hammock. Distinguishing between young hammock and artificial nest substrate as preferred resources is critical since management recommendations differ greatly for each.

Monitoring a Rare Population

In an effort to prevent extinction, the U.S. Fish and Wildlife Service (1999) drafted a recovery plan outlining priority research and management goals. In addition to improving habitat quality, 2 components of the recovery plan are to determine current status of the population and monitor population trends over time. Monitoring abundance of KLWR allows managers to evaluate the effectiveness of habitat improvement and is essential feedback for successful implementation of an adaptive management plan (Hollings 1978, Walters 1986).

Recent research suggests KLWR occupies approximately 20% of its available range (170 ha, McCleery et al. 2006b). Rare species, such as KLWR, are difficult to sample using traditional designs (i.e., simple or stratified random sampling), with low capture rates resulting in imprecise estimates of abundance even with relatively large sample sizes. In addition to being rare, KLWR uses artificial substrate (i.e., rock and debris piles) for nest sites at a high frequency (McCleery et al. 2006b). Rock and debris piles are the result of dredging and illegal dumping along abandoned roads and tend to be clustered in the landscape. The high frequency of use of this artificial resource provides anecdotal evidence that KLWR may exhibit a clustered distribution.

Adaptive cluster sampling (ACS) was developed to provide a more efficient means of sampling rare and clustered populations (Thompson 1990). After an initial random sample is taken, additional units are selected from all immediately surrounding units once a pre-defined criterion (e.g., when 1 individual is detected) is met. The result is a cluster of sampling units around the original randomly selected unit, known collectively as a 'network.' Adaptively selected units not meeting the criteria are considered edge units and are not used in the analysis (Thompson 1990). A modified version of the Horvitz-Thompson estimator is used to account for the unequal inclusion probabilities of sampling units (Thompson 1990). ACS can also be used when the initial sample is selected using stratified random sampling (SRS, Thompson 1991), which may further lower the variance of an estimate if biologically meaningful strata can be identified. The high frequency of use of artificial substrate by KLWR, suggests areas delineated by proximity to, or density of these resources, may be an effective means of stratifying and reducing estimator variance. In the absence of artificial substrate, large trees are typically used for nests (McCleery et al. 2006b), making more mature patches of forest, lacking rock and debris piles, a potentially useful stratum. With several options for stratifying, post-stratification is effective for evaluating multiple designs, differing in strata delineation, on lowering estimator variance (Thompson 2002).

Although SRS and ACS are design-unbiased estimates of a population mean, imperfect detection and the mobility of animals can introduce bias. Individual detection rates typically are <1.0 for most animals and result in underestimates of abundance. Mark-recapture models are a common and effective means of estimating detection rate (Otis et al. 1978, Williams et al. 2002) and were used successfully in past KLWR population studies (Humphrey 1988, McCleery et al. 2006b). Estimates of detection rate can be used to correct the bias of designed-based estimates of ACS and SRS (Thompson and Seber 1994).

Individual mobility inflates abundance estimates by over-estimating density, a result of the effective detection area of a sampling unit being larger than the geographical boundary of the unit (Wilson and Anderson 1985). For small mammal studies employing trapping grids, the nested-grid approach is a common method to account for positive bias (Wilson and Anderson 1985, Humphrey 1988). The nested-grid approach requires high target animal densities, large grids, and is cumbersome when employing ACS, which results in networks varying in shape.

Alternatively, radio-telemetry can be used to estimate the proportion of individuals with nests occurring outside the geographic boundary of the sampled area. Similar to estimating survival, the binomial likelihood can be used to estimate the proportion of transient individuals (Williams et al. 2002). The design-based estimate can then be corrected for bias by subtracting the proportion of transients from the naïve estimate of abundance.

OBJECTIVES

The objectives of this study were to:

1. Evaluate factors affecting the distribution of KLWR throughout the hardwood hammocks of north Key Largo using multiple competing models in an information-theoretic framework.
2. Evaluate microhabitat and nest site selection by KLWR. Microhabitat selection was examined using multiple competing models, with and without distance and a distance-habitat interaction. Two models were examined to compare the importance of young hammock and the availability of artificial nest substrate on nest site selection by KLWR.
3. Evaluate the efficiency of adaptive cluster sampling relative to stratified random sampling. Poststratification was used to evaluate the effect of 3 strata delineations on lowering estimator variance. Collectively results were used to recommend an optimal sampling design to monitor abundance of KLWR.

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CHAPTER 2

AN EVALUATION OF FACTORS RESTRICTING THE DISTRIBUTION OF THE KEY LARGO WOODRAT[1]

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[1] Winchester, C., M. T. Mengak, and S. B. Castleberry. To be submitted to Journal of Wildlife Management

ABSTRACT

The Key Largo woodrat (KLWR; *Neotoma floridana smalli*) is a federally endangered species endemic to Key Largo, Florida. Since its listing in 1984, the population has continued to decline, presumably in response to reduced habitat quality, and appears to be absent from approximately 80% of the remaining forested uplands of Key Largo. Factors restricting the distribution of KLWR are not fully understood, making development of management actions needed to recover the population difficult. The goal of this study was to evaluate habitat selection at the macro-scale to identify variables restricting the presence of KLWR throughout its available range. We formulated 3 hypotheses on mechanisms restricting the distribution of KLWR: 1) availability of quality nest habitat 2) increased abundances of mammalian predators 3) decreased quality of foraging habitat. We developed multiple, a priori models to examine the effects of variables related to each hypothesis on KLWR habitat selection. We surveyed 40, 0.56 ha sampling units with live-traps for the presence of KLWR in the summer, fall, and winter of 2005. We measured habitat variables associated with each unit and analyzed data using logistic regression in an information-theoretic framework. We estimated occupancy of KLWR to be 22.8% (SE = 6.7). Models hypothesizing KLWR was restricted by the availability of nest sites and high predator abundances were supported by the data. The abundance of natural (large overstory trees) and artificial (rock and debris piles) nest substrate were the most important predictor variables, followed by indices of feral cat and raccoon abundance. We used multi-model inference to develop a resource selection probability function (RSPF), which predicted KLWR were more likely to occur in areas with large overstory trees, higher densities of debris piles, and lower abundances of mammalian predators. The RSPF can be used to identify priority areas requiring habitat improvement to assist in the recovery of the population. The continued

protection of upland forests from anthropogenic disturbances should benefit KLWR by increasing the availability of large overstory tress used as nest sites. However, our data suggest that immediate recovery efforts should focus on providing artificial nest substrate to supplement the lack of naturally occurring nest sites and expedite the recovery of this endangered species.

INTRODUCTION

The Key Largo woodrat (KLWR; *Neotoma floridana smalli*) is a federally endangered subspecies of the eastern woodrat (*Neotoma floridana*) endemic to Key Largo, Florida. KLWR historically ranged throughout the hardwood hammocks of Key Largo but is now restricted to federal- and state-managed lands on the northern one-third of the island (Barbour and Humphrey 1982). Despite the protection of its remaining habitat from development, the KLWR population continued to decline after its listing in 1984 (McCleery et al. 2006b). The cause of the population decline is unknown, but the quality of existing habitat appears severely compromised.

Past land use, such as farming and abandoned development projects, altered the structure and composition of the forested landscape. *Neotoma floridana* are nocturnal herbivores and exhibit central-place foraging behavior (McGinley 1984). As a central place forager, KLWR require suitable nest sites for cover and quality foraging habitat within close proximity to the nest to minimize predation risk and optimize resource acquisition (Orians and Pearson 1979). Recent changes in forest structure and composition on Key Largo maybe limiting the quality of available nest sites and foraging habitat. Naturally occurring nest sites used by KLWR include the root systems of large standing or wind-blown trees, logs, and solution holes (McCleery et al. 2006b; Goodyear, unpublished report). Recently, KLWR was found to use artificial nest substrate such as rock piles created by abandoned development projects and trash from dumping activities

(McCleery et al. 2006b). Whether the observed pattern of nest use is opportunistic or indicative of a decrease in the quality of naturally occurring nest substrate is unknown.

Housing developments, roads, and a waste transfer station, within and around the remaining hardwood hammock, resulted in habitat fragmentation and increased anthropogenic food resources for raccoons (*Procyon lotor*) and feral cats (*Felis catus*), potentially increasing their densities (Hoffman and Gottschanng 1977, Prange et al. 2005). Feral cats are well documented predators of small mammals (Baker et al. 2005, Espinosa-Gayosso and Alvarez-Castaneda 2006) and raccoons have been shown to depredate small mammals, including Norway rats (*Rattus norvegicus*; Hoffman and Gottschanng 1977). Increased predation levels due to elevated predator densities could be contributing to the KLWR population decline. The increase in forest edge resulting from fragmentation has facilitated the establishment of red imported fire ants (*Solenopsis invicta*), which favor disturbed areas lacking canopy cover (Forys et al. 2002). Past studies found small mammals altered patterns of habitat selection to avoid fire ants (Smith et al. 1990, Killion and Grant 1993, Killion et al. 1995, Holtcamp et al. 1997). Although the nocturnal foraging strategy employed by KLWR may limit interactions with diurnal fire ants, food-caches and young may be susceptible to fire ant invasions of KLWR nests.

Past studies examining habitat use by KLWR focused on use of hammock at different stages of recovery from anthropogenic disturbances. Results were conflicting, with most finding KLWR more abundant in mature hammock (Hersh 1978, Barbour and Humphrey 1982; Goodyear, unpublished report), but other studies finding higher abundance in young hammock (McCleery et al. 2006b), or equal abundance among all hammock age classes (Keith and Gaines 2002, Sasso and Gaines 2002). The discrepancy in results from past research creates uncertainty as to the relevance and predictive ability of using forest age class, a course-grained variable, to identify habitat for KLWR. Given the uncertainty surrounding habitat requirements of KLWR and the factors contributing to its decline, additional information on habitat use is needed to assess the quality of existing habitat and effectively guide the recovery of the population.

The goal of this study was to identify factors restricting the presence of KLWR throughout their available range by evaluating habitat selection using an information-theoretic approach with multiple, competing models (Burhnam and Anderson 2002). A priori models were developed under 3 general hypotheses on factors restricting KLWR, which included a decrease in the availability of quality nest sites (McCleery et al. 2006b; Goodyear, unpublished report), increased predation from raccoons and feral cats (U. S. Fish and Wildlife Service 1999, B. Muzniaks, USFWS personal communication), and decreased availability of quality foraging habitat (Hersh 1978, Orians and Pearson 1979, Barbour and Humphrey 1982; Goodyear, unpublished report).

STUDY AREA

Our study area was defined as all upland forest habitat (hardwood hammocks) occurring on the Crocodile Lake National Wildlife Refuge and Dagny Johnson Key Largo State Botanical Preserve (Figure 2.1). The hardwood hammocks of this region are characterized as a closed canopy forest with a limestone substrate and a diverse assemblage of evergreen and semi-deciduous tree and shrub species (U.S. Fish and Wildlife Service 1999). The hardwood hammocks are one of the most species rich forests of North America with over 150 known tree and shrub species (Ross et al. 1992). Common canopy species include pigeon plum (*Coccoloba diversifolia*), poisonwood (*Metopium toxiferum*), west-indian mahogany (*Swietenia mahagoni*), wild tamarind (*Lyisloma latisiliguum*) and gumbo-limbo (*Bursera simaruba*). Common sub-canopy species include crabwood (*Gymnanthes lucida*), white stopper (*Eugenia axillaries*), marlberry (*Ardisia escallonioides*) and wild lime (*Zanthoxylum flavum*). Canopy closure creates a shady environment on the forest floor resulting in a sparse shrub and herb layer. As a result, the understory consists mainly of seedlings and saplings of canopy and sub-canopy species, however, wild coffee (*Psychotria nervosa*) and snowberry (*Chiococca alba*) are found commonly in the understory (U. S. Fish and Wildlife Service 1999). The hammock grades into mangroves along the coast creating a relatively shallow transitional zone that is comparatively open with high vine growth and numerous thorny plant species (Ross et al. 1992).

METHODS

Presence/Absence Determination

The 850-ha study area was divided into 0.56 ha sampling units in ArcGIS (Environmental Systems Research Institute, Redlands, CA), from which 40 were selected via a stratified random design. Prior to trapping we systematically surveyed the study area and recorded the location of rock and debris piles that were available as nest substrate with a Global Positioning System (GPS; Trimble GeoXT, Sunnyvale, California, USA). Because rock or debris piles have been shown to be important substrates for KLWR nest sites (McCleery et al. 2006b), we defined sampling units ≤ 75 m and >75 m from a rock or debris pile as strata for allocating sampling effort. Based on previous estimates of KLWR movements (McCleery et al. 2006a), 75m was believed to be a conservative distance and maximized the chance that an individual would be captured if using nearby artificial nest substrate. The number of sampling units was selected in proportion to the area available in each stratum, such that each unit had an equal inclusion probability. Within each of the 40 sampling units, we placed a trapping grid with 9 stations, in a 3 x 3 array, with 25 m spacing between stations. At each station, we placed two 10.2 x 11.4 x 38.1 cm, vented, Sherman traps with raccoon-proof door latches (model PXLF15, H. B. Sherman

Traps Inc., Tallahassee, Florida). Traps were baited with peanut butter and crimped oats. We opened traps for 4 consecutive nights checking each trap daily within the first three hours after sunrise. All captured individuals were marked with passive integrated transponders (PIT) tags (BioMark, Boise, Idaho) and #1005 Monel ear tags (National Band and Tag, Newport, Kentucky). Each grid was sampled three times (April-May, August-September, and November-December) to account for temporal variation in habitat use. A sampling unit was defined as 'used' with the presence of one unique individual on at least one of the three sampling occasions. All capture and handling of KLWR was conducted under the Federal Fish and Wildlife endangered species permit # TE0959080-1 , the special purpose permit from the State of Florida Fish and Wildlife Conservation Commission # WX05089, the research and collection permit from Florida Department of Environmental Protection # 5-05-41, and Institutional Animal Care and Use (IACUC) permit number (A2005-10044-0).

Habitat Variables

We measured 10 variables associated with each trapping grid to characterize the availability of nest substrate, quality of foraging habitat, and predator activity (Table 2.1). Prior to trapping we systematically surveyed the study area and recorded the location of rock and debris piles that were available as nest substrate with a Global Positioning System (GPS; Trimble GeoXT, Sunnyvale, California, USA). In ArcGIS, we calculated density of debris piles within 100 m of the center of each trapping grid. A 100 m buffer was chosen to include nest substrate likely available to trapped individuals based on KLWR home range estimates (McCleery et al. 2006b). We characterized structure and composition of vegetation at the 4 corner and center trap stations of each grid. We measured the density of stems in the understory, midstory, and overstory using the point-centered quarter method (Cottam and Curtis 1956). We defined understory stems as those with a diameter of 1.0 -5.0 cm, measured at 4.0 cm from the base of the stem. We defined midstory and overstory stems as having a diameter at breast height (DBH; 1.4 m from the base) of 5.1-10.0 cm and >10.1 cm respectively. We calculated mean DBH of midstory and overstory stems for each trapping grid. In the analysis, only density of understory stems was used because density and DBH were correlated within each size class. Mean DBH of overstory stems was used separately as a metric for natural nest substrate and food availability. We calculated species richness as the estimated number of plant species, in all size classes, occurring on the trapping grid using program SPECRICH (Hines 1996).

We indexed fire ant activity on each trapping grid by placing canned sardines on aluminum foil at each of the 9 trap stations. We baited stations in the morning between 0700 - 1000 and returned 3 hours later to record the presence or absence of fire ants. We surveyed each station for 3 consecutive days and conducted fire ant surveys within 30 days after the completion of the August and November KLWR trapping session. We calculated fire ant activity for each grid as the mean number of detections of fire ants from 54 presence/absence observations, resulting in an index value occurring between 0 (0/54) -1 (54/54).

We indexed raccoon and feral cat abundance by installing 37 scent stations opportunistically throughout the study area, utilizing old roads and clearings that were free of understory vegetation and easily accessible. We used fine-grained sand placed in a 1.5 m square to record raccoon and feral cat tracks. We baited stations with canned sardines in the early evening and recorded the presence or absence of feral cat and raccoon tracks the following morning. We baited each station for 3 days in the summer (between July 9 -24), fall (between September 7 -17) and winter (between November 23 – January 15), within 60 days following each KLWR trapping session. All stations in the summer and winter seasons were surveyed three times each. Due to frequent rains and time constraints during the fall survey, 11 stations were surveyed 3 times, 17 stations twice, 8 stations once, with 1 station not surveyed. For each station we indexed abundance of feral cats and raccoons separately as the sum of the mean number of detections in each season, multiplied by the mean number of detections between seasons. This resulted in an abundance index of 0-3 for each station. For each trapping grid, the index of raccoon and feral cat abundance was calculated as the mean abundance value of stations occurring within 800 m of the center of the grid. We chose an 800 m buffer around trapping grids to be consistent with published accounts of raccoon and feral cat home range estimates in an urbanized environment (Molsher et al. 2005; Chamberlain and Leopold 2002). Using ArcGIS we calculated distance to nearest anthropogenic food source (e.g., the waste transfer station and housing developments on the southern and northern edges of the study area) and identified the hammock age class of the trapping grid location. Hammock age classes were defined as mature (disturbed before 1940), medium (disturbed from 1940–1971) and young (disturbed after 1971) in a previous studies (McCleery et al. 2006b, Ross et al., unpublished report) and were available as a GIS layer from the U.S. Fish and Wildlife Service.

Statistical Analysis

We developed 20 a priori models, under 3 general hypotheses, regarding factors restricting the distribution of KLWR, as well as, a model with hammock age class as a single covariate for comparison with previous studies (Table 2.2). Models were based on published and unpublished accounts of habitat use by KLWR and habitat selection theory (Orians and Pearson 1979, U. S. Fish and Wildlife Service 1999, McCleery et al. 2006b; Goodyear, unpublished report). Models developed under the nest availability hypothesis (n = 3) examined the individual and combined effects of the availability of the 2 most frequently used types of nest substrate, overstory trees and rock and debris piles (McCleery et al. 2006b). Models developed under the increased predation hypothesis (n = 12) examined the individual and combined effects of feral cat and raccoon abundance indices, suspected predators of KLWR (Brown 1978, U. S. Fish and Wildlife Service 1999), along with metrics of escape cover. Models developed under the reduction in quality foraging habitat hypothesis (n = 5) examined metrics of food availability, ground cover, and fire ant activity as factors affecting optimal foraging (Orians and Pearson 1979). Variables were tested for co-linearity using PROC CORR (SAS Institute 1999). Correlated variables (Pearson's correlation coefficient > 0.5, $P \leq 0.05$) were not included in the same model, with the most biologically meaningful variable retained in the model. Data were analyzed using a logistic regression in SAS (PROC LOGISTIC; SAS Institute 1999). We evaluated goodness-of-fit for the most general model using the Hosmer-Lemeshow test (Hosmer and Lemeshow 2002). We ranked candidate models by calculating Akaike's Information Criterion adjusted for small sample size (AICc; Akaike 1973, Hurvich and Tsai 1989) and model weights (w_i; Burnham and Anderson 2002). The top models were defined as those with a ΔAIC_c value ≤ 4 ($\exp(-1/2\Delta_i) = 0.135$; Burnham and Anderson 2002). We estimated the amount of variation in the data accounted for by the top models by calculating Nagelkerke's R^2 value. We evaluated the importance of individual parameters by summing w_i for models containing each parameter (Burnham and Anderson 2002). We used Moran's I in ArcGIS to test for the presence of spatial autocorrelation in the top models. Multi-model inference was used to develop a Resource Selection Probability Function (RSPF) that took the form of the logistic model (Manly et al. 2002):

$$w^*(x) = \exp(ß_0 + ß_1x_1 + ß_2x_2 + \ldots ß_ix_i) / 1 + \exp(ß_0 + ß_1x_1 + ß_2x_2 + \ldots ß_ix_i)$$

Where w*(x) is the estimated probability of use of a grid given a combination of habitat covariates (x_i) and their respective, model averaged, regression coefficients ($ß_i$).

To examine the effects of imperfect detection on estimates of resource selection, we estimated the proportion of sites occupied (Ψ) and detection probability (p) using methods developed by MacKenzie et al. (2002). Detection histories were compiled over a 4-day interval for each of the 40 sampled units. For those units where KLWR was detected in multiple seasons we used the detection history for the first season in which it was encountered. We estimated Ψ and p using the six default models in program PRESENCE 2.0 (Hines 2006) which allow for group and time effects.

RESULTS

We detected KLWR on 9 of the 40 trapping grids over the three sampling sessions yielding a naïve occupancy estimate of 0.225. KLWR was detected on 7, 9 and 4 trapping grids in the April, August and November sampling periods, respectively. A total of 15 individuals were captured, 10 on grids (n = 5) within 75 m of a rock or debris pile. The occupancy model with the lowest AIC value ($w_i = 0.681$) had one group with constant *p* across the 4 survey occasions and estimated $\Psi = 0.228$ (SE = 0.067) and p = 0.658 (SE = 0.083). The next best model ($\Delta AIC = 2.73$; $w_i = 0.174$) with 2 groups and a survey-specific p estimated $\Psi = 0.227$ (SE = 0.067) and $p_1 = 0.551$ (SE = 0.166), $p_2 = 0.661$ (SE = 0.158), $p_3 = 0.882$ (SE = 0.111), $p_4 = 0.551$ (SE = 0.166).

Raccoons were detected on 95%, 84%, and 87% scent stations during the summer, fall, and winter survey periods, respectively. Feral cats were detected on 22%, 54%, and 14% of scent stations during the summer, fall, and winter survey periods, respectively. The mean abundance indices for raccoons and feral cats were higher on grids where KLWR were absent.

Grids used by KLWR were farther from anthropogenic food sources than unused grids (Table 2.3).

Fire ants were detected on 19 grids, 6 of which were utilized by KLWR, and fire ant activity was higher on unused grids (Table 2.3). Grids where KLWR were detected had larger overstory trees, higher densities of rock and debris piles, and lower densities of understory stems (Table 2.3). Mean DBH of midstory trees was similar for used and unused grids, while plant species richness was higher on unused grids (Table 2.3). Five used grids were in mature hammock, 3 in medium-aged hammock and 1 in young hammock. Three of the unused grids were in young hammock while 11 and 17 grids, respectively, were in medium-aged and mature hammock.

Six models were included in the confidence set ($\Delta AIC_c \leq 4.0$; Table 2.4). Model averaged regression coefficients that predict probability of use are positively correlated with increasing amounts of ROCK and DBH1, and negatively correlated with increasing values of CAT, RAC, and FOOD (Table 2.5). ROCK and DBH1 were the most important variables, followed by CAT, RAC, and FOOD (Figure 2.2). Based on mean w_i, models developed under the nest availability hypothesis were given the most support, followed by predation and foraging-habitat models (Figure 2.3).

Model-averaged parameter estimates from the confidence set were used to develop the following RSPF:

$$w^*(x) = \exp(-8.651683 + 0.319944(ROCK) + 0.474975(DBH1) + -2.695417(CAT) + -0.157564(RAC) + -0.000004(FOOD) / 1+ \exp(-8.651683 + 0.319944(ROCK) + 0.474975(DBH1) + -2.695417(CAT) + -0.157564(RAC) + -0.000004(FOOD)$$

The data in the best models were not spatially autocorrelated (Moran’s I index = $-0.05 \leq -0.11$).

DISCUSSION

KLWR was detected on 9 of the 40 sites surveyed with an estimated occupancy of 22.8%. The occupancy estimate was consistent with previous results (McCleery et al. 2006b) and supports previous research suggesting that KLWR distribution is limited within the remaining forested uplands of Key Largo. The presence of KLWR was best explained by the availability of natural and artificial nest substrate, where probability of use was positively correlated with increasing amounts of both artificial and natural nest substrate. Naturally occurring nest substrate utilized by KLWR include the root systems of large trees, both standing and felled, hollowed logs, and solution holes. Artificial nest substrate available to KLWR include remnant rock piles from dredging, rubble piles from demolished buildings, and trash piles (e.g., abandoned cars, washing machines, and roofing material) from illegal dumping. The most important variable predicting use was the abundance of artificial nest substrate, followed by the size of overstory trees. Results are consistent with a previous examination of KLWR nest preference in which rock and debris piles comprised 73% of all known nest sites and the root systems of trees and logs accounting for the remaining 27% (McCleery et al. 2006b).

Although stick piling is a common behavior, occasionally leading to large stick houses through successive generational use (Brown 1978), KLWR appear to rely heavily on artificial substrate for nest sites. The construction and use of free-standing stick nests was previously documented on Key Largo (Brown 1978; Goodyear, unpublished report). More recently, free-standing stick nests were found to be absent, with natural and artificial substrate, the sole resource for nest sites (McCleery et al. 2006b). The limited distribution of KLWR and the apparent reliance on artificial nest substrate suggests the quality of natural nest substrate may be deficient throughout much of their available range.

Models including indices of feral cat and raccoon abundance were supported by the data. Feral cats and raccoons are listed as potential predators of KLWR (Brown 1978, U. S. Fish and Wildlife Service 1999). Both species are common in north Key Largo, although our data demonstrate that raccoons are much more abundant than cats. The high raccoon abundance index in the hardwood hammocks is likely influenced by the presence of anthropogenic food sources available through the waste transfer station and neighboring residential communities (Hoffman and Gottschanng 1977). A model including distance to food source, along with metrics of nest substrate availability, was included in the confidence set, but the parameter estimate was very small and did not have much influence on probability of use. Feral cats likely benefit from the same food resources (Prange et al. 2005), but are also purposefully maintained by residents of Key Largo. The relatively high abundance of raccoons and feral cats may be influencing the observed reliance on artificial nest substrate, which offer greater protection than free-standing stick nests, as well as, additional cover for avoiding predators while foraging. The inclusion of raccoon and feral cat abundance indices, along with metrics of nest substrate availability, in models in the confidence set provides evidence that increased predation and decreased nest availability may be acting in concert as limiting factors.

Previous studies of KLWR habitat use focusing on hammock age class as a predictor variable yielded inconsistent results (Hersh 1978, Barbour and Humphrey 1982, Keith and Gaines 2002, Sasso and Gaines 2002, McCleery et al. 2006b; Goodyear, unpublished report). In our study, the model with hammock age class as a single covariate was not supported by the data, and KLWR were detected in all three age classes. In addition, mean DBH of overstory trees was a better predictor variable than age class, suggesting that KLWR selects areas with larger overstory trees, a characteristic of more mature hammock. In the absence of rock and debris piles, KLWR was only detected in forest patches characterized as mature. The presence of KLWR in younger age classes may be driven more by the availability of artificial nest substrate than the structure of the vegetation. McCleery et al. (2006b) found higher abundances of KLWR in young hammock and speculated that young hammock was preferred due the high density of ground cover, although a statistically significant difference in ground cover was not detected between the age classes. Young or recently disturbed patches of hammock are the result of human activities (i.e., farming and development) and are typically accompanied by rock and debris piles resulting from the activity. Therefore, the observed use of this habitat type may be confounded with the high availability of artificial nest substrate in recently disturbed hammock. In the absence of artificial substrate it seems unlikely KLWR could inhabit young hammock with few naturally occurring nest sites (i.e., large overstory trees) given their apparent reliance on nest substrate. Furthermore, density of understory stems was not included in the top models and does not appear to be restricting the distribution of KLWR. Dense ground cover is not common in the hardwood hammocks, which are characterized by a closed canopy (U. S. Fish and Wildlife Service 1999, Ross et al., unpublished report). The overall lack of dense ground cover in the hammocks may further emphasize the importance of nest substrate for avoiding predators. Despite the observed relationship with large overstory trees, occupancy in mature hammock was relatively low, suggesting that the availability of quality natural nest substrate may be lacking.

Additional variables examined in this study included the activity of fire ants, plant species richness, and the DBH of midstory trees. None of the models containing these variables were supported by the data, and therefore, do not appear to be restricting the distribution of KLWR. Fire ants were commonly observed in areas where KLWR was present, particularly disturbed areas where artificial nest substrate was available. A preference for disturbed areas was noted in a previous fire ant study on Key Largo (Forys et al. 2002) and may increase the probability of an interaction with KLWR in areas where artificial nest substrate is abundant. Because fire ants are diurnal, they may pose few problems for the nocturnally foraging KLWR. However, it seems plausible that some level of interaction is occurring in KLWR nests where individuals cache food and rear young. Fire ants are known to invade bird and sea turtle nests and have been suspected of lowering reproductive success (Allen et al. 2001, Kopachena et al. 2000). The potential for an interaction between fire ants and KLWR nests may warrant further research, particularly regarding the survival and fecundity of individuals in areas with high fire ant activity. The DBH of midstory trees and plant species richness were included in the analysis as metrics of abundance and diversity of food items. The relatively high diversity of plant species and year-round food availability in the hammock likely results in a food-rich environment for KLWR, however, selection of food resources may still occur at a finer spatial scale, while not restricting distribution.

MANAGEMENT IMPLICATIONS

The distribution of KLWR appears to be limited by the availability of nest sites. The relatively high abundances of mammalian predators may have increased the importance of artificial nest substrate. Results indicate KLWR would benefit from continued protection of the hammocks from human activities which would increase the availability of natural nest substrate as the forest matures. However, immediate habitat improvement requires the addition of artificial substrate. The addition of artificial nest substrate could be employed, along with predator reductions, in a two-way factorial design, under the framework of adaptive management, to better gauge the effectiveness of the individual and combined effects of nest substrate availability and predator abundances on the abundance and distribution of KLWR. The RSPF developed in this study can be used to identify areas in need of habitat improvement.

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Table 2.1. Variables measured at Key Largo woodrat trapping grids distributed throughout north Key Largo, Florida, 2005.

Habitat Variable	Acronym
DBH[a] trees > 10cm	DBH1
DBH trees 5.1 – 10cm	DBH2
Density of understory stems (stems per ha)	COVER
Plant species richness	SPECR
Density of rock/debris piles in 100m radius	ROCK
Fire ant activity	ANT
Index of raccoon abundance	RAC
Index of cat abundance	CAT
Distance to anthropogenic food source (km)	FOOD
Forest age class (young, medium, or old)	AGE

[a] Diameter at breast height measured 1.4m from the base

Table 2.2. A priori models developed to examine factors restricting the distribution of the Key Largo woodrat in north Key Largo, Florida, USA, 2005. The general hypotheses used to develop each model are listed with covariates included in each model.

Hypotheses/Models			
Availability of nest substrate	Predation	Foraging Habitat	Forest Age Class
ROCK	RAC	ANT	AGE
DBH1	CAT	SPECR	
ROCK DBH1	RAC CAT	DBH1 DBH2	
	RAC ROCK DBH1	COVER	
	CAT ROCK DBH1	ANT SPECR DBH1 DBH2	
	RAC CAT ROCK DBH1		
	RAC COVER		
	CAT COVER		
	RAC CAT COVER		
	FOOD		
	FOOD ROCK DBH1		
	FOOD COVER		

Table 2.3. Summary of mean values and SE for habitat variables measured in association with used and unused grids by KLWR during 2005 in north Key Largo, Florida.

	Used (n=9)	Unused (n=31)
Variable	Mean (SE)	Mean (SE)
DBH1	6.70 (0.17)	6.85 (0.07)
DBH2	16.69 (2.98)	15.25 (1.63)
COVER	4171.92 (657.25)	5100.42 (478.23)
SPECR	16.62 (1.85)	18.85 (1.75)
ROCK	5.00 (1.92)	0.87 (0.44)
ANT	0.89 (0.31)	2.13 (0.64)
RAC	1.85 (0.13)	2.14 (0.09)
CAT	0.19 (0.03)	0.27 (0.05)
FOOD	2.14 (0.39)	1.38 (0.22)

Table 2.4. Variables, number of parameters in each model (K), Akaike's Information Criterion adjusted for small sample size (AIC_c), difference of AIC_c between a model and the model with the lowest AIC_c (ΔAIC_c), model weights (w_i), and Nagelkerke's R^2 values for models with $\Delta AIC_c \leq 4.0$ for 21 a priori models used to evaluate factors restricting the distribution of the Key Largo woodrat in north Key Largo, Florida, USA, 2005.

Model	K	AIC_c	ΔAIC_c	w_i	R^2
CAT ROCK DBH1	4	36.28	0.00	0.32	0.49
ROCK DBH1	3	36.49	0.21	0.29	0.42
RAC ROCK DBH1	4	38.48	2.20	0.11	0.43
RAC CAT ROCK DBH1[a]	5	38.48	2.21	0.11	0.50
FOOD ROCK DBH1	4	38.95	2.68	0.08	0.42
ROCK	2	39.85	3.57	0.05	0.25
DBH1	2	43.75	7.48	0.01	
FOOD	2	44.27	7.99	0.01	
DBH1 DBH2	3	44.31	8.03	0.01	
RAC	2	44.48	8.20	0.01	
RAC CAT	3	44.89	8.62	0.00	
ANT	2	45.53	9.25	0.00	
COVER	2	45.94	9.66	0.00	
CAT	2	45.98	9.70	0.00	
FOOD COVER	3	46.24	9.96	0.00	
RAC COVER	3	46.38	10.11	0.00	
SPECR	2	46.48	10.20	0.00	
AGE	2	46.98	10.70	0.00	
RAC CAT COVER	3	47.07	10.79	0.00	
CAT COVER	2	47.15	10.88	0.00	
ANT SPECR DBH1 DBH2	5	48.16	11.88	0.00	

[a] Hosmer and Lemeshow goodness-of-fit $\chi^2 = 5.39$, $p > 0.72$

Table 2.5. Model-averaged parameter estimates (β) and unconditional SE for habitat variables in models included in the confidence set ($\Delta AIC_c \leq 4.0$), selected from 21 a priori models on factors restricting the distribution of KLWR in Key Largo, Florida.

Variable	β	SE
Intercept	-8.651683	4.777625
ROCK	0.319944	0.142838
DBH1	0.474975	0.281379
CAT	-2.695417	5.666697
RAC	-0.157564	0.797948
FOOD	-0.000004	0.000082

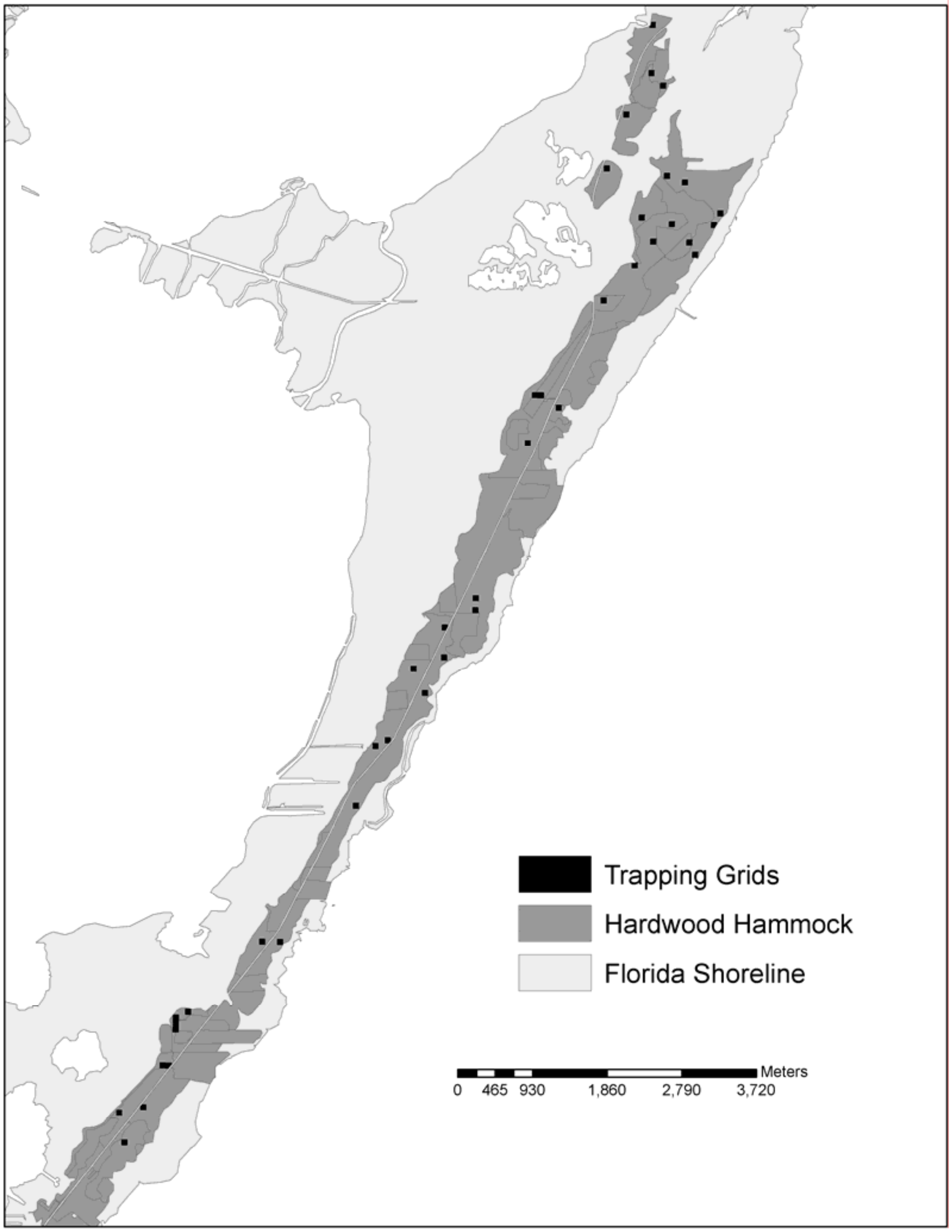

Figure 2.1. Map of study area delineating hardwood hammock from mangroves of north Key Largo, Florida, USA comprising the Crocodile Lake National Wildlife Refuge and Dagny Johnson Key Largo State Botanical Preserve and locations of 40 trapping grids used to estimate occupancy and evaluate factors restricting the distribution of the Key Largo woodrat in 2005.

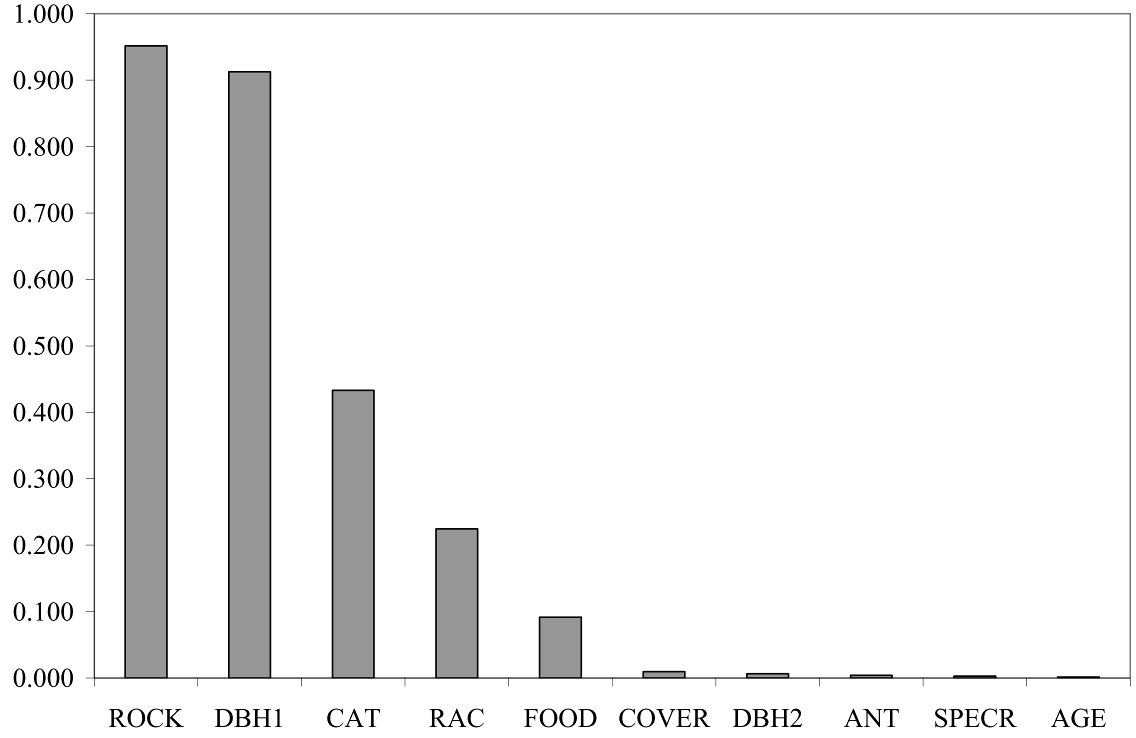

Figure 2.2. Sum of model weights (w_i) for 10 habitat variables included in 21 a priori models developed to evaluate factors restricting the distribution of the Key Largo woodrat in north Key Largo, Florida, USA, 2005.

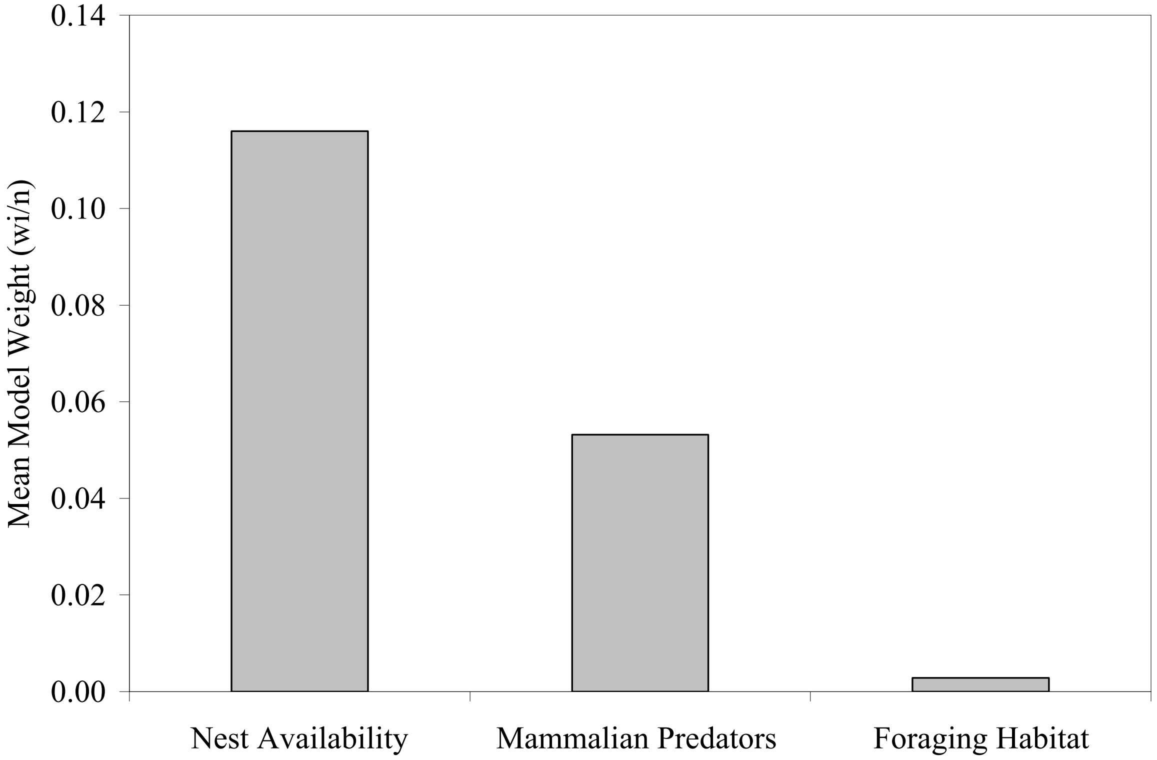

Figure 2.3. Mean model weights (w_i) for 21 a priori models developed under 3 general hypotheses on factors restricting the distribution of the Key Largo woodrat in north Key Largo, Florida, USA, 2005.

CHAPTER 3

FORAGING MICROHABITAT AND NEST SITE SELECTION BY THE ENDANGERED KEY LARGO WOODRAT[1]

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[1] Winchester, C., M. T. Mengak, and S. B. Castleberry. To be submitted to Journal of Mammalogy.

ABSTRACT

Population size of the Key Largo woodrat (KLWR; *Neotoma floridana smalli*), a federally listed species, has declined precipitously over the last 20 years. To recover the population, additional information on microhabitat and nest site selection is required. In 2005-2006, we used radiotelemetry to evaluate microhabitat selection by comparing structure and composition of vegetation in known foraging areas to the area available to foraging woodrats. We also evaluated nest site selection by comparing the abundance of artificial nest substrate and hammock age in the vicinity of identified nests with random locations. Data were analyzed with logistic regression and within an information-theoretic framework. There was little support for selection of foraging areas based on the structure and composition of vegetation. KLWR primarily used artificial substrate for nests (82%), but also used natural nest substrate (18%), mainly the root systems of overstory trees. KLWR selected nest sites in areas with higher abundance of artificial nest substrate and higher percentage of younger hammock. However, hammock age alone was a poor predictor of nest site selection. KLWR appear to be generalists with respect to microhabitat, but display a high degree of selection for artificial nest substrate, which may offer greater protection from predators. The results of our study suggest the quality of KLWR habitat would be improved by the addition of artificial nest substrate and the continued protection of the hammocks from human activities to increase the availability of natural nest substrate.

INTRODUCTION

The Key Largo woodrat (KLWR; *Neotoma floridana smalli*) is a federally listed endangered subspecies of the eastern woodrat (*Neotoma floridana*) endemic to Key Largo, Florida. KLWR historically ranged throughout the hardwood hammocks of Key Largo, but due to habitat loss, is restricted to federal and state managed lands on the northern one-third of the island (Barbour and Humphrey 1982). Despite the protection of its remaining habitat from development, the KLWR population has continued to decline and is currently at a high risk of extinction (McCleery et al. 2005). The cause of the population decline is unknown, but may be related to degraded quality of existing habitat.

Prior to protection, the hardwood hammocks of north Key Largo underwent a variety of human-induced disturbances (Ross et al., Florida International University, unpublished data; A. H. Wiener, The National Audubon Society, unpublished data). Farming, fires, and abandoned development projects altered the structure and composition of the forested landscape, possibly lowering the quality of habitat for KLWR. As a result, the hardwood hammocks currently are composed of patches differing in seral age. Heterogeneity in hammock age was the focus of past studies examining KLWR habitat use, which were conducted at relatively large spatial scales (Brown 1978, Hersh 1978, Barbour and Humphrey 1982, Keith and Gaines 2002, Sasso and Gaines 2002, McCleery et al. 2006a). Results of previous studies examining KLWR habitat selection of hammock age classifications were conflicting, with some finding KLWR more abundant in mature hammock (Hersh 1978, Barbour and Humphrey 1982), but other studies finding higher abundance in young hammock (McCleery et al. 2006a), or equal abundance among all hammock age classes (Keith and Gaines 2002, Sasso and Gaines 2002; N. C. Goodyear, North Key Largo Study Committee, unpublished data). Given the uncertainties of KLWR habitat requirements, additional information on habitat selection is required to improve habitat quality and effectively recover the population.

Animals select habitats at a variety of spatial scales (Johnson 1980, Weins 1989). Identifying microhabitat selection may provide valuable information on habitat preferences that are obscured at larger spatial scales (Castleberry et al. 2002). Sasso and Gaines (2006) evaluated KLWR microhabitat selection by comparing habitat around used and unused trap locations and suggested larger canopy trees and open understory were most important. However, traps may bias observed habitat selection due to imperfect detection and from attracting animals into poor quality habitats. Radiotelemetry can be used to estimate habitat selection without the biases of trap response (Douglas 1989). McCleery et al. (2006b) used radiotelemetry to estimate selection of hammock age class by foraging individuals but measured availability at the scale of the study area and did not infer fine-scale selection.

Identifying habitat selection, where use is disproportionate to availability, is critically dependent on accurate and biologically meaningful definitions of availability (Johnson 1980, McClean et al. 1998). Within areas defined as available to an individual, resources typically are assumed to be equally available. Under this assumption, the distribution of resources relative to the individual is not considered. For central place foragers, resources farther from the central place may be less available, as predation risk and energy expended increase with increasing distance from the central place (Orians and Pearson 1979). Due to increased risks at far distances, central-place foraging theory predicts individuals will select fewer resources that are at greater distances from the central place (Schoener 1979). *Neotoma* species are nocturnal herbivores which exhibit central place foraging behavior (McGinley 1984). Woodrats typically maintain one or several nests that serve as day-time refugia. Little information is available on KLWR foraging behavior but individuals are known to maintain small home ranges, incorporating one or several nests (McCleery et al. 2006a). If both habitat availability and selectivity vary with distance from the nest, habitat selection models incorporating distance of foraging locations along with habitat should be more informative (Rosenberg and McKelvey 1999).

Nest sites, which are used as protective cover and refugia during inactive periods, are critical resources for KLWR (McCleery et al. 2006a; N.C. Goodyear, North Key Largo Study Committee, unpublished data). KLWR use a variety of nest types, including large free-standing stick nests, natural cavities in the limestone substrate, root systems of large trees, and artificial substrate, such as rock and trash piles (McCleery et al. 2006a; N.C. Goodyear, North Key Largo Study Committee, unpublished data). In the early 1980's, when free-standing stick nests were more common, the abundance of stick nests was not equal among hammock age classes. Stick nests were found less commonly in mature hammock, where natural cavities created by large trees were more available (N.C. Goodyear, North Key Largo Study Committee, unpublished data). Recently, free-standing stick nests were found to be absent, with KLWR restricting use to natural and artificial cavities for nest sites (McCleery et al. 2006a). McCleery et al. (2006a) found the majority of nests consisted of artificial substrate and occurred more frequently in younger hammock, and suggested that young hammock was a preferred nesting habitat. However, higher proportional use of artificial nest substrate may indicate a lack of quality natural nest sites. In addition, artificial substrate is likely to occur in recently disturbed areas, and be confounded with younger patches of hammock. Distinguishing between hammock age and artificial substrate as resources selected for nest sites is critical for developing appropriate management actions. The goal of this study was to examine foraging microhabitat and nest site selection using an information-theoretic approach to identify important resources for KLWR and provide direction on improving habitat quality.

STUDY AREA

Our study area was defined as all upland forest habitat (hardwood hammocks) occurring on the Crocodile Lake National Wildlife Refuge and Dagny Johnson Key Largo State Botanical Preserve (Figure 3.1). The hardwood hammocks of this region are characterized as closed canopy forest with a limestone substrate, containing over 150 species of evergreen and semi-deciduous tree and shrub species (USFWS 1999). Common canopy species include pigeon plum (*Coccoloba diversifolia*), poisonwood (*Metopium toxiferum*), west-indian mahogany (*Swietenia mahagoni*), wild tamarind (*Lyisloma latisiliguum*) and gumbo-limbo (*Bursera simaruba*). Common sub-canopy species include crabwood (*Gymnanthes lucida*), white stopper (*Eugenia axillaries*), marlberry (*Ardisia escallonioides*) and wild lime (*Zanthoxylum flavum*). Canopy closure creates a shaded environment on the forest floor resulting in a sparse shrub and herb layer. As a result, the understory consists mainly of seedlings and saplings of canopy and sub-canopy species, however, wild coffee (*Psychotria nervosa*) and snowberry (*Chiococca alba*) are found commonly in the understory (USFWS 1999). The hammock grades into mangroves along the coast creating a relatively shallow transitional zone that is comparatively open with high vine growth and numerous thorny plant species (Ross et al. 1992).

METHODS

Capture and Handling

We used a combination of random and non-random sampling to capture KLWR (Figure 3.1). Using a stratified random design and proportional allocation, we established 40 trapping grids in 2 strata, delineated by distance (≤ 75 or > 75 m) to artificial nest substrate. Each grid consisted of 9 stations and in a 3 x 3 array with 25 m spacing. At each station we placed two 10.2 x 11.4 x 38.1 cm, vented Sherman traps with raccoon-proof door latches (Model PXLF15;

H. B. Sherman Traps Inc., Tallahassee, Florida). Traps were baited with peanut butter and crimped oats. We opened traps for 4 consecutive nights checking each trap daily within the first three hours after sunrise. Each grid was sampled three times (Apr-May, Aug-Sep and Nov-Dec) during 2005. We supplemented random sampling by targeting areas where KLWR were known to occur (McCleery et al. 2006a, B. Muznieks, U.S. Fish and Wildlife Service, personnel communication) and areas with signs of woodrat activity (i.e., stick piling). We placed 3 traps at the target location for 3 consecutive days and checked traps daily within the first 3 hours after sunrise. All captured individuals were sexed, weighed, and marked with passive integrated transponder (PIT) tags and #1005 Monel ear tags (National Band and Tag Company, Newport, KY). Individuals weighing ≥ 180 g were selected opportunistically for radio-collaring with the goal of maintaining an equal sex ratio and maximizing sample dispersion throughout the study area. Individuals were restrained by hand and radiocollared with 9 g radio-transmitters (AVM Instrument Co., Colfax, California, USA), which were fixed around the neck with cable ties. Radiocollared individuals were released at the capture site within the first 3 hours after sunrise. All capturing and handling was conducted under Federal Fish and Wildlife Endangered Species Permit # TE0959080-1, State of Florida Fish and Wildlife Conservation Commission Special Purpose Permit # WX05089, Florida Department of Environmental Protection Research and Collection Permit # 5-05-41, and University of Georgia Animal Care and Use Permit # A2005-10044-0.

Nest Use and Availability

We located nests of each individual during the day, 3 times/month, by homing with a TRX-1000S VHF receiver (Wildlife Materials, Murphysboro, Illinois USA) and a folding, 3-element Yagi antenna. We recorded type of substrate used and location with a Global

Positioning System (GPS; Trimble GeoXT, USA). Nest substrate was categorized as artificial (rock or debris piles) or natural (downed logs, root systems of standing or wind-blown trees). We determined the availability of artificial substrate by searching the study area and recording the location of all rock and debris piles with a GPS. We calculated the density of debris piles in each hammock age class in ArcGIS (Environmental Systems Research Institute, Redlands, CA) using Hawth's Analysis Tools (Beyer 2004). Telemetry data revealed foraging individuals moved a mean distance of 45 m from the nest. Therefore, we quantified availability of artificial nest substrate as the total number of rock and debris piles (DEBRIS) occurring within a 45-m radius around each nest in ArcGIS. To evaluate selection of hammock age, we assigned a value of 1-4 to disturbed, young, medium, and mature hammock respectively. At each nest location, we calculated the weighted average of hammock age (AGE) occurring within 45 m of the nest in ArcGIS using Hawth's Analysis Tools, resulting in a value between 1 and 4. Hammock age class was available as a GIS layer from the USFWS (McCleery et al. 2006a; Ross et al., Florida International University, unpublished data). Random points were generated throughout the study area equal to the number of nests identified, with DEBRIS and AGE calculated within 45 m of each point.

Foraging Microhabitat Selection

We estimated locations of foraging individuals using 2 observers taking simultaneous bearings from permanently established telemetry stations. Stations were placed within 100 m of nests occupied by radiocollared woodrats and station coordinates were determined using a GPS. We located individuals within the first 6 hours after sunset, 2-5 times per week, and no more than once in a 24 hr period. We used a rotating schedule, beginning with a different individual each night, to ensure data were not biased by time, and only recorded foraging locations for those individuals judged to be active (i.e., discontinuous signal). We calculated foraging locations from telemetry station coordinates and azimuths using program LOCATE III (Nams 2006). Radiocollar batteries typically lasted >4 months allowing us to collect 30-45 locations/individual using a rotating schedule. We estimated telemetry error using 2 observers taking simultaneous bearings on test collars placed at randomly selected points within each individuals foraging range. To minimize bias from telemetry error, we removed estimated foraging locations that were out of the range of the equipment (>500 m from telemetry station).

We examined foraging microhabitat selection by comparing the area used by each radiotagged woodrat for foraging to the area available to each individual for foraging. To represent the foraging area of each radiotagged individual we randomly selected 5 foraging locations, with the constraint that locations be >25 m apart,. Three individuals had foraging ranges that were too small to include 5 locations >25 m apart. For those 3 individuals, we used 4 locations for 2 individuals and 3 locations for the other individual. We defined the area available to each individual for foraging as a circle with a radius equal to the distance from the nest, or nest centroid in the case of multiple nests, to the farthest estimated foraging location (Castleberry et al. 2001). Within the available area of each individual, we selected random locations equal to the number of randomly selected foraging locations. For each foraging and available location we measured the structure and composition of vegetation using the point-centered quarter method (Cottam and Curtis 1956) within a fixed radius of 12.5 m around each point to ensure vegetation measurements were independent within an individuals foraging area (Table 3.1). At each point, we calculated the density of understory stems (1.0 – 5.0 cm diameter; measured at 4 cm above ground), and the mean diameter at breast height (DBH; 1.4 m above stem base) of midstory (5.1 – 10.0 cm DBH) and overstory stems (>10.1 cm DBH). We recorded the species of each stem encountered at each point and estimated overall species richness using program SPECRICH (Hines 1996). We measured distance from each foraging and available location to the nest or nest centroid in ArcGIS. Habitat variables chosen were based on previous accounts of KLWR habitat use (Sasso and Gaines 2006; N.C. Goodyear, North Key Largo Study Committee, unpublished data) and those hypothesized to be indicative of cover and food availability.

Model Development and Evaluation

Two competing models were developed to determine the effect of artificial nest substrate and hammock age on KLWR nest site selection. Both artificial nest substrate and patches of young hammock were used frequently by KLWR in a previous study (McCleery et al. 2006a). Twenty two a priori models were developed to examine the importance of structure and composition of vegetation and distance traveled on KLWR microhabitat selection. We developed models with distance from nest and distance-habitat interactions as covariates based on central-place foraging theory (Orians and Pearson 1979, Schoener 1979, Rosenberg and McKelvey 1999).

Microhabitat and nest site selection data were analyzed with logistic regression in SAS (PROC LOGISTIC; SAS Institute 1999). We compared model likelihood’s using Akiake’s Information Criterion (AIC) and calculated Akiake weights (w_i) for each model (Burnham and Anderson 2002). We calculated Nagelkerke's R^2 for each model to evaluate the amount of variation in the response variable accounted for by the covariate. When appropriate, we used multi-model inference (MMI) to estimate parameters by calculating model-averaged estimates of regression coefficients and unconditional standard errors for covariates included in the top models ($\Delta AIC \leq 4$; Burnham and Anderson 2002). Variable importance for nest site and microhabitat models was determined by summing w_i for all models that included each variable

(Burnham and Anderson 2002). We developed a resource selection function (RSF) for nest site selection which took the exponential form:

$$w^*(x) = \exp(\beta_1 x_1)$$

where $w^*(x)$ is the relative probability of use of a nest site based on the estimated slope (β) and magnitude of the habitat covariate (x).

RESULTS

We radiocollared 42 individuals, 20 females and 22 males, during the 12-month study period. Three individuals disappeared after initial release and no data on nest or microhabitat use were collected. An additional 4 were lost within the first two weeks after release and only nest use data were collected for these individuals. We retrieved radiocollars for the 4 missing individuals and remains of one individual, which was depredated by an unknown predator. We failed to recapture the remaining three individuals and fates are unknown. We identified an additional female KLWR found dead in its nest after 8 weeks of radio-tracking. A necropsy was performed but cause of death was not determined. Another female KLWR was depredated by a large (>1.5 m) everglades rat-snake (*Elaphe obsoleta rossalleni*) after 8 weeks of radio-tracking.

We identified 66 unique nests used by 39 individuals. Mean number of nests used/individual was 1.93 (SE = 0.22) for all individuals, with males averaging 2.25 (SE = 0.19) and females averaging 1.63 (SE = 0.16) nests. Artificial nest substrate was used more frequently than natural substrate, with the proportion of natural/artificial substrate tending to increase with hammock age (Table 3.2). Natural nest substrate utilized by KLWR included the root systems of standing and wind-blown trees (n = 9), a log (n = 1), and piles of dead vegetation and vines in a heavily disturbed area (n = 2). Artificial substrate varied, with individuals utilizing piles of rocks and cement (n = 25), a standing building (n = 2), abandoned boats and cars (n = 7), and miscellaneous debris (e.g., roofing material, truck tire, metal debris; n = 20). Artificial substrate was more abundant in disturbed and young hammock (Table 3.2). Areas within 45 m of nest sites had higher amounts of artificial nest substrate (mean = 3.32 debris piles within a 45-m radius, SE = 0.42) than random areas (mean = 0.12 debris piles within a 45-m radius, SE = 0.07) and a higher percentage of young hammock (used sites mean AGE = 2.39 SE = 0.79; random sites AGE = 3.08, SE = 1.01). There was considerable support for the DEBRIS model, while AGE performed poorly as a predictor of nest site selection (Table 3.3). Using the top model, the following RSF was developed:

$$w^*(x) = \exp(\ 2.181(\mathrm{DEBRIS}))$$

where relative probability of use increased with increasing amounts of artificial nest substrate.

We obtained foraging locations on 35 radiotagged individuals tracked for an average of 13 weeks/individual, resulting in a mean of 33.0 locations/individual. Mean telemetry error from test collars was 29.5 m (SE = 6.0). Microhabitat measured around foraging locations was similar to available locations, with the largest differences observed for DIST and DEN1 (Table 3.4). There was considerable model uncertainty, with 14 models having a ΔAIC ≤ 4 (Table 3.5). There was little support for selection of microhabitat variables alone, as models not including DIST as a main effect performed poorly (ΔAIC > 55.0; Table 3.5). In addition, R^2 values were low for all models (< 0.235; Table 3.5). The sum of model weights suggested DIST was the most important variable predicting microhabitat use, followed by DEN1, DBH3, SPEC, and DBH2 respectively (Figure 3.2). Model averaged parameter estimates were negative for DBH3 and DIST, and positive for DEN1, DBH2, and SPEC (Table 3.6).

DISCUSSION

Our results suggest that KLWR select nest sites in areas with high densities of artificial nest substrate regardless of hammock age. Over 80% of KLWR nests were in artificial substrate, with the remainder occurring in large logs or the root systems of overstory trees. Use of artificial nest substrate by KLWR was noted previously, with rock and debris piles assumed to offer greater protection from predators than freestanding stick nests (N.C. Goodyear, North Key Largo Study Committee, unpublished data). Hammock age class alone was a poor predictor of nest site selection. KLWR utilized disturbed areas and young hammock for nest sites, but likely due to the higher availability of artificial substrate in these areas. Debris piles are the result of dredging and illegal trash disposal in recently disturbed areas, and are therefore confounded with younger hammock. In the absence of debris piles, KLWR rarely occurred in younger hammock.

Consistent with the most recent report of KLWR nest use (McCleery et al. 2006a), freestanding stick nests were found to be absent from north Key Largo. Stick nests require energy to build and maintain and likely increase predation risk due to the extra excursions required for stick collecting. Given the potential benefits of reduced energy and predation risk associated with artificial substrate, it is not surprising KLWR opportunistically choose to use these resources when available. However, the large majority of available habitat (>75%) does not contain artificial substrate. The low use of natural substrate, the absence of freestanding stick nests, and the overall low occupancy of KLWR in areas without artificial nest substrate suggest quality natural nest substrate is lacking. Natural nest substrate identified in this study and previously (McCleery et al. 2006a; N.C. Goodyear, North Key Largo Study Committee, unpublished data) consists of features characteristic of mature hammock (i.e., downed logs and large trees with exposed root systems). Areas of mature hammock (disturbed prior to 1940) comprise approximately 50% of the hardwood hammock, yet occupancy and nest use in mature hammock was relatively low in our study and in previous studies (McCleery et al. 2006a). The forested uplands of north Key Largo have undergone a variety of major disturbances from human activities over the last century, with most of the area under cultivation for fruit crops up until 1935 (Ross et al., Florida International University, unpublished data). Although the forest has recovered to some degree, the overall maturity level may still be too low, lacking trees large enough to provide adequate substrate for constructing natural nests.

The negative consequences of a reduction of quality nest sites may be amplified by unnaturally high predator abundances. The habitat available to KLWR is bordered by heavily urbanized areas which offer an abundance of anthropogenic food sources for mammalian predators, possibly increasing their abundances (Hoffman and Gottschanng 1977, Prange et al 2003). Recent trapping efforts and surveys of neighboring developments determined raccoons and feral cats were abundant in North Key Largo (B. Muznieks, USFWS, personnel communication). However, information on cause-specific mortality for KLWR is lacking. We documented 3 mortalities during our study, consisting of a non-predator related mortality of unknown cause, predation by an everglades rat snake (*Elaphe obsoleta rossalleni*), and predation by an unknown predator (either avian or mammalian based on the condition of the remains). Previous reports of 2 additional mortalities included one road-kill and one predation by an unknown mammalian predator, likely feral cat (*Felis catus*), raccoon (*Procyon lotor*) or opossum (*Didelphis virginiana*; B. Muznieks, USFWS, personnel communication). Additional information is required to assess the effect of predators on nest site selection by KLWR.

KLWR were detected more frequently in close proximity to the nest. Efficient foraging and limited long-range movement is predicted by optimal foraging theory as a means to reduce predation risk (Orians and Pearson 1979). Thus, the limited movement exhibited by KLWR is not surprising, however, there was little support for selection of microhabitat or varying degrees of selection at different distances from the nest, as was expected based on central place foraging theory (Schoener 1979) and applied research (Rosenberg and McKelvey 1999). Telemetry data suggests KLWR used a variety of microhabitats independent of ground cover and canopy or midstory structure at all available distances from the nest. *Neotoma floridana* are considered generalist herbivores and able to utilize a wide variety of habitats and food resources (Rainey 1956, Wiley 1980, Wagle and Feldhamer 1997). The hardwood hammock habitat of Key Largo contains a high diversity of tree species (>150 species; Ross et al., Florida International University, unpublished data) and likely offers an abundance of year-round food sources due to the sub-tropical climate. As a generalist herbivore in a resource-rich environment, KLWR may not be limited by the structure and composition of vegetation and able to exploit a variety of microhabitats in close proximity to the nest.

MANAGEMENT IMPLICATIONS

Our results suggest KLWR may not be restricted by the quality of foraging habitat, but more likely by the availability of suitable nest substrate and would likely benefit from restoration of previously disturbed areas. Disturbed areas commonly contain rock and debris piles that are selected as nest sites. The root systems of overstory trees are utilized as nest sites, but at a low frequency. Continued protection of the forest will allow the hammock to mature, possibly increasing the availability of quality natural nest substrate. Frequent use of artificial substrate suggests KLWR would benefit in the short term from additions of artificial substrate, which would increase the availability of quality nest sites while the forest continues to recover from past disturbances. In addition, the species would benefit from further research on factors limiting the use of natural substrate, particularly the effects of predators.

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Table 3.1. Habitat variables measured within a 12.5 m radius around KLWR foraging locations and random points within available foraging areas. Data were collected in north Key largo, Florida, 2005-06.

Variable	Code	Definition
Understory density	DEN1	Density (stems/ha)of stems <5 cm diameter
Midstory DBH[a]	DBH2	Diameter (cm) at breast height of trees >5.1 cm
Overstory DBH	DBH3	Diameter (cm) at breast height of trees >10.1 cm
Species richness[b]	SPEC	Estimated number of shrub and tree species
Distance	DIST	Distance (m) from nest or nest centroid to foraging and random location

[a] Diameter at breast height (1.4 m)

[b] Estimated using program SPECRICH

Table 3.2. Number of KLWR nests, utilizing natural and artificial substrate, in 4 hammock age classes (disturbed, young, medium, and mature) and number of debris piles available in each class. Data were collected on 39 radio-collared individuals between 2005 - 2006 on north Key Largo, Florida.

	Disturbed (92 ha)	Young (87 ha)	Medium (327 ha)	Mature (430 ha)	Total (937 ha)
Natural substrate used	2	0	6	4	12
Artificial substrate used	14	11	23	6	54
Number of artificial substrate (per ha)	78 (0.85)	46 (0.53)	79 (0.24)	37 (0.09)	240 (0.26)

Table 3.3. Model, number of parameters in the model (K), Akaike's Information Criterion (AIC), difference in AIC value between the model and the model with the lowest AIC value (ΔAIC), Akaike weights (w_i), Nagelkerke's R^2, parameter estimates (β), SE, and 95% confidence intervals for 2 competing models on factors effecting nest selection by KLWR. DEBRIS and AGE were measured around nest sites (n = 66) and random locations (n = 66) throughout the study area in north Key Largo, Florida, 2005-2006.

Model	K	AIC	ΔAIC	w_i	R^2	β	SE	95% UCI	95% LCI
DEBRIS	2	99.23	0	1.00	0.65	2.181	0.446	1.307	3.055
AGE	2	169.30	70.07	0.00	0.17	-0.825	0.212	-1.241	-0.410

Table 3.4. Mean (±SE) of variables measured at KLWR foraging and available locations. Data were collected on 35 radiocollared individuals in north Key Largo, Florida, 2005-2006.

Variable	Used (n = 170)	Random (n = 170)
	$\bar{x}$ (SE)	$\bar{x}$ (SE)
Den1	0.90 (0.09)	0.71 (0.07)
DBH2	6.72 (0.05)	6.81 (0.07)
DBH3	15.50 (0.31)	15.85 (0.43)
Spec	9.17 (0.33)	9.27 (0.47)
Dist	44.58 (3.99)	78.78 (5.55)

Table 3.5. Variables, number of parameters in the model (K), Akaike's Information Criterion (AIC), difference in AIC value between the model and the model with the lowest AIC value (ΔAIC), Akaike weights (w_i), and Nagelkerke's R^2 for 22 a priori models used to predict relative probability of use of microhabitat by foraging KLWR. Data were collected on 35 radio-collared individuals in north Key Largo, Florida, 2005-2006.

Model	AIC	ΔAIC	wi	R^2
DIST DEN1 DEN1*DIST	415.265	0.000	0.170	0.229
DIST DEN1	415.478	0.213	0.153	0.222
DIST	415.617	0.352	0.143	0.215
DIST DBH3 DEN1	416.847	1.582	0.077	0.224
DIST SPEC DEN1 DIST*SPEC DIST*DEN1	417.091	1.826	0.068	0.236
DIST DBH3	417.101	1.836	0.068	0.217
DIST SPEC DEN1	417.216	1.951	0.064	0.223
DIST SPEC	417.632	2.367	0.052	0.215
DIST DBH2 DBH3	418.109	2.844	0.041	0.220
DIST SPEC SPEC*DIST	418.327	3.062	0.037	0.219
DIST DBH3 DEN1 DBH3*DIST DEN1*DIST	418.391	3.126	0.036	0.232
DIST DBH2 DBH3 DBH2*DIST DBH3*DIST	418.471	3.206	0.034	0.232
DIST DBH3 DBH3*DIST	418.621	3.356	0.032	0.218
DIST SPEC DBH2 DBH3	420.064	4.799	0.015	0.220
DIST SPEC DBH2 DBH3 SPEC*DIST DBH2*DIST DBH3*DIST	421.466	6.201	0.008	0.235
DEN1	470.448	55.183	0.000	0.019
DEN1 DBH3	471.967	56.702	0.000	0.021
SPEC DEN1	471.978	56.713	0.000	0.021
DBH3	474.918	59.653	0.000	0.002
SPEC	475.272	60.007	0.000	0.000
SPEC DBH3 DBH2	475.887	60.622	0.000	0.006
DBH2 DBH3	477.812	62.547	0.000	0.006

Table 3.6. Model averaged parameter estimates and unconditional SE's for microhabitat variables and interaction terms included in the top models (ΔAIC<4) from a candidate set of 22 a priori models of KLWR microhabitat selection.

Variable	B	SE
DIST	-0.023169	0.007090
DEN1	0.020486	0.175704
DBH2	0.003536	0.027569
DBH3	-0.003758	0.016340
SPEC	0.004222	0.013798
DIST*DEN1	0.002090	0.003578
DIST*SPEC	-0.000109	0.000735
DIST*DBH2	-0.000295	0.000626
DIST*DBH3	-0.000059	0.000171

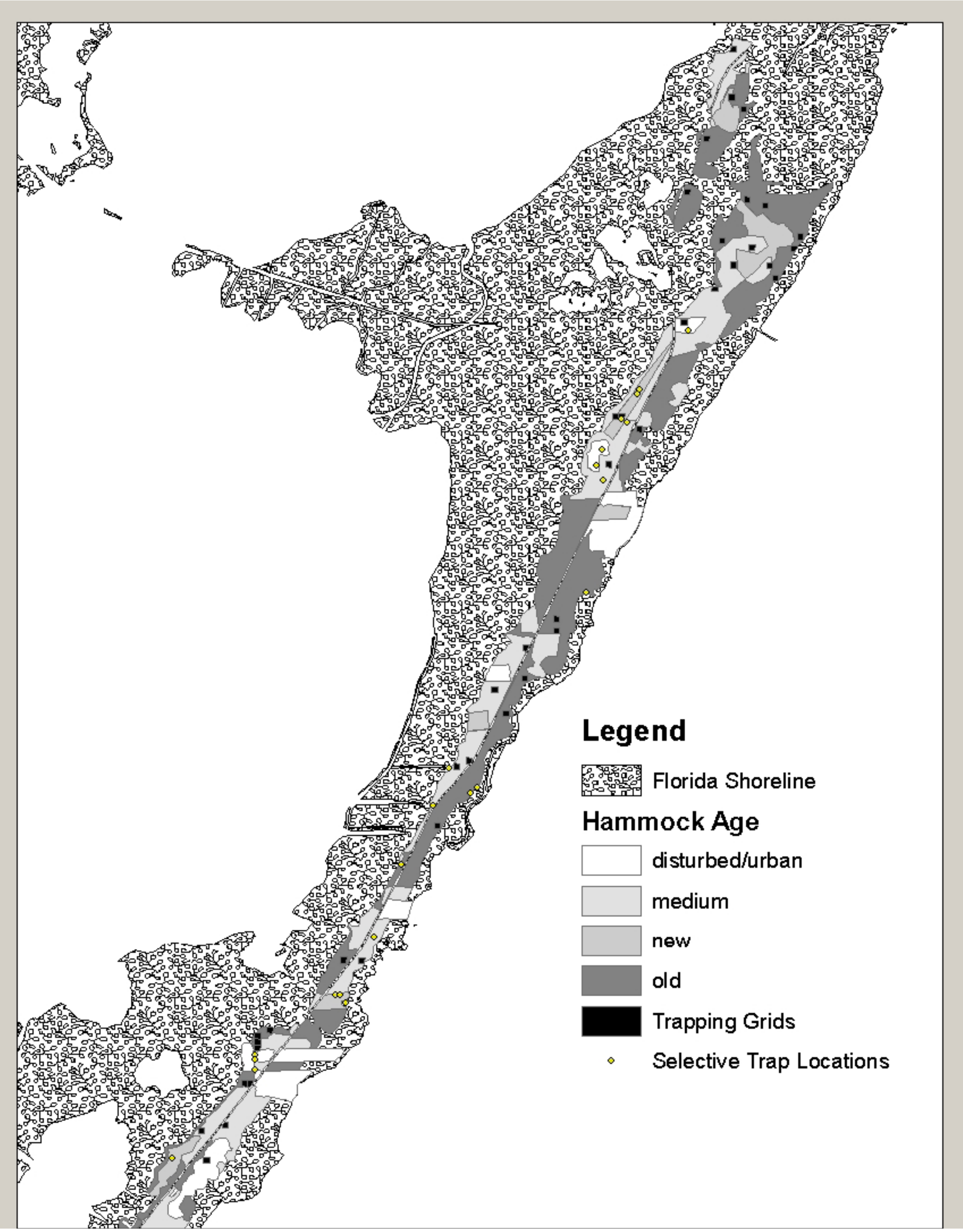

Figure 3.1. Map of north Key Largo, Florida depicting study area (hardwood hammock), hammock age classes (new, medium old and disturbed/urban), and locations of random trapping grids and selectively trapped areas.

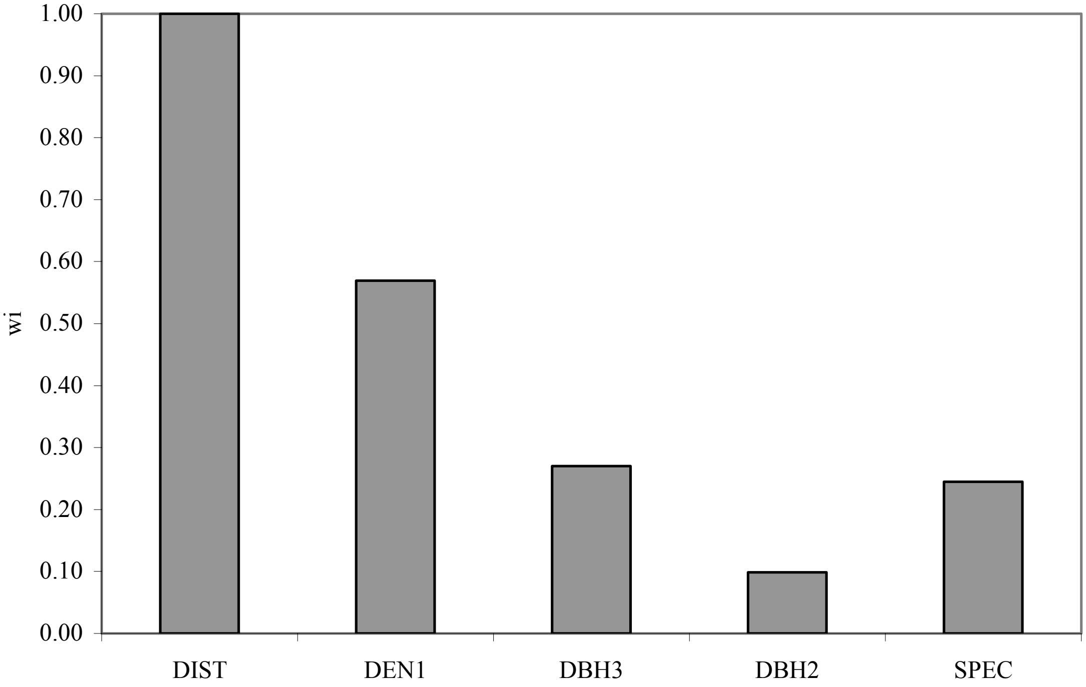

Figure 3.2. Sum of model weights (w_i) for 5 habitat variables in 22 a priori models developed to examine microhabitat selection by KLWR in north Key Largo, Florida, 2005-2006.

CHAPTER 4

KEY LARGO WOODRAT ABUNDANCE ESTIMATE USING EQUAL PROBABILITY AND ADAPTIVE SAMPLING[1]

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[1] Winchester, C., M. T. Mengak, and S. B. Castleberry. To be submitted to Journal of Wildlife Management.

ABSTRACT

The endangered Key Largo woodrat (KLWR; *Neotoma floridana smalli*) is endemic to Key Largo, Florida. Monitoring abundance of KLWR is necessary to provide feedback on the response of the population to prescribed management actions, but is challenging due to the rarity and clustered distribution of the species across the landscape. Our objectives were to evaluate the efficiency of adaptive cluster sampling (ACS) compared to equal-probability, stratified-random sampling (SRS) for estimating abundance of KLWR and to identify the most effective stratified design using post-stratification. We established 40, 0.56 ha trapping grids using a stratified random design and adaptively sampled around grids on which at least 1 unique individual was captured. We captured 11 individuals on 40 random grids and an additional 22 individuals on 33 adaptive grids. Despite the increased capture rate, ACS was found to be less efficient than SRS, with an estimator variance twice as high as the SRS variance with equal sample sizes. Post-stratification effectively lowered estimator variance and was used to identify three strata based on nest substrate availability and forest age as the most efficient stratified random design. Accounting for bias due to movement (20%) and imperfect detection (-4%), the most precise estimate of abundance was 188 (95% CI 0-400). Sample sizes required to estimate abundance over a range of precision levels, using the optimal design identified, suggest monitoring total population size will be costly, even to detect large fluctuations in abundance. Small scale monitoring of improved habitat along with representative controls may be more cost-effective for evaluating the success of prescribed management.

INTRODUCTION

The Key Largo woodrat (KLWR; *Neotoma floridana smalli*) is a federally endangered subspecies of the eastern woodrat (*Neotoma floridana*) endemic to Key Largo, Florida. The

KLWR historically ranged throughout the hardwood hammocks of Key Largo, but is now restricted to state and federally protected lands on the northern one-third of the island (Barbour and Humphrey 1982). Despite the protection of its remaining habitat from development, the KLWR population continued to decline after listing in 1984 (McCleery et al. 2006). The cause of the population decline is unknown, but the quality of existing habitat appears severely compromised. In an effort to prevent extinction, the U.S. Fish and Wildlife Service drafted a recovery plan outlining priority research and management goals (USFWS 1999). In addition to improving habitat quality, 2 major components of the recovery plan are to determine current status of the population and monitor population trends over time. Monitoring abundance of KLWR allows managers to evaluate the effectiveness of habitat improvement and is essential feedback for the successful implementation of an adaptive management plan (Hollings 1978, Walters 1986).

Recent research suggests KLWR occupy approximately 20% of its available range (170 ha; McCleery et al. 2006). Rare species, such as the KLWR, are difficult to sample using traditional designs (i.e., simple or stratified random sampling), with low occupancy resulting in imprecise estimates of abundance even with relatively large sample sizes (e.g. trapping grids). In addition to being rare, KLWR frequently uses artificial substrate (i.e., rock and debris piles) for nest sites (McCleery et al. 2006). Rock and debris piles are the result of dredging and illegal dumping along abandoned roads and tend to be clustered in the landscape. The high frequency of use of this artificial resource provides anecdotal evidence that KLWR may exhibit a clustered distribution.

Adaptive cluster sampling (ACS) was developed to provide a more efficient means of sampling rare and clustered populations (Thompson 1990). After an initial random sample is taken, additional units are selected from all immediately surrounding units when a pre-defined criterion is met (e.g., when 1 individual is detected). The result is a cluster of sampling units around the original randomly selected unit, known collectively as a 'network.' Adaptively selected units not meeting the criteria are considered edge units and are not used in the analysis (Thompson 1990). A modified version of the Horvitz-Thompson estimator is used to account for the unequal inclusion probabilities of sampling units (Thompson 1990). ACS also can be used when the initial sample is selected using stratified random sampling (SRS; Thompson 1991), which may further lower the variance of an estimate if biologically meaningful strata can be identified. The high frequency of use of artificial substrate by KLWR suggests areas delineated by proximity to or density of these resources may be an effective means of stratifying, thereby, lowering the estimator variance. In the absence of artificial substrate, large trees typically are used for nests (McCleery et al. 2006), making more mature patches of forest, lacking rock and debris piles, a potentially useful stratum. With several options for stratifying, post-stratification is effective for evaluating multiple designs, differing in strata delineation, on lowering estimator variance (Thompson 2002).

Although SRS and ACS are design-unbiased estimates of a population mean, imperfect detection and the mobility of animals can introduce bias (Otis et al. 1978, Wilson and Anderson 1985). Individual detection rates typically are <1.0 for most animals and result in underestimates of abundance. Mark-recapture models are a common and effective means of estimating detection rate (Otis et al. 1978, William et al. 2002) and were used successfully in past KLWR population studies (Humphrey 1988, McCleery et al. 2006). Estimates of detection rate can be used to correct the bias of designed-based estimates of ACS and SRS (Thompson 1994).

Mobility of individuals inflates abundance estimates by over-estimating density, a result of the effective detection area of a sampling unit being larger than the geographical boundaries of the unit (Wilson and Anderson 1985). For small mammal studies employing trapping grids, the nested-grid approach is a common method used to account for positive bias (Wilson and Anderson 1985, Humphrey 1988). However, the nested-grid approach requires high target animal densities, large grids, and is cumbersome when employing ACS, which results in networks varying in shape. An alternative method to account for the effects of movement is the use of radiotelemetry to estimate the proportion of individuals with nests occurring outside of the geographic boundaries of the sampled area. Similar to estimating survival, the binomial likelihood can be used to estimate the proportion of transient individuals (Williams et al. 2002). The design-based estimate can then be corrected for bias by subtracting the proportion of transients from the naïve estimate of abundance.

The goals of our study were to evaluate the efficiency of ACS relative to SRS in providing a point estimate of KLWR abundance. Estimates of detection rate and proportion of transient individuals were used to correct for bias in each estimate. In addition, post-stratification was used to evaluate the effectiveness of 3 strata delineations on reducing the variance for SRS.

STUDY AREA

Our study area was defined as all upland forest habitat (hardwood hammocks) occurring on the Crocodile Lake National Wildlife Refuge and Dagny Johnson Key Largo State Botanical Preserve. The hardwood hammocks of this region are characterized as closed canopy forest with a limestone substrate, contaning over 150 species of evergreen and semi-deciduous tree and shrub species (USFWS 1999). Common canopy species include pigeon plum (*Coccoloba* *diversifolia*), poisonwood (*Metopium toxiferum*), west-indian mahogany (*Swietenia mahagoni*), wild tamarind (*Lyisloma latisiliguum*) and gumbo-limbo (*Bursera simaruba*). Common sub-canopy species include crabwood (*Gymnanthes lucida*), white stopper (*Eugenia axillaries*), marlberry (*Ardisia escallonioides*) and wild lime (*Zanthoxylum flavum*). Canopy closure creates a shaded environment on the forest floor resulting in a sparse shrub and herb layer. As a result, the understory consists mainly of seedlings and saplings of canopy and sub-canopy species, however, wild coffee (*Psychotria nervosa*) and snowberry (*Chiococca alba*) are found commonly in the understory (USFWS 1999). The hammock grades into mangroves along the coast creating a relatively shallow transitional zone that is comparatively open with high vine growth and numerous thorny plant species (Ross et al. 1992).

METHODS

Sampling Design

Stratified random sampling – The 850-ha area of hardwood hammock was divided into 0.56 ha sampling grids (units) from which 40 were selected via a stratified random design (Figure 4.1). Because rock or debris piles have been shown to be important substrates for KLWR nest sites (McCleery et al. 2006), we defined sampling units ≤75 m and >75 m from a rock or debris pile as strata for allocating sampling effort. We recorded the locations of all known rock and debris piles with a Global Positioning System (GPS; Trimble GeoXT, USA) prior to the selection of sampling units and created a layer in a Geographic Information System (ArcGIS; Environmental Systems Research Institute, Redlands, CA USA) to delineate stratum boundaries. Sampling units were selected in proportion to the area available in each stratum, such that each unit had an equal inclusion probability.

Adaptive cluster sampling – Using the initial SRS, we set the criterion for adaptive sampling at one unique individual captured on a unit and defined the neighborhood as all immediately surrounding units large enough to contain a 50 m^2 trapping grid (Figure 4.2). We applied a stopping rule of 1 adaptive addition to each initial unit meeting the criteria to control the final sample size. Sampling units added adaptively were allowed to cross stratum boundaries.

Trapping and Handling

We sampled 73 units, 40 random and 33 adaptive, between April 27 and June 1, 2005. Within each unit we placed a trapping grid with 9 stations, in a 3 x 3 array, with 25 m between stations. At each station we placed two 10.2 x 11.4 x 38.1 cm, vented Sherman traps (model PXLF15, H. B. Sherman Traps Inc., Tallahassee, Florida) with raccoon (*Procyon lotor*)-proof door latches. Traps were baited with peanut butter and crimped oats. We opened traps for 4 consecutive nights, checking each trap daily within the first 3 hours after sunrise. All captured individuals were weighed, sexed, and marked with passive integrated transponders (PIT) tags (BioMark, Boise, Idaho) and #1005 Monel ear tags (National Band and Tag, Newport, Kentucky). All capture and handling of KLWR was conducted under the Federal Fish and Wildlife endangered species permit # TE0959080-1 , State of Florida Fish and Wildlife Conservation Commission Special Purpose Permit # WX05089, Florida Department of Environmental Protection Research and Collection Permit # 5-05-41, and University of Georgia Institutional Animal Care and Use permit # A2005-10044-0.

Radiotagging and Nest Location

To account for potential bias in abundance estimates due to movement, we radiotagged 10 individuals (5 males and 5 females) on 8 sampling units. We selected individuals opportunistically with a minimum weight of 180 g for attachment of 9 g radiotransmitter collars

(AVM Instrument Company, Colfax, California, USA). Eight individuals were radiocollared and located during the day, twice each week, between April and July 2005. Two individuals were radiocollared and located twice each week between September-November 2005. We assumed nest use did not differ for these 2 individuals. We located nests by homing with a TRX-1000S VHF receiver (Wildlife Materials, Murphysboro, Illinois USA) and a folding, 3-element Yagi antenna and recorded each nest location with a hand-held GPS unit. If multiple nests were used by individuals, we defined primary nest sites as those utilized ≥80% of the time. Locations of primary nest sites were downloaded into ArcGIS to determine if they occurred within or outside of the network on which the individual was captured.

Abundance Estimation

We estimated abundance using the design-based estimator for SRS (Thompson 2002). For ACS, abundance was estimated with the modified Horvitz-Thompson estimator using initial intersection probabilities (Thompson and Seber 1996). We used the ratio of the estimator variances [var(SRS)/var(ACS)] to evaluate the efficiency of ACS, setting the sample size of SRS equal to the final sample size (including edge units) for ACS (Thompson and Seber 1996). With this ratio, a value >1 indicates greater efficiency using ACS. We estimated capture and recapture probabilities of the trappable population in program MARK using 8 closed-population models which allow for the effects of behavior, capture heterogeneity, and time (White and Burnham 1999). We used model averaged estimate of abundance provided by MARK to estimate detection rate. Due to the low number of captures, all individuals were pooled for analysis. To correct for imperfect detection, the estimates of abundance from the design-based estimators were divided by the estimated detection rate (Thompson and Seber 1994). To account for bias due to movement, we calculated the maximum-likelihood estimate for the proportion of individuals with primary nest sites occurring outside of the sampled area. Design-based estimates of abundance were corrected for bias due to movement by subtracting the estimated percentage of individuals with nests occurring outside of the sampled area.

Poststratification

To ensure more accurate strata delineations for post-stratification analysis, we searched sampling units and the remaining study area more thoroughly, and recorded locations of rock and debris piles with a GPS. We evaluated the effectiveness of stratification of the original 40 samples by comparing the variance of the estimated population total under three designs differing in strata delineations. In the first design, strata were delineated as originally intended (≤75 m or >75 m from rock or debris pile) but with greater accuracy. In the second design, 2 strata were delineated based on the density of rock or debris piles (≤3 or >3) within a 75 m radius of the grid center. In the third design, 3 strata were defined, where density of rock and debris piles was used to delineate 2 strata, as in design 2, and areas having ≤3 debris piles further delineated as mature forest or otherwise. A GIS layer delineating forest age class was made available by the USFWS and was developed using data from previous studies (Ross et al. 1995, McCleery et al. 2006). All strata delineations were based on previous studies of habitat use by KLWR (Hersh 1978, Barbour and Humphrey 1982, Goodyear 1985, McCleery et al. 2006) with the goal of dividing the study area into low and high density areas, thereby, lowering estimator variance.

The variance equation from Thompson (2002) was used to calculate the post-stratified variance of the population total. Efficiency of each design was estimated as the ratio of the original SRS estimator variance to the estimator variance for post-stratification, where a value greater >1 indicates increased efficiency due to post-stratification. Using the post-stratified design with the lowest variance, we calculated expected samples size to estimate abundance within ranges varying between 80-20% of the actual abundance (Thompson 2002).

RESULTS

With the initial stratified random sample (n=40), 11 individuals were detected on 7 sampling units. Thirty-three units were added adaptively and resulted in an additional 22 individuals. Of the 33 grids added adaptively, 23 were edge units and did not result in additional captures. The best closed-population model in program MARK included the effects of behavior and heterogeneity in capture probability (M_{bh}; Table 2.1). The model-averaged estimate of abundance for the sampled population was 34.4 (SE=2.32) resulting in an estimated detection rate of 0.96 (95% CI 0.85-1.0; Table 2.2). Two of the 10 individuals were found to have nest sites outside of the sampled area resulting in an estimated 0.20 (SE=0.06) positive bias from movement. Total population size, corrected for imperfection detection and movement, was 321 (95% CI 13-629) for SRS and 323 (95% CI 0-652) for ACS. The ratio of var(SRS)/var(ACS) was 0.47 with sample size equal to the final sample size for ACS (n=73).

Additional rock and debris piles were detected with more extensive ground-truthing, resulting in 6 misclassified sampling units. Post-stratification was an effective means of accounting for initial error with strata delineations, and resulted in lower estimator variances for all 3 designs examined, with the third design, delineating 3 strata, found to be the most efficient (Table 2.3). The most precise estimate of abundance using post-stratification, corrected for detection rate and movement, was 188 (95% CI 0 – 400). With this design, required sample sizes to estimate abundance within 20-80% ($\alpha = 0.10$) of the actual value varied between 181-58, respectively (Figure 4.3).

DISCUSSION

Adaptive Cluster Sampling was developed as an alternative to traditional sampling designs, which assume a random distribution of the target population, to more effectively estimate population size for rare and clustered populations (Thompson 1990). Despite its intuitive appeal and success with simulated populations, ACS has not been widely used to sample biological populations (Smith et al. 2004). Limited use was linked to a general lack of knowledge of the methods and to problems encountered when applied to real populations. Although use of adaptive sampling has increased in the last 10 years, applications to real populations produced mixed results with respect to efficiency (Lo et al. 1997, Strayer et al. 1997, Smith et al. 2003, Philippi 2005, Noon et al. 2006). The efficiency of ACS over traditional designs, as measured by a decrease in estimator variance given equal cost, is dependent on both the degree of rarity and spatial distribution of the target population (Thompson and Seber 1996, Smith et al. 2004). Our results from ACS resulted in a higher number of individuals per sampling unit than SRS, however, the variance of the SRS estimator was significantly lower and was, therefore, a more efficient design for estimating abundance of KLWR.

In addition to population distribution, efficiency of a sampling design can be strongly influenced by the criteria set to initiate adaptive sampling and the neighborhood definition, both of which affect the final sample size (Brown 2003). Liberal criteria and neighborhood definitions can lead to a large final sample size, making ACS impractical for most monitoring situations. Final sample size has been effectively controlled with the use of stopping rules, which often bias estimates of abundance due to inaccurate estimates of network inclusion probabilities (Brown 1994, Lo et al. 1997, Brown and Manly 1998, Salehi and Seber 2002). However, if small, the bias introduced with a stopping rule may be outweighed by the increase in efficiency gained through reduced sample size and cost (Lo et al. 1997). We applied a stopping rule of one adaptive addition, but final sample size was still relatively high. Our high final sample size was likely attributed to the criteria for initiating adaptive sampling being too liberal. Increasing the criteria to 2 individuals would have resulted in 1 less individual and 17 fewer sampling units, lowering the cost of the study (i.e. # of trapping grids required) by 23%.

More problematic than high final sample size, was the high estimator variance of ACS, which resulted in a less precise estimate of abundance than SRS despite an 83% increase in sample size. For ACS to generate a relatively low estimator variance, the population must be rare and clustered, resulting in a high within-network and low between-network variance (Thompson and Seber 1996). The least efficient design employing ACS would result from no additional detections on sampling units added adaptively, such that network size is 1 and within-network variance is zero for all networks (Thompson and Seber 1996). In our study, 3 of the 7 units that met the criteria for adaptive sampling did not result in additional detections, indicating a portion of the population was not clustered. The zero within-network variance of these 3 units likely contributed to the high ACS estimator variance. The 4 sampling units which resulted in additional detections on adaptively selected grids, demonstrated variation in degree of clustering with 2, 5, 10, and 13 individuals captured on networks comprised of 2, 3, 4, and 5 units, respectively. The latter 3 networks accounted for 85% of individuals captured while comprising 16% of the final sample. The observed clustering of KLWR in these areas may best be explained by the high availability of nest substrate. All sampling units in the 3 most productive networks had ≥3 rock or debris piles within a 75 m radius. Rock and debris piles were identified as important nest substrate for KLWR in a recent study (McCleery et al. 2006) and are largely clustered throughout north Key Largo, as a result of concentrated human activities (i.e., dredging and illegal dumping along abandoned roads). In areas without clusters of rock and debris piles (approximately 95% of the study area), KLWR may not be clustered, making ACS inefficient for the large majority of the study area.

The results of post-stratification offered additional evidence to support the influence of areas with rock and debris clusters on KLWR densities. Initial inaccurate strata delineations resulted in a high estimator variance, approaching that of a simple random sample. The effect of accurate strata delineations on estimator variance was demonstrated with poststratification, resulting in a ~ 40% decrease in the variance. Post-stratifying by areas with higher density of available nest substrate (i.e., ≥3 rock and debris piles in 75 m), areas with mature forest but low artificial substrate density, and all other areas gave the most precise abundance estimate, reducing the original SRS estimator variance by more than one-half. Mature forest was considered a valuable stratum delineation because KLWR use the root systems and downed logs of large overstory trees as nest sites, although at a lower frequency than rock and debris piles (McCleery et al. 2006). The utility of mature hammock as a statum was supported by the fact that all individuals captured on sampling units >75 m from debris piles were in mature hammock. Given the relative inefficiency of ACS and the improved efficiency of SRS with post-stratification, the optimal design identified in this study is SRS with three strata delineated by density of artificial substrate and forest age. Despite the increased precision gained with the above strata delineations, estimating abundance of the entire population would still be costly. Sample size calculations suggest 100, 0.56 sampling units would be required to estimate abundance of KLWR within 50% of the actual value, with 90% confidence, using the most efficient design.

The combined effects of imperfect detection and transient individuals introduced an appreciable amount of bias into the abundance estimate of KLWR. Because KLWR exhibit high capture and recapture probabilities (i.e., highly trappable), the effect of imperfection detection on bias was relatively minor with an estimated 4% negative bias. Although estimated bias due to imperfect detection was low, the extra cost for estimating detection rate with marked animals is minimal and ensures more accurate estimates of abundance, especially if detection rate varies over time. Transient individuals resulted in considerable positive bias and should be accounted for in future estimates of KLWR abundance. Two males in our study moved 35 m and 46 m, respectively, from nests to the edges of the networks on which they were captured. Radiotelemetry appears to be a viable option for estimating bias due to movement, but incurs additional costs. The nested-grid approach can be used to account for bias due to movement without additional cost, but requires larger trapping grids than those used in this study (Wilson and Anderson 1985). Humphrey (1988) used the nested-grid approach to estimate density of KLWR, but had difficulty gaining reliable estimates due to the strict data requirements of the method. To successfully employ the nested-grid approach, large grids and relatively high and uniform densities are optimal (Wilson and Anderson 1985). Given the problems encountered by Humphrey (1988) when densities of KLWR were likely higher than we encountered, the nested-grid approach may not be a viable option for this rare population.

MANAGEMENT IMPLICATIONS

Our results suggest that estimating abundance of the entire KLWR population is likely to be cost-prohibitive. However, if management goals require information on the status of the entire population and if sufficient resources are available, the optimal design identified in this study, with required sample size estimates, outlines a working method to estimate the total population, and allows for costs to be anticipated for a desired level of precision. To gain accurate estimates of abundance using a design-based estimator, both detection rate and percent transients should be estimated. The use of mark-recapture models and radiotelemetry in this study demonstrate a method for accounting for imperfect detection and movement bias, providing a more reliable estimate of abundance.

Depending on management goals, it may be more cost-effective to restrict monitoring to smaller spatial scales, particularly as habitat improvement is likely to be applied incrementally at small scales. Small-scale monitoring, under the framework of adaptive management, would still provide essential feedback on the response of KLWR to habitat improvement, while greatly reducing the effort required to monitor the total population.

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Table 4.1. Eight closed population models from program MARK including the null model (M_0), models with a behavioral response (Mb), heterogeneity in capture probability (M_h), time effects (M_t), and models with all combinations of time, heterogeneity and behavior effects (M_{tb}, M_{th}, M_{bh}, and M_{tbh}), with number of parameters in each model (K), Akaike's Information Criterion adjusted for small sample size (AIC_c), difference of AIC_c between a model and the model with the lowest AIC_c (ΔAIC_c), model weights (w_i), and model deviance.

Model	AICc	ΔAICc	w_i	Model Likelihood	K	Deviance
M_{bh}	-3.52	0.00	0.79	1.00	6	16.99
M_{tbh}	-0.62	2.90	0.18	0.24	14	0.96
M_t	5.51	9.03	0.01	0.01	4	30.38
M_h	5.79	9.31	0.01	0.01	4	30.65
M_{tb}	6.37	9.89	0.01	0.01	7	24.64
M_b	7.11	10.63	0.00	0.00	3	34.10
M_0	8.11	11.63	0.00	0.00	2	37.19
M_{th}	9.29	12.81	0.00	0.00	4	34.16

Table 4.2. Model-averaged estimates of mixture (n=2 groups; pi), capture (p) and recapture probabilities (c), and abundance (N) from program MARK for 33 KLWR captured on 17 sampling units, 7 random and 10 adaptive, for 4 consecutive days April-May, 2005 on north Key Largo, Florida.

Parameter	Estimate	SE	95% LCI	95% UCI
Probability of Mixture (pi)	0.201	0.089	0.078	0.428
Group 1, day 1 (p)	0.654	0.223	0.215	0.929
Group 1, day 2 (p)	0.659	0.231	0.206	0.935
Group 1, day 3 (p)	0.506	0.309	0.083	0.920
Group 1, day 4 (p)	0.693	0.260	0.171	0.961
Group 2, day 1 (p)	0.483	0.095	0.308	0.662
Group 2, day 2 (p)	0.445	0.127	0.226	0.687
Group 2, day 3 (p)	0.565	0.185	0.229	0.850
Group 2, day 4 (p)	0.582	0.207	0.208	0.881
Group 1, interval 1 (c)	0.040	0.096	0.000	0.852
Group 1, interval 2 (c)	0.040	0.096	0.000	0.850
Group 1, interval 3 (c)	0.071	0.127	0.002	0.769
Group 2, interval 1 (c)	0.879	0.071	0.664	0.964
Group 2, interval 2 (c)	0.894	0.077	0.633	0.976
Group 2, interval 3 (c)	0.851	0.078	0.630	0.950
Population Size (N)	34.414	2.324	29.859	38.968

Table 4.3. Comparison of abundance estimates (N) and estimator variance for KLWR sampled April-June, 2005 in Key Largo, Florida from stratified random sampling as originally defined, with strata (K) misclassified, versus 3 designs using post-stratification with strata accurately delineated.

Design	K	N[a]	Variance	Efficiency[b]
Original SRS	2	386	23273.42	
Design 1[c]	2	261	13832.38	1.68
Design 2[d]	2	238	13520.37	1.72
Design 3[e]	3	223	11077.97	2.10

[a] Estimate of abundance not accounting for bias due to movement or imperfect detection

[b] Efficiency measured as ratio of original SRS variance to post-stratified variance

[c] Two strata delineated as ≤75 m or >75 m from rock or debris piles

[d] Two strata delineated as ≤3 or >3 debris piles

[e] Three strata delineated as >3 debris piles, ≤3 debris piles in mature hammock, and otherwise

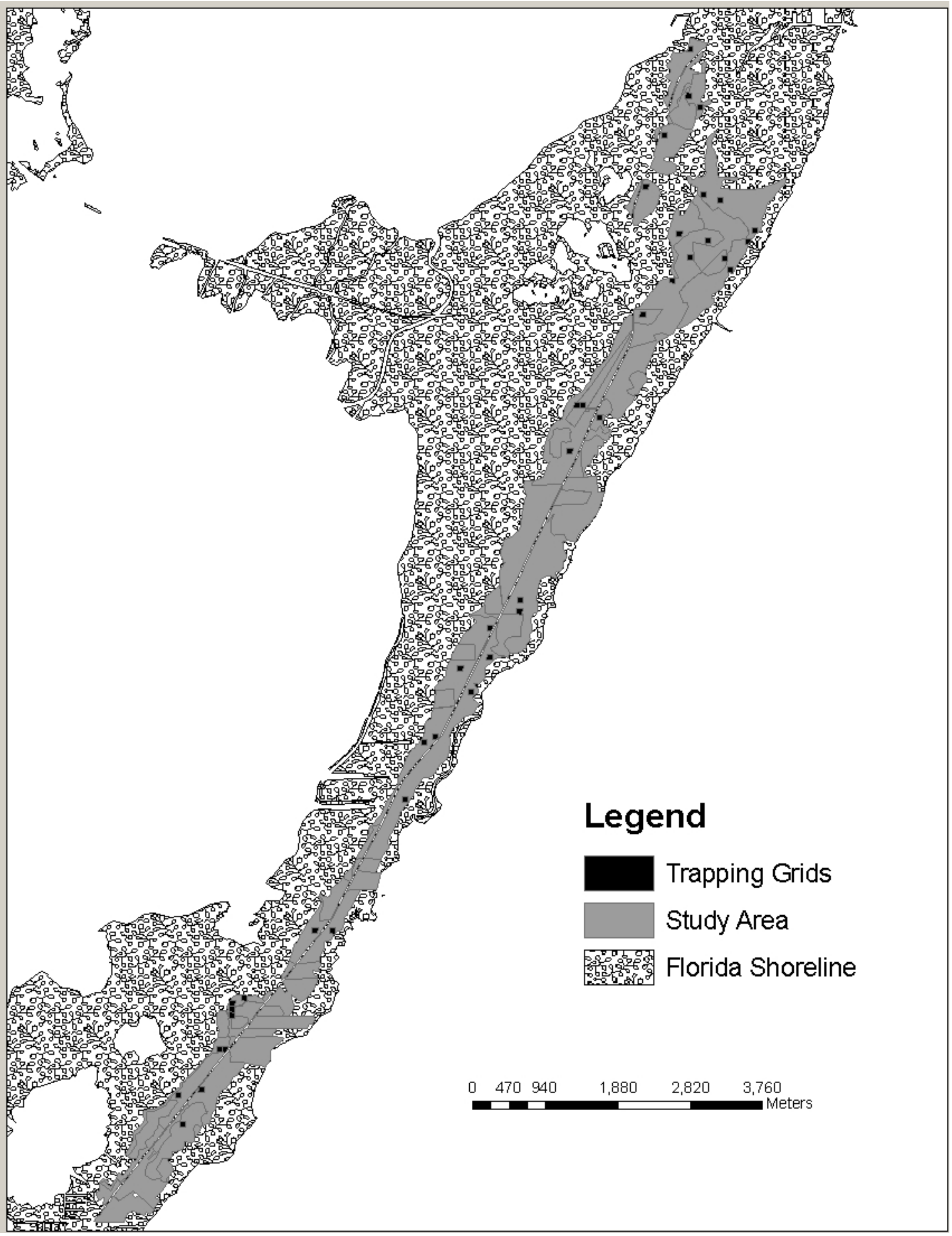

Figure 4.1. Map of study area delineating hardwood hammock from mangroves of north Key Largo, Florida and locations of 40 trapping grids used to estimate abundance of the Key Largo woodrat, 2005.

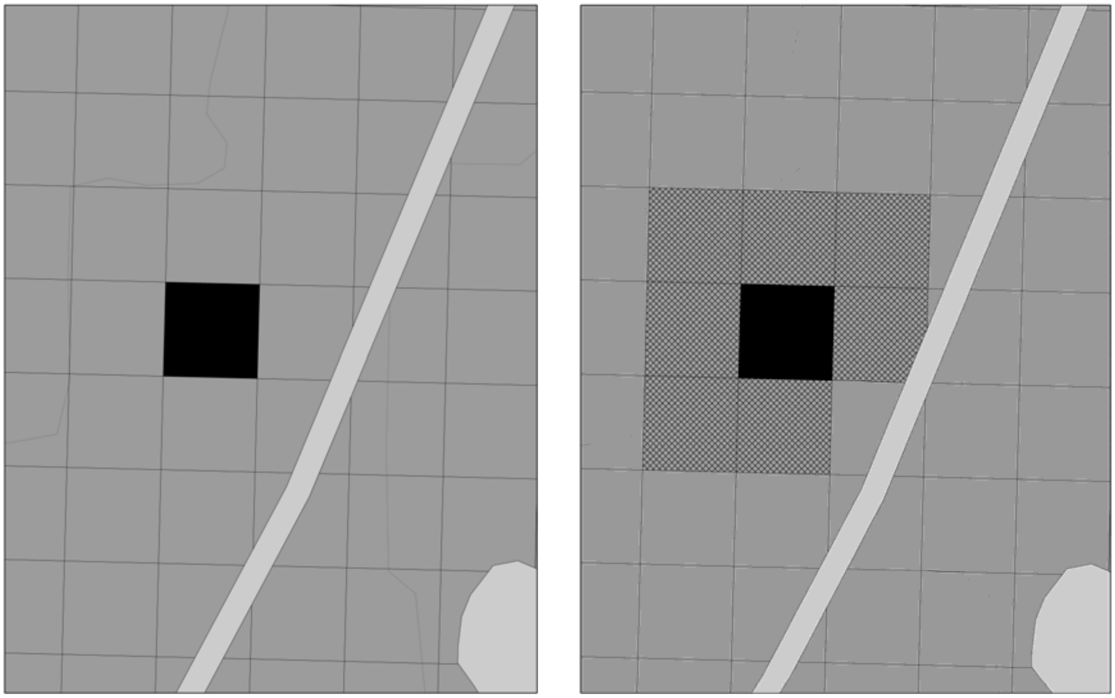

Figure 4.2. Example of adaptive cluster sampling (cross-hatched squares) around an initial random sample (black square) that met the pre-defined criteria of 1 unique individual, with the neighborhood for adaptive sampling defined as all immediately surrounding units that could support a 3 x 3 trapping grid with 25 m spacing. The resulting network consisted of 8 sampling units. The sampling unit on the bottom right corner was intersected by a road, which was considered a barrier to movement, and was not included as part of the network.

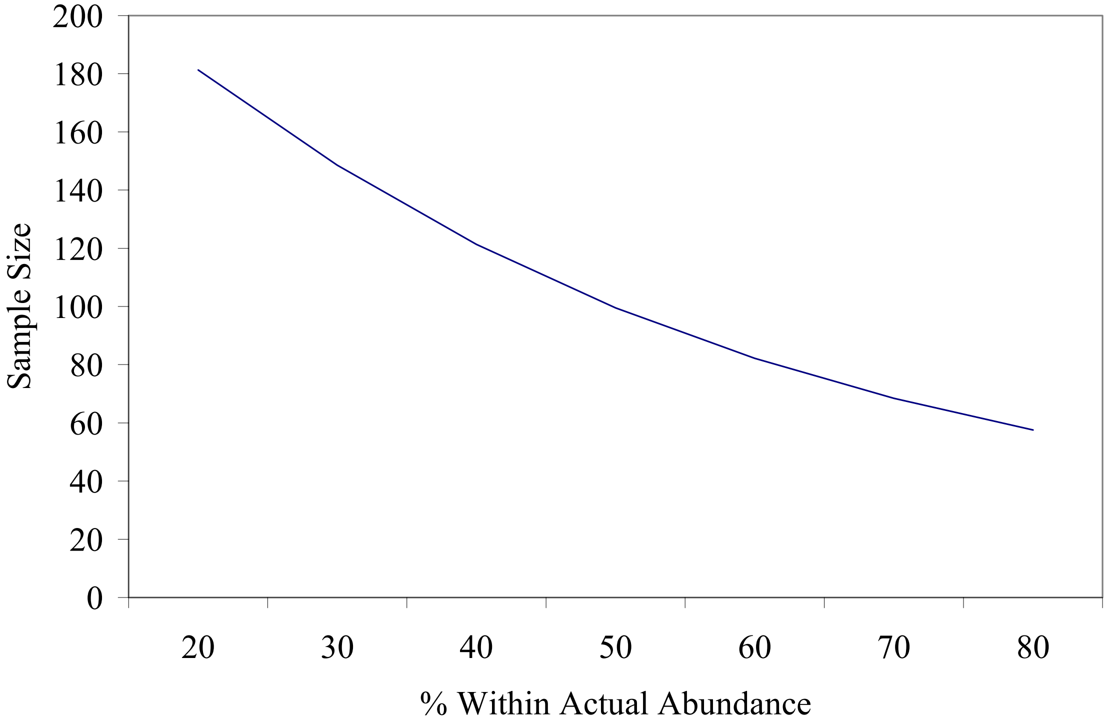

Figure 4.3. Sample size (number of trapping grids) required to estimate abundance of KLWR within 20-80% of the actual value ($\alpha = 0.10$) using a stratified random design, with three strata delineated by density of debris piles and forest age class. Sample sizes were calculated based on strata-specific sample variances from post-stratification of data collected on 40 trapping grids on north Key Largo, Florida.

CHAPTER 5

CONCLUSIONS

Despite the protection of its remaining habitat from development after being listed as federally endangered in 1984, the Key Largo woodrat (KLWR, *Neotoma floridana smalli*) population has continued to decline (McCleery et al. 2006). The cause of the decline is unknown, but several factors, such as degraded habitat quality, increased predator densities, and reduced availability of quality nesting substrate, may be contributing. Given the uncertainty regarding the specific causes of decline, the goals of my research were to examine factors potentially restricting the KLWR population, including predator activity, structure and composition of vegetation, and availability of nest sites, by identifying macro- and microhabitat selection. Nest site selection was evaluated to distinguish between the importance of artificial nest substrate and hammock age as KLWR nesting habitat. In addition, I compared the efficiency of adaptive cluster sampling (ACS; Thompson 1990) to stratified random sampling to recommend an optimal design to monitor the abundance of KLWR, and evaluate the effectiveness of recovery efforts. Collectively, the results were used to recommend a management strategy to recover the population and monitor the population's response to management actions.

My KLWR occupancy estimate (22.8%) was consistent with previous estimates (McCleery et al. 2006), and supports previous conclusions that the population maintains a limited distribution within the remaining hardwood hammocks of Key Largo. In evaluating macrohabitat selection, I found that KLWR selected areas with higher abundances of artificial and natural nest substrate. The abundance of artificial substrate (i.e., rock and debris piles) was the most important variable in predicting the occurrence of KLWR, followed by areas with larger overstory trees, of which the root systems are used as nest sites. In addition, KLWR selected areas with lower abundances of feral cats (*Felis catus*) and raccoons (*Procyon lotor*). The distribution of KLWR, therefore, appears restricted by a reduction in quality nesting habitat, which may be exacerbated by high mammalian predator abundance. Although a mammalian predator effect was supported by the data, manipulative studies are needed to better determine if predator reductions would effectively increase KLWR occupancy.

Observed nest use was consistent with macrohabitat selection results, where individuals mainly used the root systems of overstory trees and artificial substrate for nest sites. Artificial nest substrate was used more frequently, however, and KLWR was found to select nest sites based on the availability of artificial nest substrate. The high proportional use of artificial substrate suggests naturally occurring nest sites are in short supply, possibly due to past disturbances to the hammocks (i.e., farming and development). Due to the relatively short time since disturbance (70-80 years), current hammock conditions may not be fully representative of mature hammock, lacking trees large enough to provide adequate substrate for constructing natural nests. Additionally, artificial substrate likely offers greater protection from predators than natural substrate. Thus, high predator densities may be furthering the reliance on rock and debris piles for refugia. Although KLWR nested in younger hammock, the observed association appears to be confounded with the abundance of artificial substrate available in younger patches of hammock, as hammock age class alone was a poor predictor of nest site selection.

I found no evidence that KLWR selected foraging locations based on the structure or composition of vegetation. Consistent with central-place foraging theory (Schoener 1979), I found that KLWR forage more frequently closer to the nest. The limited range of movements, and lack of habitat selection within the home range, suggests individuals may not be restricted by food resources or microhabitat, and can readily obtain necessary resources without incurring high risks associated with long distance movements.

ACS was less efficient than stratified random sampling as a method to estimate total population size. Delineating three strata based on the density of artificial substrate and hammock age using poststratification, effectively lowered estimator variance and produced the most precise estimate of total population size. Despite the reduction in estimator variance, gaining meaningful estimates of total population size with the optimal sampling design will be costly. Given that habitat improvement is likely to be conducted at smaller spatial scales, monitoring total population size may be unnecessary. Conducting habitat improvement under the framework of adaptive management, where the effects of prescribed management at treatment sites are measured against control sites, would provide feedback on the effectiveness of management in recovering the population without the costs of monitoring total population size.

Based on my results, KLWR would likely benefit from the continued protection of the hardwood hammocks which would promote the maturation of the hammock, increasing the availability of large overstory trees used as nest sites. However, immediate habitat improvement may require the addition of artificial nest substrate while the forest recovers from past disturbances. Given the evidence of a potential negative effect of high predator densities, experimental predator reductions, coupled with the addition of artificial nest substrate, may be the most effective means of improving habitat quality, while allowing for greater insight into the roles of each factor on limiting the KLWR population. Monitoring the effects of habitat improvement at small spatial scales would likely be the most cost effective approach for evaluating the success of prescribed management.

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