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Factors affecting settling, survival, and viability of black bears reintroduced to Felsenthal National Wildlife Refuge, Arkansas

Brandon J. Wear, Rick Eastridge, and Joseph D. Clark

Abstract We used radiotelemetry and population modeling techniques to examine factors related to population establishment of black bears (*Ursus americanus*) reintroduced to Felsenthal National Wildlife Refuge (NWR), Arkansas. Our objectives were to determine whether settling (i.e., establishment of a home range at or near the release site), survival, recruitment, and population viability were related to age class of reintroduced bears, presence of cubs, time since release, or number of translocated animals. We removed 23 adult female black bears with 56 cubs from their winter dens at White River NWR and transported them 160 km to man-made den structures at Felsenthal NWR during spring 2000–2002. Total movement and average circuity of adult females decreased from 1 month, 6 months, and 1 year post-emergence ($F_{2,14}$ =19.7, P<0.001 and $F_{2,14}$ =5.76, P=0.015, respectively). Mean first-year post-release survival of adult female bears was 0.624 (SE=0.110, $SE_{interannual}$= 0.144), and the survival rate of their cubs was 0.750 (SE=0.088, $SE_{interannual}$=0.109). The homing rate (i.e., the proportion of bears that returned to White River NWR) was 13%. Annual survival for female bears that remained at the release site and survived >1-year post-release increased to 0.909 (SE=0.097, $SE_{interannual}$=0.067; Z=3.5, P<0.001). Based on stochastic population growth simulations, the average annual growth rate (λ) was 1.093 (SD=0.053) and the probability of extinction with no additional stockings ranged from 0.56–1.30%. The bear population at Felsenthal NWR is at or above the number after which extinction risk declines dramatically, although additional releases of bears could significantly decrease time to population reestablishment. Poaching accounted for at least 3 of the 8 adult mortalities that we documented; illegal kills could be a significant impediment to population re-establishment at Felsenthal NWR should poaching rates escalate.

Key words black bear, Felsenthal, poaching, population model, reintroduction, *Ursus americanus*, White River

The Interior Highlands, Gulf Coastal Plain, and Delta regions of Arkansas once supported large numbers of black bears (*Ursus americanus*; Holder 1951). Unregulated market hunting coupled with extensive habitat loss led to the extirpation of bears throughout most of the state by the early 1900s (Smith et al. 1991). One small population of bears persisted and has since recovered in what is now White River National Wildlife Refuge (NWR), located in the Delta region (Holder 1951, Smith 1985, Smith et al. 1991, White 1996).

The Gulf Coastal Plain of Arkansas was not inhab-

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ited by a reproducing population of bears, although occasional sightings of solitary animals were reported. Bear populations in Arkansas and elsewhere in the Southeast often exist in small isolated fragments of wooded habitat (Pelton 1991) and may be more vulnerable to extinction because of demographic or stochastic events (Levins 1970, Griffith et al. 1989, Hanski 1996). Bears may not be well adapted to colonize fragmented habitats by natural dispersal (van Manen 1991, Clark et al. 2002). Although the considerable dispersal capabilities of male black bears have been well documented (Kemp 1976, Young and Ruff 1982, Schwartz and Franzmann 1992), female black bears typically do not disperse, residing instead within a portion of their mother's home range (Kemp 1976, Alt 1978, Rogers 1987a, Schwartz and Franzmann 1992). In addition, bears have relatively low reproductive rates (Bunnell and Tait 1981), a characteristic that limits population growth and natural colonization ability (Hanski 1991, Hastings 1991). Thus, reintroduction may be necessary to facilitate the recolonization process (Clark et al. 2002).

Homing or philopatry (i.e., return to place of origin) in black bears following translocation is common (Rogers 1973, Beeman and Pelton 1976, Rogers 1987b), and bears suffer increased associated mortality (Rogers 1986, Fies et al. 1987, Stiver 1991, Comly 1993, Riley et al. 1994, Eastridge and Clark 2001). Thus, homing is a major obstacle to black bear reintroduction success (Eastridge and Clark 2001, Clark et al. 2002). Older bears, females, the presence of cubs, abundant food, large translocation distances, and physiographic barriers (e.g., rivers, mountain ranges) are thought to reduce homing propensity (Beeman and Pelton 1976, Singer and Bratton 1980, McArthur 1981, Rogers 1986, Fies et al. 1987, Clark et al. 2002). A winter-den technique was evaluated in Tennessee, whereby denning females with neonates were removed from their dens and transported to dens at the reintroduction site (Eastridge and Clark 2001). That technique was shown to reduce total distance moved from the release sites, net distance moved, mean daily distance moved, circuity, and mortality rates (Eastridge and Clark 2001).

Although occasional sightings of solitary animals had been reported in the Gulf Coastal Plain of Arkansas, sightings of females and cubs had not been documented. Consequently, the Arkansas Game and Fish commission (AGFC) and the United States Fish and Wildlife Service (USFW) proposed a plan to translocate bears to Felsenthal NWR in the Gulf Coastal Plain from White River NWR, where similar habitat and flooding conditions occurred (Smith 1985, White 1996, Oli et al. 1997). Our objectives were to 1) estimate survival, recruitment, and homing rates of translocated female black bears and evaluate relationships between those parameters and age class, presence of cubs, and time elapsed following translocation and 2) evaluate relationships between population viability and number of bears translocated, time following translocation, and the manner in which variance estimates were applied in stochastic simulations of extinction risk.

Study area

White River NWR was in the Lower Mississippi Riverine Forest Province (Bailey 1995) and encompassed portions of Arkansas, Desha, Monroe, and Phillips counties in eastern Arkansas (Figure 1). This 65,000-ha refuge varied from 5–16 km in width and contained a 145-km portion of the White River. Flooding inundated approximately 75% of White River NWR annually, most often during winter and spring.

Felsenthal NWR was in the Southern Mixed Forest Province (Bailey 1995) and encompassed portions of Ashley, Bradley, and Union counties in southeastern Arkansas (Figure 1). Felsenthal NWR was about 26,000 ha in size, with about 4,000 ha comprised of upland forest communities and >16,000 ha comprised of bottomland hardwoods. Felsenthal Lock and Dam, approximately 5 km north of Louisiana, formed the 6,000-ha Felsenthal Pool, which increased to >14,500 ha during winter flooding.

Vegetation on both refuges occurred along a continuum of decreasing flood tolerance from lowest to highest elevations (Fredrickson and Heitmeyer 1988). Bottomlands contained willow oak (*Quercus. phellos*), water oak (*Q. nigra*), Nuttall oak (*Q. nuttallii*), sweetgum (*Liquidambar styraciflua*), and sugarberry (*Celtis laevigata*). Overcup oak (*Q. lyrata*) and bald cypress (*Taxodium distichum*) occurred on poorly drained sites, especially backwater areas, oxbows, and depressions. The upland sites in and adjacent to Felsenthal NWR were dominated by loblolly pine (*Pinus taeda*). White River NWR was surrounded by agriculture.

Climate was similar on the 2 refuges, with abundant rainfall throughout the year. Mean annual pre-

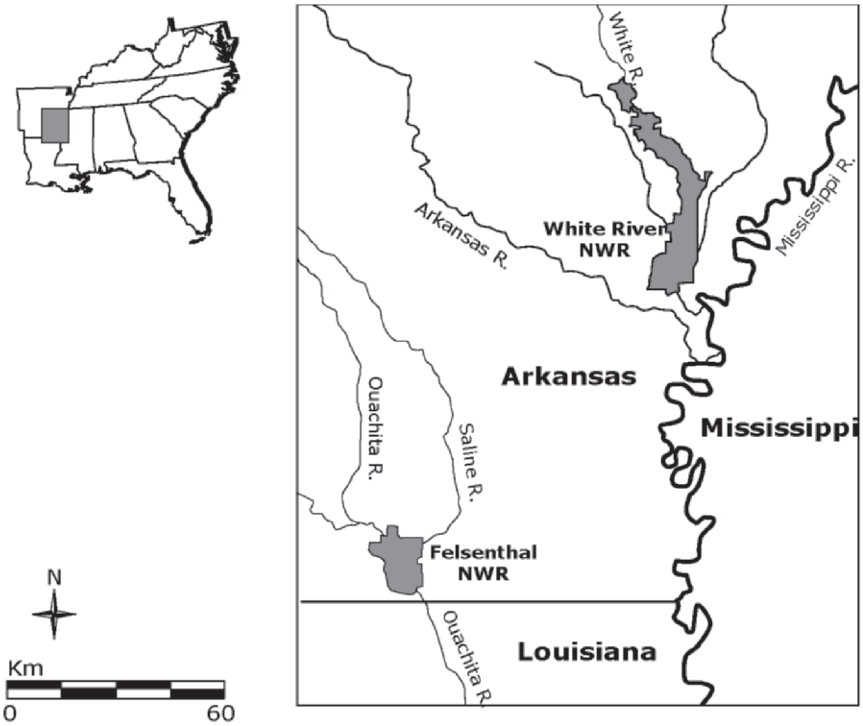

Figure 1. Location of White River National Wildlife Refuge and Felsenthal National Wildlife Refuge, Arkansas.

cipitation at White River and Felsenthal was 131 and 143 cm, respectively (National Oceanic and Atmospheric Administration [NOAA] 2000). Winter and spring were the wettest seasons, with March precipitation on both areas averaging about 14 cm.

Methods

Capture and handling

We captured bears at White River NWR and at nearby Montgomery Island and Big Island during summer 1998–2001 with Aldrich spring-activated foot snares (Aldrich Animal Trap Company, Clallam Bay, Wash.). We equipped snares with swivels and automobile hood springs to minimize injuries to captured animals (Johnson and Pelton 1980). We immobilized captured bears with a 2:1 mixture of ketamine hydrochloride (Ketaset®, Fort Dodge Animal Health, Fort Dodge, Ia.) and xylazine hydrochloride (Rompun®, Bayer Corporation, Shawnee Mission, Kan.). We administered the immobilization drug intramuscularly with a push pole at a dosage of 4.4 mg of ketamine hydrochloride and 2.2 mg of xylazine hydrochloride per kg of estimated body mass. We monitored body temperature, pulse, and respiration throughout each immobilization.

We classified adult female bears that showed vaginal swelling or a pinkish vaginal discharge as being in estrus (Wathen 1983). We considered bears in estrus or not lactating during summer trapping to be prospective candidates for winter translocation and fitted these bears with MOD-500 radiocollars (Telonics Incorporated, Mesa, Ariz.). We extracted the first upper premolar from each bear for aging by cementum annuli analysis (Willey 1974). At the conclusion of handling each bear, we administered yohimbine hydrochloride (Lloyd Laboratories, Shenandoah, Ia.), an antagonist to xylazine hydrochloride, through the sublingual vein at a dosage of 0.2 mg/kg of body mass. We conducted all procedures according to University of Tennessee Animal Care Protocol #906.

Translocation

We tracked females trapped and radiocollared the previous summer to their winter dens during February and March, 2000–2002, to document presence of cubs. Because >90% of black bears at White River NWR used elevated tree cavities for winter dens (Oli et al. 1997), accessibility of denned females was determined in large part by the overall stability of the den tree, dimensions of den entrance and cavity, height of opening, and distance to den floor.

Once translocation candidates had been identified, we returned to the den site in March to early April, ascended the den tree, and immobilized the female using either push pole, blowgun, or dart rifle. We removed the cubs and secured and removed the adult female in a cargo net or safety harness. We applied mentholated salve to the bodies of cubs and to the nose of the adult female to help prevent cub abandonment (Eastridge 2000) and intramuscularly injected the adult female with oxytocin to promote lactation. Finally, we placed the cubs with the female in an aluminum cage, and transported the family group by vehicle to Felsenthal NWR.

We normally held bears overnight at Felsenthal NWR and reimmobilized them the following day. We placed the female and cubs in wooden den boxes (1.2 m L × 0.9 m W × 0.9 m H, with a 56-cm-diameter circular opening at the front) that we constructed and placed in various locations at the release site. We established those den sites in areas above the 100-year flood stage in elevation and isolated from areas of high public use.

Radiotelemetry

We radiomonitored recently translocated bears each day to determine the date of den emergence, as evidenced by increased movements. For the first month following den emergence, we radiolocated translocated bears daily by ground or aerial telemetry. We gradually reduced the frequency to weekly radiolocations for each bear.

We performed ground telemetry with a model TR-4 receiver (Telonics Incorporated, Mesa, Ariz.) and a 5-element Yagi antenna (Wildlife Materials Inc., Carbondale, Ill.) using triangulation and the loudest signal method (Springer 1979). For each location we obtained 3 azimuths that formed angles between 30° and 150°, collecting those azimuths within a time interval of <50 minutes. If triangles formed by the 3 azimuths were >2 ha in size, we collected additional azimuths. Those procedures helped identify spurious azimuths and significant animal movement while azimuths were being collected (Schmutz and White 1990). We obtained most radiolocations between 0800 and 2000 hours.

We obtained aerial locations from fixed-wing aircraft with an H-antenna (Telonics Incorporated, Mesa, Ariz.) attached to each wing strut and connected with coaxial cable to a switch box and telemetry receiver inside the cabin. We obtained locations by flying the aircraft toward the loudest signal. When the aircraft was directly over the bear, we recorded the position with a Global Positioning System.

We estimated the accuracy of our bear radiolocations in 2000 and 2001 by placing radiocollars throughout the study area in locations similar to actual bear locations. We estimated the location of each test collar with the same procedures described above for ground and aerial telemetry. We then calculated distance from the actual location to the estimated location to obtain an error distribution (Schmutz and White 1990, Zimmerman and Powell 1995).

Movements

Translocations could result in mortality, homing, or settling; we used several parameters to quantify and categorize these types of movements. Mean daily movement, a measure of rate of travel, was calculated by dividing total movement by number of days the bear required to move that distance. We determined net movement, a measure of release-site fidelity or settling, by calculating the straight-line distance between the starting point and the ending point. We determined circuity, a measure of directed movement, by dividing net movement by total movement. For example, a circuity value of 0 indicated the animal returned to its starting point or never left it (i.e., net movement = 0), whereas a value of 1 indicated the animal moved directly away from its starting point along a linear pathway (i.e., net movement = total movement; Eastridge and Clark 2001). We calculated total movement and mean daily movement with the Animal Movement extension (Hooge and Eichenlaub 1997) in ArcView® (Environmental Systems Research Institute, Incorporated, Redlands, Calif.). To characterize movement behavior subsequent to release, we compared circuity, total movement, net movement, and daily movement by time after den emergence (i.e., 1 month, 6 months, and 1 year) with analysis of variance, only using bears for which an entire year of telemetry data had been collected. We used log transformations of total movement, net movement, and daily movement to approximate normality. We estimated mean bearings of bear movements subsequent to release and calculated Raleigh's z-values to determine whether those bearings differed from random (White and Garrott 1990, Hooge and Eichenlaub 1997), to evaluate whether bears tended to return in the direction of origin (i.e., White River NWR). We used linear regression to assess relationships between movement characteristics and bear age to determine whether older bears were less likely to exhibit homing behavior.

Survival

We used the Kaplan-Meier staggered-entry procedure (Pollock et al. 1989) to estimate annual survival of adult females during the period <1 year after release, and survival of those adult females that survived >1-year post-release. Bears that lost radiocollars, bears whose signals were lost before the completion of the study, and bears that permanently left the area were censored at that time (Pollock et al. 1989). We compared survival over the 3 years of study with a log-rank test using a corrected Bonferonni critical value ($P = 0.05/3$) and compared survival <1 year post-release with survival >1-year post release with a Z-test (Pollock et al. 1989). We estimated homing rates by re-estimating first-year post-release survival but treating surviving bears that permanently left the study area as mortalities. We then subtracted that rate from the survival rate previously estimated to determine the homing rate.

We visually observed bear family groups at regular intervals to assess survival of translocated cubs and litters. We performed cub counts at 2 and 4 months post-release, when cubs were approximately 4 and 6 months of age, respectively. We considered sightings reported by the public to be valid counts if an individual female bear and her litter were sighted on multiple occasions, the same number of cubs was reported each time, and locations were consistent with telemetry data.

Population growth and viability

We used a population model (RISKMAN, version 1.9.000; Ontario Ministry of Natural Resources, Toronto, On.) to estimate population growth (λ) and probability of extinction. RISKMAN employed a Monte Carlo approach to estimate the uncertainty of population trajectories given estimates of population size and demographic parameters (Taylor et al. 2001). The model was based on estimates of cub survival, litter survival, subadult male and female survival, adult male and female survival, litter production rate, and the probability of producing 1-, 2-, 3-, or 4-cub litters. Litter survival was the probability that $\geq$1 cub in a litter survived. Litter production rate was the probability that unencumbered females (i.e., without the previous year's cubs) would produce a litter. Taylor et al. (1987, 2001) suggested using interannual variation of rate parameters in RISKMAN rather than pooled variation; therefore, we used interannual standard errors ($SE_{interannual}$) for vital rate parameters for our stochastic simulations. We estimated male survival rates, litter production rates, and litter size probabilities from data collected at White River NWR (R. Eastridge, AGFC, unpublished data). Male survival at White River was estimated with Cormack-Jolly-Seber techniques (Burnham et al. 1987, Lebreton et al. 1992), and recruitment statistics were based on visual cub counts (R. Eastridge, AGFC, unpublished data).

Estimates of variance of population parameters typically are a product of both parameter uncertainty and environmental variability (White 2000). RISKMAN allowed the partitioning of that variance into parameter and environmental components. Parameter uncertainty is mimicked in RISKMAN by selecting parameter values from a user-defined distribution and applying those values at the beginning of each simulation trial. Environmental variation is mimicked by reselecting parameter values for each year of the simulation trial. We partitioned variance at 75% for parameter uncertainty and 25% for environmental variability (75:25) and the reverse (25:75) to evaluate sensitivity of model outcomes to different variance proportionments (Taylor et al. 2001, 2002). We used the covariance option in RISKMAN to simulate non-independence of parameter variances because environmental variation likely would affect both survival and reproductive rates of all age classes, and the covariance option would allow for this and other correlated effects in the stochastic trials. We did not include density-dependent effects in the simulations.

We performed 5,000 stochastic simulations for an initial population size of 28 (the estimated 2003 population of adults and cubs with no additional stockings). We used the estimated standing age distribution as the starting condition to estimate λ over a 10-year period. We estimated the proportion of trials in which the population declined to extinction during a 50-year period. We then performed 1,000 stochastic simulations to estimate extinction rates given different initial population sizes, with stable age distribution starting conditions.

Results

Translocation

We translocated 23 adult female black bears with a cumulative total of 56 cubs from White River NWR to Felsenthal NWR (approximately 160 km). We translocated 6 adult females and 15 cubs, 4 adult females and 10 cubs, and 13 adult females and 31 cubs to Felsenthal NWR from 25–29 March 2000, 27 February–13 March 2001, and 12 March–4 April 2002, respectively. Other than cubs, no male bears were translocated.

Movements

We obtained 75 locations of test radiocollars from June 2000 through August 2001. Proportions of test locations obtained by aerial and ground telemetry approximated the proportion of bear locations collected by each of those methods during the study. The mean distance from the estimated location to the actual location was 109 m for aerial and ground telemetry combined; 95% of the estimated locations were within 324 m of the actual location.

Total movement of translocated adult female bears during the first month after den emergence averaged 42.7 km (SE=14.1) and ranged from 2–251 km. Daily movements, net movements, and circuity for the first month averaged 923 m (SE=324), 26.6 km (SE=9.4), and 0.47 (SE=0.06), respectively.

Total movement of surviving adult females 6 months after emergence averaged 222.6 km (SE=49.5) and ranged from 52–747 km. Mean daily movement, net movement, and circuity the first 6 months averaged 1,132 m (SE=253), 34.2 km (SE=7.7), and 0.14 (SE=0.03), respectively.

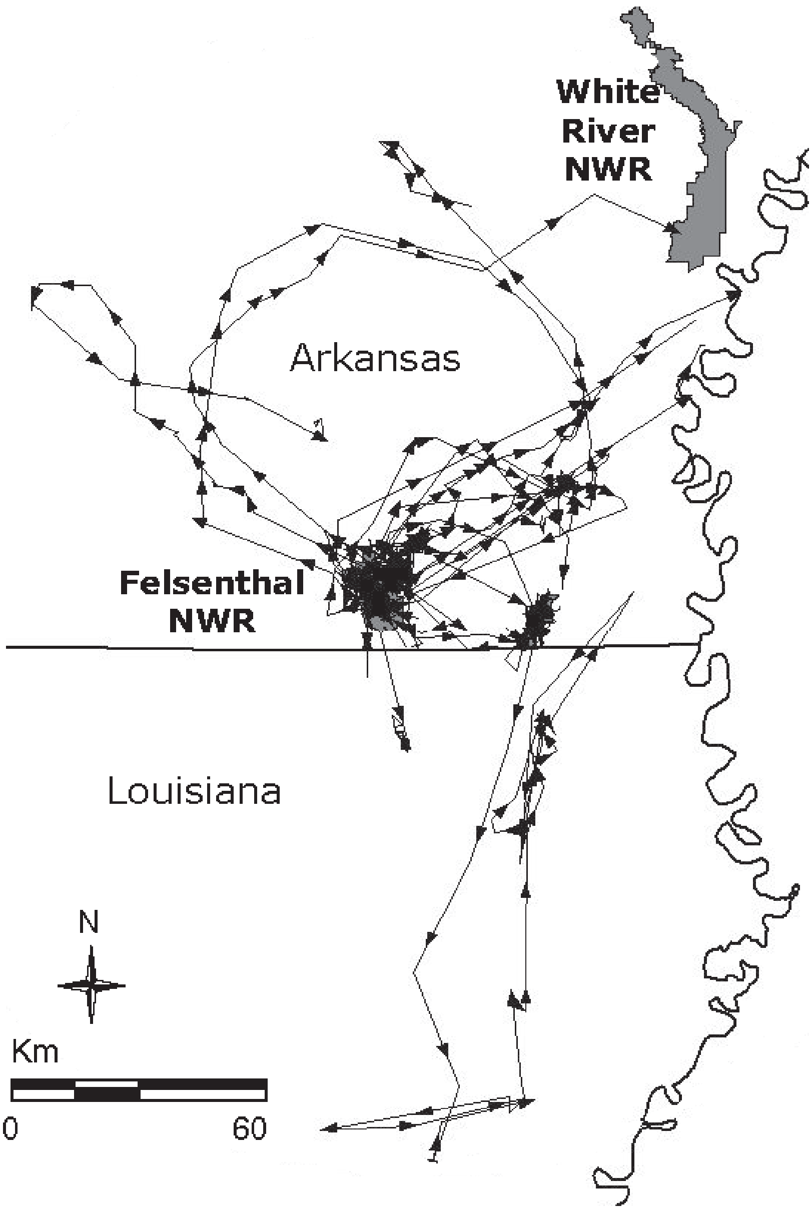

Figure 2. Post-release movements of black bears reintroduced to Felsenthal NWR, Arkansas, 2000–2002.

Total movement of adult females that survived to 1 year after emergence averaged 311.2 km (SE=116.3) and ranged from 110–750 km. Mean daily movement, net movement, and circuity after 1 year averaged 782 m (SE=291), 33.1 km (SE=13.1), and 0.09 (SE=0.04), respectively. Total movements and circuity differed by time since emergence ($F_{2,14}=19.7$, $P<0.001$ and $F_{2,14}=5.76$, $P=0.015$, respectively), but net movements ($F_{2,14}=1.60$, $P=0.236$) and daily movements ($F_{2,14}=3.24$, $P=0.070$) did not. Only 4 of 20 radiocollared bears had mean bearings that differed from random (Raleigh's $z>2.04$; $P<0.05$). Of those 4, however, mean bearings were in the direction of White River NWR (about 40°), ranging from 20.2–68.2° (Figure 2). Circuity and net movement after 6 months were less for older bears ($r^2=0.423$, $P=0.012$ and $r^2=0.423$, $P=0.012$, respectively). We detected no other age-movement relationships ($P\geq0.104$).

Survival

We estimated survival of 23 adult female bears monitored between March 2000 and January 2003. Three of 8 mortalities were the result of poaching, and poaching was the suspected cause of an addi-

Table 1. Survival rates (*S*) of bears reintroduced to Felsenthal National Wildlife Refuge, Arkansas, 2000–2003.

	Cub survival			Litter survival			Adult female (<1 year after release)			Adult female (>1 year after release)		
Year	Deaths	*S*	SE	Deaths	*S*	SE	Deaths	*S*	SE	Deaths	*S*	SE
2000	3	0.625	0.171	0	1.000	0.000	1	0.833	0.170	0	1.000	0.000
2001	0	1.000	0.000	1	0.500	0.354	2	0.333	0.193	1	0.800	0.207
2002	3	0.786	0.110	0	1.000	0.000	4	0.595	0.155	0	1.000	0.000
Overall	6	0.750	0.088	1	0.900	0.090	7	0.624	0.110	1	0.909	0.097
$SE_{interannual}$			0.109			0.167			0.144			0.067

tional death. One bear was killed due to a vehicle collision, and cervisitis contributed to the death of another. We were unable to determine causes of the remaining 2 mortalities.

First-year post-release survival of adult female bears translocated in 2000, 2001, and 2002 was 0.833 (SE=0.170), 0.333 (SE=0.193), and 0.595 (SE=0.155), respectively (Table 1), but differences by year were not detected ($\chi^2_{0.015}\leq 1.857$, 1 df, $P\geq 0.173$). Overall, first year survival was 0.624 (SE=0.110, $SE_{interannual}$=0.144). Annual second-year survival for all female bears that survived >1-year post-release was 0.909 (SE=0.097, $SE_{interannual}$=0.067), which was higher than survival during the first year after release (Z=3.50, $P<0.001$). Likewise, >1-year post-release survival did not differ by year ($\chi^2_{0.015}\leq 0.60$, 1 df, $P\geq$ 0.438). Cub survival was 0.750 (SE=0.088, $SE_{interannual}$=0.109), and litter survival was 0.900 (SE=0.090, $SE_{interannual}$ = 0.167); neither differed by year ($\chi^2_{0.015}\leq 1.61$, 1 df, $P\geq 0.205$ and $\chi^2_{0.015}\leq 2.0$, 1 df, $P\geq$ 0.157, respectively). When we excluded bears that successfully returned to White River NWR (n=4) in survival rate estimates, the overall survival rate declined to 0.492 (SE=0.101; i.e., 49.2% of the bears reintroduced to Felsenthal NWR survived and remained there or elsewhere in the Gulf Coastal Plain after 1 year). Conversely, 13.3% ($\bar{x}$ = 0.133, SE = 0.140) of the reintroduced bears returned to White River NWR.

Population growth and viability

Based on the 2003 standing age distribution and population parameter estimates (Table 2), our simulations resulted in a mean population size after 10 years of 74.3 (SD=39.5; 75:25 variance proportionment) bears at Felsenthal NWR, with a geometric mean λ of 1.093 (SD=0.053). Variance was high; lower and upper 95th percentiles were 18 and 171, respectively. The median population size after 10 years (69) was lower than the mean because upper estimates of population size inflated the mean but had less effect on the median. Given the standing age distribution, the probability of declining to extinction during a 50-year period ranged from 0.56–1.30%, depending on variance proportionment (25:75 and 75:25, respectively; Figure 3). Extinction was possible throughout the 50-year

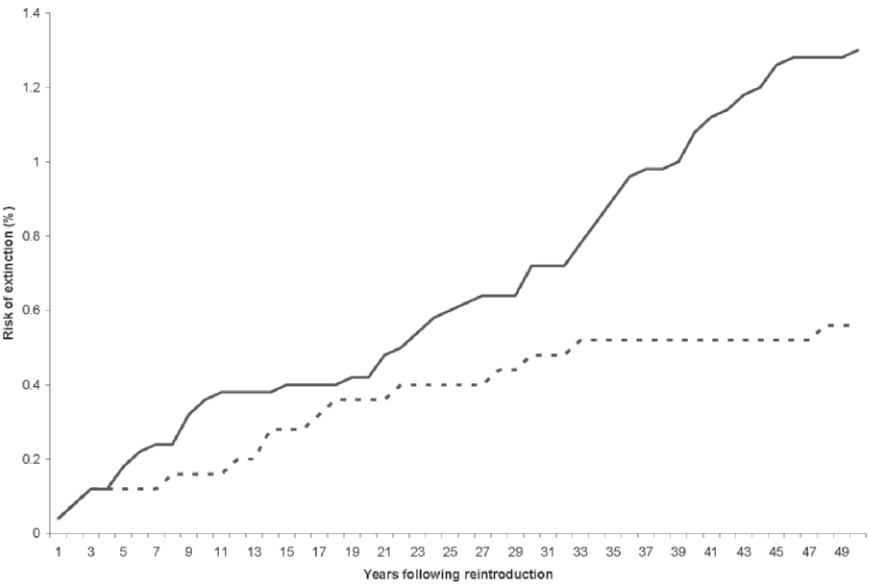

Figure 3. Probabilities of extinction for bears reintroduced to Felsenthal NWR, Arkansas, given a starting population size of 28 and standing age distribution. The dark solid line represents a variance proportionment of 75% attributable to the parameter estimate and 25% to environmental variation (75:25), and the dotted line represents a 25:75 proportionment.

Table 2. Black bear population parameter estimates used for population modeling, Felsenthal National Wildlife Refuge, Arkansas, 2003.

Parameter	$\bar{x}$	$SE_{interannual}$
Cub-of-the-year (COY) survival[a]	0.750	0.109
Litter COY survival[a]	0.900	0.167
Subadult (1–3) survival (M) [b]	0.690	0.150
Subadult (1–3) survival (F) [a]	0.909	0.067
Adult (4+) survival (M)[b]	0.690	0.150
Adult (4+) survival (F)[a]	0.909	0.067
Litter production rate[b]	0.733	0.019
Probability of COY litter = 1[b]	0.416	
Probability of COY litter = 2[b]	0.324	
Probability of COY litter = 3[b]	0.232	
Probability of COY litter = 4[b]	0.028	
Mean litter size[b]	1.872	0.096

[a] Based on data from this study.

[b] Based on data from bears at White River NWR (R. Eastridge, AGFC, unpublished data).

period for the 75:25 variance proportionment but declined after about 30 years for the 25:75 proportionment.

The extinction probability was successively less with greater starting population sizes, but the severity of that effect declined after an initial population size of about 20 was reached (Figure 4). Estimates of extinction risk with 25% of the variance attributable to parameter uncertainty and 75% to environmental variation were lower than the reverse proportionment (Figure 4), but the relationship with initial population sizes was similar.

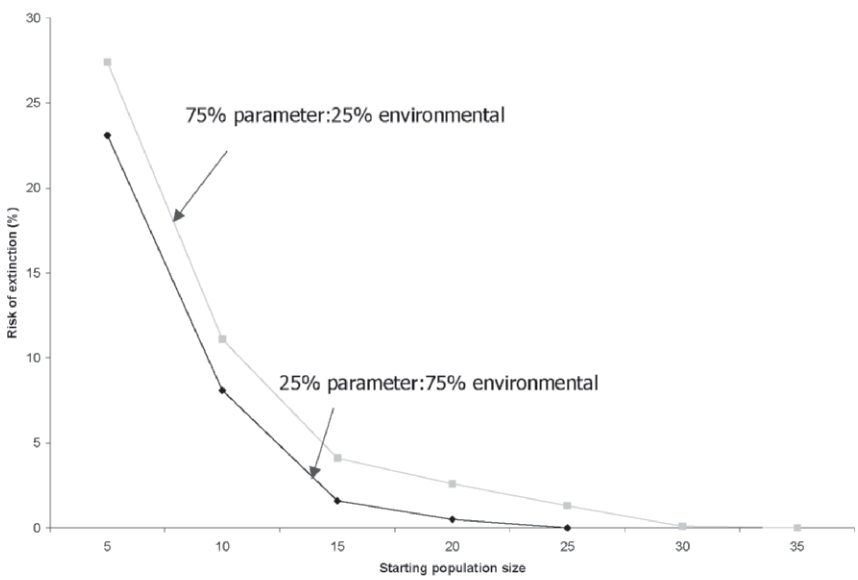

Figure 4. Probabilities of extinction for bears reintroduced to Felsenthal NWR, Arkansas, given different starting population sizes (stable age distributions) and a variance proportionment of 75% attributable to the parameter estimate and 25% to environmental variation (light line), and 25% attributable to the parameter estimate and 75% to environmental variation (dark line).

Discussion

Time since reintroduction was an important aspect of bear movement dynamics. Generally, bears that returned home or exhibited homing behavior did so within the first month following den emergence. Philopatric females traveled quickly and in a directed path, not appearing to be constrained by landscape attributes or land cover types. In contrast, bears that extensively moved but did not return home tended to follow rivers, creeks, or bayous. Non-homing bears generally reduced mean daily movements during the first month. Reintroduced bears in Tennessee exhibited similar restricted movements after 9 months, with most restricting their movements within 6 months after translocation (Eastridge 2000).

The presence of cubs also was an important factor in reintroduction success. In all but 2 instances, females that attempted to return home during our study had no cubs when observed. Furthermore, when the 2 females with surviving cubs that left Felsenthal NWR were recaptured and translocated back to Felsenthal, they eventually established home ranges there. In previous studies of bears translocated in the Southeast, females with cubs also exhibited reduced post-release movements and demonstrated greater affinity to the reintroduction site than unencumbered females (Comly 1993, Eastridge and Clark 2001).

Mean first-year survival of adult females translocated to Felsenthal NWR (0.624) was lower than that of winter-released bears in Tennessee (0.88; Eastridge and Clark 2001) but was greater than the survival rates of hard-released (i.e., nondenning bears with no acclimation period at the release site) black bears in Wisconsin

Most female bears at White River National Wildlife Refuge denned in large trees. Photo by Clint Turnage.

(0.56, Massopust and Anderson 1984) or Virginia (0.37, Comly 1993). First-year post-release survival of female bears translocated in 2001 (0.333) was noticeably lower than first-year survival estimates from 2000 (0.833) and 2002 (0.595), even though statistical differences were not detected. In 2001 we translocated bears earlier in the year because extensive winter flooding at White River NWR forced many females from their winter dens. The translocated bears had younger, possibly less-resilient cubs. Those cubs suffered high mortality rates after translocation, which may have resulted in greater post-release movements and increased mortality among adult females.

In all instances ($n=5$), number of yearlings with adult females in winter dens corresponded with number of cubs observed during cub counts the previous autumn. Our visual assessments were an effective alternative to affixing radiotransmitters to young cubs (LeCount 1987, Echols et al. 1999). We were able to identify a period of time when cubs suffered higher mortality, although the cause of death could not be determined. All cubs we observed ≥4 months after release (6 months of age) survived to yearling age. That finding was similar to previously reported black bear cub survivorship (Erickson 1959, Rogers 1977, LeCount 1987).

Although we did not have sufficient data to estimate the proportion of variance in model inputs attributable to parameter uncertainty versus environmental variation, our absolute estimates of extinction were relatively insensitive to the variance proportionment in RISKMAN. However, the pattern of extinction varied with variance proportionment; extinctions were possible throughout the 50-year period, with the 75:25 proportionment compared with most extinctions occurring within 30 years with the 25:75 proportionment. That was because the 75:25 simulations were largely determined by the initial randomized starting values and, consequently, extinction probabilities did not decrease as the population size increased over time, as were the 25:75 simulations. Regardless of variance proportionment, extinction probabilities decreased most rapidly when we increased the starting population size in our simulations from 5 to 20 adults with cubs and was less affected by starting populations >20.

We documented 1 instance when females bred and produced cubs at Felsenthal NWR, out of 3 possible occasions in 2002 and 1 of 1 in 2003 ($\bar{x}=$ 0.500, $SE_{interannual}=0.333$). Those females were present at Felsenthal NWR and were not encumbered with cubs during the previous breeding seasons; thus, breeding necessarily occurred at Felsenthal. We documented 2 unmarked males on Felsenthal NWR during our study, and we speculate that nontranslocated males sired those cubs. One male, incidentally captured by a landowner, was too old (>3) to have been reintroduced as a cub. We captured another male while attempting to snare and recollar a translocated female. Microsatellite DNA analysis indicated that this young male, though not one of the cubs translocated to Felsenthal, was closely related (T. King, United States Geological Survey, personal communication) and was probably a natural immigrant from the White River NWR population. We did not document any non-reintroduced adult females at

Felsenthal NWR. The low litter production rate was not unexpected, given the low numbers of breeding-age males present at Felsenthal. When we modeled extinction risk with the 0.50 estimate of litter production during the first 10 years of population reestablishment, the risk of extinction during that period only slightly increased (0.16 to 0.38).

We based our population projections on the assumption that habitat conditions would result in constant survival and production rates over time. We did not model the potential for declining habitat quantity and quality in the region in the future. Likewise, we did not model density effects, which could alter vital statistics as the population approaches carrying capacity. Our estimates of risk, therefore, are useful for comparing various management scenarios and less useful as absolute measures of risk of population decline below specific thresholds.

Management implications

The den boxes proved to be an effective alternative to locating natural den sites at the reintroduction area. Females with cubs generally remained in the den boxes for the few weeks prior to the normal time of den emergence. We were limited by the availability of dry sites for den-box placement at Felsenthal NWR, but we plan to test den boxes placed on trees in inundated areas.

Numerous studies have emphasized the importance of releasing an adequate number of founding individuals to ensure population establishment (Griffith et al. 1989, Smith and Clark 1994, Saltz 1995, Wolf et al. 1996, Pelton and van Manen 1997), but there is a point of diminishing returns at which additional stockings become less effective (Eastridge and Clark 2001). It appears that the greatest effect of increased stockings of bears at Felsenthal NWR has already been achieved. However, time to achieve population reestablishment goals could be accelerated with additional stockings, largely due to the relatively low population growth rate.

At least 3 of 8 adult female mortalities were the result of poaching. We consider this level of poaching to be a potential obstacle for bear population reestablishment at Felsenthal NWR. Although >70% of local residents surveyed prior to this study supported bear reintroduction (R. Eastridge, AGFC, unpublished data), even a minimal number of opponents could have a dramatic impact on the success of the project if poaching escalates. Education programs should strive to inform a generally supportive public that the actions of a few threaten the bear reintroduction program.

Finally, extensive post-release movements of translocated bears should be expected and explained to the public. The release site should be viewed only as a starting point, with settling anywhere in the region accepted as a criterion for successful reintroduction. In our study the local news media tended to emphasize the nonsuccessful releases (i.e., mortality, homing), thus detracting from the overall success of the project.

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