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Effectiveness of supplemental stockings for the endangered Key Largo woodrat

Robert A. McCleery *, Roel R. Lopez, Nova J. Silvy, William E. Grant

Department of Wildlife and Fisheries Sciences, Texas A&M University, Mail Stop 2258, TAMU, Nagle Hall, College Station, TX 77843, USA

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Abstract

We evaluated the effectiveness of supplemental stockings being proposed in the recovery of the federally endangered Key Largo woodrat (KLWR, *Neotoma floridana smalli*) using a stage-based, stochastic model. Supplemental stockings were evaluated with a population model using current trapping and telemetry data along with published and unpublished KLWR data. Model simulations predicted the KLWR had >70% probability of terminal extinction over the next 10 years even with the use of supplemental annual stockings. Supplemental stockings of KLWRs (⩽20 females) appear to delay the extinction of the species, but negative population trajectories accelerated after stockings cease. Model results illustrated the importance of determining limiting factors on the population prior to the use of supplemental stockings as a recovery option for this endangered woodrat population.

Keywords: Endangered species; Key Largo woodrat; Population viability analyses; Relocations; Supplementation

1. Introduction

Relocating species to increase population numbers and help prevent extinction has become an integral part of endangered species management (Griffith et al., 1989; Price, 1991; Fischer and Lindenmayer, 2000). Animal relocations for conservation purposes usually take the form of translocations (movement of animals from one part of their range to another), reintroductions (establishment of a species in part of its former range) or supplementations (adding animals to an existing population, Fischer and Lindenmayer, 2000). Relocations of endangered species can be exceedingly expensive (Kleiman et al., 1991; Griffith et al., 1989; Fischer and Lindenmayer, 2000). Additionally, translocations, reintroductions and supplementations of endangered species are often unsuccessful (Griffith et al., 1989; Black et al., 1997; Van Zant and Wooten, 2002), especially when the source population is captive-bred (Black et al., 1997; Fischer and Lindenmayer, 2000). It has been argued that population viability analyses (PVA) or other modeling techniques can be used as a planning tool in endangered species recovery (Griffith et al., 1989; Durant and Mace, 1994; Kleiman et al., 1991; Seal, 1991) but few studies illustrate how simulation models can be used in the planning stages endangered species relocations (Southgate and Possingham, 1995). Compared to other alternatives for making conservation decisions, PVAs provide a rigorous methodology that can incorporate different types of data, uncertainties and natural variation, and provide outputs or predictions that are relevant to conservation goals (Akcakaya and Sjogren-Gulve, 2000). PVA results also can incorporate uncertainties using sensitivity analyses based on ranges of parameters, which gives a range of extinction risk estimates and other assessment endpoints (Akcakaya, 2000). The use of commercially-available simulation packages (e.g., RAMAS Metapop, Vortex) are an inexpensive way for agencies to evaluate

* Corresponding author. Tel.: +1 979 845 8263; fax: +1 979 845 3786.
E-mail address: iamnotfunny@yahoo.com (R.A. McCleery).

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relocations before committing to them. Here we describe the use of a population model used to evaluate supplemental stocking a priori in the recovery of the federally endangered Key Largo woodrat (*Neotoma floridana smalli*, KLWR).

1.1. Background

The endangered KLWR is a federally-listed sub-species endemic to Key Largo, Florida. Since 1973, the KLWR has been confined to approximately 850 ha of tropical hardwood hammock forest on the northern third of the island (US Department of the Interior [USDI] 1973; Barbour and Humphrey, 1982; US Fish and Wildlife Service, USFWS, 1999). Most of these 850 ha are within the bounds of 2 protected areas: the Dagny Johnson Key Largo Hammock Botanical State Park and the Crocodile Lake National Wildlife Refuge (Frank et al., 1997). Still, even within these protected areas the KLWR has suffered at least two decades of decline (McCleery, 2003). Population trend data suggest a precipitous decline in the population with current estimates at <100 individuals (McCleery, 2003). Feral cats (*Felis catus*, Humphrey, 1992; Frank et al., 1997; USFWS, 1999), fire ants (*Solenopsus* spp., Frank et al., 1997), habitat fragmentation (Frank et al., 1997; USFWS, 1999), competition with black rats (*Rattus rattus*, Hersh, 1981; Humphrey, 1992; Frank et al., 1997; USFWS, 1999), and a combination of the above (Frank et al., 1997) have been suggested, but there is little or no data to support these hypotheses. In 2003, the US Fish and Wildlife Service (USFWS) decided to take management action and begin a captive breeding program with a goal of supplementing the wild KLWR population (Dean, 2003). In response, we developed a stage-based, stochastic population model representing the dynamics of the KLWR to: (1) evaluate effectiveness of KLWR supplemental releases a priori; (2) estimate the KLWR's risk of extinction; (3) conduct a sensitivity analysis to identify model parameters that account for the greatest uncertainty in the model and to aid in the planning of future field research.

2. Methods

2.1. Study area

Key Largo is the first and largest in a chain of islands (keys) that extends from the southern tip of Florida. Our study area on Key Largo was limited to KLWR habitat (845 ha) found along an 11-km stretch of protected hardwood hammock forest on the northern third of the island (Fig. 1). Hardwood hammock habitat on the island of Key Largo is unique with a high abundance of West Indian plants and trees (Strong and Bancroft, 1994; USFWS, 1999). Some common trees found in Key Largo's hammocks are gumbo-limbo (*Buresa simaruba*), poisonwood (*Metopium toxiferum*), wild tamarind (*Lysiloma bahamensis*), pigeon plum (*Cocoloba diversifo-*

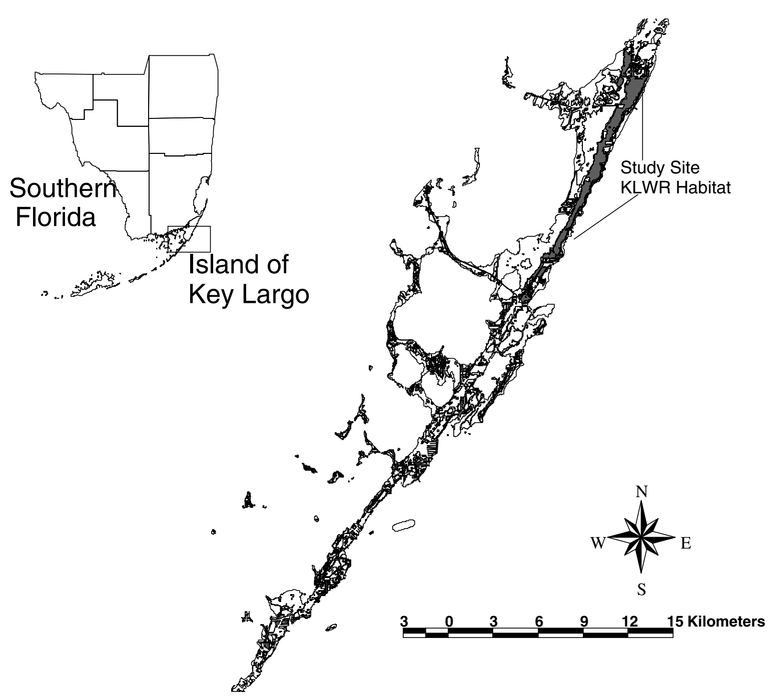

Fig. 1. The island of Key Largo, Florida, and Key Largo woodrat study site comprised of the known Key Largo woodrat range.

lia), willow bustic (*Bumelia salicifolia*), and Jamaican dogwood (*Piscidia fostidissimum*).

2.2. Model overview

We constructed a PVA using a stage-based (juveniles, <6 months; adults, >6 months, Hersh, 1981), stochastic population model that represented the female KLWR population using RAMAS Metapop software (Applied Biomathematics, Version 4, Akcakaya, 1998). We chose RAMAS Metapop because of its availability and accuracy (Brook et al., 2000). The stage matrix of the model was,

$$\begin{matrix} Fj & Fa \\ Sj & Sa \end{matrix}$$

where *Fj* and *Fa* were juvenile and adult fecundity (females born per female per year), and *Sj* and *Sa* were juvenile and adult survival (portion of the stage class surviving annually), respectively. We incorporated both demographic stochasticity (natural changes in births, deaths, and sex ratios) and environmental stochasticity (changes in the environment over time, such as rainfall, food availability, fires, etc.) into the model (Lacy, 2000). We incorporated demographic stochasticity in model simulations by sampling the number of survivors from a binomial distribution (to account for the small population size, Akcakaya, 1991) and the number of offspring from a Poisson distribution (because KLWR's have a fecundity >1, Akcakaya, 1991). We modeled environmental stochasticity by randomly sampling fecundity and survival from lognormal distributions with means taken from a mean-stage matrix and standard deviations taken from a "standard deviation matrix" (Akcakaya, 1991). Additionally, we chose to model the KLWR population using the exponential growth option in RAMAS Metapop because density-independence provides a conservative assessment with small populations (Ginzburg et al., 1990).

2.3. Parameter estimation

We estimated model parameters from trapping and telemetry data (McCleery, 2003) along with previously published and unpublished KLWR data. Where data were sparse we used published data on other *N. floridana* spp. All demographic estimates except for initial abundance were varied 30% for high and low parameter estimates. High parameter estimates were considered the "best case" scenario for biologically plausible demographic rates.

Annual adult survival estimates were determined using a Mayfield estimator (Krebs, 1999) from radiotelemetry data collected in March–December 2002 and validated with trapping data (Frank et al., 1997; Sasso, 1999) using a Jolly–Seber estimator (Krebs, 1999). We calculated juvenile survival rates using trapping data (Frank et al., 1997; USFWS, Unpublished data 2000). During simulations, juvenile survival rates were applied for the first 6 months of life. Variance (SD) for adult and juvenile survival rates were set at 5% and varied 30% for high and low parameter estimates.

Fecundity (F) for adults and juveniles was determined from the product of the female sex ratio (R), maternity (M), and adult/juvenile survival (S_a, S_j) as described by Akcakaya and Root (2002) (e.g., $F = R^*M^*S$). We used a sex ratio of 54% females from trapping data (McCleery, 2003). Maternity (number of embryos and litters produced annually) was determined from published accounts of *N. floridana* (Rainey, 1956). For given fecundity estimates (adult versus juvenile), the appropriate survival estimates were used. Juveniles were considered sexually active only 6 months, that is juvenile maternity was 1/2 of adult maternity (M) (Hersh, 1981). Fecundity variances for adults and juveniles were determined as described by Burgman et al. (1993).

Initial abundances were estimated from trapping and trend data population estimates (McCleery, 2003). The high initial abundance estimate was calculated as the high end of the 95% confidence interval (CI) from the 2002 trapping data (McCleery, 2003) and assumed to be the highest biological plausible estimate, the medium was the population estimate given after trapping on 60 1-ha grids (McCleery, 2003), and the low estimate was taken from an average 2002–2003 trend data population estimates (McCleery, 2003). We assumed a stable-age distribution for our simulations.

2.4. Model use

We evaluated 4 levels of KLWR supplementations: 0, 5, 10, and 20 female KLWRs supplemented annually from the captive breeding program. For each scenario, KLWRs were introduced from year 1 to 5. Model scenarios were as follows:

1. Scenario 0 (S0) = no change.
2. Scenario 5 (S5) = supplementation of 5 female KLWRs annually for 5 years.
3. Scenario 10 (S10) = supplementation of 10 female KLWRs annually for 5 years.
4. Scenario 20 (S20) = supplementation of 20 female KLWRs annually for 5 years.

For each scenario, we ran 10,000 simulations over a 10-year period. We used 2 criteria to assess KLWR viability: population trajectory and risk of terminal extinction. To validate the model, we first estimated the 1996 populations population with a Jolly–Seber density estimator (Krebs, 1999) using 1996 trapping data (759 female KLWRs, Frank et al., 1997). Then simulated the population, initialized with the 1996 estimate, using

Table 1
Low, medium, and high parameter estimates used in population viability analysis for Key Largo woodrat population

Parameter	Low	Medium	High
Survival	Age = mean Juvenile = 0.086 Sj Adult = 0.173 Sj	Age = mean Juvenile = 0.123 Sj Adult = 0.247 Sa	Age = mean Juvenile = 0.160 Sj Adult = 0.321 Sa
Survival SD	Age = survival SD Juvenile = 0.078 Adult = 0.065	Age = survival SD Juvenile = 0.06 Adult = 0.05	Age = survival SD Juvenile = 0.042 Adult = 0.035
Fecundity	Age = maternity Juvenile = 0.095 Adult = 0.383	Age = maternity Juvenile = 0.135 Adult = 0.547	Age = maternity Juvenile = 0.176 Adult = 0.711
Fecundity SD	Not varied	Age = fecundity SD Adult = 0.186 Juvenile = 0.079 Average (10% CV)	Not varied
Initial abundances	Age = initial abundance Juvenile = 5 Adult = 13	Age = initial abundance Juvenile = 16 Adult = 41	Age = initial abundance Juvenile = 27 Adult = 71
Supplementation	Age = no. introduced Adult = 5 females	Age = no. introduced Adult = 10 females	Age = no. introduced Adult = 20 females

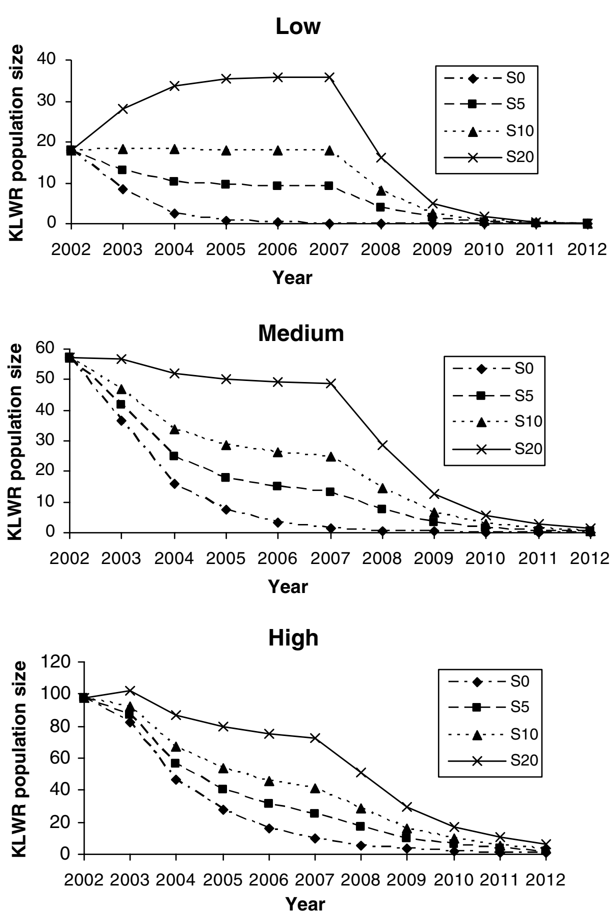

Fig. 2. Population trajectories for simulations of the KLWR population with varying numbers of supplementations (0 females [S0], 5 females [S5], 10 females [S10], and 20 females [S20]) using low, medium, and high parameter estimates, Key Largo, Florida, 2002–2012.

high, medium and low parameters estimates (Table 1). Ending simulation results (2002) were compared to actual populations estimates (McCleery, 2003). We also varied the aforementioned variables while holding others constant to identify sensitive variables in the KLWR model (Table 1, Akcakaya, 2000).

3. Results

3.1. Model validation

Model simulations initialized with 1996 population estimates yielded average 2002 populations of 45.26 (SD = 30.29), 9.35 (SD = 12.20), and 1.09 (SD = 4.09) using high, medium, and low parameter estimates, respectively (Fig. 4). Actual population estimates from 2002 trapping data were similar (14.0–57.3 female KLWRs, McCleery, 2003, Fig. 4). Additionally, our sensitivity analysis showed no parameters had a substantial impact (<4%) on the risk of extinction. Adult survival and adult fecundity were the most sensitive parameters with a >0.028 change in risk of terminal extinction between high and low parameters (Fig. 3), whereas standard deviation estimates of juvenile and adult survival showed almost negligible (<0.004) changes in the terminal extinction risks between high and low parameters.

3.2. KLWR viability

All 4 scenarios simulated with high, medium, and low parameters yielded an average KLWR population size of <10 individuals by the end of the simulation period

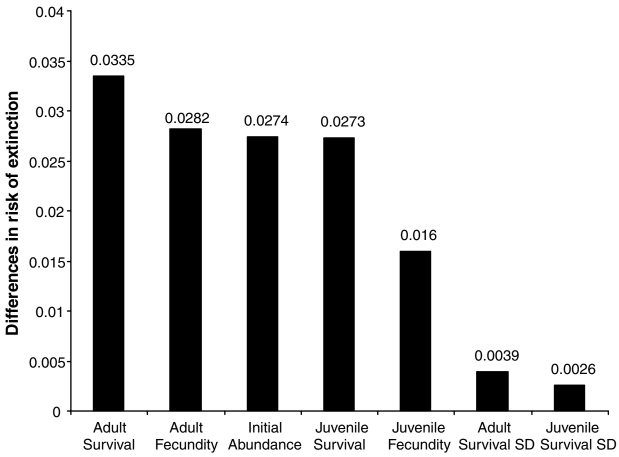

Fig. 3. Sensitivity analysis (difference in risk extinction between high and low parameter values) of model results to 6 parameters for the KLWR population, Key Largo, Florida, 2002.

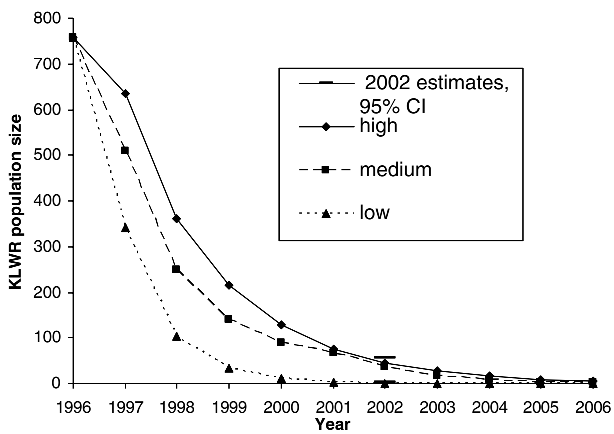

Fig. 4. Simulated population trajectories for the KLWR population using low, medium, and high parameter estimates from 1996 to 2006. Note, 2002 KLWR population estimates.

(10 years) (Fig. 2). The risk of terminal extinction was high (>98%) for scenarios using low and medium parameters without introduction. Supplementing the population with a high number of captive-reared KLWR (e.g., 20) managed to reduce the rate of population decline over the 5-year introduction period, however, when supplementation ceased population declines accelerated. Using high parameter estimates and supplementations the risk of terminal extinction was effectively decreased to <36%.

4. Discussion

4.1. Model validation

Our model appeared to be quite robust with the risk of terminal extinction varying no more than 4% when model parameters were varied by 60%. The model and parameter estimations also proved to be a good predictor of the 2002 population estimates when using initial abundances determined from 1996 KLWR population estimates.

4.2. KLWR viability

The KLWR is at high risk of extinction with the model predicting extinction of the species within 10 years (Fig. 2, Table 1). Even using best case scenario estimates (high parameters, Table 1) the KLWR had >70% of extinction over the next 10 years. Simulations suggest that captive breeding and supplementation of the wild population as proposed by the USFWS will have minimal long-term impacts on the recovery of the KLWR. Population trajectories using all three parameter levels declined after 2007 when supplementations ceased. However, supplementation of the population may prove an effective tool for preventing extinction of the KLWR population if continued annually over the period of interest. If optimistic parameters prove valid and $\geqslant$20 females are added to the population annually the risk of terminal extinction was substantially decreased (<7.5% risk). Study results suggest even assuming low parameters estimates the supplementation of 20 females annually (S20), might have some limited management potential (Fig. 2). The value of supplementing the population may be a short-term prevention of extinction allowing researchers more time to improve habitat and further examine limiting factors acting on the KLWR population. Still, it is important to note the model assumes the KLWRs necessary for breeding will not be removed from the wild, and supplementation of KLWRs will have the same survival and fecundity rates as wild KLWRs. Thus, model results should be viewed with caution because model estimates likely overestimated the effectiveness of supplementations.

4.3. PVA as a planning tool

The use of PVAs could prove instrumental in planning the recovery of the KLWR and other endangered species. Our model results suggest that if supplementation is used as a management tool it should be intensive (>20 captive breed female KLWRs released annually). Results also suggest the success of recovering the KLWR with supplementation is limited. Considering the limitations of supplemental stockings and the knowledge that captive breeding will be expensive, woodrat conservationists should determine if there is adequate support, resources and facilities to produce over 20 female woodrat annually. Additionally, because the model predicted the KLWR population was unable to sustain itself after supplementations ended, resources should be focused on identifying and removing potential limiting factors, such as habitat quality, fires ants, and feral cats. Finally, the model also illustrates the overall success of relocations will increase if planning and managing for an endangered species is done before the species is in a critical situation and management options are limited or have a limited chance of success (Griffith et al., 1989).

4.4. Sensitivity analysis and model improvement

An additional benefit of using a PVA approach to investigate the feasibility of various types of relocations is the ability to determine gaps and weakness of population parameters through the use of a sensitivity analysis. For instance, we found adult survival, adult fecundity, juvenile survival, and initial abundance resulted in the greatest differences in the model output. To improve the precision of PVA estimates and to enhance its effectiveness as a management tool, we make several recommendations. We recommend continuous monitoring of the population size with the trapping protocol used to obtain trend data (McCleery, 2003) and the addition of transects throughout north Key Largo. With the same amount of effort, transects have been shown to be more effective than grids at capturing small mammals at low densities (Pearson and Ruggiero, 2003). We also would monitor nests of sexually active KLWR females in an attempt to trap, radio tag, and estimate juvenile survival. Maternity data should be collected from females in captivity and incorporated into the KLWR model. Lastly, we believe future model improvements will help conservationists in evaluating supplementation and other possible management strategies that might help prevent the extinction of the KLWR.

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