Received: 19 April 2024 | Revised: 6 August 2024 | Accepted: 10 September 2024

DOI: 10.1002/ecs2.70044

ECOSPHERE
AN ESA OPEN ACCESS JOURNAL

ARTICLE

Methods, Tools, and Technologies

# Long live the cat: Ocelot population viability in a planned reintroduced population in Texas, USA

**Lindsay A. Martinez**[1] | **Jason V. Lombardi**[2] | **Israel D. Parker**[3] | **Forrest East**[3] | **Tyler A. Campbell**[4] | **Roel R. Lopez**[3]

[1]Department of Rangeland, Wildlife, and Fisheries Management, Texas A&M University, College Station, Texas, USA

[2]Caesar Kleberg Wildlife Research Institute, Texas A&M University-Kingsville, Kingsville, Texas, USA

[3]Texas A&M Natural Resources Institute, Texas A&M University, College Station, Texas, USA

[4]East Foundation, San Antonio, Texas, USA

**Correspondence**
Lindsay A. Martinez
Email: lindsay.martinez14@gmail.com

**Present addresses**
Lindsay A. Martinez, East Foundation, San Antonio, Texas, USA; and Jason V. Lombardi, Wildlife Health Laboratory, California Department of Fish and Wildlife, Rancho Cordova, California, USA.

**Funding information**
U.S. Fish and Wildlife Service; Texas A&M University Department of Rangeland, Wildlife, and Fisheries Management

**Handling Editor:** Joseph D. Holbrook

## Abstract

Reintroductions are often needed to recover carnivore populations and restore ecological processes. Felids are common subjects of reintroduction efforts, but published population models informing felid reintroduction plans are uncommon, and poor planning has sometimes caused issues in felid reintroduction programs. In the United States, ocelots (*Leopardus pardalis pardalis*) are classified as endangered, and recovery requires population expansion into historic habitat. A multi-organization effort is underway to establish a new ocelot population in Texas by releasing ocelots into an area of 478 $km^2$ of suitable habitat in ocelots' historic but now unoccupied range. In this study, we used population viability analyses to compare different ocelot reintroduction strategies for the identified reintroduction area. Based on a potential ocelot breeding program's limitations, we modeled reintroduction using a founding population of no more than six ocelots and no more than four ocelots released per year for no more than 15 subsequent years. Within these limitations, we assessed projected population abundances and extinction risks after 30 years for 20 different reintroduction strategies. We found that long-term releases are necessary to establish a viable population; under conservative model assumptions, releasing six ocelots in the initial year and then releasing four individuals annually for an additional 10–15 years is necessary for attaining a projected population greater than 36.62 ocelots (baseline) with <6% extinction risk. We also found that ocelot population abundance is about equally sensitive to post-release mortality and inbreeding depression. This highlights the importance of not only supporting reintroduced ocelots' survival but also managing for high genetic diversity in the reintroduction program. Further, we found that realistic but more liberal assumptions on the carrying capacity of the reintroduction area and the age of first reproduction for ocelots increase projected population abundances (53.95 individuals and 61.26 individuals, respectively), and thus reintroduction success. The model's sensitivity to carrying capacity suggests that long-term habitat protection and expansion are among the most important management actions to support ocelot reintroduction. Our study establishes the first population viability model for an ocelot reintroduction plan anywhere across the species' wide geographic range, and it reinforces several key considerations for wildlife reintroduction efforts worldwide.

**KEYWORDS**
captive propagation, carnivore reintroduction, *Leopardus pardalis*, population viability analysis, reintroduction strategies, Texas

## INTRODUCTION

Carnivores play an important role in ecosystem function, as their direct and indirect interactions with prey species can cause cascading impacts through ecosystems (Dickman et al., 2014; Ripple et al., 2014). During a time of global declines in carnivore populations (Crooks et al., 2011) that impact other species and ecosystem function (Hoeks et al., 2020), reintroduction efforts may sometimes be needed to recover carnivore populations and restore their important ecological roles (Ripple et al., 2014; Wolf & Ripple, 2018).

Wolf and Ripple (2018) found that there is a large potential for future worldwide carnivore reintroduction efforts. This potential, along with historically successful reintroductions and efforts to identify common reasons for success or failure in reintroductions by the International Union for the Conservation of Nature (IUCN, 2013; Soorae, 2013) and others (Wilson, 2018), may inspire and support increased use of reintroduction to recover carnivore populations.

In the United States, over the past 30 years, several carnivore species have been successfully reintroduced in areas where the species had been extirpated. This includes gray wolves (*Canis lupus*) to Yellowstone National Park (Ripple & Beschta, 2012), fishers (*Pekania pennanti*) to Washington state (Happe et al., 2020), bobcats (*Lynx rufus*) to New Jersey (Matos, 2020) and Cumberland Island, Georgia (Diefenbach et al., 2013), river otters (*Lutra canadensis*) to Pennsylvania (Serfass et al., 2003), and Canada lynx (*Lynx canadensis*) to Colorado (Devineau et al., 2010). Key factors for success in these American reintroductions have included releasing carnivores into large, contiguous patches of suitable habitat that provide sufficient prey populations, managing human threats to carnivores, securing a sufficient source stock of individuals for the reintroduction, and conducting post-release monitoring to inform adaptive management of reintroduction. Internationally, efforts for captive breeding and reintroduction of Iberian lynx (*Lynx pardinus*) have demonstrated that captive breeding and reintroduction programs can support recovery of endangered felids when conducted in coordination with abatement of historic threats to the species, genetic management of the source stock, habitat and prey management and protection, and public outreach promoting human-carnivore coexistence (Simón et al., 2012).

Despite the above examples of successful reintroductions, an assessment of historic global reintroduction efforts thus far shows that many wildlife reintroductions do not result in survival and reproduction of the released animals, much less establishment of self-sustaining populations (Fischer & Lindenmayer, 2000; Jule et al., 2008; Stepkovitch et al., 2022). Carnivore reintroductions specifically can face challenges from carnivores' ecological needs for large amounts of prey and habitat (Miller et al., 2013; Wolf & Ripple, 2016) plus their often-slow life histories (Stier et al., 2016). A variety of other ecological factors can also negatively impact carnivore reintroduction efforts, such as predation by existing carnivore populations (e.g., in reintroduced European mink [*Mustela lutreola*] Maran, 2013), hybridization with sympatric species (e.g., in reintroduced red wolves [*Canis rufus*]), or disease outbreaks (e.g., viral infections in reintroduced red wolf pups; Bartel & Rabon, 2013).

Additionally, the socioeconomic dimensions of human-carnivore coexistence (Bruskotter et al., 2017; Miller et al., 1999, 2013) make carnivore reintroductions challenging. For example, red wolf (Hinton et al., 2017) and Mexican wolf (*Canis lupus baileyi*, Cruz Romo et al., 2013) reintroductions have revealed that anthropogenic mortality of reintroduced carnivores can be high when managers fail to secure local community support and tolerance for the reintroduced carnivores. As an added human dimensions factor in the United States, private landowners may worry that if endangered animals are reintroduced directly to or disperse to their lands, Endangered Species Act regulations protecting the species will lead to legal restrictions on land use (Hansen et al., 2018).

Finally, reintroduction success depends not just on natural or human factors present in the environment but also on management and planning decisions shaping the reintroduction process. Lack of proper reintroduction planning has limited the success of some historic felid reintroduction efforts. For example, a low number of founders used for some Eurasian lynx reintroductions has led to low genetic diversity and slow population growth in reintroduced lynx populations (Linnell et al., 2009; Mueller et al., 2022).

The IUCN Species Status Commission created the guidelines for reintroductions and other conservation translocations (2013) to provide recommendations for overcoming the various social and ecological challenges facing conservation reintroductions. The IUCN's recommended reintroduction planning process includes using models such as population viability analyses (PVAs) to evaluate whether the establishment of a viable population is feasible in a selected reintroduction area given constraints on habitat, food resources, and source animals, for example. Models are needed not just for assessing reintroduction feasibility but also for comparing different possible release plans (Seddon et al., 2007). Release plans should identify the necessary intensity, frequency, and duration of releases plus the optimal age and sex ratios of released individuals (Armstrong & Reynold, 2012; IUCN, 2013). Modeling can also be used to identify key factors that drive population growth and viability to inform the planning of additional management practices, such as habitat management, supplement feeding, or control of competitors or predators (IUCN, 2013; Seddon et al., 2007). Finally, once a reintroduction has been implemented, ecological monitoring data collected from reintroduced animals and the habitat should be incorporated into models to reassess population trajectory and revise reintroduction plans, as needed, through an adaptive management process (Armstrong & Reynold, 2012; IUCN, 2013; Seddon et al., 2007).

The ocelot (*Leopardus pardalis* spp.) is a medium-sized felid found from southern Brazil and Uruguay to northern Mexico and the extreme Southern United States, including the state of Texas. The ocelot is classified as a species of least concern range-wide but is considered endangered in multiple range countries, including the United States and Mexico (Hunter, 2015). In the United States, ocelot (*L. p. pardalis*) populations have declined significantly due to historic habitat loss and fragmentation, road mortality, loss of genetic diversity, and unregulated killing during the early-mid 20th century (U.S. Fish and Wildlife Service, 2016). In the United States today, ocelots occur in only two known breeding populations in extreme coastal southern Texas along the Gulf of Mexico (Lehnen et al., 2021; Lombardi et al., 2021; Sergeyev et al., 2022; Sergeyev, Campbell, et al., 2023; Veals et al., 2022). Scientists theorize there may be less than 100 individuals remaining in Texas, of which 80% are thought to exist within the Ranch Ocelot Population, which occurs on extensive private working ranchlands, while a smaller number of individuals (estimated <20) exist in the Refuge Ocelot Population found in and around Laguna Atascosa National Wildlife Refuge (Lombardi et al., 2021, 2022). Though ocelot population densities and home range sizes vary across the territorial and solitary ocelot's range (de Oliveira et al., 2010), the current population density estimate across one portion of private coastal ranchlands in Texas is 17.6 ocelots/100 $km^2$ (Lombardi et al., 2022).

In the known geographic range of ocelots in Texas, Veals et al. (2022) found that the maximum estimated area of high-quality ocelot habitat—described as large patches of woody cover containing dense low vegetative cover (Lombardi et al., 2021, 2022; Sergeyev, Campbell, et al., 2023)—is only 1515 $km^2$. This small occupied range may make ocelots in Texas vulnerable to localized extirpation due to potentially catastrophic events such as severe and persistent drought (Haines, Tewes, & Laack, 2005), disease outbreak (U.S. Fish and Wildlife Service, 2016), wildfire, or major tropical cyclone flooding (Onorato et al., 2010). The reintroduction of an additional ocelot population in Texas is likely needed to supplement the existing populations and achieve abundance thresholds established for ocelot recovery from endangered status in the United States (U.S. Fish and Wildlife Service, 2016). Lombardi et al. (2021) used contemporary ocelot geolocation data to inform a landscape structure suitability analysis for ocelots, revealing at least 24,430 $km^2$ of suitable landscape structure of woody cover for ocelots across the southernmost 18 counties of Texas. With the knowledge of a large amount of suitable landscape structure of woody cover for ocelots in Texas, Martinez et al. (2024) identified the same landscape structure across a larger area of Texas and incorporated Light Detection and Ranging (LiDAR) data to assess fine-scale vegetative cover suitable for ocelots and ultimately identify a suitable area for reintroduction of an additional population.

This recent ocelot habitat suitability assessment was part of a larger effort by private landowner, federal, state, zoological, nonprofit, and academic partners in Texas to plan to reintroduce an ocelot population to a part of their historic, but now unoccupied, range in Texas (Lombardi et al., 2022; Martinez et al., 2024; RecoverTexasOcelots.org). Planning has included creating a policy to reintroduce endangered ocelots to historic habitats in Texas without impacting the property rights and land uses of Texas' private landowners, who make up over 97% of the state's land base (Leslie, 2016). Additionally, partner organizations are preparing to establish an ocelot breeding and behavioral preparation program in Texas that will create a source of ocelots for reintroduction without having to rely on translocation of ocelots from other range countries (Translocation Team, 2009; U.S. Fish and Wildlife Service, 2016).

To aid the ocelot breeding and reintroduction planning process, we used PVAs to model the outcomes of potential ocelot reintroduction strategies. Our objectives were to evaluate the feasibility of establishing a new ocelot population in Texas at the ocelot reintroduction area identified by Martinez et al. (2024) given a small number of available individuals for release, compare different release strategies, and assess the impact of different model input parameters on population persistence and growth to inform possible management actions supporting the reintroduction.

# METHODS

## Study system

We evaluated reintroduction strategies at an identified area within the ocelot's historic but now unoccupied range in southern Texas. The area (Figure 1) was chosen based on ecological factors such as the extent of suitable landscape structure of woody cover and fine-scale vegetative cover for ocelots, remoteness from likely threats to ocelots (including high-traffic roadways, projected urban development, and major tropical storm surges), and sociopolitical dimensions like private land ownership patterns and likely landowner tolerances to reintroduction of an endangered carnivore (Martinez et al., 2024). In the reintroduction area, Martinez et al. (2024) identified 478 $km^2$ of contiguous suitable landscape structure of woody cover that was at least 1 km from high-traffic roadways (with annual average daily traffic exceeding 1000 vehicles per day) using methods from Lombardi et al. (2021), who found that ocelots in Texas use large, low-density, and unfragmented patches of woody cover. Ocelots also use/select for low (<2 m) dense vegetation for resting and hunting sites (Sergeyev, Campbell, et al., 2023; Sergeyev, Tanner, et al., 2023). Martinez et al. (2024) used LiDAR data to identify 363 $km^2$ of suitable fine-scale vegetative cover for ocelots within the area of suitable landscape structure based on a requisite amount of canopy height and vegetation density (0.5–1.0 m above ground) identified in Sergeyev et al. (2022), Sergeyev, Campbell, et al. (2023), and Sergeyev, Tanner, et al. (2023).

The chosen reintroduction area has no existing ocelot populations but is known to support populations of various small- and medium-sized mammals, birds, and herpetofauna that could serve as ocelot prey as well as populations of bobcats and coyotes (Watts, 2015), which have a similar body size and diet to ocelots and co-occur with existing ocelots in Texas (Lombardi et al., 2023; Sergeyev et al., 2022; Sergeyev, Campbell, et al., 2023; Sergeyev, Tanner, et al., 2023). Located approximately 100 km from existing ocelot populations along the Texas Gulf Coast, the reintroduction area is within the Texas-Tamaulipan Thornscrub ecoregion of southern Texas and consists of mostly large, privately owned working cattle ranches with sparse human populations and infrastructure (Martinez et al., 2024). Ocelots do not depredate cattle, and small livestock animals such as poultry that are impacted by ocelots in other parts of their range (Gálvez et al., 2023) are not raised in this area. Vegetation types in the reintroduction area include woody savannas with diverse mixed-dense patches of shrub communities containing whitebrush (*Aloysia gratissima*), lime prickly ash (*Zanthoxylum fagara*), cat-claw acacia (*Acacia greggii*), blackbrush (*Acacia rigidula*), honey mesquite (*Prosopis glandulosa*), and a variety of cacti species (French et al., 2022; Leivers et al., 2023). The region has a semiarid subtropical climate with inconsistent rainfall and episodic drought (Norwine & Kuruvilla, 2007). Given that the longest observed ocelot dispersal is 50 km (Booth-Binczik, 2007) and that major (inter)state highways exist between the reintroduction area and existing populations (Martinez et al., 2024), connectivity between the existing ocelot populations and the reintroduction area is not expected. While connectivity could allow for genetic exchange with the genetically depressed existing ocelot populations, the Ocelot Recovery Plan (U.S. Fish and Wildlife Service, 2016) calls for the establishment of additional geographically distinct populations to reduce ocelot extinction risk in Texas. The Recovery Plan suggests using translocations to augment existing populations (U.S. Fish and Wildlife Service, 2016). Translocations between the reintroduced and existing populations can also be used to manage genetic diversity in ocelot populations in Texas (U.S. Fish and Wildlife Service, 2016).

## Model overview

We used Vortex (version 10.5.5), a standard software used for PVAs (Lacy & Pollak, 2021). In Vortex, known or estimated species demographic parameters (e.g., rates of survival, reproduction, inbreeding depression, and dispersal) and environmental conditions (e.g., stochastic catastrophic events and carrying capacity) are used to produce models of population dynamics and extinction processes. The goal of a Vortex model is to explore how different management strategies can impact population viability and to identify which demographic factors are the most impactful for population growth and/or persistence (Lacy & Pollak, 2021).

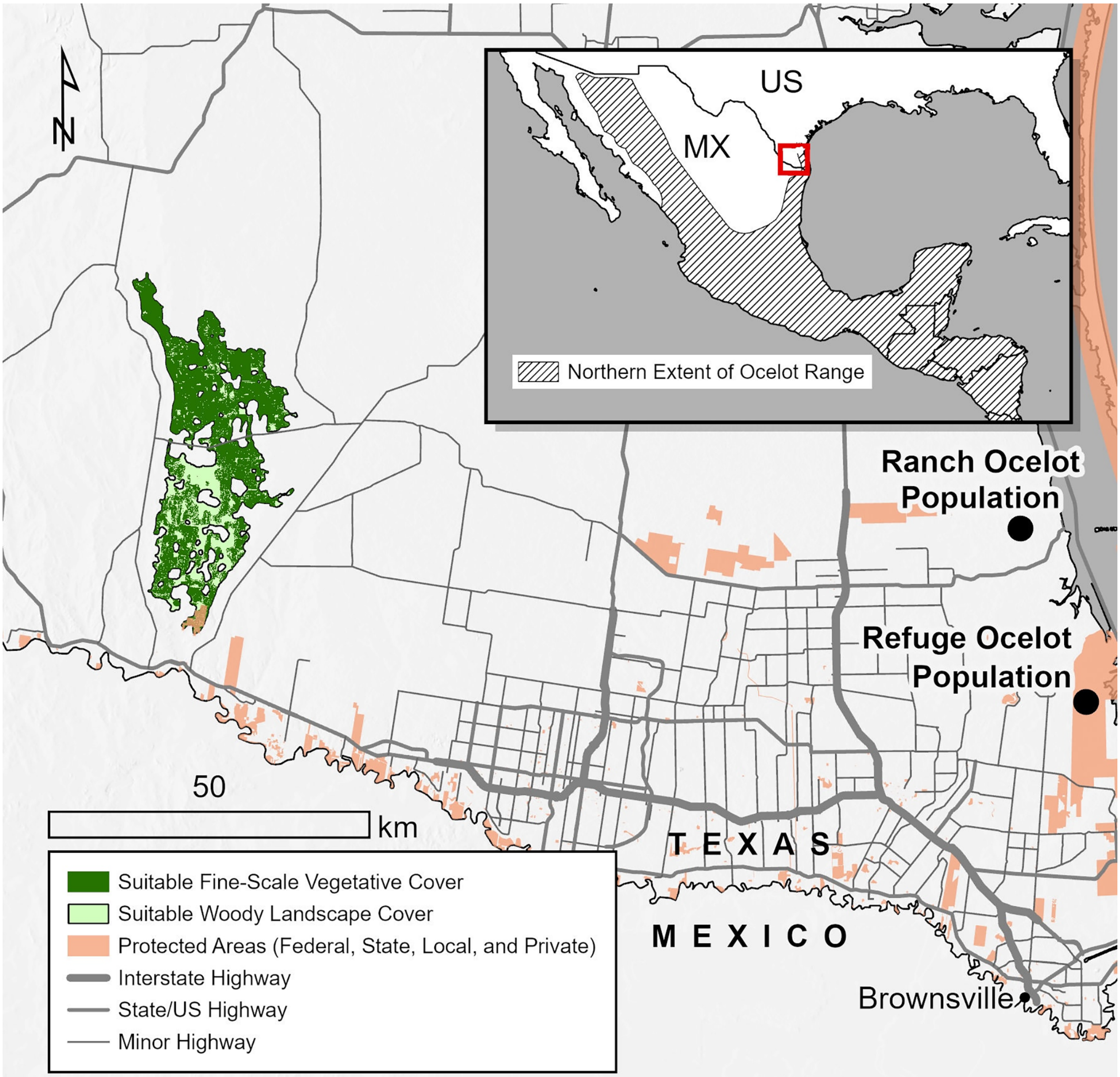


**FIGURE 1** The potential ocelot (*Leopardus pardalis pardalis*) reintroduction area identified by Martinez et al. (2024) has contiguous suitable fine-scale vegetative cover for ocelots (totaling 363 km$^2$) within a patch of contiguous suitable woody landscape cover (totaling 478 km$^2$). This patch is approximately 100 km west of the approximate locations of existing ocelot populations in Texas. Note that while other suitable fine-scale cover and suitable landscape cover for ocelots exist around the area, no cover of either type that is noncontiguous to the patch is shown.

A reintroduction population model should include initial and supplement releases of individuals at the reintroduction area; post-release mortality of reintroduced animals; and—amongst released individuals who survive—demographic rates of mortality, dispersal, and reproduction (Knight, 2012).

## Parameter estimates

In our model (conceptualized in Figure 2), we input ocelot demographic parameters (Table 1) identified by Haines, Tewes, Laack, Grant, et al. (2005), who conducted a baseline PVA of ocelots in southern Texas, though we updated several of the parameters based on more recent study of ocelots and the ecology of the identified reintroduction area (Martinez et al., 2024). First, we maintained the age of first reproduction as three for females and four for males and the probabilities of litter sizes of one (62%), two (37%), or three kittens (1%) as in Haines, Tewes, Laack, Grant, et al. (2005), but we increased male and female maximum age of reproduction from 11 to 13 years. Haines, Tewes, Laack, Grant,

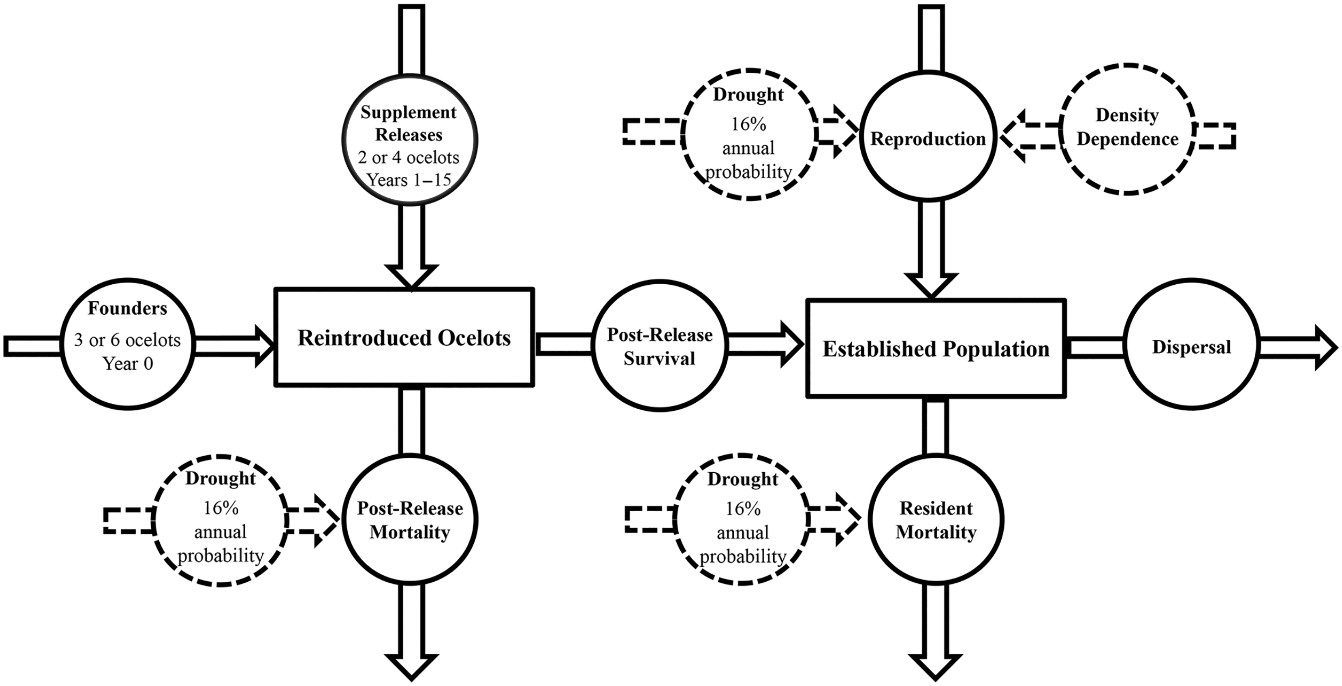


**FIGURE 2** Conceptualization of the population model to evaluate future reintroduction strategies for establishing an additional ocelot (*Leopardus pardalis pardalis*) population in unoccupied ocelot habitat in southern Texas. The model uses ocelot demographic parameters identified by Haines, Tewes, Laack, Grant, and Young (2005) and includes population inputs (arrows into boxes) such as the initial release of ocelots at the reintroduction area, supplement releases of ocelots for up to 15 additional years, and reproduction of established ocelots. Population losses (arrows out of boxes) include mortality of released or established individuals and dispersal out of the population. Severe drought has negative impacts on both ocelot survival and reproduction and was modeled as occurring in 16% of years. Density dependence impacts reproduction in that the number of adult females breeding in a year decreases when the population is at high density.

et al. (2005) initially suggested that ocelots can reproduce beyond 10 years of age under favorable conditions, and there is evidence of a 13-year-old wild ocelot in Texas birthing and raising consecutive litters of kittens (Lombardi et al., 2022; Masters, 2022; Sergeyev et al., 2022; Sergeyev, Campbell, et al., 2023; Sergeyev, Tanner, et al., 2023).

For dispersal parameters, which in the Vortex model reflect permanent individual exit from the reintroduced population, we followed Haines, Tewes, Laack, Grant, et al. (2005), which assumed that ocelots may disperse at 2–3 years old (Haines, Tewes, & Laack, 2005). We modeled a 2.5% dispersal rate for males and females 2–3 years old. Haines, Tewes, Laack, Grant, et al. (2005) used a 5% dispersal rate for males 2–3 years old in a population near carrying capacity, but we anticipate dispersal will be lower from the reintroduction area due to the lack of an existing population and the extensive availability (478 $km^2$) of suitable woody landscape structure for ocelots, of which 363 $km^2$ is also suitable fine-scale vegetative cover (Martinez et al., 2024). Although the reintroduction area has extant coyote and bobcat populations (Watts, 2015) and potentially mountain lion (*Puma concolor*) populations (Elborch & Harveson, 2022), ocelots have been documented to coexist with these species in Texas (Lombardi et al., 2023; Sergeyev et al., 2022; Sergeyev, Campbell, et al., 2023; Sergeyev, Tanner, et al., 2023) and northeastern Mexico (Branney et al., 2023).

We defined severe droughts as a catastrophic environmental event that could impact ocelots in the reintroduction area, which has a drier climate than other areas of the ocelot's geographic range (Lehnen & Lombardi, 2023). As in Haines, Tewes, Laack, Grant, et al. (2005), we used the Modified Palmer Drought Index (MPDI) (National Centers for Environmental Information, 2022) from the past 100 years to calculate the probability of severe drought (defined as MPDI $<-2.3$) at the reintroduction area. We determined there is a 16% annual risk of severe drought at the reintroduction area and used estimates from Haines, Tewes, Laack, Grant, et al. (2005) on the negative impacts of severe drought on ocelot survival (10% lower in a drought year than in a non-drought year) and reproduction (25% lower in a drought year).

Finally, we modeled an initial population of 0 at the reintroduction area and a carrying capacity of 63 ocelots. In Vortex, any population growth is truncated at the carrying capacity. The estimation of carrying capacity was

**TABLE 1** Demographic parameters used in a baseline ocelot (*Leopardus pardalis pardalis*) population model constructed by Haines, Tewes, and Laack (2005) and Haines, Tewes, Laack, Grant, et al. (2005) compared with those used in the ocelot reintroduction population model that included updated parameters based on expert opinion and recent literature on ocelot ecology and biology.

| Parameter | Baseline model | Reintroduction model |
|---|---|---|
| Inbreeding depression | | |
| Lethal equivalents (measure of inbreeding depression) | 3.14 | 3.14[a,b] |
| Probability of lethal equivalents due to recessive alleles | 50 | 50 |
| Reproduction | | |
| Reproduction correlated with survival | Yes | Yes |
| Long-term polygamous mating system | Yes | Yes |
| Age first female reproduction | 3 | 3[b] |
| Age first male reproduction | 4 | 4[b] |
| Maximum age of reproduction | 11 | 13 |
| Sex ratio at birth | 50:50 | 50:50 |
| Maximum litter size | 3 | 3 |
| Percent of adult males breeding | 50 | 50 |
| Percent of females with litter/year (SD) at low population density | 85 (10) | 85 (10)[b] |
| Percent of females with litter/year (SD) at high population density | 65 (10) | 65 (10)[b] |
| Probability litter size of 1 | 62 | 62 |
| Probability litter size of 2 | 37 | 37 |
| Probability litter size of 3 | 1 | 1 |
| Mortality | | |
| Female probability of mortality first-year post-release | 50 | 33[a,b] |
| Female probability of mortality at year 0–1 (SE) | 29 (5) | 29 (5) |
| Female probability of mortality at year 1–2 (SE) | 13 (5) | 13 (5) |
| Female probability of mortality at year 2–3 (SE) | 22 (5) | 22 (5) |
| Female probability of mortality at years 3+ (SE) | 13 (2) | 13 (2) |
| Male probability of mortality first-year post-release | ... | 33[a,b] |
| Male probability of mortality at year 0–1 (SE) | 29 (5) | 29 (5) |
| Male probability of mortality at year 1–2 (SE) | 13 (2) | 13 (2) |
| Male probability of mortality at year 2–3 (SE) | 37 (10) | 37 (10) |
| Male probability of mortality at years 3+ (SE) | 13 (2) | 13 (2) |
| Severe drought impacts | | |
| Annual probability of severe drought | 0.11 | 0.16 |
| Percent reduction in reproduction due to severe drought | 25 | 25 |
| Percent reduction in survival due to severe drought | 10 | 10 |
| Dispersal | | |
| Age range of dispersers (male and female) | 2–3 | 2–3 |
| Annual probability of dispersal | 5 | 2.5[a] |
| Study site | | |
| Initial population size | 38 | 0 |
| Carrying capacity (SD) | 38 (4.4) | 63.8[b] (4.4) |

*Note*: The reintroduction model is used to plan the establishment of an additional ocelot population in a part of the historic but unoccupied geographic range of the ocelot in southern Texas, USA.

[a]Subject to ±50% sensitivity analyses.

[b]Varied in liberal models.

based on the identification of 363 km$^2$ of suitable fine-scale vegetative cover for ocelots at the reintroduction area (Martinez et al., 2024) and a recent ocelot population density estimate in Texas of 17.6 ocelots/100 km$^2$ (Lombardi et al., 2022). Carrying capacity was calculated based on the extent of fine-scale vegetative cover rather than prey densities because ocelots are a dietary generalist that has been observed in Texas consuming various species of small to potentially large mammals (e.g., rodents, rabbits, armadillo [*Dasypus novemcintus*] and white-tailed deer [*Odocoileus virginiana*]), herpetofauna, and birds (Booth-Binczik et al., 2013). As in Haines, Tewes, Laack, Grant, et al. (2005), we used a percentage of adult females breeding (AFB) of 85% at low population density (as calculated by Vortex; Lacy & Pollak, 2021) and 65% at high population density to reflect possible density dependency impacts on the reproduction of a territorial felid population that is approaching carrying capacity and thus space limitation.

## Model use

We evaluated a suite of 20 ocelot release strategies (Table 2) in the PVA. Design of release strategies was based on protocols for reintroductions of captive-bred Iberian lynx (Rueda et al., 2021). All modeled release strategies reflected a conservative assumption that a newly established ocelot breeding program can provide only a limited number of individuals for release (<6) in any year. Within this limitation, release strategies varied in both the size of the founding population released in the first year and the structure of supplement releases in subsequent years (Table 2). For the founding population, the model tested all possible release strategies under either a small initial release of three ocelots (including one male and two females) or a larger initial release of six ocelots (two males and four females). The skew toward females was used because one male ocelot can breed with multiple females (U.S. Fish and Wildlife Service, 2016). Next, we varied the structure of supplement releases

**TABLE 2** Summary of the 20 possible strategies assessed in our model of ocelot (*Leopardus pardalis pardalis*) reintroduction into an unoccupied habitat in Texas.

| | Founding population | Supplement releases | | | |
|---|---|---|---|---|---|
| Reintroduction strategy | Total (female, male) | Length supplement release period (years) | Supplement release interval (years)[a] | No. released per supplement release event (female, male) | Total no. (male, female) released during supplement period |
| (1ab) No supplementation | 6 (4, 2) or 3 (2, 1) | … | … | … | 0 |
| (2ab) 5-year, low supplementation | 6 (4, 2) or 3 (2, 1) | 5 | 1 | 2 (1, 1) | 10 (5, 5) |
| (3ab) 5-year, high supplementation | 6 (4, 2) or 3 (2, 1) | 5 | 1 | 4 (2, 2) | 20 (10, 10) |
| (4ab) 10-year, low supplementation | 6 (4, 2) or 3 (2, 1) | 10 | 1 | 2 (1, 1) | 20 (10, 10) |
| (5ab) 10-year, high supplementation | 6 (4, 2) or 3 (2, 1) | 10 | 1 | 4 (2, 2) | 40 (20, 20) |
| (6ab) 15-year, low supplementation | 6 (4, 2) or 3 (2, 1) | 15 | 1 | 2 (1, 1) | 30 (15, 15) |
| (7ab) 15-year, high supplementation | 6 (4, 2) or 3 (2, 1) | 15 | 1 | 4 (2, 2) | 60 (30, 30) |
| (8ab) 15-year back-load high | 6 (4, 2) or 3 (2, 1) | 1–5 | 1 | 2 (1, 1) | 26 (13, 13) |
| | | 6–15 | 3 | 4 (2, 2) | |
| (9ab) 15-year back-load low | 6 (4, 2) or 3 (2, 1) | 1–5 | 1 | 2 (2, 2) | 18 (9, 9) |
| | | 6–15 | 5 | 4 (2, 2) | |
| (10ab) 15-year front-load | 6 (4, 2) or 3 (2, 1) | 1–5 | 1 | 4 (2, 2) | 28 (14, 14) |
| | | 6–15 | 3 | 2 (1, 1) | |

*Note*: Reintroduction strategies varied by the size of the founding population released in the first year (either 6 or 3 ocelots), as well as the structure of supplement releases after the initial year. Ten supplement release strategies were modeled that varied by: total length of the supplement release period (0, 5, 10, or 15 years), interval (in years) between supplement releases, and number of male and female ocelots used per supplement release (high supplementation of four ocelots or low supplementation of two ocelots). Most strategies included an annual supplement release, but in 15-year "back-load" and "front-load" strategies, supplement releases only occur in 3- or 5-year intervals for the last 10 years of the supplement release period. Back-load strategies use low supplementation (two ocelots per supplement release) in the first 5 years of the supplement release period and high supplementation (four ocelots per supplement release) in the last 10 years of the supplement release strategy, while the front-load strategy uses high supplementation for the first 5 years and low supplementation for the last 10 years of the supplement release period.

[a]Supplement release intervals: 1, supplement every year; 3, supplement every 3 years; 5, supplement every 5 years.

following the release of founders in the initial year. We modeled 10 possible supplementation strategies, including no supplement releases, low supplementation strategies of two individuals per annual supplement release (one male and one female), and high supplementation strategies of four individuals (two males and two females) per annual supplement release.

Based on the discussion of feasible breeding and reintroduction activities with ocelot reintroduction planners (including wildlife veterinarians and wildlife scientists from involved partner organizations including Caesar Kleberg Wildlife Research Institute, U.S. Fish and Wildlife Service, East Foundation, and Lindner Center for Conservation and Research of Endangered Wildlife at the Cincinnati Zoo and Botanical Garden), we modeled supplement releases for periods of 5, 10, and 15 years. Finally, in addition, to supplement release strategies where ocelots are released every year of the supplementation period, we also modeled non-annual strategies in which supplement releases do not occur every year during the supplement release periods but at lengthier intervals (every 3 or 5 years) for the last 10 years of a 15-year supplement release period. These strategies assessed the impacts of variable annual availability of ocelots for release and declining or increasing intensity of releases over time. Two non-annual release strategies were characterized as "back-load," in which two ocelots are released per year in years 1–5 of supplement releases and then four ocelots are released every three years (back-load high) or every 5 years (back-load low) in years 6–15. In the "front-load" strategy, meanwhile, four ocelots are released per year for years 1–5 of supplementation, and only two ocelots are released every three years for years 6–15.

In our model, all released ocelots were between 1 and 2 years old. Releasing 1–2-year-olds has proven successful for Iberian lynx reintroductions (Rueda et al., 2021), perhaps because young animals have more behavioral plasticity than older individuals and because young animals have not yet established a home range that they may attempt to return to upon release (Gross et al., 2010; Tetzlaff et al., 2019; Thomas et al., 2023). We used the estimation from Haines, Tewes, Laack, Grant, et al. (2005) that ocelots' age of first reproduction in the wild is 3 for females and 4 for males. As such, in our model, reproduction could not occur for multiple years after the initial release of 1-year-old ocelots.

Since captive-bred ocelots have never been reintroduced to the wild, we did not have a measurement of post-release mortality for ocelots. High post-release mortality is often documented in reintroduction programs (in the cases in which mortality figures are measured and published at all), particularly those using captive-bred individuals, due to the stresses of transfers and poor adaptation to the reintroduction site (Jule et al., 2008). Our model utilized published measures of mortality from four captive-bred Eurasian lynx reintroduction programs, which reported 32%–70% post-mortality of released individuals (Jule et al., 2008). Ocelot reintroduction planning partners suggested that successful methods used for other species, such as raising captive-bred animals in a quasi-natural behavioral preparation program in which they can learn to explore, hunt, and conduct other natural behaviors necessary for life in the wild before eventual soft release to the reintroduction area, will likely improve survival upon release (Devineau et al., 2011; Rueda et al., 2021; Tetzlaff et al., 2019; Thomas et al., 2023). In our model, we used a 33% probability of mortality (including death or dispersal out of the reintroduction area) for the first year after an ocelot is released but conducted a sensitivity analysis to assess results with a 50% higher post-release mortality (i.e., 49.5% post-release mortality of ocelots) than the baseline estimate and with a 50% lower mortality (i.e., 16.5% post-release mortality) than the baseline. In the model, post-release mortality operated as an addition to the normal demographic rates of mortality for 1-year-old ocelots from Haines, Tewes, Laack, Grant, et al. (2005).

We also conducted a sensitivity analysis on dispersal and the impacts of inbreeding (measured by the number of lethal equivalents [LEs] in ocelots and modeled by Vortex as the mortality of individuals with two copies of a lethal allele). We assessed the impacts of both a 50% higher and 50% lower probability of dispersal than the baseline for males and females 2–3 years old. We also varied lethal equivalents by 50% to test low inbreeding (1.57 LEs) and high inbreeding (4.71 LEs) compared with the baseline of 3.14 LEs, which is the default value in Vortex based on a study of captive mammals (Lacy & Pollak, 2021) and was assumed by Haines, Tewes, Laack, Grant, et al. (2005) for ocelots. All sensitivity analyses were used to explore how the selected factors, which may be targeted for management in the breeding and reintroduction programs, impact persistence of the reintroduced population.

We used conservative inputs to build the baseline PVA of ocelot reintroduction. To further assess the impact of selected factors in model results and to explore possible best-case scenarios given the use of the most effective release strategy, we also assessed the results of models using several more liberal inputs. First, for the best-performing release strategy, we tested a model using a carrying capacity of 84 ocelots rather than 63 to represent ocelot use of the full extent of 478 $km^2$ of woody landscape cover size. This was based on the notion that ocelots will use open, mixed, and dense cover within woody patches with suitable landscape structure metrics and adjacent herbaceous cover (Lombardi et al., 2021, 2022; Sergeyev et al., 2022; Sergeyev, Campbell, et al., 2023; Sergeyev, Tanner, et al., 2023). The baseline (conservative) estimate of carrying capacity was based only on the 363 $km^2$ of suitable low vegetative cover, which exists within the patch of suitable woody cover (Martinez et al., 2024).

We also assessed a model with 0 LEs and one with 0% post-release mortality to further explore the impacts of eliminating inbreeding and post-release mortality through management of the reintroduction program. In another liberal model, we decreased the age of first reproduction for ocelots to two years old for both males and females given that ocelots can physiologically reproduce at two years of age (Seager & Demorest, 1978) and may do so in an environment with sufficient space for new territories plus no or few older, reproducing ocelots. Finally, we used a more liberal input regarding the impacts of density dependence in one model by setting the percent of adult females breeding to 95% at low population density and 75% at high population density, compared with the baseline conservative inputs of 85% and 65%, respectively.

All release strategies were modeled for 500 iterations over a 30-year management timeline that the U.S. Fish and Wildlife Service uses for assessments of species status and recovery planning. For each release strategy, we calculated extinction risk as the percent of the 500 iterations in which one or both sexes went extinct from the population within 30 years. For each strategy, we also measured the average population abundance after 30 years across all iterations in which the population did not, under that strategy, go extinct.

# RESULTS

The number of ocelots released in the initial year had an impact on population abundance and extinction risks after 30 years (Table 3). For all strategies, the larger founding population (four females, two males) resulted in lower extinction risks and higher population abundances compared with the smaller founding population (two females, one male).

All strategies (Table 2) that included some supplement releases of ocelots after the initial year had lower extinction risks compared with the strategies without supplement releases, which had extinction risks of at least 87%. The highest-performing strategies were the 10- and 15-year high supplementation strategies that both included annual releases of four ocelots per year. These strategies performed similarly; with an initial first-year release of six ocelots, the 10-year high supplementation strategy resulted in a population of 36.62 ocelots with a 6% extinction risk after 30 years while the 15-year high supplementation strategy resulted in a population of 41.35 with a <1% extinction risk. Meanwhile, shorter (five-year) supplement release periods and inconsistent supplement releases (i.e., front-load or back-load strategies where ocelots are not released annually throughout the supplementation) resulted in population extinction risks of at least 19% after 30 years.

We compared results from baseline reintroduction models with results from the sensitivity analyses where we varied mortality in the first-year post-release, dispersal, and inbreeding impacts (number of LEs) by 50% (Table 4). Across all possible strategies, extinction risk was most sensitive to post-release mortality, while population abundance

**TABLE 3** Percent extinction risk (risk), population abundance (*N*), and population abundance standard deviation (SD) outcomes after 30 years for 20 potential future ocelot (*Leopardus pardalis pardalis*) reintroduction strategies for the reintroduction area identified in southern Texas.

| Reintroduction strategy | Founding population: Four females, two males | | | Founding population: Two females, one male | | |
|---|---|---|---|---|---|---|
| | Risk (%) | *N* | SD | Risk (%) | *N* | SD |
| No supplementation | 87 | 13.27 | 10.02 | 99 | 9.14 | 3.44 |
| 5-year, low supplementation | 54 | 19.14 | 13.86 | 72 | 16.01 | 11.42 |
| 5-year, high supplementation | 28 | 24.79 | 16.10 | 38 | 24.98 | 16.17 |
| 10-year, low supplementation | 27 | 25.23 | 15.99 | 40 | 23.21 | 15.93 |
| 10-year, high supplementation | 6 | 36.62 | 16.21 | 5 | 33.60 | 16.66 |
| 15-year, low supplementation | 11 | 29.17 | 15.96 | 14 | 25.77 | 15.67 |
| 15-year, high supplementation | <1 | 41.35 | 14.53 | <1 | 42.63 | 14.26 |
| 15-year back-load high | 24 | 25.66 | 16.20 | 38 | 20.36 | 14.17 |
| 15-year back-load low | 32 | 22.48 | 15.12 | 48 | 18.98 | 13.42 |
| 15-year front-load | 19 | 29.73 | 16.29 | 25 | 25.72 | 15.97 |

**TABLE 4** Population abundance and extinction risk results for sensitivity analyses of ocelot (*Leopardus pardalis pardalis*) reintroduction strategies with variation in inputs for dispersal (for age/sex classes that disperse), mortality (for the first year after release), and lethal equivalents (a measure of inbreeding in population viability analyses) compared with the baseline model.

| Model | Baseline | +50% dispersal | −50% dispersal | +50% mortality | −50% mortality | +50% LEs | −50% LEs |
|---|---|---|---|---|---|---|---|
| No supplementation | 13.27 (87) | 12.19 (93) | 18.98 (88) | 16.04 (90) | 15.39 (88) | 11.49 (92) | 16.67 (87) |
| 5-year, low supplementation | 19.14 (54) | 19.63 (59) | 20.43 (48) | 18.54 (66) | 22.33 (50) | 16.15 (57) | 26.69 (53) |
| 5-year, high supplementation | 24.79 (28) | 25.26 (29) | 28.34 (23) | 24.32 (40) | 29.33 (17) | 23.06 (33) | 30.57 (20) |
| 10-year, low supplementation | 25.23 (27) | 22.46 (31) | 27.97 (23) | 21.85 (43) | 28.02 (20) | 22.26 (31) | 29.86 (24) |
| 10-year, high supplementation | 36.62 (6) | 36.40 (6) | 36.41 (4) | 31.04 (13) | 39.90 (2) | 32.75 (6) | 39.01 (4) |
| 15-year, low supplementation | 29.17 (11) | 28.79 (16) | 32.30 (12) | 25.03 (26) | 32.40 (5) | 25.74 (13) | 33.65 (12) |
| 15-year, high supplementation | 41.35 (<1) | 42.09 (1) | 44.25 (<1) | 36.42 (3) | 45.71 (0) | 39.82 (<1) | 45.26 (<1) |
| 15-year back-load high | 25.66 (24) | 23.29 (27) | 26.40 (21) | 21.62 (38) | 27.59 (15) | 22.51 (26) | 28.54 (23) |
| 15-year back-load low | 22.48 (32) | 22.02 (38) | 24.99 (26) | 19.39 (45) | 26.56 (23) | 20.01 (39) | 27.62 (32) |
| 15-year front-load | 29.73 (19) | 27.78 (21) | 30.66 (17) | 24.45 (30) | 33.35 (1) | 25.19 (19) | 33.33 (16) |

*Note*: Population abundance (percent extinction risk) is indicated for each model. LEs are lethal equivalents. Reintroduction strategies were modeled over 30 years, and all strategies used for sensitivity analyses begin with a founding population of six ocelots (four females, two males) and vary by structure of supplement release.

was about equally sensitive to both post-release mortality and inbreeding and least sensitive to dispersal. In all strategies modeled, 50% reductions in post-release mortality, dispersal, or inbreeding impacts had positive impacts on both extinction risks and final population abundances. Extinction risk was <4% for 10-year and 15-year high-supplementation strategies starting with six ocelots and with 50% lower dispersal, mortality, or lethal equivalents. However, even with 50% reductions in dispersal, post-release mortality, or inbreeding depression, extinction risk was >10% for all other strategies, except for the 15-year low supplementation strategy starting with six ocelots and with 50% lower post-release mortality (10% extinction risk).

Our tests of liberal "best-case scenario" models using the best-performing 15-year high supplementation strategy starting with a founding population of six ocelots (four females, two males) showed that projected population abundance increases if the model is less conservative, while extinction risk within 30 years remains almost nonexistent for this strategy (Table 5). Ocelot reproduction at two years of age was the most impactful of the parameters assessed in the liberal model; this model produced a final population abundance of 61.26 ocelots, just under the conservative estimate of a carrying capacity of 63. The second most impactful factor was estimated carrying capacity. The model projected 53.95 ocelots after 30 years if the carrying capacity was 84, which would reflect a more liberal estimate of carrying capacity based on ocelot use of the full extent of suitable woody patch structure and likely different woody cover types throughout their diel cycle. Eliminating post-release survival, eliminating inbreeding depression, or lowering the impacts of density dependence on the percent of adult females breeding had similar impacts on final population abundance that were slightly lower than the impacts of a higher carrying capacity.

# DISCUSSION

While many carnivore reintroductions have been found to fail or have not been evaluated at all (Jule et al., 2008), historically successful reintroductions have helped identify lessons learned for future reintroduction programs. Many carnivore reintroduction case studies point to the importance of conducting reintroductions in areas that are ecologically and socially suitable for reintroduction, meaning the areas provide for animals' ecological needs while also minimizing natural or human risks (Diefenbach et al., 2013; Happe et al., 2020; Simón et al., 2012). Other studies highlight the importance of identifying and managing ecological threats to reintroduced populations, such as predation (Maran, 2013), disease, or hybridization (Bartel & Rabon, 2013), before the reintroduction happens, if possible, or in an adaptive management process as those threats arise. Population models can be used to evaluate the impacts of ecological factors in reintroduction sites as well as in situ management strategies targeting these factors (Seddon et al., 2007). Finally, studies suggest using wild-born (Diefenbach et al., 2013; Jule et al., 2008) and locally adapted animals (Diefenbach et al., 2013) for reintroduction and employing strategies such as soft release (Devineau et al., 2011) or supplementary

**TABLE 5** Percent extinction risk (Risk %) and population abundance (*N*) result for models of the 15-year, high supplementation strategy beginning with a founding population of six ocelots (four females, two males) using liberal model inputs.

| Liberal model | Risk % | *N* |
|---|---|---|
| Baseline | <1 | 41.35 |
| 2-year-olds reproduce | 0 | 61.26 |
| Carrying capacity 84 ocelots | <1 | 53.95 |
| 0% post-release mortality | 0 | 48.39 |
| Reduced impacts from density dependence | <1 | 48.49 |
| 0 lethal equivalents | <1 | 46.46 |

*Note*: Liberal model inputs included male and female age of first reproduction as 2 years old, carrying capacity of 84 ocelots, no post-release impacts on mortality, low-density dependence impacts with 95% adult females breeding at low population density and 75% adult females breeding at high density, and no inbreeding depression.

feeding (Lopez-Bao et al., 2008) to increase post-release survival and site fidelity amongst the reintroduced individuals.

In its guidelines for planning, implementing, and managing wildlife reintroductions, the International Union for Conservation of Nature (IUCN, 2013) recommends addressing many of the above factors. It also recommends using population modeling to design release strategies that will maximize the chance of successful population establishment given ecological conditions and life history of the species. Models of release strategies are important for identifying the number and age/sex classes of individuals needed for the reintroduction to be successful. They can also help identify how long releases need to occur to establish a population, how to avoid inbreeding or other genetic problems in the reintroduced population, and how to not cause harm to source populations (IUCN, 2013). Ultimately, modeling is needed to design effective reintroduction strategies that can lead to successful population establishment and avoid pitfalls such as low population growth or loss of genetic diversity (Buk et al., 2018; Mueller et al., 2022).

It is known that reintroduction success is often greater when there are more individuals released for multiple years at multiple locations within a reintroduction area (IUCN, 2013; Wilson, 2018). In cases where reintroductions are supported by translocation of wild-sourced individuals, this can be complicated due to the potential impacts of removing individuals from wild-source populations (IUCN, 2013). In the case of the ocelot, translocation of wild individuals from extant populations to the reintroduction site in Texas is not currently being assessed due to the small size and genetic erosion of the existing populations in Texas (U.S. Fish and Wildlife Service, 2016) and the expected logistical challenges of securing a sufficient number of wild ocelots from populations in other range countries. Rather, a captive breeding program will be established in Texas with the specific purpose of supplying ocelots for reintroduction in Texas (Ocelot Reintroduction Study Captive Propagation Team, 2023). Our goal was to model different possible release strategies given a limited source stock of available founders sourced from the breeding program to explore the feasibility of reintroduction, compare possible strategies, and identify key factors impacting population viability.

The planned breeding program for ocelots in Texas is conservatively predicted to be able to provide six ocelots in an initial year and then up to four annually based on available breeders and space at a to-be-established captive breeding facility (Ocelot Reintroduction Study Captive Propagation Team, 2023). Our model showed that releasing only a limited number of individuals (six in the first year and four in subsequent years) does have the potential to establish a viable population large enough to contribute to ocelot recovery from endangered status in the United States if releases occur for 10–15 years.

We found that the most successful 15-year reintroduction strategies we assessed resulted in projected populations of 41.35 ocelots (conservative scenarios) or 53.95–61.26 ocelots (more liberal, best-case scenarios) with less than ≤1% extinction risk after 30 years. The conservative and best-base scenarios both result in population sizes that exceed the size of the existing Refuge Ocelot Population (<20 ocelots) and are comparable with the largest known portion of the Ranch Ocelot Population (33–55 ocelots; Lombardi et al., 2022). In total, reintroduction has the potential to be a large positive development for ocelot population abundance in Texas relative to the species' current abundance in the northern periphery of its range. To reach this potential, a variety of other biological, ecological, and sociopolitical factors must be accounted for during reintroduction program implementation, as these factors are likely to impact long-term program success. Further, program partners must secure long-term operational resources to continue supplement releases for a decade-plus.

Unsurprisingly, our model showed that increasing the number of released ocelots has positive impacts on population growth and persistence. Over time, expanding the size of the breeding program and the number of breeding facilities can make it possible to release a greater number of ocelots and improve the likelihood of population persistence and growth toward carrying capacity. Releasing 8–10 individuals each year at a reintroduction area is recommended in the Iberian lynx reintroduction program (Iberlince, 2016), and this is a possible target that can likely increase the chance of successfully establishing a viable reintroduced ocelot population.

We also compared the influence of different factors (inbreeding depression, post-release survival, carrying capacity, and life history) on projected population abundance. In the best-case scenario liberal models of reintroduction, we found that the reintroduced population of ocelots could nearly reach the estimated carrying capacity of 63 individuals if ocelots begin to reproduce at two years of age in the reintroduction area. We believe this may occur in the absence of mature, established ocelots in the reintroduction area. Further, we found that the population is projected to reach 53.95 ocelots if the carrying capacity depends on the total area of available suitable landscape structure of woody cover at the reintroduction area, not just available suitable fine-scale vegetative cover with areas of suitable woody cover. We believe that it is reasonable to expect that the carrying capacity at the reintroduction area may be greater than the baseline estimates of 63 ocelots based on the suitable landscape structure of woody cover available within the area (Lombardi et al., 2021; Martinez et al., 2024). Further, in the calculation of available habitat in the ocelot reintroduction area, Martinez et al. (2024) did not include any habitat within 1 km of a high-traffic roadway, any noncontiguous woody cover to the rest of the woody patch, or any herbaceous or bare land cover types, though ocelots are likely to also use these areas (Lombardi et al., 2022; Sergeyev, Campbell, et al., 2023; Sergeyev, Tanner, et al., 2023; Veals et al., 2022).

Given that the model was most sensitive to the age of first reproduction and to carrying capacity, it is important to monitor ocelot reproduction at the reintroduction area and total extent of habitat use to inform values for carrying capacity and reproduction inputs in future models of ocelot reintroduction. As one example, given that high-traffic paved roads greatly impact ocelot mortality (Blackburn et al., 2021), home range placement, and resource use (Veals et al., 2022), further study should be conducted to refine the extent of suitable habitat and the estimated carrying capacity of the reintroduction area based on presence of roads. Regarding management priorities, age of first reproduction cannot be influenced by managers and may be dependent on socio-spatial dynamics within the reintroduction area. Therefore, the most important management strategy to support reintroduction likely will be habitat management, including preservation of existing contiguous patches of suitable cover and habitat expansion.

Our sensitivity analysis also showed that reducing post-release mortality, dispersal, or inbreeding have positive impacts on population persistence, though the effects of each of these are smaller than those for increasing carrying capacity. We found that ocelot mortality in the first year post-release is more impactful than dispersal on extinction risk across all possible release strategies. As such, reintroduction planners should also design, implement, evaluate, and adapt methods to increase ocelots' post-release survival to maximize the chance of population persistence. Possible strategies for increasing post-release survival may include management of behavioral preparation programs for ocelots fated for release, use of soft release procedures at the reintroduction area, and post-release support systems (Sasmal et al., 2015; Tetzlaff et al., 2019). One such post-release support system is a supplement feeding program with live prey held in enclosures; cats can enter and naturally hunt in the enclosure if they are unable to capture free-ranging prey due to low prey availability or inexperience hunting. These systems have proven successful in other wild cat reintroduction efforts such as Iberian lynx in Spain and Portugal (Lopez et al., 2024; Lopez-Bao et al., 2008; Serra et al., 2024).

Inbreeding depression, represented in our model by the number of lethal equivalents in the population, had impacts only slightly lower than those from post-release mortality on population abundance after 30 years. This shows that genetic diversity is nearly as important as high post-release survival for long-term persistence and growth of a reintroduced population. Along with managing post-release survival, wildlife reintroduction program managers should develop methods to reduce the risk of inbreeding in the reintroduced population because it can limit future population growth due to the accumulation of lethal alleles in the population (Leus & Lacy, 2009). In the selection of individuals for release, managers should prioritize animals with high heterozygosity levels and low relatedness to other individuals in the reintroduction area. Long-term supplement releases of unrelated individuals may also promote genetic diversity in the reintroduced population.

Our model showed that an initial release of founders as well as at least 10 years of consistent supplement releases of two to four ocelots are likely needed to establish a viable ocelot population. These strategies will require the release of 46–66 ocelots over 11–16 years. Although the implementation of a successful reintroduction can be challenging, our study indicates that population abundance varies between conservative baseline scenarios and more liberal best-case scenarios. Therefore, if inputs in the baseline do prove overly conservative, population abundances may be greater than those predicted in the model.

While we were interested in exploring model results beyond 30-year timelines, model variance was high after 30 years and made interpretation difficult. Further, demographic parameters such as survival and reproductive rates still must be measured in the reintroduction area to provide accurate parameters for model inputs and make meaningful projections of long-term outlooks for the reintroduction. We used previously published demographic parameters on ocelot survival and reproduction from existing ocelot populations in Texas (Haines, Tewes, Laack, Grant, et al., 2005), which are approximately 100 km from the proposed reintroduction area. Since all released ocelots in the reintroduction program will be collared before release, the reintroduction program provides an excellent opportunity for monitoring demographic parameters in ocelots in the reintroduction area. Updated information on these parameters is likely to impact population model outcomes (Haines, Tewes, Laack, Grant, et al., 2005). Accurate demographic information must be used to properly update the reintroduction model and further assess possible reintroduction strategies, including the need for future releases or other management actions (Miller-Butterworth et al., 2021). Monitoring of ocelot movements and habitat use in the reintroduction area can also be used in the future to create a spatially explicit ocelot reintroduction model that can account for factors such as habitat availability, dispersal, and interactions between ocelot territories (as in Haines et al., 2006; Lehnen et al., 2021; Lombardi et al., 2021; Veals et al., 2023).

While we used this PVA to coarsely explore model parameters and release strategies across 30 years, we plan to adapt this model in the future to account for observed biological and ecological dynamics at the release site to more accurately determine long-term projections for the reintroduced population. Assessing ocelot population abundance using large-scale, long-term camera trap monitoring will also allow for basic testing of the PVA model's predictive accuracy as the reintroduction proceeds (Miller-Butterworth et al., 2021).

While it is important to identify release strategies that may lead to program success, the identification of strategies not feasible for successful reintroduction is also necessary. We found that if the ocelot breeding program's output makes it difficult to continue releases annually for at least 10 years following the initial release or difficult to release multiple ocelots every year, the reintroduction program will likely no longer be sustainable. Under the 5-year strategies and the non-annual back-load and front-load strategies, the population is estimated to have at least a 19% probability of extinction within 30 years. Program managers can use our model's adaptive framework to predict population trajectory under different breeding program outputs and release strategies to inform management decisions regarding further commitment of resources toward the reintroduction program.

Comparisons of the different release strategies evaluated in our model and our sensitivity analyses provide recommendations for the design of an ocelot reintroduction program in Texas, and they support several important findings from worldwide reintroduction efforts. First, our model showed that supplement releases are critical for the viability of reintroduced populations that start with a small number of founders. Initial releases containing a small number of founders alone will not succeed in establishing a population; our model for ocelots showed that populations' probabilities of extinction were over 87% in cases where no supplement releases were included. Further, supplement releases require a long-term commitment; in the case of ocelot reintroduction, supplement releases should occur for at least 10–15 years for reintroduction to be viable while inconsistent, infrequent, or short-term supplement release strategies are less likely to succeed in establishing a population. Also in the ocelot model, inbreeding and post-release mortality had similar impacts, highlighting that genetic diversity can limit reintroduced populations' growth and viability. As such, in addition to securing a sufficient and sustained number of source animals for release and implementing management actions to increase animals' chance of survival upon release, managers should manage for high genetic diversity in the reintroduced population. They can do so by selecting founders who have high levels of heterozygosity, are unrelated to other founders, and have uncommon alleles. Finally, in our model, population abundance and extinction risk were most sensitive to carrying capacity, further supporting that reintroduction sites should be as large and connected as possible to maximize available habitat and potential carrying capacity.

## AUTHOR CONTRIBUTIONS

Lindsay A. Martinez, Jason V. Lombardi, Israel D. Parker, Forrest East, Tyler A. Campbell, and Roel R. Lopez contributed to the conceptual design of the population model. Israel D. Parker and Forrest East ran the models. Lindsay A. Martinez, Jason V. Lombardi, and Israel D. Parker led the writing of the manuscript. All authors contributed to the development of the manuscript drafts, and all gave their approval for publication.

## ACKNOWLEDGMENTS

We thank all project partners from the Texas Ocelot Reintroduction Study for their support in informing this work and supporting other elements of the ocelot reintroduction planning effort. The U.S. Fish and Wildlife Service and the Texas A&M University Department of Rangeland, Wildlife, and Fisheries Management provided financial support for this study. Thank you to SE Lehnen, AM Veals, and four peer reviewers for providing feedback on earlier versions of this manuscript. This publication is number 100 of the East Foundation and number 23-115 of the Caesar Kleberg Wildlife Research Institute.

## CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

## DATA AVAILABILITY STATEMENT

Model input and output data (Martinez, 2024) are available from Figshare: https://doi.org/10.6084/m9.figshare.25521439.v1.

## REFERENCES

Armstrong, D. P., and M. H. Reynold. 2012. "Modeling Reintroduced Populations: The State of the Art and Future Directions." In *Reintroduction Biology: Integrating Science and Management*, edited by J. G. Ewen, D. P. Armstrong, K. A. Parker, and P. J. Seddon. Oxford: Wiley-Blackwell. 528 pp.

Bartel, R. A., and D. R. Rabon, Jr. 2013. "Re-Introduction and Recovery of the Red Wolf in the Southeastern USA." In *Global Re-Introduction Perspectives: 2013. Further Case Studies from around the Globe*, edited by P. S. Soorae. Gland and Abu Dhabi: IUCN/SSC Re-Introduction Specialist Group and Environment Agency-Abu Dhabi. xiv + 282 pp.

Blackburn, A., L. J. Heffelfinger, A. M. Veals, M. E. Tewes, and J. H. Young. 2021. "Cats, Cars, and Crossings: The Consequences of Road Networks for the Conservation of an Endangered Felid." *Global Ecology and Conservation* 27: e01582.

Booth-Binczik, S. D. 2007. "Report from the Field: Monitoring Ocelot Dispersal with Satellite Telemetry." *Endangered Species Update* 24: 110–12.

Booth-Binczik, S. D., R. D. Bradley, C. W. Thompson, L. C. Bender, J. W. Huntley, J. A. Harvey, and J. L. Mays. 2013. "Food Habits of Ocelots and Potential for Competition with Bobcats in Southern Texas." *Southwestern Naturalist* 58: 403–410.

Branney, A. B., S. Carvajal, J. V. Lombardi, W. C. Stasey, M. E. Tewes, and A. Caso. 2023. "Jaguars, Ocelots, Coatimundis… Oh My: Species Composition and Temporal Overlap of a Diverse Carnivore Guild in the Sierra of Tamaulipas, México." *Global Ecology and Conservation* 44: e02493.

Bruskotter, J. T., J. A. Vucetich, M. J. Manfredo, G. R. Karns, C. Wolf, K. Ard, N. H. Carter, et al. 2017. "Modernization, Risk, and Conservation of the World's Largest Carnivores." *BioScience* 67: 646–655.

Buk, K. G., C. V. van der Merwe, K. Marnewick, and P. J. Funston. 2018. "Conservation of Severely Fragmented Populations: Lessons from the Transformation of Uncoordinated Reintroductions of Cheetahs (*Acinonyx jubatus*) into a Managed Metapopulation with Self-Sustained Growth." *Biodiversity and Conservation* 27: 3393–3423.

Crooks, K. R., C. L. Burdett, D. M. Theobald, C. Rondinini, and L. Boitani. 2011. "Global Patterns of Fragmentation and Connectivity of Mammalian Carnivore Habitat." *Philosophical Transactions of the Royal Society B* 366: 2642–51.

Cruz Romo, J. L., R. A. Wolf Webels, B. A. Gonzales, Z. Y. Gonzales, C. Aguilar, C. Garcia Chaves, N. E. Lara Diaz, and C. A. Lopez Gonzales. 2013. "Re-Introduction of the Mexican Wolf in the Sierra Madre Occidental, Mexico." In *Global Re-Introduction Perspectives: 2013. Further Case Studies from around the Globe*, edited by P. S. Soorae. Gland and Abu Dhabi: IUCN/SSC Re-Introduction Specialist Group and Environment Agency-Abu Dhabi. xiv + 282 pp.

de Oliveira, T. G., M. A. Tortao, L. Silveira, C. B. Kasper, F. D. Mazim, M. Lucherini, A. T. Jácomo, J. B. G. Soares, R. V. Marques, and M. Sunquist. 2010. "Ocelot Ecology and Its Effect on the Small-Felid Guild in the Lowland Tropics." In *Biology and Conservation of Wild Felids*, edited by D. W. Macdonald and A. J. Loveridge, 559–580. Oxford: Oxford University Press.

Devineau, O., T. M. Shenk, G. C. Doherty, G. C. White, and R. H. Kahn. 2011. "Assessing Release Protocols for Canada Lynx Reintroduction in Colorado." *Journal of Wildlife Management* 75: 623–630.

Devineau, O., T. M. Shenk, G. C. White, P. F. Doherty, Jr., P. M. Lukacs, and R. H. Kahn. 2010. "Evaluating the Canada Lynx Reintroduction Program in Colorado: Patterns in Mortality." *Journal of Applied Ecology* 47: 524–531.

Dickman, C. R., A. S. Glen, M. E. Jones, M. E. Soulé, E. G. Ritchie, and A. D. Wallach. 2014. "Strongly Interactive Carnivore Species: Maintaining and Restoring Ecosystem Function." In *Carnivores of Australia: Past, Present, and Future*, edited by A. S. Glen and C. R. Dickman, 301–322. Collingwood: CSIRO Publishing.

Diefenbach, D., L. A. Hansen, C. Miller-Butterwoth, J. H. Bohling, R. J. Warren, and M. J. Conroy. 2013. "Re-Introduction of Bobcats to Cumberland Island, Georgia, USA: Status and Lessons Learned after 25 Years." In *Global re-Introduction Perspectives: 2013. Further Case Studies from around the Globe*, edited by P. S. Soorae. Gland and Abu Dhabi: IUCN/SSC Re-Introduction Specialist Group and Environment Agency-Abu Dhabi. xiv + 282 pp.

Elborch, L. M., and P. M. Harveson. 2022. "It's Time to Manage Mountain Lions in Texas." *Wildlife Society Bulletin* 46: e1361.

Fischer, J., and D. B. Lindenmayer. 2000. "An Assessment of the Published Results of Animal Relocations." *Biological Conservation* 96: 1–11.

French, J. T., N. J. Silvy, T. A. Campbell, and J. M. Tomeček. 2022. "Divergent Predator Activity Muddies the Dynamic Landscape of Fear." *Ecosphere* 13: e3927.

Gálvez, N., J. Infante-Varela, T. G. de Oliveira, J. C. Cepeda-Duque, L. A. Fox-Rosales, D. Moreira, J. C. Huaranca, et al. 2023. "Small Wild Felids of South America: A Review of Studies, Conservation Threats, and Research Needs." In *Neotropical Mammals: Hierarchical Analysis of Occupancy and Abundance*, edited by S. Mandujano, E. J. Narano, and G. P. Andrade Ponce, 13–41. Cham: Springer.

Gross, K., G. Pasinelli, and H. P. Kunc. 2010. "Behavioral Plasticity Allows Short-Term Adjustment to a Novel Environment." *The American Naturalist* 176: 456–464.

Haines, A. M., M. E. Tewes, L. L. Laack, J. S. Horne, and J. H. Young. 2006. "A Habitat-Based Population Viability Analysis for Ocelots (*Leopardus pardalis*) in the United States." *Biological Conservation* 132: 424–436.

Haines, A. M., M. E. Tewes, and L. L. Laack. 2005. "Survival and Cause-Specific Mortality of Ocelots in Southern Texas." *Journal of Wildlife Management* 69: 255–263.

Haines, A. M., M. E. Tewes, L. L. Laack, W. E. Grant, and J. Young. 2005. "Evaluating Recovery Strategies for an Ocelot (*Leopardus pardalis*) Population in the United States." *Biological Conservation* 126: 512–522.

Hansen, M. E., S. M. Bennett, J. Morales, and R. M. Yeagley. 2018. *Cooperative Conservation: Determinants of Landowner Engagement in Conserving Endangered Species*. Center for Growth and Opportunity at Utah State University. Policy Paper 2018.003. https://doi.org/10.13140/RG.2.2.19684.22400

Happe, P. J., K. T. Jenkins, R. M. McCaffery, J. C. Lewis, K. L. Pilgrim, and M. K. Schwartz. 2020. "Occupancy Patterns in a Reintroduced Fisher Population during Reestablishment." *Journal of Wildlife Management* 84: 344–358.

Hinton, J. W., G. C. White, D. R. Rabon, Jr., and M. J. Chamberlain. 2017. "Survival and Population Size Estimates of Red Wolf." *Journal of Wildlife Management* 81: 417–428.

Hoeks, S., M. A. J. Huigbregts, M. Busana, M. B. J. Harfoot, J.-C. Svenning, and L. Santini. 2020. "Mechanistic Insights into the Role of Large Carnivores for Ecosystem Structure and Functioning." *Ecography* 43: 1752–63.

Hunter, L. 2015. *Wild Cats of the World*. London: Bloomsbury Publishing.

Iberlince. 2016. "Protocolo de liberaciones de lince ibérico." http://www.iberlince.eu/images/docs/3_InformesLIFE/ProtocoloLiberaciones_M.Iberlince.pdf.

IUCN. 2013. *Guidelines for Reintroductions and Other Conservation Translocations*. Version 1.0. Gland: IUCN Species Survival Commission viiii + 57 pp.

Jule, K. R., L. S. Leaver, and S. E. G. Lea. 2008. "The Effects of Captive Experience on Reintroduction Survival in Carnivores: A Review and Analysis." *Biological Conservation* 141: 355–363.

Knight, T. M. 2012. "Using Population Viability Analysis to Plan Reintroductions." In *Plant Reintroduction in a Changing Climate: Promises and Perils*, edited by J. Maschinski and K. E. Haskins, 155–170. Washington, D.C.: Island Press.

Lacy, R. C., and J. P. Pollak. 2021. *VORTEX: A Stochastic Simulation of the Extinction Process. Version 10.5.5*. Brookfield, IL: Chicago Zoological Society.

Lehnen, S. E., and J. V. Lombardi. 2023. "Climate Envelope Modeling for Ocelot Conservation Planning: Peering inside the Black Box." *Ecosphere* 14: e4477.

Lehnen, S. E., M. A. Sternberg, H. M. Swarts, and S. E. Sesnie. 2021. "Evaluating Population Connectivity and Targeting Conservation Action for an Endangered Cat." *Ecosphere* 12: e03367.

Leivers, S., T. Campbell, M. Bodenchuk, and J. Tomeĉek. 2023. "Behavior of Wild Pigs toward Conspecific Carcasses: Implications for Disease Transmission in a Hot, Semiarid Climate." *Transboundary and Emerging Diseases* 4195199: 1–10.

Leslie, D. M., Jr. 2016. *An International Borderland of Concern: Conservation of Biodiversity in the Lower Rio Grande Valley* (No. 2016-5078). U.S. Geological Survey. http://dx.doi.org/10.3133/sir20165078

Leus, K., and R. C. Lacy. 2009. "Genetic and Demographic Management of Conservation Breeding Programs Oriented towards Reintroduction." In *Iberian Lynx Ex Situ Conservation: An Interdisciplinary Approach*, edited by A. Vargas, C. Breitenmoser, and U. Breitenmoser, 56–71. Madrid: Fundación Biodiversidad.

Linnell, J. D. C., U. Breitenmoser, C. Breitenmoser-Wursten, J. Odden, and M. von Arx. 2009. "Recovery of Eurasian Lynx in Europe: What Part Has Reintroduction Played?" In *Reintroduction of Top-Order Predators*, edited by M. W. Hayward and M. J. Somers, 72–91. Chichester: Blackwell Publishing Ltd.

Lombardi, J., H. L. Perotto-Baldivieso, M. Sergeyev, A. M. Veals, L. Schofield, J. H. Young, and M. E. Tewes. 2021. "Landscape Structure of Woody Cover Patches for Endangered Ocelots in Southern Texas." *Remote Sensing* 13: 4001.

Lombardi, J. V., M. Sergeyev, M. E. Tewes, L. R. Schofield, and R. N. Wilkins. 2022. "Spatial Capture-Recapture and LiDAR-Derived Vegetation Metrics Reveal High Densities of Ocelots on Texas Ranchlands." *Frontiers of Conservation Science* 3: 1003044.

Lombardi, J. V., T. J. Yamashita, A. Blackburn, J. H. Young, Jr., M. E. Tewes, and C. J. Anderson. 2023. "Examining the Spatial Structure of Woody Cover within a Highway Road Effect Zone for Ocelots in Texas." *Urban Ecosystems* 26: 1057–69.

Lopez, G., M. Lopez-Parra, L. Fernandez, R. A. G. Garrote, M. Garcia-Tardio, T. Del Rey-Wamba, E. Rojas, et al. 2024. "Iberian Lynx Conservation Programme in the Last 22 Years." *Catnews Special Issue* 17: 18–23.

Lopez-Bao, J. V., A. Rodrigues, and F. Palomares. 2008. "Behavioral Response of a Trophic Specialist, the Iberian Lynx, to Supplementary Food: Patterns of Food Use and Implications for Conservation." *Biological Conservation* 141: 1857–67.

Maran, T., and Põdra, M. 2013. "Establishing New Populations of European Mink in Hiiumaa and Saaremaa Island, Estonia." In *Global Re-Introduction Perspectives: 2013. Further Case Studies from around the Globe*, edited by P. S. Soorae. Gland and Abu Dhabi: IUCN/SSC Re-Introduction Specialist Group and Environment Agency-Abu Dhabi. viii + 284 pp.

Martinez, L. 2024. "Martinez et al. Ocelot Reintroduction Model Outputs." 2024 Revision. Figshare. Dataset. https://doi.org/10.6084/m9.figshare.25521439.v1.

Martinez, L. A., J. V. Lombardi, G. Powers, A. D. Anderson, T. Campbell, and R. R. Lopez. 2024. "Assessing Ecological and Socio-Political Factors in Site Selection for Ocelot Reintroduction in Texas." *Conservation Science and Practice* 6: e13113.

Masters, B. 2022. "American Ocelot. [Film]." PBS Nature.

Matos, R. I. 2020. "Home Range and Habitat Use: Analysis on the State Endangered Bobcat (*Lynx rufus*) in Northwestern New Jersey." Master's thesis, Montclair State University.

Miller, B., K. Ralls, R. P. Reading, J. M. Scott, and J. Estes. 1999. "Biological and Technical Considerations of Carnivore Translocation: A Review." *Animal Conservation* 2: 59–68.

Miller, S. D., B. N. McLellan, and A. E. Derocher. 2013. "Conservation and Management of Large Carnivores in North America." *International Journal of Environmental Studies* 70: 383–398.

Miller-Butterworth, C. M., D. R. Diefenbach, J. E. Edson, L. A. Hansen, J. D. Jordan, T. M. Gingery, and A. L. Russell. 2021. "Demographic Changes and Loss of Genetic Diversity in Two

Insular Populations of Bobcats (*Lynx rufus*)." *Global Ecology and Conservation* 26: e04157.

Mueller, S. A., S. Prost, O. Anders, C. Breitenmoser-Würsten, O. Kleven, P. Klinga, M. Konec, et al. 2022. "Genome-Wide Diversity Loss in Reintroduced Eurasian Lynx Populations Urges Immediate Conservation Management." *Biological Conservation* 266: 109442.

National Centers for Environmental Information. 2022. "Historical Palmer Drought Indices." [Data Set]. National Oceanic and Atmospheric Administration. https://www.ncei.noaa.gov/access/monitoring/historical-palmers/.

Norwine, J., and J. Kuruvilla. 2007. *The Changing Climate of South Texas 1900–2100 – Problems and Prospects, Impacts and Implications*. Kingsville, TX: Crest-Resacca, Texas A&M University-Kingsville.

Ocelot Reintroduction Study Captive Propagation Team. 2023. "Manual for Ocelot (*Leopardus pardalis*) Breeding and Reintroduction for Recovery in the United States, Version 1." https://recovertexasocelots.org/manual-for-breeding-and-reintroduction.

Onorato, D., C. Belden, M. Cunningham, D. Land, R. McBride, M. Roelke, D. W. MacDonald, and A. J. Loveridge. 2010. "Long-Term Research on the Florida Panther (*Puma concolor coryi*): Historical Findings and Future Obstacles to Population Persistence." In *Biology and Conservation of Wild Felids*, edited by D. W. Macdonald and A. J. Loveridge, 453–469. Oxford: Oxford University Press.

Ripple, W. J., and R. L. Beschta. 2012. "Trophic Cascades in Yellowstone: The First 15 Years after Wolf Reintroduction." *Biological Conservation* 145: 205–213.

Ripple, W. J., J. A. Estes, R. L. Beschta, C. C. Wilmers, E. G. Ritchie, M. Hebblewhite, J. Berger, et al. 2014. "Status and Ecological Effects of the World's Largest Carnivores." *Science* 343: 1241484.

Rueda, C., J. Jiménez, M. J. Palacios, and A. Margalida. 2021. "Exploratory and Territorial Behavior in a Reintroduced Population of Iberian Lynx." *Scientific Reports* 11: 14148.

Sasmal, I., K. Honness, K. Bly, M. McCaffery, K. Kunkel, J. A. Jenks, and M. Phillips. 2015. "Release Method Evaluation for Swift Fox Reintroduction at Bad River Ranches in South Dakota." *Restoration Ecology* 23: 491–98.

Seager, S. W. J., and C. M. Demorest. 1978. "Reproduction of Captive Wild Carnivores." In *Zoo and Wild Animal Medicine*, edited by M. E. Fowler, 667–706. Philadelphia, PA: W.B. Saunders Company.

Seddon, P. J., D. P. Armstrong, and R. F. Maloney. 2007. "Developing the Science of Reintroduction Biology." *Conservation Biology* 21: 303–312.

Serfass, T. L., R. P. Brooks, L. M. Rymon, and O. E. Rhodes, Jr. 2003. "River Otters in Pennsylvania, USA: Lessons for Predator Reintroduction." In *Proceedings of the European Otter Conference "Return of the Otter in Europe – Where and How."* Journal of the International Otter Survival Fund, No. 2. Scotland: IOSF Broadford, Isle of Skye.

Sergeyev, M., D. A. Campbell, J. D. Holbrook, J. V. Lombardi, M. E. Tewes, and T. A. Campbell. 2023. "Selection in the Third Dimension: Using LiDAR Derived Canopy Metrics to Assess Individual and Population-Level Habitat Partitioning of Ocelots, Bobcats, and Coyotes." *Remote Sensing in Ecology and Conservation* 10: 264–278.

Sergeyev, M., J. D. Holbrook, J. V. Lombardi, M. E. Tewes, and T. A. Campbell. 2022. "Behaviorally Mediated Coexistence of Ocelots, Bobcats, and Coyotes Using Hidden Markov Models." *Oikos* 2023: e09480.

Sergeyev, M., E. P. Tanner, M. J. Cherry, J. V. Lombardi, M. E. Tewes, and T. A. Campbell. 2023. "Influence of Abiotic Factors on Habitat Selection of Sympatric Ocelots and Bobcats: Testing the Interactive Range Limit Theory." *Frontiers in Ecology and Evolution* 11: 1166184.

Serra, R., S. Carreira, J. Fernandez, J. A. Godoy, R. Gonzalez, I. Gutierrez, M. J. Perez, et al. 2024. "The Iberian Lynx Ex-Situ Conservation Programme: From Birth to Release." *Catnews Special Issue* 17: 24–28.

Simón, M. A., J. M. Gil-Sánchez, G. Ruiz, G. Garrote, E. B. McCain, L. Fernández, M. López-Parra, et al. 2012. "Reverse of the Decline of the Endangered Iberian Lynx." *Conservation Biology* 26: 731–36.

Soorae, P. S., ed. 2013. *Global Re-introduction Perspectives: 2013. Further Case Studies from around the Globe*. Gland, Abu Dhabi: IUCN/SSC Re-Introduction Specialist Group and Environment Agency-Abu Dhabi. xiv + 282 pp.

Stepkovitch, B., R. T. Kingsford, and K. E. Moseby. 2022. "A Comprehensive Review of Mammalian Carnivore Translocations." *Mammal Review* 52: 554–572.

Stier, A. C., J. F. Samhouri, M. Novak, K. N. Marshall, E. J. Ward, R. B. Holt, and P. S. Levin. 2016. "Ecosystem Context and Historical Contingency in Apex Predator Recoveries." *Science Advances* 2: e1501769.

Tetzlaff, S. J., J. H. Sperry, and B. A. DeGregorio. 2019. "Effects of Antipredator Training, Environmental Enrichment, and Soft Release on Wildlife Translocations: A Review and Meta-Analysis." *Biological Conservation* 236: 324–331.

Thomas, S., V. van der Merwe, W. D. Carvalho, C. H. Adania, R. Černe, T. Gomeričić, M. Krofel, et al. 2023. "Evaluating the Performance of Conservation Translocations in Large Carnivores across the World." *Biological Conservation* 279: 109909.

Translocation Team, A Subcommittee of the Ocelot Recovery Team. 2009. *Plan for Translocation of Northern Ocelot (Leopardus pardalis albescens) in Texas and Tamaulipas*. Albuquerque, NM: U.S. Fish and Wildlife Service Southwest Region.

U.S. Fish and Wildlife Service. 2016. *Recovery Plan for the Ocelot (Leopardus pardalis)*. Albuquerque, NM: U.S. Fish and Wildlife Service Southwest Region.

Veals, A. M., J. D. Holbrook, A. Blackburn, C. J. Anderson, R. W. DeYoung, T. A. Campbell, J. H. Young, Jr., and M. E. Tewes. 2022. "Multiscale Habitat Relationships of a Habitat Specialist over Time: The Case of Ocelots in Texas from 1982 to 2017." *Ecosphere* 13: e4204.

Veals, A. M., J. D. Holbrook, M. J. Cherry, T. A. Campbell, J. H. Young, Jr., and M. E. Tewes. 2023. "Landscape Connectivity for an Endangered Carnivore: Habitat Conservation and Road Mitigation for Ocelots in the US." *Landscape Ecology* 38: 363–381.

Watts, G. W. 2015. "Bobcat Response to Drought in a Semi-Arid Ecosystem and the Influence of Microhabitat Structure on Detection." Master's thesis, Texas A&M University-Kingsville.

Wilson, S. M. 2018. "Lessons Learned from Past Reintroduction and Translocation Efforts with an Emphasis on Carnivores." LifeLynx. https://www.lifelynx.eu/wp-content/uploads/2018/10/Lessons-Carnivore-Reintroduction-Efforts-Final-Version-4.0-2018.pdf.

Wolf, C., and W. J. Ripple. 2016. "Prey Depletion as a Threat to the World's Large Carnivores." *Royal Society of Open Science* 3: 160252.

Wolf, C., and W. J. Ripple. 2018. "Rewilding the World's Large Carnivores." *Royal Society of Open Science* 5: 172235.

**How to cite this article:** Martinez, Lindsay A., Jason V. Lombardi, Israel D. Parker, Forrest East, Tyler A. Campbell, and Roel R. Lopez. 2024. "Long Live the Cat: Ocelot Population Viability in a Planned Reintroduced Population in Texas, USA." *Ecosphere* 15(11): e70044. https://doi.org/10.1002/ecs2.70044