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Better off in the wild? Evaluating a captive breeding and release program for the recovery of an endangered rodent

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ABSTRACT

The critical question for the success of all captive breeding and release programs (CBRPs) is the same: will the benefit of augmenting or reestablishing a population with captive animals outweigh the loss of taking individuals from the wild? Yet, few studies have simultaneously evaluated the impact of removal of animals for captive breeding on the source population and the potential contribution of the released animals to the augmented populations. We used the endangered Key Largo woodrat (*Neotoma floridana smalli*, KLWR) as a model system to simultaneously examine the effect of animal removal, captive breeding, and reintroduction on the dynamics and persistence of a wild population. We used mark-recapture and telemetry data, as well as zoo records from a recent CBRP for the endangered KLWR to parameterize a matrix population model and to simulate the response of the KLWR population to alternative captive breeding and release strategies. Our results suggest that a CBRP as practiced previously would not contribute to KLWR recovery; instead, removal of wild KLWR for captive breeding could harm the population. Captive breeding programs will not contribute to the recovery of KLWR unless survival of released animals and breeding success of captive individuals are improved. Our study provides a framework for simultaneous consideration of animal removal from the wild, breeding success in captivity and survival of released animals for a comprehensive evaluation of captive breeding programs.

1. Introduction

Over the last four decades humans have directly threatened at least one fifth of the planet's vertebrate species with extinction (Hoffmann et al., 2010). One management strategy that has been used to help prevent the extinction of rare or threatened species is captive breeding and release programs (CBRPs; Snyder et al., 2002). Under these programs, animals are removed from the wild and placed in a controlled captive environment where they are bred and their offspring are reared. Eventually, some or all of this captive population is released into its habitats to augment struggling populations or reestablish expatriated ones. CBRPs have yielded some high profile successes (e.g., California condor [*Gymnogyps californianus*] and the black-footed ferret [*Mustela nigripes*]), but such programs often fail to achieve the desired outcome (Fischer and Lindenmayer, 2000; Mathews et al., 2005; Snyder et al., 2002).

The critical question for the success of all CBRPs is the same: will the benefit of augmenting or reestablishing a population with captive animals outweigh the loss of taking individuals from the wild? One way to address this question is to use population models. Specifically, matrix population models provide a flexible framework for evaluating dynamics and persistence of biological populations, and for evaluating effects of alternative management strategies on population dynamics; these models can be used for evaluating the efficacy of expensive CBRPs before they are initiated or modified (Caswell, 2001; Ezard et al., 2010; Morris and Doak, 2002; Hostetler et al., 2013; Seddon et al., 2007). However, most modeling studies of reintroduction programs have focused on either evaluating the impacts to the source population of animal removal for captive breeding, or predicting the influence of the released individuals on the dynamics of augmented populations (Armstrong and Reynolds, 2012). Rarely have studies evaluated the potential costs (removal of wild animals) and benefits (increased population size or viability) of a CBRP simultaneously within a single modeling framework (Bustamante, 1996). Yet, it is only by weighing these costs and benefits that we can critically determine the overall benefit of starting or continuing a CBRP.

Concerned by the threat of extinction, in 2002 the U.S. Fish and Wildlife Service established captive breeding colonies and a release program for Key Largo woodrat (*Neotoma floridana smalli* [KLWR], McCleery et al., 2005, 2006; McCleery et al., 2013; Winchester

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et al., 2009). This cryptic, nocturnal subspecies of the eastern woodrat (*Neotoma floridana*) has been isolated in the approximately 972 ha of remaining hardwood hammock forests on the northern 1/3 of Key Largo, Florida where its population is believed to have steadily declined since the 1970s (McCleery et al., 2005; U.S. Department of the Interior, 1973, 1984). In fact, a population viability analysis (PVA) suggested a 70% probability of extinction by 2012 (McCleery et al., 2005). The causes for the KLWR's decline remain unknown but has been attributed to altered habitats (McCleery et al., 2007), predation (Winchester et al., 2009) and reduced recruitment during drier years (McCleery et al., 2013).

Captive breeding facilities were established at Lowry Park Zoo (Tampa, Florida, USA) and later at Disney's Animal Kingdom (Orlando, Florida, USA) in 2002 (Alligood et al., 2011). Subsequently, a release program was designed to put captive-bred KLWRs into their native hammock habitats. In an effort to augment the population, 41 KLWRs were released into the wild over a period of two years (McCleery et al., 2013). The survival rates of released KLWRs during the first 3 months were exceedingly low, with only a few released KLWRs surviving long enough to contribute to the growth of the wild population through reproduction. In fact, most of the released animals were lost to predation shortly after their release (McCleery et al., 2013). Low survival rates of released KLWRs have been attributed to inadequate anti-predator and vigilance behaviors of released individuals (McCleery et al., 2013). These results effectively halted the KLWR CBRP until the program could be thoroughly evaluated. However, it may be possible to address the behavioral shortcoming, and improve survival of released animals through prerelease conditioning programs and/or *in situ* captive-breeding program (Kock et al., 2007; Seddon et al., 2007).

We used the endangered KLWR as a model system to develop a comprehensive modeling framework that allows simultaneous consideration of animal removal from the wild for captive breeding, captive breeding success, and the influence of released animals on the dynamics and persistence of the wild population. We then use this framework for evaluating the KLWR CBRP, and for identifying strategies that can ensure success of CBRP. Accordingly, our objectives were to: (1) determine if CBRP as practiced previously would benefit or harm KLWR recovery; (2) evaluate population-level effects of alternative removal, captive breeding and reintroduction strategies; and (3) determine if improved recruitment rates of captive KLWR and improved survival of released KLWRs can improve a CBRP through growth of the wild population.

2. Materials and methods

2.1. Study area

KLWRs are isolated in an approximately 972 ha, 14-km stretch of protected tropical hardwood hammocks on the northern third of the island of Key Largo, the first and largest in a chain of islands (keys) extending from the southeastern tip of peninsular Florida. The hardwood hammock habitats on the island of Key Largo are unique, with a high diversity of mast producing trees and shrubs of West Indian origin (Karim and Main, 2009; Strong and Bancroft, 1994). Common trees found in the hammocks of Key Largo include gumbo-limbo (*Bursera simaruba*), poisonwood (*Metopium toxiferum*), wild tamarind (*Lysiloma bahamensis*) and pigeon plum (*Coccoloba diversifolia*). The climate of Key Largo is sub-tropical, exhibiting marked wet and dry seasons. Rainfall amounts and patterns can be variable but the region averages 1179 mm of rainfall annually, most of which occurs from May through September (Bancroft et al., 2000).

2.2. Population model

2.2.1. Parameter estimation

Estimates of KLWR abundance have been varied. In 2002, McCleery et al. (2006) estimated the wild KLWR population to be 30–182 individuals. A more recent study using data from 2008 to 2011 estimated annual abundance between 78 and 693 individuals; however, the confidence intervals of these estimates ranged from 0 to 1164 KLWRs, indicating poor precision of those estimates (Potts et al., 2012). Due to uncertainty in the estimates of population size and the disparity in the estimates, we repeated our analyses using a range of initial abundance of 150, 300 and 500 KLWRs.

We used estimates of apparent survival (φ) and recruitment (f) rates for wild-born KLWRs reported by McCleery et al. (2013) based on Capture-Mark-Recapture (CMR) analyses. KLWR recruitment varies seasonally as well as annually, with peaks in the spring and fall and little reproduction over the winter (Sasso and Gaines, 2002; McCleery et al., 2013). To account for this variation, we used seasonal and annual estimates of f from McCleery et al. (2013; Table 1).

To estimate true survival (S) of the wild, zoo and released populations we used radio-telemetry data and captive breeding records reported by McCleery et al. (2013). However, instead of utilizing the non-parametric estimates reported in that study, we used parametric estimates of survival, because parametric survival models allow projections of survival and its variance to desired time intervals (Lee and Wang, 2003). We used R version 2.12.2 (R Development Core Team 2011) statistical software (survival package; Therneau and Lumley, 2011) and evaluated the fit of four different parametric models for survival (exponential, lognormal, Weibull, and log logistic) based on Akaike Information Criterion (AIC; Table 2). For the zoo and wild populations, the exponential model was the best fitting model, and we used estimates based on this model to parameterize our population model. Alternatively, the best model for the released population was the lognormal model, which allows the hazard rate (the instantaneous rate of mortality) to vary over time. In this case, the hazard rate increased rapidly until approximately the 9th day after release, after which it declined slowly. To account for the varying hazard rate we estimated different survival probabilities from the lognormal model for the first two months and the next two months (Table 1). We set true survival rate for zoo population to an exponential log-hazard scale and true survival of released animals to a lognormal log-hazard scale.

To estimate recruitment in the KLWR zoo population we used specimen reports from Disney Animal Kingdom (Orlando, FL) and Lowery Park Zoo (Tampa, FL) that detailed any changes in the health or reproductive status of each woodrat (McCleery et al., 2013). We acquired records on 58 individuals at Disney (47 born at the facility) and 33 individuals at Lowry Park Zoo (24 born at facility) from April 2002 to December 2011.We calculated the mean number of offspring per individual for every two month interval (time step of the model) that they were in captivity and used this as an estimate of recruitment rate for the population model.

Finally, we assumed that the difference between true survival and apparent survival reflected emigration rate (i.e., $E = S_{wild} - \varphi$), and that this rate was the same for released and wildborn individuals.

2.2.2. Model structure

For modeling purposes, we created a population composed of four interacting subpopulations: (1) wildborn (*wild*) population on Key Largo, (2) captive (*zoo*) population for captive breeding, (3) released, 0–2 months post release (rel_1), and (4) released, 2–4 months post release (rel_2) (Fig. 1). We split the released rats