LOWER KEYS MARSH RABBIT AND THE SILVER RICE RAT:

STEPS TOWARDS RECOVERY

A Thesis

by

NEIL DESMOND PERRY

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

MASTER OF SCIENCE

August 2006

Major Subject: Wildlife and Fisheries Sciences

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Approved by:

Chair of Committee,	Roel R. Lopez
Committee Members,	Philip A. Frank
	Donald S. Davis
Interim Head of Department,	Delbert M. Gatlin, III

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ABSTRACT

Lower Keys Marsh Rabbit and Silver Rice Rat: Steps Towards Recovery.

(August 2006)

Neil Desmond Perry, B.S., University of Massachusetts

Chairs of Advisory Committee: Dr. Roel R. Lopez

Extensive development has destroyed and fragmented wildlife habitat in the Lower Florida Keys. The Lower Keys marsh rabbit (LKMR; *Sylvilagus palustris hefneri*) and the silver rice rat (SRR; *Oryzomys argentatus*) are listed by the United States Fish and Wildlife Service (USFWS) and the Florida Fish and Wildlife Conservation Commission (FFWCC) as endangered species. Both species depend on coastal prairies, freshwater marshes, and intertidal salt-marsh zones. The objective of this study was to meet specific, species-level recovery goals and to add reliable information that may modify or support current recovery plans. Specifically, I (1) evaluated the use of LKMR reintroduction to suitable habitat, (2) examined characteristics of habitat used by LKMR, and (3) surveyed the Lower Florida Keys for SRRs, documenting current range and examining survey results for the past decade.

I reintroduced 7 rabbits (3 males, 4 females) to suitable habitat on Water Key, and monitored their survival and release-site fidelity. All reintroduced rabbits survived and some reproduced, suggesting these translocation techniques are a viable tool for recovery. On Boca Chica Key, I radio-collared 13 LKMRs and compared vegetation characteristics between core-use and avoided areas within home ranges. Binary logistic regression associated rabbit use with high vegetation heights (7–8 dm), low canopy coverage (≤10%), high bunchgrass densities (2.5–3.8/sq m), and forb presence (>5%), supporting the hypothesis that LKMRs may be detrimentally impacted by hardwood encroachment into salt-marsh habitats. For LKMR recovery, I recommend management to resist hardwood encroachment, together with active predator control.

I surveyed 36 locations on 18 islands for SRRs, capturing rats on 12 islands, including 2 on which SRRs had not previously been found. Comparisons of my data with historic data suggest SRRs either have increased in abundance over the past decade or that previous trapping efforts were not effective. Abundance of SRRs does not appear to be significantly different from that of populations of rice rats on the mainland. The USFWS and FFWCC should consider revising the conservation status of the SRR; however, it still should be regarded as a unique evolutionary unit with a very limited potential range.

DEDICATION

To Mom and Dad

Your support is unbounded, I'm proud of you both!

ACKNOWLEDGEMENTS

Beyond his support as a member of my committee, Phil Frank conceptualized and ensured funding for this project, also providing essential logistical support as the project leader for the Florida Keys Refuge System. Thank you for your advice and encouragement—I am grateful to have had such a meaningful graduate school experience. Thanks to my wonderful technicians Leslie Witter and Amanda Crouse for enduring long days of muck-walking mosquito biting fun in the sun; it was my pleasure to work with such talented people. Thanks to Craig (rabbit man) Faulhaber for teaching me so much about the keys and the rabbit. And to the rest of my cohorts at Texas A&M University, especially Tomas P. Dixon, Clay W. Roberts, Israel Parker, and Dominique Watts, thank you for your help in the field and your critical insights for my projects. To my committee, past, present, and unofficial, Roel Lopez, Philip Frank, Donald Davis, Nova Silvy, and Rodney Honeycutt, thank you for your guidance, support and sufficient rope to hang myself with. I also thank Texas A&M University, the Texas Agriculture Experiment Station, the staff at 210 Nagle Hall, and particularly Jennifer Baker for always knowing how to fix my mistakes. I thank the staff at the National Key Deer Wildlife Refuge for being so accommodating and fun to work with. I thank the Vero Beach Ecological Services office for their financial support, and particularly Philip Hughes for your insight, suggestions, and advice regarding my projects. Thanks to Heather Cole, your support and encouragement made the latter half of my graduate school experience all the more rewarding. Very special thanks to my Mom and Dad for a lifetime of support and encouragement.

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CHAPTER I

INTRODUCTION

BACKGROUND

Exploding human population growth and subsequent development is depleting wildland habitats and rapidly changing conditions for the wildlife communities they support (Forys et al. 1996, Vitousek et al. 1997). Many wildlife populations have been fragmented and disconnected, with local subpopulations becoming less viable because of recent isolation (Johst et al. 2002). As remaining natural areas continue to be fragmented, conservation biologists must focus on preservation of ecological and evolutionary dynamics of small and fragmented populations (Lande 1988, Caughley 1994, Frankham 1998). Forced to contend with limited knowledge of rare and endangered species, wildlife managers are implementing conservation strategies based on limited data, often using demographic and life history data from research of similar, more thoroughly studied species (Petit and Pors 1996). As such, conservation plans should incorporate efforts to gain reliable knowledge about life history strategies and regulatory factors of the species of concern. A valid monitoring program, incorporating opportunities to acquire knowledge pertinent to conservation, is a priority during the development of a conservation or recovery plan (Walters 1986, United States Fish and Wildlife Service [USFWS] 1999). Further, conservation plans should include plasticity—they must be dynamic and absorb information gained during execution of conservation measures.

Format and style follows the Journal of Wildlife Management.

Four decades of extensive residential and commercial development has destroyed and fragmented important habitat for much of the fauna of the Lower Florida Keys (Forys et al. 1996, USFWS 1999). Endemic fauna of the Lower Florida Keys includes the Lower Keys marsh rabbit (LKMR; *Sylvilagus palustris hefneri*; Lazell 1984), and the silver rice rat (SRR; *Oryzomys argentatus*; Spitzer and Lazell 1978). The USFWS and the Florida Fish and Wildlife Conservation Commission (FFWCC) list these species as endangered populations (USFWS 1990, 1991, 1999). Both species depend on coastal prairies, freshwater marshes, and intertidal salt-marsh zones (Negus et al. 1961, Goodyear 1987, Forys and Humphrey 1999*a*). Over the past 40 years, many of these areas were converted to ocean-side canal-based human developments. Though habitat destruction has been largely thwarted by recent conservation legislation, there are persistent, indirect threats to these species (Forys et al. 1996, USFWS 1999). Road kill and predation by free roaming feral cats (*Felis domesticus*) are 2 common factors affecting survivorship of LKMR (Forys and Humphrey 1999*a*). Although data are lacking, SRR also likely fall prey to feral cats (Forys et al. 1996). Black rats (*Rattus rattus*), introduced to most islands in the Florida Keys as early as 300 years before present (DePourtales 1877), compete directly for resources with SRR and likely depredate neonate LKMR and SRR (Dunson and Lazell 1982, Mitchel 1996, Forys et al. 1996). Imported red fire ants (*Solenopsis invicta*), a recent colonizer of the Florida Keys, also threaten neonate small mammals (Forys et al. 2002). Finally, encroachment of hardwood mangroves and buttonwoods, a result of resent changes in human land-use practices, also may be limiting the suitability of coastal prairies for LKMRs. A combination of all of these factors associated with human development is likely the cause of continued reductions in range and numbers for both of these endangered populations (Forys et al. 1996, Forys and Humphrey 1999*b*).

Study Area

The Lower Florida Keys are a densely spaced group of islands near the western end the Florida Keys archipelago (Figure 1.1). The Lower Florida Keys was isolated following a dramatic sea level rise between 15,000 and 10,000 years before present (YBP) (Mueller and Winston 1997), resulting in the evolution of a distinct endemic community. The human population of the Florida Keys has increased dramatically in the past century, from approximately 6,000 to 80,000 (Monroe County Growth Management Division 1992). The majority of development in the Lower Florida Keys, however, has occurred on Key West. Much of the land area of other human inhabited islands remains undeveloped (40%). Nonetheless, development has directly eliminated habitat critical to the Lower Florida Keys unique endemic community (Forys et al. 1996).

Adaptive Management for Endangered Species Recovery

Recovery efforts for rare and endangered species are inherently difficult to monitor and evaluate (Gibbs 2000). Species declines are often not detected until the population has already declined greatly (Westemeier et al. 1998). Contending with a limited population in an altered ecosystem, the cause of a population decline is often difficult to assess with certainty (Forys and Humphrey 1999*b*). To improve the base of knowledge, many conservation managers are adopting an adaptive management approach to land management and endangered species conservation (Walters 1986, USFWS 1999). Adaptive management plans incorporate multidimensional efforts targeting factors that likely reduce population and ecosystem viability. Evaluation of these efforts via monitoring programs is the backbone of adaptive management plans, revealing failures, providing managers with reliable knowledge, and eventually providing evidence of recovery success.

Population trends, however, can be difficult to track (Westemeier et al. 1998, Elkinton 2000). Failure to detect population trends, including those associated with management efforts could adversely affect rare species like LKMR and SRR. As managers attempt to improve habitat conditions and, thereby, promote an increase in the abundance of LKMR and SRRs, it is essential that these efforts incorporate evaluations (USFWS 1999). Wildlife managers must evaluate the effectiveness of each recovery action with scientifically valid methodologies. These results should be considered as future recovery actions are planned and/or modified.

In 1999, the USFWS completed its South Florida Multi-Species Recovery Plan (USFWS 1999). Initial recovery objectives for LKMR and SRR were to determine distribution, conduct research relevant to recovery efforts, and to execute and evaluate recovery actions. Species-specific sections below include some pertinent background information and explain how I proposed to meet specific recovery objectives for LKMR and SRR.

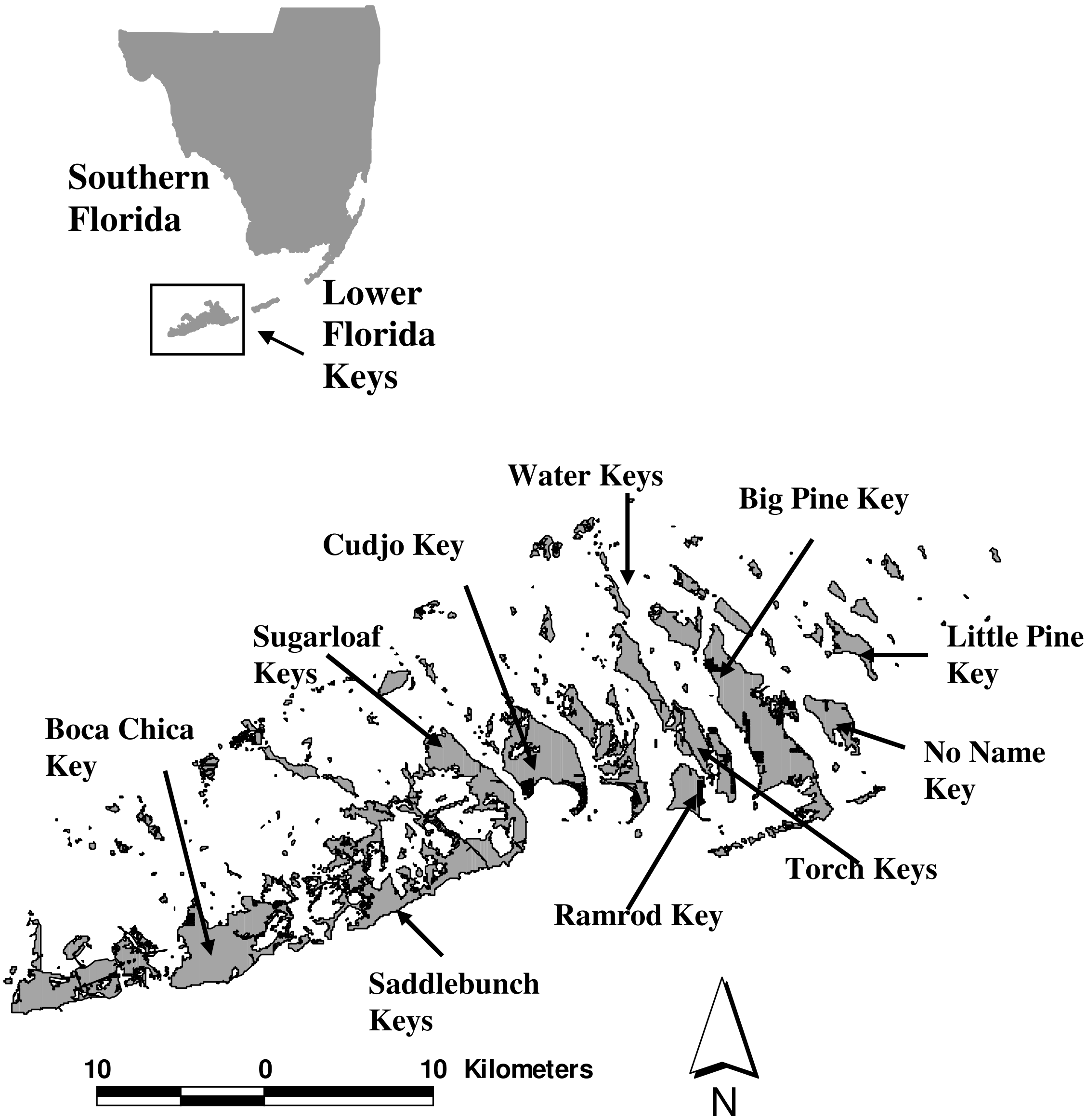

Figure 1.1. The Lower Florida Keys, Florida. The SRR and the LKMR are endemic to the Lower Florida Keys and their known ranges extend from Boca Chica Key to the west and Little Pine Key to the east.

LOWER KEYS MARSH RABBIT

Reintroduction

The USFWS (1999) proposed reintroduction as part of a recovery plan for the LKMR. Between January and April 2002, 11 LKMRs were reintroduced to Little Pine Key, an isolated and restored island with a large area (13.1 ha) of suitable habitat (Faulhaber 2003). A 12th animal was released in October of 2002. High survivorship during the first 5 months and evidence of reproduction (juvenile pellets found May 2003, personal observation), suggest that reintroduction to Little Pine Key has been initially successful (Faulhaber 2003). Identified in the USFWS species recovery plan as a good reintroduction site, Water Key has at least 4.3 ha of suitable habitat and can likely harbor 10–20 LKMR (USFWS 1999, Faulhaber 2003). Further research on the utility of translocations for LKMR is needed.

Habitat Characteristics

Conservation managers hypothesize that hardwood encroachment of buttonwood and mangrove trees may be a limiting factor for LKMR population viability in the Lower Florida Keys (USFWS 1999). Though there has been some research regarding habitat use by LKMR, none of the radio-telemetry based research to date has included night-time tracking (Forys 1995, Faulhaber 2003). Reliable information on LKMR habitat use is essential as managers seek to restore and/or maintain the coastal-prairie ecosystem that supports LKMRs. In 2004, the US Navy began efforts to explore means to maintain low woody vegetation (e.g., mangroves, buttonwood) surrounding the Key West Naval Air Station runway system in Boca Chica, Florida. The objective of the

Navy was to identify methods of maintaining runway safety standards while minimizing adverse affects to LKMRs. Deemed one of the most important of 3 recognized LKMR sub-populations (Forys 1995, Crouse 2005), a large proportion of Boca Chica Key's LKMR population exists in close proximity to the runway system. The preferred option, after clearing high vegetation, is the planting of coastal wet-prairie plant species to reduce woody encroachment rates, and ideally enhance exiting habitat conditions for LKMRs. Research is needed on the effects of proposed vegetation management strategies.

SILVER RICE RAT

Distribution Survey

In 1991, the USFWS listed the SRR as an endangered vertebrate population and designated critical habitat in 1993 (USFWS 1999). The last comprehensive distribution survey for SRRs was completed between 1995–1996 (Forys et al.1996). Following the recommendations of the USFWS species recovery plan, my primary objective was to survey and provide an updated distribution of SRR. I also examined population trends using published and unpublished historic data.

RESEARCH OBJECTIVES

1. Evaluate the reintroduction program for LKMR
    a. Translocate 7 LKMR to Water Key, Great White Heron National Wildlife Refuge.
    b. Monitor the persistence of the introduced population.
2. Conduct a radio-telemetry project to evaluate habitat characteristics of LKMR

a. Compare vegetative characteristics within core-use and non-core use areas.

b. Evaluate the hypothesis that coastal hardwood encroachment is detrimental to LKMR population viability

3. Conduct a distribution survey for SRR in the Lower Florida Keys.

    a. Conduct trapping surveys on 15 islands to determine current distribution.

    b. Examine trends and the status of the LKWR population.

CHAPTER II

REINTRODUCTION OF LOWER KEYS MARSH RABBITS TO WATER KEYS

SYNOPSIS

The Lower Keys marsh rabbit was listed as endangered by the United States Fish and Wildlife Service (USFWS) in 1990. Despite legal protection, a population viability analysis predicted the LKMR metapopulation would become extinct within 30–40 years without active management (Forys and Humphrey 1999b). In 2002, a pilot study that reintroduced 13 LKMRs to Little Pine Key, an isolated island with a relatively large area of suitable habitat (13 ha), was conducted to assess the effectiveness of reintroductions in the recovery of LKMR (Faulhaber 2003). High survival (81%) during the first 5 months and evidence of reproduction suggested reintroduction was a feasible management tool. To further evaluate the translocation techniques used, I introduced 7 LKMRs to Water Key, an island with a moderate area of suitable habitat (10 ha). Survivorship on Water Key during the first 5 months (100%) and evidence of reproduction validates these translocation techniques as a viable tool for recovery biologists. Long-term success of this reintroduction program will depend on availability of translocation candidates and possibly an *in-situ* captive breeding program.

INTRODUCTION

Exploding human population growth and subsequent development has increased the conversion of wild lands and the wildlife communities they support (Vitousek et al. 1997). Many wildlife communities have been fragmented and disconnected, with sub-populations becoming less viable because of recent isolation (Johst et al. 2002). As remaining natural areas continue to be fragmented, conservation biologists must focus on the ecological and evolutionary dynamics of small populations (Lande 1988, Caughley 1994, Frankham 1998). This becomes particularly important in the recovery of an endangered species and restoration of critical habitats. Endangered species that exist in fragmented landscapes results in a myriad of challenges for conservation biologists.

Following four decades of increased commercial and residential development, the endemic fauna of the Lower Florida Keys has faced extensive habitat fragmentation (Forys et al. 1996). The Lower Florida Keys are a densely spaced group of islands within the Florida Keys archipelago, isolated following a dramatic sea level rise between 10,000–15,000 years ago (Mueller and Winston 1997). In the past 30 years important natural areas have been lost, replaced with development and road systems that have left remaining native areas fragmented. Human development also has created new, less direct threats (e.g., introduction of non-native predators) which continue to affect the fauna and flora of the Florida Keys (Lazell 1984, Forys and Humphrey 1999*b*). Indeed, many local endemics are listed as threatened or endangered by state and federal agencies.

The LKMR is a sub-species of marsh rabbit endemic to the Lower Florida Keys (Lazell 1984). The LKMR was listed as an endangered subspecies by the USFWS and the Florida Fish and Wildlife Conservation Commission in 1990 (FFWCC; USFWS 1990, 1999). The LKMRs typically occupy wet areas with dense cover (Forys 1995, Faulhaber 2003) including salt-marsh, buttonwood transition zones, and freshwater marshes. Road kill and predation by free roaming feral cats (*Felis domesticus*) are 2 common factors affecting survivorship of LKMRs (Forys and Humphrey 1999*a*). A population viability analysis has predicted the LKMR could be extinct in as few as 30–40 years (Forys and Humphrey 1999*b*). Management of remaining habitat is essential for the persistence and recovery of the LKMR populations.

The LKMR habitat in the Lower Keys does not occur in large contiguous areas (Faulhaber 2003). Thus, LKMR exist as a metapopulation in distinct habitat patches scattered throughout the Lower Keys. Persistence of some populations will depend on emigration from others; their roles as sinks and sources can and do vary depending on patch-specific demographics (Hanski and Gilpin 1991, McCullough 1996). In the past 30 years, more than half of the LKMR habitat has been destroyed by residential and commercial development (USFWS 1999). What habitat remains is fragmented; new dispersal obstacles hindering migration between patches (Forys 1999*a*, Faulhaber 2003). As such, translocation of LKMRs to suitable habitat within their historic range could be an important tool for recovery biologists—a means to establish and/or maintain LKMR sub-populations. In 2002, 13 LKMRs were reintroduced to Little Pine Key as an experimental evaluation of translocation techniques (Faulhaber 2003). No rabbits died or suffered any noticeable injuries during translocation and exhibited excellent fidelity to release sites (Faulhaber 2003). As many as 3 reproductive events occurred in the first year after monitoring (Faulhaber 2003). Results from his study indicated (1) adult rabbits can be moved with little or no mortality, (2) translocated adult rabbits can establish themselves and survive in a new environment, and (3) translocated rabbits will eventually breed.

My primary objective was to reintroduce the LKMR to Water Key using the techniques developed by Faulhaber (2003). In doing so, I hoped to validate the efficacy of his techniques and re-establish a sub-population of LKMR on a protected island.

Study Area

Reintroduction Site.--The USFWS species recovery plan for the LKMR identifies Water Key, a 92-ha island, as a good reintroduction site (Fig. 2.1, USFWS 1999). Water Key has approximately 6 hectares of suitable habitat and can likely harbor 20-40 LKMRs (USFWS 1999, Faulhaber 2003). Water Key is located approximately 100 m north of Big Torch Key, a 605-ha island currently unoccupied by LKMR, but with a substantial amount of suitable LKMR habitat (Fig. 2.1). Big Torch could serve as a dispersal zone for LKMRs from Water Key and is being considered as the next location for a LKMR reintroduction (Phillip Hughes, USFWS, personal communication). Water Key is part of the Great White Heron National Wildlife Refuge managed by the USFWS, and it also is a Florida Keys Wilderness Area, part of the National Wilderness Preservation System designated by the U. S. Congress through the Wilderness Act of 1964.

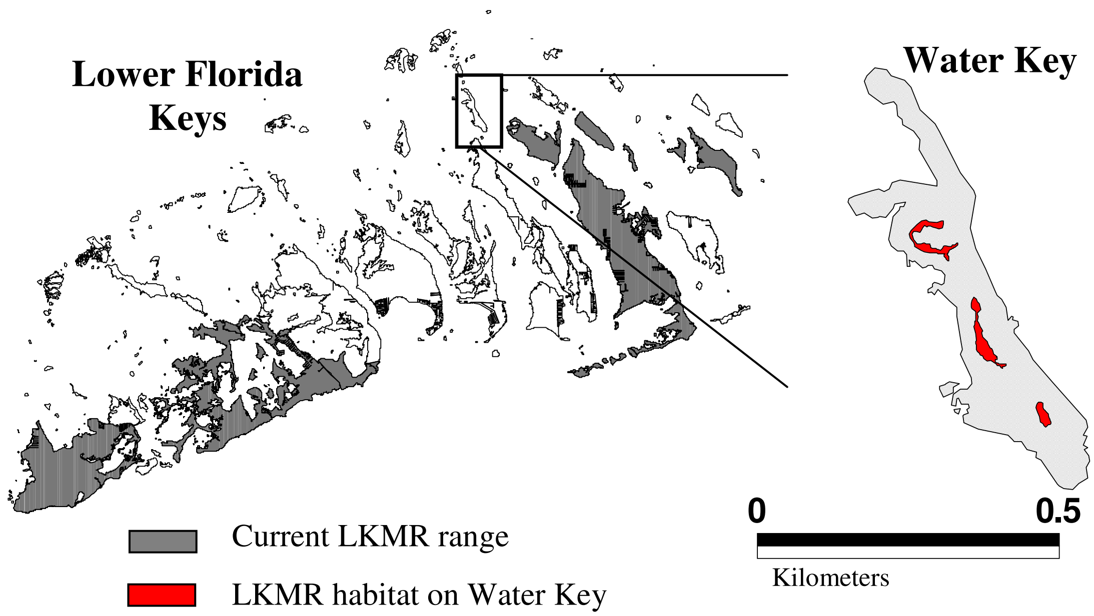

Figure 2.1. Lower Keys marsh rabbits (LKMR) were translocated to a patch of suitable habitat in the center of Water Key in 2004; 2 additional patches may serve as dispersal zones for future generations of LKMR.

Source Areas.--Annual fecal pellet monitoring data were used to identify source populations with consistently high indices of pellet abundance of LKMR over the past 3 years to avoid long-term effects to source populations (Forys 1999*a*, Faulhaber 2003). Furthermore, rabbits were taken from a variety of patches from 2 different geographic groups (Boca Chica and Sugarloaf keys, Fig. 2.2) throughout the LKMR range to reduce inbreeding potential (Lomolino 1986, Hedrick and Kalinowski 2000, Mansfield and Land 2002). The habitat patch I trapped on Sugarloaf Key includes 22 ha of contiguous LKMR habitat and maintains reasonable connectivity to other nearby patches. The 3

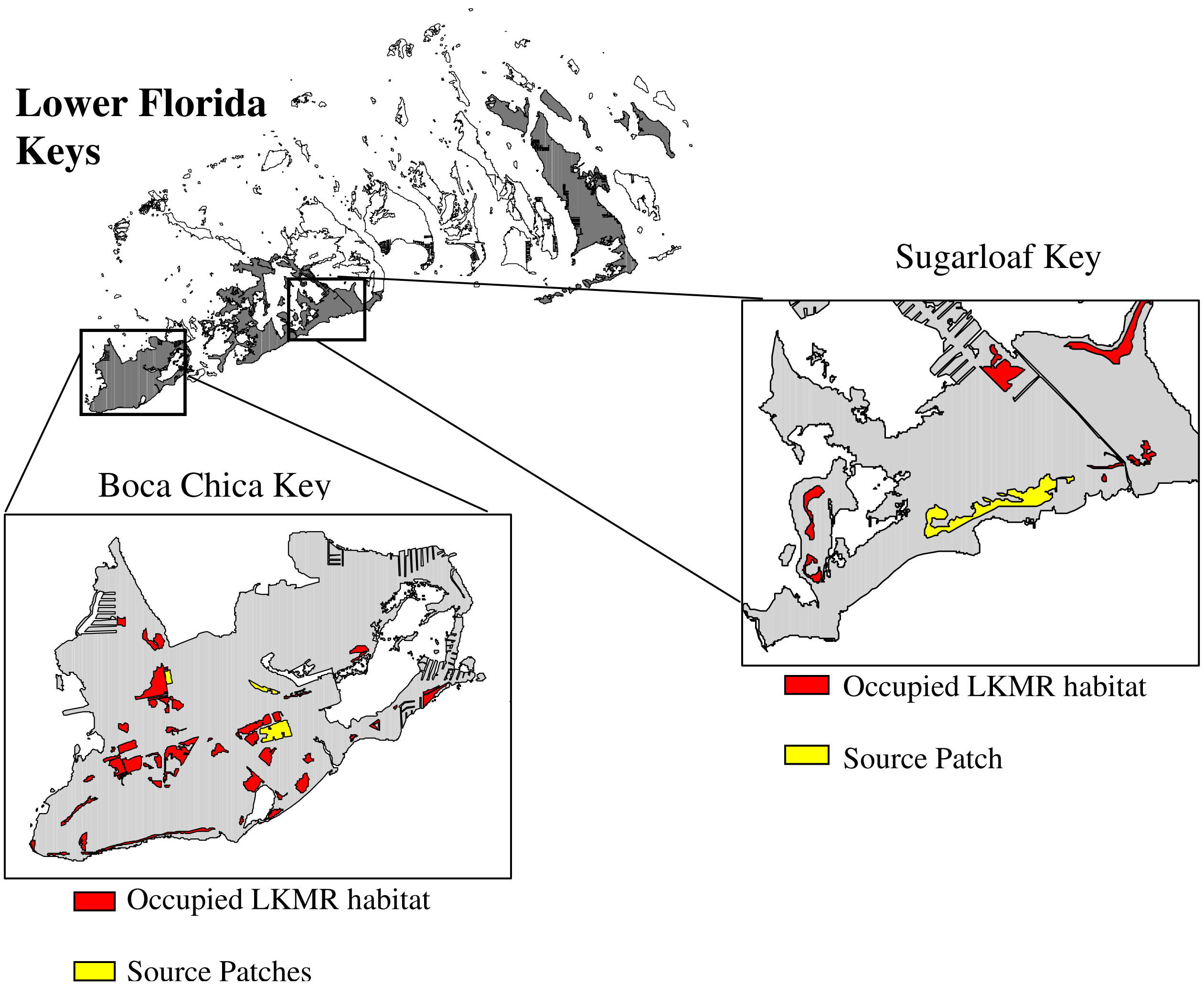

Figure 2.2. Translocated Lower Keys marsh rabbits (LKMR) trapped from 3 areas on Boca Chica Key and 1 area on Sugarloaf Key in the Lower Keys of Florida, USA. Trapping was preceded by ocular indices of fecal pellets suggesting high densities of LKMRs.

habitat patches trapped on Boca Chica Key were not as large (12, 3, and 2 ha) but were in close proximity to other patches, all of which have maintained consistent populations of LKMR over the past decade (Forys 1995, Faulhaber 2003).

METHODS

Translocation

All LKMRs were trapped using 2-door 60 x 18 x 18 cm Tomahawk (Woodstream Corporation, Lititz, Pennsylvania, USA) traps. Traps were opened at night and checked early the next morning as suggested by the American Society of Mammalogists (1998). Traps were set without bait, disguised with vegetation, and drift fences were used to direct rabbits towards traps (Faulhaber 2003). Upon removal of a rabbit from a trap, a hood was placed over the rabbit's eyes to reduce subsequent handling stress. Rabbits were fitted with neoprene collars and transmitters with an estimated life of 200 days (Advanced Telemetry Systems, Isanti, Minnesota, USA; 16-22 g/radio; <2% of body weight). Each transmitter had a mortality sensor, negating the need for visual locations. Only adults and subadults greater than 1,000 g were considered for translocation. Other factors considered when selecting individuals for translocation included age, presence of ectoparacites, and any external wounds or physical impairment. Animals deemed inappropriate for transport based on these visual criteria were released at the site of capture.

Rabbits were transported in a padded "pet carrier" to lessen the risk of injury. We did not anaesthetize the rabbits because rabbits rely heavily on their speed and reflexes to evade predators, and detaining the rabbits until the anesthesia wore off would subject them to more stress. Releases were "hard," directly into the habitat patch with no acclimation pen or supplemental food. Using these methods, Faulhaber (2003) observed no injury or mortality related to transport and found that survival of translocated rabbits was comparable to that of a control group in source populations.

Post-release Monitoring

Rabbits were tracked during daylight hours for each of the first 3 days after release, and then 1-2 times weekly until the transmitter failed or the rabbit died. We attained visual confirmation of rabbits when feasible. Locations were recorded with a WAAS-enabled Magellan global positioning system (Thales Navigation Inc., Santa Clara, California, USA). Locations were entered into a geographic information system using Arc View (Version 3.3; Environmental Systems Research Institute, Redlands, California, USA).

Fidelity to the release area was evaluated by measuring the mean distance from point of release to all telemetry locations after a rabbit established a stable range. During each trip to the island, researchers surveyed for juvenile fecal pellets to document breeding events. Juvenile pellets can be distinguished by size (Forys 1995). Mucous, composition, and color of pellets can indicate freshness, facilitating the discernment of different reproductive events.

Table 2.1. Seven Lower Keys marsh rabbits (3 males, 4 females) translocated to Water Key from June–July 2004. All rabbits, except one, outlived their transmitters.

Rabbit ID	Source island	Date moved (2004)	Gender	Mortality	Collar last heard
W1	Sugarloarf	24 May	female	--	20 Jan 2005
W2	Sugarloarf	25 May	male	--	20 Jan 2005
W3	Boca Chica	9 Jun	male	--	25 Oct 2004
W4	Boca Chica	13 Jun	female	--	21 Jun 2004
W5	Boca Chica	29 Jun	male	1 Feb 2005	--
W6	Boca Chica	6 Jul	female	--	20 Jan 2005
W7	Boca Chica	7 Jul	female	--	1 Feb 2005

RESULTS

Translocation

Seven rabbits (3 males, 4 females) were translocated to Water Key from June–July 2004 (Table 2.1). There were no injuries or mortalities associated with trapping or transport of LKMRs.

Post-release Monitoring

Rabbit W4 was last heard 7 days after release (Table 2.1). Rapid signal fading in successive relocations suggested the transmitter's battery failed; the rabbit's fate was not determined. All other rabbits survived the first 6 months after release, after which transmitters began to fail. The only mortality recorded during this study was estimated to have occurred 7 months after release. The carcass was discovered at least 2 weeks after death, making the cause of mortality difficult to determine.

Five of the 6 animals tracked showed site fidelity (Fig. 2.3). Rabbit W5 was found to use the area around his release periodically, exhibiting an unusually large daytime range. Rabbit W5 was found dead within 10 m of his release site. No animals dispersed away from the release area.

In December 2004, a juvenile LKMR was seen near the location of W6, confirming a breeding event. Juvenile pellets also were found in January 2005, though they were not fresh, and their proximity suggested they may have been associated with the above reproductive event. In March 2005, juvenile and subadult pellets were found near the north end of the release area, within the range of rabbit W7. This was at least 300 m north of the juvenile rabbit sighting and the previously found juvenile pellets, suggesting a separate breeding event. Thus, I surmise that at least 2 breeding events occurred within 6 months of reintroduction.

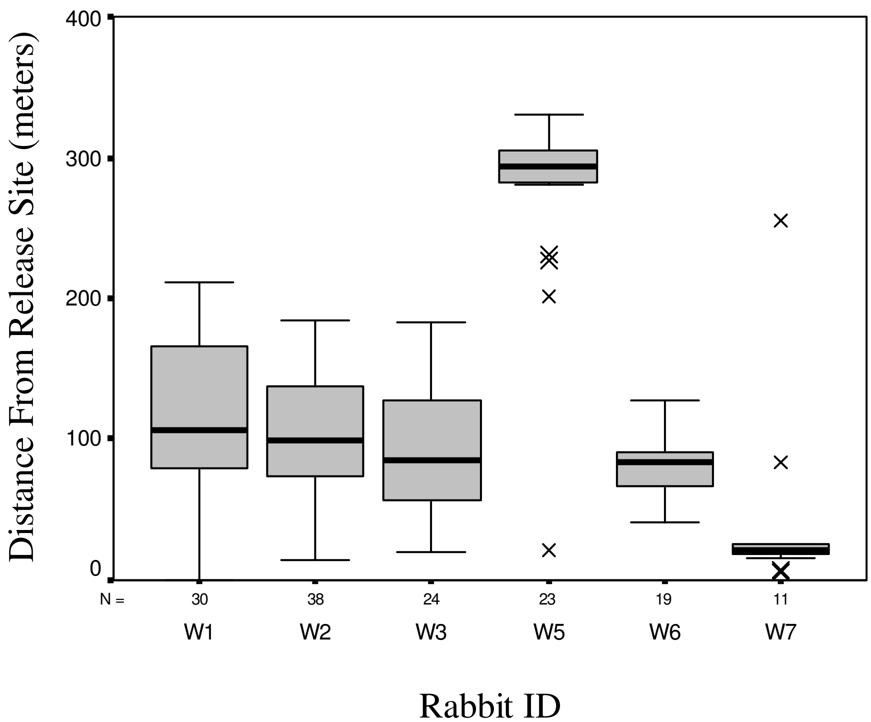

Figure 2.3. Mean distance from release sites (mean = solid black line, grey boxes include upper and lower quartile, 95% CI = error bars) for translocated Lower Keys marsh rabbits, Water Key, Florida. The number of locations used for each is listed under the X-axis; only points recorded after the animals established typical home ranges were included.

DISCUSSION

Translocation

Removal of individuals from habitat patches has not resulted in any decrease in pellet abundance (USFWS, unpublished data). I did not remove more than 2 rabbits from any patch (1 male, 1 female), to limit impacts on local demographics. Three weeks of trapping from an unrelated study in the smallest source patch on Boca Chica Key revealed 5 adults and 3 juveniles in approximately 10% of the patch, suggesting high densities 6 months after removal of 2 individuals (unpublished data).

Post-release Monitoring

Preliminary results of this translocation can be deemed successful based on the following criteria: (1) high survivorship of translocated LKMR's, (2) release-site fidelity, and (3) successful reproduction. Survival of all monitored rabbits after >5 months was better than Faulhaber's (2003) reintroduction group (81%) and his control group (79%). This might suggest that habitat quality on Water Key was superior to Little Pine Key. Reintroduced rabbits exhibited release-site fidelity, supporting good selection of release areas. Though reproduction was not detected for 5 months after release, my survey data suggest that at least 2 of 3 monitored females on Water Key did reproduce within the first 6 months. By comparison, 2 breeding events were detected on Little Pine Key within the first 6 months (Faulhaber 2003). Moreover, some breeding events may have been missed, as juvenile pellets are small and difficult to detect.

Animals were collected from throughout the current range of the LKMR to reduce risks of inbreeding, but given such small founder numbers, demographic stochasticity could confound breeding opportunities (Hedrick and Kalinowski 2000). Ideally, managers could augment the existing populations with more releases over time (Mills and Allendorf 1996). However, on Little Pine Key, Faulhaber (2003) recorded evidence of patch saturation. That is, an established female dispersed from a patch after introduction of a new female. Introduction of animals at inappropriate times could pose risk to subordinate females with a litter. Also, the introduced animal could be forced into sub-optimal habitat with reduced resources and increased exposure to predation, thus reducing its breeding opportunities (VanZant and Wooten 2003). Managers might consider executing a well monitored follow-up translocation. Successful establishment of translocated rabbits into established sub-populations may expand the utility of translocations. That is, animals could be used to augment populations throughout the rabbits range.

Though the goal of reintroductions is long-term persistence of established sub-populations, variable occupancy of these habitat patches may be inevitable. Both the Water Key and the Little Pine Key sub-populations are on the periphery of a complex of currently occupied LKMR habitat patches. Metapopulation theory contends that sub-populations on the periphery of a population cluster have an increased likelihood of extirpation due to limited immigration (Brown and Kodric-Brown 1977, Lomolino 1986, McCullough 1996). Conversely, given the relatively large areas of quality habitat and their protected status, these new island populations may persist without natural augmentation from adjacent patches; they could become functional metapopulation sources. In fact, at least 1 dispersing animal has colonized another island near Little

Pine Key since the reintroduction (USFWS, unpublished data). Future monitoring may provide insight into the demographics of these populations and their relationship with the LKMR metapopulation. Supplemental translocations may be necessary to ensure persistence of LKMRs on these islands.

This study validated the utility of translocation as a tool for LKMR recovery biologists. The LKMR are tolerant to hard release and translocated individuals had high survivorship following release. All translocated animals that I monitored established stable ranges, most exhibiting fidelity to release sites, and I documented reproduction. The LKMR have a small range and are limited to distinct patches of suitable wetland habitats, much of which is now fragmented by human development (Forys 1995, Faulhaber 2003). Reintroductions to suitable habitat, thus increasing the number of occupied patches and/or ensuring the persistence of existing patches, will improve the long-term viability of the subspecies (Den Boer 1968). Using the protocols detailed in this manuscript, managers should include translocation in future LKMR recovery plans.

MANAGEMENT IMPLICATIONS

Monitoring persistence of this established population should be facilitated via the LKMR annual monitoring program, coordinated by USFWS recovery biologists. The monitoring protocol ensures that the release area and the 2 other potential habitat patches on Water Key will be surveyed annually for LKMR presence. Genetic health of the Water Key and Little Pine Key populations should be monitored and evaluated at some point in the future. This could be executed using fresh pellets or by trapping and collection of serological samples.

Future reintroduction areas are limited; Little Pine and Water keys were the only 2 areas that we met all of our reintroduction criteria. Unfortunately the proximate cause of extirpation in many areas is not certain, making restoration priorities difficult to deduce. Nonetheless, managers should try to identify proximate causes of extirpation at other potential release sites and execute some measure of habitat enhancement or restoration before reintroduction is conducted. Further, these measures must have some certainty of permanence. For example, removing feral cats from an area may yield a short-term reintroduction success, but re-colonization by feral and free roaming cats may limit the perseverance of the sub-population. Hence, for this example, long-term cat control must be ensured before animals are reintroduced.

Availability of individuals for translocation is essential for this reintroduction program. Restoration of reintroduction areas should take second priority to habitat enhancement and protection of source areas, ensuring that they can maintain or, preferably, increase densities of rabbits in currently occupied patches. Further, Little Pine Key and Water Key both contain good LKMR habitat and have maintained survivorship levels as good as or better than the translocation source populations. These islands could become source populations for future translocations.

CHAPTER III

HABITAT CHARACTERISTICS OF THE LOWER KEYS MARSH RABBIT

SYNOPSIS

The endangered Lower Keys marsh rabbit (LKMR; *Sylvilagus palustris hefneri*) is largely limited to a narrow ecotone in the Lower Florida Keys referred to as coastal transition zone. Late successional stages of this habitat zone, characterized by dense over story of hardwoods and decreased grasses and forbs, may be unsuitable for LKMRs. My study objective was to identify microhabitat characteristics associated with rabbit occurrence. From October 2004–February 2005, I trapped and radio-collared 13 rabbits (8 M, 5 F). I compared (1) canopy cover, (2) vegetation height, (3) horizontal shrub cover, (4) ground cover, and (5) vegetative composition between core areas (50% kernel estimates) and non-core areas (remaining areas; minimum convex polygon less core areas) using binary logistic regression. Best model fits (90% concordance) predicted rabbit co-occurrence with high vegetation (7–8 dm), low canopy coverage (≤10%), high bunchgrass densities (2.5–3.8/sq m), and forb presence (>5%). These results support the hypothesis that LKMRs select areas with less canopy cover and high levels of ground cover. Future management measures should consider these habitat characteristics in drafting guidelines.

INTRODUCTION

The LKMR is a subspecies of marsh rabbit endemic to the Lower Florida Keys. The LKMR was listed as an endangered population by the United States Fish and Wildlife Service (USFWS) and the Florida Fish and Wildlife Conservation Commission in 1990 (FFWCC, USFWS 1999). Marsh rabbits typically occupy wet transitional areas with dense vegetative cover (Layne 1974) including coastal wet-prairie and buttonwood transition zones (Forys 1999*a*). These unique vegetation types have been largely fragmented by development in recent years (Forys 1999*b*); for example, many coastal wetlands have been dredged and converted to canal-based developments that afford views and access to the water. Futhermore, recent research (e.g., Litvaitis and Villafuerte 1996, USFWS 1999, Forys 1999*a*) and observations of land managers (P. Frank, USFWS, National Key Deer Refuge, personal communication) also suggest that hardwood encroachment and recession of herbaceous vegetation in the coastal transition zones have increased over the past 3 decades. Lacking adequate ecological drivers (i.e., disturbances), buttonwood hardwoods (dominant overstory species) grow vigorously in this environment, creating shade and dense leaf-litter that inhibits persistence and regeneration of understory vegetation. Increased fire suppression following increased development in the 1960’s and 1970’s likely has resulted in less suitable habitat for LKMRs (Bergh and Wisby 1996). Increased fragmentation and resulting hardwood encroachment of coastal wetlands are attributed to LKMR population declines observed over the past 3 decades (USFW 1999).

Previous research on LKMR microhabitat selection has included daytime form site characteristics (Faulhaber, in press) or vegetative composition associated with the presence of fecal pellets (Forys 1995). Limitations in these studies include a lack of night-time observations when LKMR are most active and foraging. Habitat restoration or mitigation programs require an understanding of necessary vegetative structure selected by LKMR. Such information can provide guidelines for restoration efforts and aid in the recovery of this species. For example, the US Navy is in initial stages of a project to remove overstory vegetation adjacent to runway systems on Boca Chica Key, Naval Air Station Key West (NASKW) to improve safety conditions and achieve compliance with NAVFAC P-80.3, *Facility Planning Factor Criteria for Navy and Marine Corps Shore Installations*. Areas to be cleared include occupied LKMR habitat. Though removal of hardwood trees may benefit LKMRs, the Navy is exploring means to restore and maintain herbaceous and grass vegetation, thus achieving their safety goals and enhancing LKMR habitat. However, guidelines defining target vegetative structures, based on quantitative assessments of LKMR habitat use, would ensure habitat mitigation suitable for LKMRs.

My objectives were to identify vegetation characteristics that correlate with rabbit occurrence on Boca Chica Key and to examine the hypothesis that hardwood encroachment is detrimental to LKMR habitat. Results from my study should provide guidelines for habitat restoration and management of transition zones throughout the LKMR range, including areas adjacent to runways of the NASKW.

Study Area

Boca Chica Key is an approximately 250 ha island that includes the western most terminus of the LKMR range. Boca Chica Key is comprised of 36 patches of LKMR habitat, 21.4% of which are occupied. I chose 4 patches of coastal transition zone habitat for this project (patch 9 area = 6.3 ha, patch 8 area = 4.3 ha, patch 160 area = 2.8 ha, and patch 14 area = 1.4 ha; patch ID #s from Faulhaber 2003; Fig. 3.1), each including a patchy distribution of marsh grasses, herbaceous vegetation, and overstory of buttonwoods and mangrove trees. A narrow (5–80 m) ecotone, frequently found between the coastal mangroves and upland forests of the Florida Keys, coastal transition zones occur between 1–3 m above sea level. Much of this habitat is subject to tidal inundation during annual spring peaks in high tides (Forys 1999*b*). The transition zone is floristically simple, dominated by relatively few species of grasses and forbs. These include cord grasses (*Spartina spp*.), sea daises (*Borrichia spp*.), glassoworts (*Salicornia spp*.), coastal dropseed (*Sporobolis virginicus*) and rushes (familiy *Cyperacea*). The dominant hardwood tree in the costal transition zones is the buttonwood (*Conocarpus erectus*), lower elevations include white mangrove (*Laguncularia racemosa*), red mangrove (*Rhizophora mangle*), and black mangrove (*Avicennia germinans*) trees. Occupied LKMR patches were selected because they were within the NASKW project area.

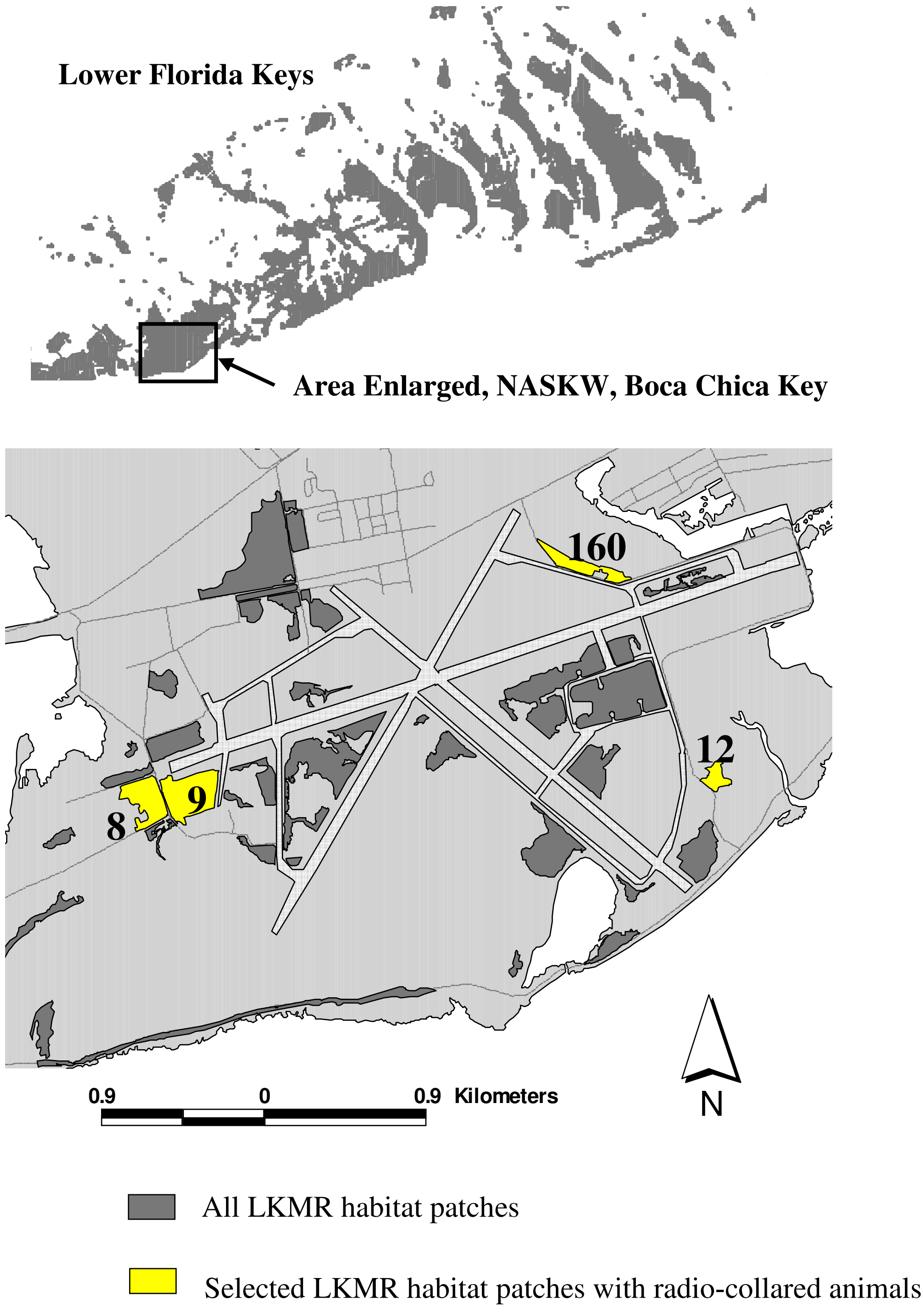

Figure 3.1. Study area for Lower Keys marsh rabbit (LKMR) habitat selection study, 2004–2005, Boca Chica Naval Air Station, Key West.

METHODS

I trapped LKMR using 2-door, 60 cm x 18 cm x 18 cm Havahart (Woodstream Corporation, Lititz, Pennsylvania, USA) trap from October 2004–January 2005. Adults ≥ 900 g were fitted with neoprene collars and battery-powered mortality-sensitive transmitters with an estimated life of 200 days (Advanced Telemetry Systems, Isanti, Minnesota, USA; 16-22 g/radio; < 2% of body weight). For each radio-collared rabbit, I recorded sex, capture location, and body mass. I relocated radio-marked rabbits via homing (White and Garrott 1986) approximately 3 times/week during randomly determined intervals (24-hour period was divided into 6 equal 4-hour segments; 1 [4-hour] segment was randomly selected, and during that time all rabbits were located). Telemetry locations were entered into a GIS using ArcView (Version 3.2; Environmental Systems Research Institute, Redlands, California, USA).

I evaluated LKMR microhabitat selection using radio telemetry. Using pooled telemetry locations at each site, I determined minimum convex polygons (MCP) and core use areas (50% kernel estimates) for LKMR's. I projected 100 random points within core use areas and 100 points in non-core areas (i.e., MCP less core use polygons) across all patches. I compared vegetative characteristics at each point (core use and non-core use points) by sampling (1) canopy cover, (2) vegetation height, (3) horizontal shrub cover, (4) ground cover, and (5) vegetative composition. I estimated canopy cover (%) with an ocular tube (James and Shugart 1970) from the center of each point. Vegetation height (dm; Robel et al. 1970) and horizontal shrub cover (Griffith and Youtie 1988) were recorded by averaging range pole measurements from each of the 4 cardinal directions. Ground cover (%) was estimated with a Daubenmire frame (Daubenmire 1959). I also recorded abundance of bunch grasses (number of tussocks), and dominant species of forbs, grasses, and trees within a 4-m radius of each point.

I evaluated the selection of LKMR microhabitat characteristics using binary logistic regression (0 = random point, 1 = core point). Predictor variables were based on vegetation characteristics of random locations outside the 50% kernel versus random locations from within the kernel. Highly correlated variables ($r^2 \geq 0.70$) were identified (Pearson-product moment correlation) prior to model building, in which case 1 of the pair of predictors was eliminated from analysis. Models were evaluated using an information-theoretic approach via Akaike's Information Criterion (AIC; Burnham and Anderson 2002). Additionally, the importance of model parameters was evaluated by summing AIC weights of each subset in which the parameter appears (Burnham and Anderson 2002). Logistic regression analyses were performed with Statistica 6 (StatSoft, Inc., Tulsa, Oklahoma, USA).

RESULTS

I captured and fitted 13 adult LKMRs with radio collars (8 males, 5 females) and used 385 locations with an average of 30 (SD = 17; Range = 8–63) locations per animal (Table 3.1). Core areas and non-core areas (i.e., MCP area less core-use areas) were as follows: patch # 9 (core area = 0.5 ha, non-core = 4.2 ha), patch # 8 (core area = 0.9 ha, non-core = 3.9 ha), patch # 160 (core area = 0.4 ha, non-core = 1.5 ha), and patch # 14 (core area = 0.2 ha, non-core = 0.7 ha). From these range estimates, I sampled a total of 200 locations; 100 random points within combined core areas and 100 points in combined non-core areas used in habitat model construction. After eliminating correlated variables, I constructed a global model that included the following variables: visual obstruction, canopy coverage, bunchgrass density, horizontal obstruction (≥1.5 m), bare ground (%), grass (%), forbs (%), litter (%), and site. Seven competing models (i.e., ≤ 2 AIC units apart) were generated from this global model (Table 3.2). Visual obstruction, canopy coverage, bunchgrass density, and site best explained LKMR presence (Table 3.3). Specifically, LKMRs were most likely to be present in areas with relatively higher visual obstruction values, less canopy coverage, and more bunchgrasses (Figure 3.3). Mean visual obstruction values for core- and non-core-use areas were 7.36 (SE = 0.253) and 2.94 (SE = 0.268), respectively. Mean canopy coverage percentage for core- and non-core-use areas were 5.90 (SE = 1.32) and 36.55 (SE = 3.59), respectively. Mean bunchgrass abundance (tussocks per 4-m radius plot) for core- and non-core-use areas were 15.99 (SE = 1.49) and 2.74 (SE = 0.47), respectively (Fig. 3.2). Also, spatial variation existed when predicting presence (i.e., the effect of model parameters differed by site).

Table 3.1. List of adult Lower Keys marsh rabbits trapped and fitted with radio transmitters from October 2004–January 2005.

ID	Sex	Mass (g)	Patch	Radio locations
2	Male	1,010	8	23
3	Male	1,020	8	63
4	Male	1,200	9	57
5	Female	1,250	9	13
6	Female	1,175	9	38
7	Female	1,220	9	8
8	Male	1,000	9	22
9	Female	1,150	9	10
10	Female	1,280	160	37
11	Male	1,480	160	36
12	Male	1,150	14	31
13	Male	1,280	160	25
14	Male	1,180	160	22

Table 3.2. Global model, competing subsets, and associated AIC diagnostics predicting LKMR occurrence on Boca Chica Key, Florida, 2004-2005.

Model	Variables[a]	K	-2log	AIC	ΔAIC	EXP	ωi	Avg.
1	vo, cc, bgd, bg, gr, fb, st	10	63.91	83.91	0.00	1.00	0.13	1.00
2	vo, cc, bgd, bg, fb, st	9	66.01	84.01	0.10	1.00	0.13	1.00
3	vo, cc, bgd, bg, lt, fb, st	10	64.82	84.82	0.91	0.96	0.13	1.05
4	vo, cc, bgd, gr, fb, st	9	67.02	85.02	1.11	0.95	0.13	1.06
5	vo, cc, bgd, ho (≥1.5 m), gr, fb, st	10	65.71	85.71	1.80	0.91	0.12	1.09
6	vo, cc, bgd, ho (≥1.5 m), fb, st	11	63.75	85.75	1.84	0.91	0.12	1.10
7	vo, cc, bgd, bg, gr, lt, st	11	63.89	85.89	1.99	0.91	0.12	1.10
global	vo, cc. bgd, ho, bg, gr, fb, lt, st	12	63.71	87.71	3.80	0.83	0.11	1.21

[a]vo = visual obstruction; cc = % canopy coverage; bgd = bunch grass density; ho = horizontal obstruction; bg = % bare ground; lt = littler; gr = %grass; fb = % forb; st = site

Table 3.3. Vegetative characteristics (mean ± SE) used in predicting LKMR occurrence. Variable importance was expressed as the sum of weights (ω') from each model subset in which that variable appears.

		Mean ± SE	
Variables	∑ω's	Presence	Absence
visual obstruction (dm)	0.89	7.36 ± 0.253	2.94 ± 0.268
% canopy coverage	0.89	5.90 ± 1.32	36.55 ± 3.59
bunchgrass density (per plot)	0.89	15.99 ± 1.49	2.74 ± 0.47
Site	0.89	N/A	N/A
% forbs (Daubenmire score)	0.77	2.70 ± 0.133	1.94 ± 0.09
% bare ground (Daubenmire score)	0.52	1.43 ± 0.05	1.84 ± 0.12
% grass (Daubenmire score)	0.51	2.54 ± 0.12	1.92 ± 0.10
% litter (Daubenmire score)	0.25	2.85 ± 0.80	3.50 ± .014
Horizontal obstruction (≥1.5 m)	0.24	2.34 ± 0.15	2.62 ± 0.17

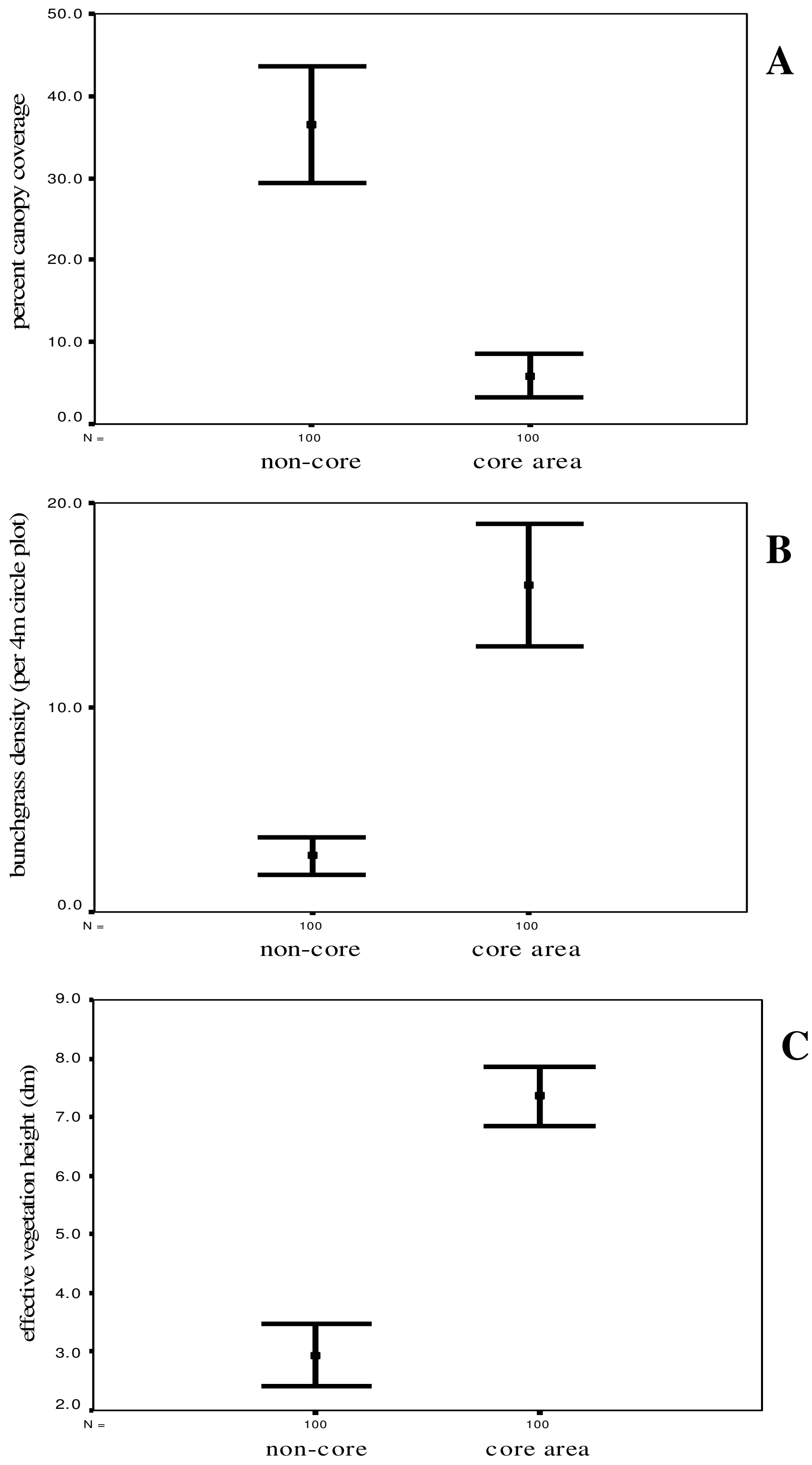

Figure 3.2. Mean values of 3 important core-use area selection variables: (percent canopy coverage [A], bunchgrass density [B], and effective vegetative height [C]) for Lower Keys marsh rabbits on Boca Chica Key, Naval Air Station Key West, Florida, winter 2004–2005. Whiskers indicate a 95% confidence interval.

DISCUSSION

The LKMRs used habitats with dense structure of low (≤1m) forbs and grasses with little to no overstory. The LKMRs avoided areas with mature buttonwoods, high canopy cover and ground level biomass. However, my study included no measures of fitness and/or demographic variables and the number of radio-collared animals (n = 13) was small. Further, the majority of LKMR locations were recorded during the winter dry season; continued research may elucidate a seasonal response to habitat use by LKMR. However, Forys (1995) detected no significant shift in daytime habitat between season. Nevertheless, model fit was good and results suggest that vegetative structure and composition are important predictors of LKMR presence.

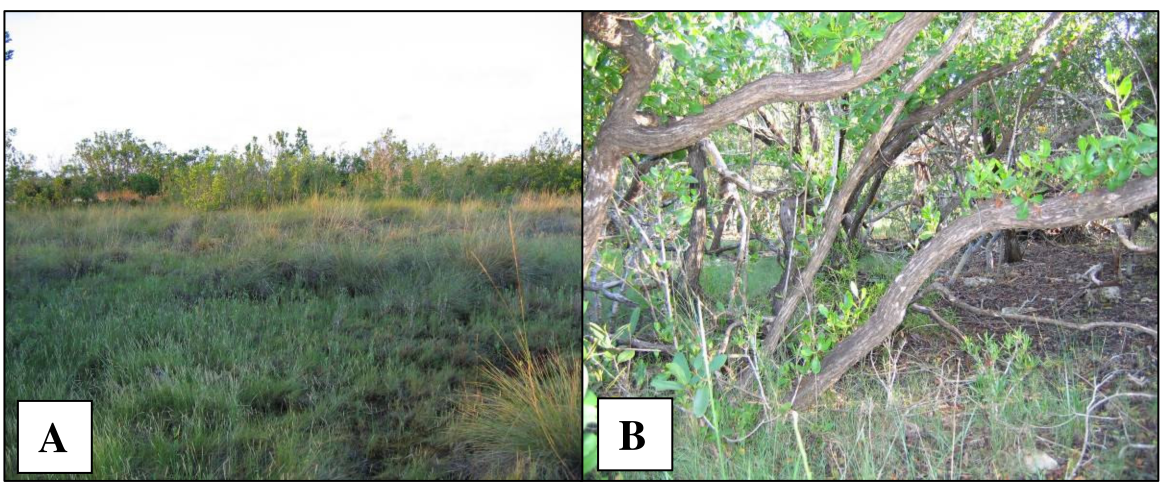

Figure 3.3. Photo taken from a Lower Keys marsh rabbit core-use area (50% kernel), Boca Chica Key, Naval Air Station Keys West, Florida, 2005; right photo taken from the same habitat patch in a marsh rabbit non-use area (i.e., minimum convex polygon); an example of the old growth buttonwood form.

My results supported the hypothesis that hardwood encroachment (i.e., higher percentages of canopy cover) is detrimental to LKMR habitat suitability. Specifically, study results suggest that LKMRs avoid areas with a canopy coverage exceeding 30%, though causative factors likely include a consequential reduction of forage and low ground cover (Litvaitis and Villafuerte 1996). Similar mechanisms are having deleterious impacts on New England cottontail (*S. transitionalis*) populations; reduced timer harvest and a gross succession of old-field habitats to hardwood forests has greatly restricted the range of this once widespread lagomorph. Accounts of historic habitat structure in the Florida Keys are not detailed, nor are there data to elucidate historic regimes of important ecosystem drivers (i.e., fire or hurricanes). Nonetheless, human land use (or non-use) practices have changed the Florida Keys landscape in recent decades, likely contributing to an evident increase in hardwood encroachment (Bergh and Wisby 1996). To address this, managers should consider a suitable management regime, possibly including habitat restoration and use of prescribed fire as a maintenance tool.

MANAGEMENT IMPLICATIONS

My study explored gross vegetative groups (i.e., forbs and bunchgrasses), finding that high incidence of both strongly predicted LKMR presence. Past research found no correlation with any specific floristic component of this ecosystem and LKMR presence (Forys 1995). I recommend habitat mitigation should restore a matrix of coastal-prairie floristic components, maximizing the herbaceous plant varieties and avoiding large (i.e.,

$\geq$0.25 ha) monotypic patches. Given the small diversity of common plants associated with transition zones, this is a logistically realistic guideline.

Though ecosystem managers ideally target a natural habitat regime, the Florida Keys may be too fragmented and altered for such a goal to be realistic. Rather, managers should focus on conserving the remaining diversity of ecosystem components. Arguably, LKMRs play a vital role in the dynamics of this ecosystem, providing an important food source for many native predators (e.g., eastern diamondback rattlesnakes [*Crotalus adamanteus*], American alligators [*Alligator mississippiensis*], raptors). Restoration and management of the coastal transition zone ecosystem, essential LKMR habitat, may be a necessary measure to restore a viable LKMR population.

CHAPTER IV

DISTRIBUTION AND POPULATION TRENDS OF SILVER RICE RATS IN THE LOWER FLORIDA KEYS

INTRODUCTION

In 1991, the United States Fish and Wildlife Service (USFWS) listed the silver rice rat (SRR, *Oryzomys palustris natator*) as an endangered vertebrate population (USFWS 1991) and critical habitat was designated in 1993 (USFWS). Endemic to the Florida Keys, the SRR was considered one of the rarest rodents in the world (Figure 4.1; Mitchell 1996). The SRR are found almost exclusively in saltmarsh habitats, though the first 2 specimens were caught in a freshwater marsh on Cudjoe Key (Spitzer and Lazell 1978); no animals have been found in freshwater since. Semi-aquatic and capable swimmers, SRRs forage in the intertidal zones, feeding on fish, crabs, grasses and forbs (Esher et al. 1978, Loxterman 1998, Forys 1996). Their ranges are exceedingly large for an animal of their size, females ranging from 2.0–8.5 ha and males from 3.4–11.0 ha (Mitchell 1996).

The last comprehensive distribution survey for SRR was completed in 1996 (Forys et al. 1996, Mitchell 1996), SRRs were recorded on 11 islands in the Lower Keys: Howe, Water, Middle Torch, Big Torch, Summerland, Raccoon, Johnston, Cudjoe, Upper Sugarloaf, Lower Sugarloaf, and Saddlebunch keys. Previous surveys in the 1980s also recorded trapping a SRR on Little Pine Key (Goodyear 1987). However, given their ability to travel long distances and proficiency at swimming, it is possible

SRRs have expanded their range. Conversely, SRRs also might have declined since these historic surveys.

The density of SSRs and mainland subspecies of rice rats is often compared and results suggest that SRRs tend to exist in much lower densities (Goodyear 1984, Mitchell 1995, Forys et. al. 1996. USFWS 1999). Data used to derive the comparisons, however, are limited as is the duration of many of these studies. Further, all comparisons use minimum number alive (MNA) and cite appropriate sources (e.g., Krebs 1966) but estimates of edge effect (Krebs 1999) varies and are not always explicit. In fact, researchers that report the highest density of rice rats used to compare with SRR densities (18.1/ha, Smith and Vrieze 1979) openly admit that their estimates do not include estimates of edge effect and "may be overestimates and not directly comparable to other population studies on the species." This referenced study was used to federally list SRRs as endangered and is cited in almost all reports and publications that claim SRRs exist in lower densities (Goodyear 1984, Forys et. al. 1996, USFWS 1999). None of these mention this comparative short-coming.

Thus, the objective of my study was to (1) conduct a distribution survey for SRR, and (2) to examine population trends over the last 25 years, and (2) compare SRR abundance estimates of the past 10 years and trap success rates with 4 published mainland rice rat population studies.

Study Area

The Lower Florida Keys form the end of a string of limestone islands extending >60 km in a southwesterly direction from Little Duck Key (24°41'N, 81°14'W) near the southern tip of peninsular Florida to Key West (24°33'N, 81° 49'W). The climate is tropical with a wet (May–November) and a dry (December–April) season. The maximum recorded range for SRR extended between Little Pine Key and the Saddlebunch Keys, including most islands with suitable habitat.

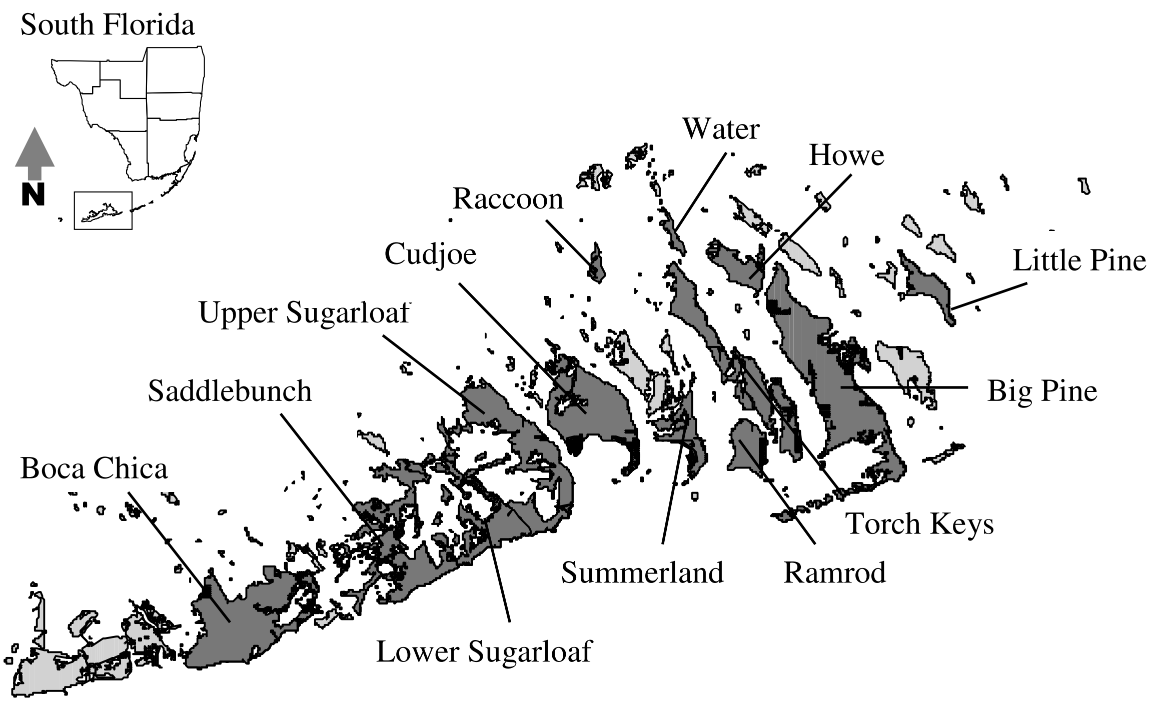

Figure 4.1. Islands surveyed for silver rice rats in the Lower Florida Keys, 2004-2005.

Vegetation Types

Elevation rarely exceeds 2 m in the Lower Florida Keys, but small variations in elevation yield distinct vegetative communities transitioning with increasing elevation from mangrove swamps, to salt-marsh/buttonwood transition zones (brackish wetlands), to tropical hardwood hammocks and/or pinelands (MacGarry MacAulay et al. 1994). Silver rice rats have been predominantly found in mangrove, dwarf mangrove/ salt-marsh habitats, and high salt marsh/buttonwood transition zones. These habitats are influenced by gradients of moderate to high tidal inundation, which defines the floristic and structural vegetative components of the Florida Keys coastal mangrove and wet prairie ecosystems. On 2 occasions, SRR have been captured in freshwater marshes (Spitzer and Lazell 1978). Freshwater marshes, however, are ephemerally wet and uncommon or absent, from many of the islands SRR are found on. Logistically limited, I did not focus any effort on freshwater marshes.

METHODS

Current Distribution

I surveyed 36 sites across the historic range of SRRs including the following 18 keys: Boca Chica, Geiger, East Rockland, Saddlebunch, Big Torch, Cudjoe, Little Pine, Little Torch, Raccoon, Ramrod, Water Keys, Saddlebunch, Upper Sugarloaf, Lower Sugarloaf, Summerland, Big Pine, Howe, and Middle Torch keys. I trapped SRR using vented 7.5 x 8.8 x 22.5 cm Sherman live traps (Sherman Traps, Inc., Tallahassee, Florida, USA). Traps were set in transects and grids with spacing of 22 m, as described by Forys et al. (1996). All transect and grid locations were recorded with a handheld global positioning system (GPS) unit and entered into a GIS database using ArcView GIS (Version 3.3; Environmental Systems Research Institute, Redlands, California, USA). There were typically 4 night trapping sessions at each transect or grid. Standard measures were recorded for each animal including lengths of ear, right hind foot, tail, head, and body. Weight, gender, and reproductive condition of each animal was recorded. Each animal was given a uniquely numbered monel, self-piercing ear tag.

Trap-night-effort (TNE) was calculated by subtracting the number of misfired traps and the number of traps with non-target animals (i.e., black rats [*Rattus rattus*]) from the total number of traps set each night.

Population Trends

Recent Trends.—I repeated historic trapping efforts on four grids in 2004, one was repeated again in 2006. Previous surveys on these grids were conducted periodically between 1997 and 2000. I compared general results from these grids with my results from replicating these grids, using the same traps and techniques (Phil Frank, USFWS, National Key Deer Refuge, unpublished data).

Historic Trap Success.—Trap success data (total SRR capture events/ TNE) were collected from published and unpublished accounts (Goodyear 1984; Goodyear 1987; Forys et. al. 1996; Phil Frank, unpublished data). Success was compared temporally and by island.

Population Comparisons

To examine historic density estimate comparisons which conclude that SRRs exist in lower densities than mainland sub-species' (*O. p. coloratus* and *palustris*), I review publications and reports of SRR and mainland rice rat populations. I offer comparisons of trap success and abundance estimates using a naive MNA when available (Krebs 1999). I compare SRR data collected over the past decade (since 1995) including only grids on which SRRs were trapped. Only my data from grids (excluding transects) in which SRRs were trapped are used in these comparisons.

RESULTS

Current Distribution

Total trap-night-effort in 2004 and 2005 was 5,110 (after subtracting misfires and non-target captures) on 36 grids and transects (Table 4.1). I trapped 120 individual SRRs (83 males, 38 females) with 216 SRR capture events. Black rats were caught 10 times. Overall trapping success was 4.2% for SRRs and 0.2% for black rats. I captured SRRs on 12 islands: Big Pine, Big Torch, Cudjoe, Howe, Lower Sugarloaf, Middle Torch, Raccoon, Ramrod, Saddlebunch, Upper Sugarloaf, and Water keys. The first record of SRRs on 2 of these islands, I captured 1 sub-adult female and 1 adult male on Big Pine and Ramrod keys, respectively. No SRR were trapped on Boca Chica, Geiger, East Rockland, Little Pine, or Big Coppit keys (Fig. 4.2).

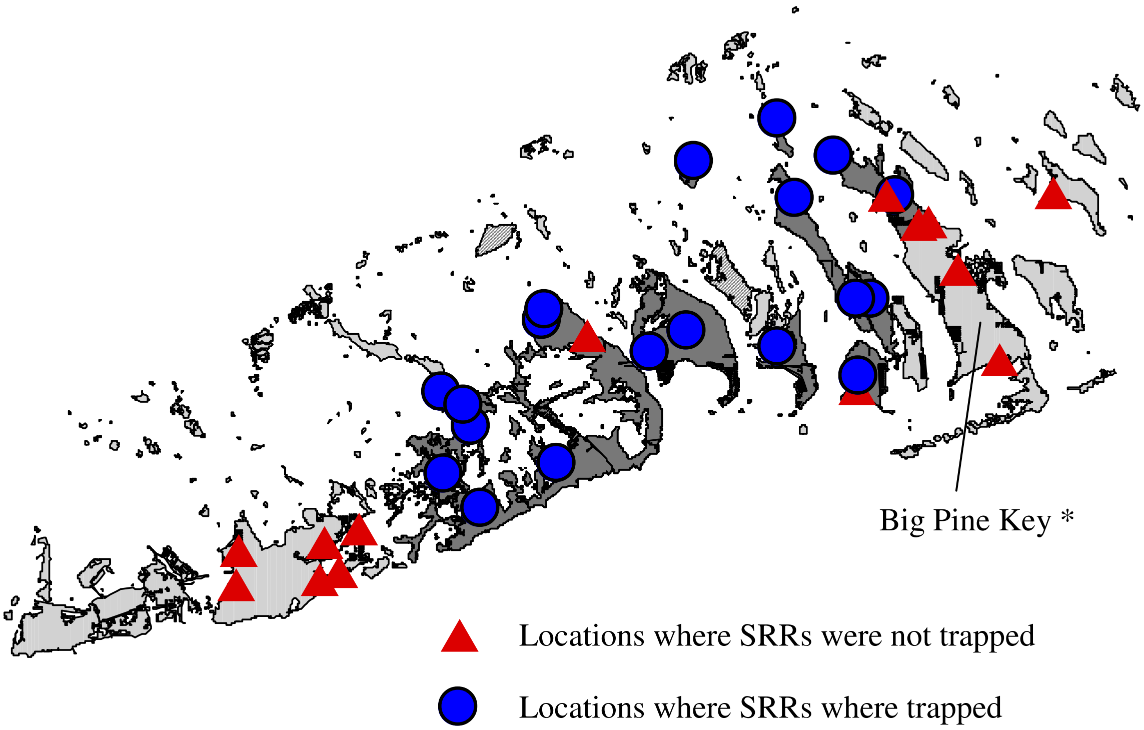

Figure 4.2. Current known range of silver rice rats in 2004-2005. Dark shaded Islands indicate SRR presence during the 2004-2005 surveys. Islands in hatch, Johnston and Cutoe keys, were not surveyed but likely harbor silver rice rats.

* **Note:** Only the northern portion of Big Pine Key is shaded. One silver rice rat was trapped at the northern most end of the island; extensive surveys on the rest of the island yielded no silver rice rats.

Table 4.1. Summary of silver rice rat (SRR) trapping efforts and results by island in the Lower Florida Keys, 2004-2006. RR = *Rattus rattus* (black rats).

Island	Grids/ transects	TNE[a]	Total SRR captures	Total SRRs	Captures/ TNE	Total RR
Big Coppit	1	233	0	0	0.000	0
Big Pine	7	496	1	1	0.002	0
Big Torch	1	453	62	32	0.137	0
Boca Chica	3	572	0	0	0.000	0
Cudjoe	3	488	24	13	0.049	1
East Rockland	1	221	0	0	0.000	0
Geiger	2	286	0	0	0.000	4
Howe	1	169	17	7	0.101	0
Little Pine	1	240	0	0	0.000	1
Lower Sugarloaf	1	76	5	2	0.066	0
Middle Torch	2	194	39	17	0.201	0
Raccoon	1	109	6	4	0.055	0
Ramrod	2	149	1	1	0.007	0
Saddlebunch	5	854	86	53	0.101	5
Summerland	1	220	3	2	0.014	0
Upper Sugarloaf	3	333	4	3	0.012	0
Water	1	238	4	2	0.017	1
Total	36	5,110	216	120	0.042	12

[a] TNE = Trap-night-effort; TNE is calculated by subtracting traps that have been sprung, but are empty and traps with non-target animals from the total number of traps set in a given night.

Population Trends

Comparisons of range-wide SRR tapping success revealed low success in the 1980s and increased, sustained trap success throughout much of the 1990s and 2004–2006 (Fig. 4.3). Comparisons with repeated SRR grid surveys from 1997–2006 revealed no apparent decline or increase in the ratio of success to effort over the past decade (Table 4.2). A trapping session in 1997 on Saddlebunch Keys yielded extremely high trapping success (SRR captures/ TNE = 35%, total individual SRRs = 106).

Population Comparisons

Three studies are often used as benchmarks to compare SRR population dynamics with those of mainland populations: Smith and Vrieze 1979, Wolf 1985, and Forys and Dueser 1993. Other studies on mainland rice rat population dynamics include: Negus et. al. 1961 and Kruchek 2004. In 1996, Forys reported the highest abundance of SRRs, at 2.3/ha, to that date. However, since then trapping has yielded 21.9/ha (1997), to 4.2/ha (1999-2000), to 7.0/ha (2004–2006; Table 4.3). Average SRR MNA abundance for the past decade (including Forys 1996 to date) is 8.7/ha (SE = 4.5, n = 4). Average mainland rice rat MNA abundances, including 4 population studies, is 11.1/ha (SE = 2.5, n = 4). Trap success of silver rice rats over the past decade has been generally higher than reported for mainland sub-species (Table 4.3.). Average SRR trap success since 1996 was 7.8 % (SE = 2.9, n = 4). Average mainland rice rat trap success since 1961 was 5.6 (SE = 1.2, n = 3). Small and variable sample sizes prohibit reasonable statistical comparison—nonetheless, it is difficult to conclude that SRRs tend to exist in lower densities than mainland populations, at least for the past decade.

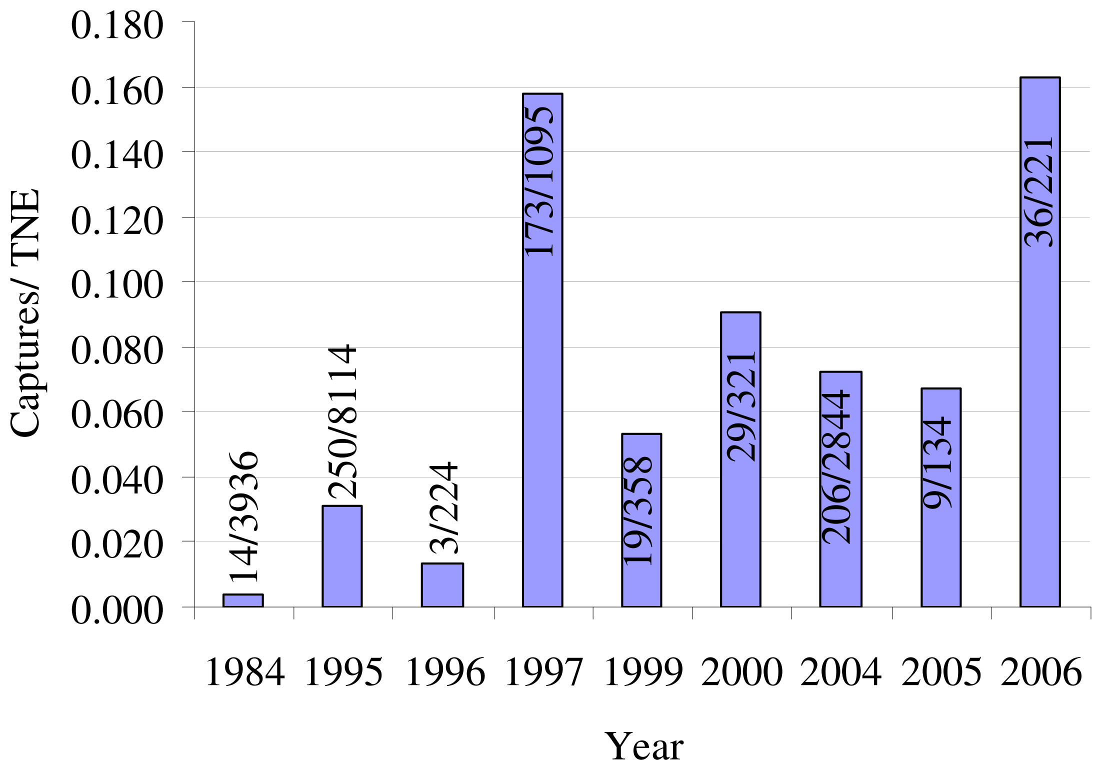

Figure 4.3. Historic and recent trap success for silver rice rats (SRR) in the Florida Keys. Includes all grids on Big Torch, Cudjoe, Johnston, Middle Torch, Raccoon, Saddlebunch, Sugarloaf, Summerland, and Water keys (i.e., all islands on which SRRs have been documented more than once). Labels on bars indicate SRR captures/Total trap night effort on included grids for that year. Note: trapping results from 2006 are not published and were not included elsewhere in this chapter.

Table 4.2. A comparison of repeated silver rice rat (SRR) grid trapping efforts (TNE) at 4 locations on 3 islands in the years 1997, 1999, 2000, 2004, and 2006, Lower Florida Keys. Includes total number of black rats (BR) caught at each grid.

Year	Island	Grid	Grid design	Total SRR	SRR captures	Total BR	Total TNE[a]	Captures/ TNE
2006	Big Torch	BT	3x20	16	36	0	221	0.163
2004	Big Torch	BT	3x20	16	26	0	232	0.112
1999	Big Torch[a]	BT	3x20	7	9	0	227	0.040
1997	Big Torch[a]	BT	3x20	13	14	1	229	0.061
2004	Cudjoe	CJ1	3x20	5	15	0	219	0.068
2000	Cudjoe[a]	CJ1	3x20	4	6	1	107	0.056
2004	Cudjoe	CJ2	3x20	3	4	1	233	0.017
1997	Cudjoe[a]	CJ2	3x20	2	2	0	227	0.009
2004	Saddlebunch[a]	SB	3x20	11	16	2	211	0.076
2000	Saddlebunch[a]	SB	3x20	12	23	7	214	0.107
1997	Saddlebunch[a]	SB	6x10	106	129	24	364	0.350

[a] indicates unpublished data provided by Phil Frank, United States Fish and Wildlife Service, National Key Deer National Wildlife Refuge, Big Pine Key, Florida, USA.

Table 4.3. Naïve density estimates and trap success rates for SRR (between 1996 and 2006) compared with results from mainland rice rat population studies (between 1961 and 1997).

Reference	Year	Number of grids	Region	Naïve density (SRR/ha)	Trap night effort	Trap success %	Trap spacing (m)
Negus et. al. (1961)	1957-1960	19	Mississippi coast	NA	8605	4.4	15
Smith and Vrieze (1979)	1975-1976	14	Florida Everglades	18.07	NA	NA	15
Wolf (1985)	1979-1982	32	Mississippi coast	6.82[a]	NA	4.7	15
Forys and Dueser (1993)	1989	10	Virginia coast	10.61 [a]	NA	7.9	15
Forys et. al. (1996)	1995-1996	24	Florida Keys	2.29 [a]	9960	3.0	15-22
Kruchek (2004)	1996-1997	16	Texas coast	10.50	NA	NA	15
Frank (Unpublished)	1997	3	Florida Keys	21.92	1095	15.8	22
Frank (Unpublished)	1999-2000	3	Florida Keys	4.16	601	3.8	22
Perry (this thesis)	2004-2006	11	Florida Keys	6.41	2440	8.7	22

[a]Methodologies unclear, may have utilized a boundary strip (Krebs 1999) based on home range estimates of animals trapped; MNAs taken from Forys et. al. (1996).

DISCUSSION

Distribution

My results suggest that the range of the SRR has not decreased in the past 10 years. The only island previously considered occupied by SRR on which I failed to detect SRRs in 2004–2005 was Little Pine Key. However, only 1 SRR has ever been collected on Little Pine Key (Goodyear 1984), in the mid 1980s. Based on current definitions of suitable SRR habitat it seems Little Pine Key should be capable of supporting a population of rice rats, however, our trapping effort on Little Pine Key (TNE = 240) should have been sufficient to detect SRR presence. Given SRRs ability to disperse over water, the historic Little Pine Key record may have been a result an isolated colonization event (Brown and Kodric-Brown 1977, Esher et al. 1978). Nevertheless, I should not rule out alternative possibilities including (1) that I failed to detect a persistent SRR sub-population or (2) that a Little Pine SRR sub-population was eradicated by unknown factors. Potential depressive factors for SRRs on Little Pine Key may include: high pressure from upland black rats (Goodyear 1992), isolation from other SRR populations, and impacts of a feral hog population that was removed in the mid 1990s (Tom Wilmers, USFWS Biologist, National Key Deer Widllife Refuge, personal communication).

Though there is substantial suitable habitat on Boca Chica, Geiger, East Rockland or Big Coppit keys, and the distance between occupied habitats and these islands is small, we failed to detect SRRs. Last surveyed in 1985, SRRs have never been detected on these islands (Wolf 1987). Two possible reasons for this are: (1) The channels separating these islands and occupied islands sustain heavy boat traffic and strong tidal currents (personal observation), and/or (2) suitable habitat on Geiger and Big Coppit—habitats that would likely be facilitative corridors for SRR colonization—are small, fragmented by roads, and near development. Any or all of these factors may hinder dispersal. Trapping efforts on these keys (Trap nights = 1,442) should have been more than sufficient to detect the presence of SRRs. Further, success on other keys validates the effectiveness of the trapping techniques used (Table 4.1).

My surveys detected SRRs on the northern end of Big Pine Key and on Ramrod Key; 2 islands on which, despite significant efforts in the past (Goodyear 1987; Mitchell 1996; Forys et al. 1996), SRRs have not been previously recorded. Given their relatively close proximity to occupied islands, these could have been dispersal events (Loxterman 1998). In fact, the animal on Big Pine Key was a subadult, the age class at which most mammalian species tend to disperse. Follow up surveys on Big Pine Key failed to detect SRRs. Indeed these and the record from Little Pine Key could all be evidence of SRRs propensity to disperse over large distances of open water. However, it also is possible that populations on these islands persist at low numbers or are functional sinks in an island metapopulation. Future surveys may elucidate the nature of these records.

Population Trends

Comparisons with data from 1997-2000 reveal few significant changes in SRR abundances, based on indices of SRR trap events/ TNE (Table 4.2). These data should be considered with caution because these surveys were conducted at different times of the year. Further, inherent fluctuations associated with rodent populations renders the detection of trends difficult (Elkington 2000). However, the evidence presented herein does not reveal a decline in the SRR range or abundance. Based on my results, the SRR population has remained stable throughout its known range for the past 10 years. Reasons for this increase and sustainability in trapping success are challenging to evaluate. Trapping efforts and techniques employed in early surveys may have been inappropriate (documentation is not explicit), or the population may have been in the midst of a population bottleneck (Wang et. al. 2005). Standardized surveys, repeated annually would provide more definitive SRR trend data.

Population Comparisons

My data, and those collected by USFWS biologists over the past decade suggest that the Florida Keys population of SRRs may not exhibit significantly different population dynamics from mainland populations, prompting the question: how do SRR differ from mainland populations? Taxonomically, SRRs have been the focus of much debate (Spitzer and Lazel 1978; Barbour and Humphrey 1982; Goodyear and Lazell 1986; Humphrey and Setzer 1989; Goodyear 1991; Humphrey 1992); authors disagree if the SRR should be deemed a species or a subspecies. However, there has been little debate regarding population dynamics of the SRR. General conclusions regarding SRRs compared to mainland populations of rice rats include: (1) that SRRs tend to naturally exist in lower densities, (2) have slower reproductive rates, (3) have relatively higher survivorship, and (4) are in a state of overall decline (Goodyear 1987, Forys et al. 1996, USFWS 1999, Wang et. al. 2005). However, these conclusions regarding SRR population dynamics are based on a paucity of data, collected during sporadic, unsystematic, and often unrepeatable efforts. Conclusions from these anecdotal accounts should be interpreted with appropriate caution; they should support the development of hypothetical questions that can be tested appropriately. My results suggest that at least 2 of these assumptions of SRR population dynamics are false—SRR populations (and range) appear stable and SRR densities are likely not significantly different from those of mainland populations.

MANAGEMENT IMPLICATIONS

The distribution of SRRs has not changed dramatically over the past 20 years and populations have not exhibited any indication of decline in the past 10 years. Much of the remaining salt-marsh and mangrove habitats that these animals depend on have been protected via acquisition from conservation-oriented land management organizations and general wetland protective legislation. As such, the USFWS might consider evaluating the endangered status of the sub-species. Re-designation as a threatened population may be more appropriate for the SRR population, still recognizing the population as a distinct evolutionary unit with a limited endemic range, and better representing the actual status of the population.

CHAPTER V

CONCLUSIONS

LOWER KEYS MARSH RABBIT

Based on the results presented in Chapter II of this thesis and the work of Faulhaber 2006, it is evident that reintroduction of Lower Keys marsh rabbit (LKMR, *Sylvilagus palustris hefneri*; Lazell 1984) into suitable habitat, using animals captured in the wild, is a viable management tool. Translocation can be used to reintroduce LKMRs into restored potential habitat or to manage for genetic diversity in isolated sub-populations. Managers should consider follow-up translocations to augment the size and promote genetic health of founder populations, especially in the first few years after reintroduction (Ramey et al. 2000).

Future application of LKMR translocations depends on the perseverance of existing source populations. Managers should focus resources on habitat enhancement in source populations, ensuring that they can maintain or, preferably, increase densities of rabbits in currently occupied patches; increasing the number of available candidates for translocation. The LKMR reintroduction program must be integrated into a comprehensive management strategy involving land acquisition to secure suitable release sites, control of exotic predators (especially free-roaming cats), and habitat restoration and enhancement of both occupied and potential habitat.

Though ecosystem managers ideally target a natural habitat regime, the Florida Keys may be too fragmented and altered for such a goal to be realistic. Rather, managers should focus on conserving the remaining diversity of ecosystem components.

Arguably, LKMRs play a vital role in the dynamics of this ecosystem, providing an important food source for many native predators (e.g., eastern diamondback rattlesnakes [*Crotalus adamanteus*], American alligator [*Alligator mississippiensis*], and raptors). Restoration and management of the coastal transition zone ecosystem, essential LKMR habitat, may be a necessary measure to restore a viable LKMR population. My research (Chapter III) supports the hypothesis that hardwood encroachment by buttonwood trees may be detrimental to LKMRs. Restoration of these habitats may require clearing of buttonwood trees, however, a prescribed fire regime may adequately maintain suitable LKMR habitat.

SILVER RICE RAT

The distribution of silver rice rat (SRR, *Oryzomys argentatus*; Spitzer and Lazell 1978) has not changed dramatically over the past 20 years and populations have not exhibited any decline in the past 10 years. Much of the remaining salt-marsh and mangrove habitats in which these animals are found have been protected via acquisition from conservation oriented land management organizations and general wetland protective legislation. As such, the United States Fish and Wildlife Service might consider evaluating the endangered status of the sub-species. Re-designation as a threatened population may be more appropriate for the SRR population, still recognizing the population as a distinct evolutionary unit with a limited endemic range, and better representing the actual status of the population.

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