Demographic Models and Reality in Reintroductions: Persian Fallow Deer in Israel

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Abstract: *Because most reintroduced species are rare, data on their dynamics are scarce. Consequently, reintroduction programs often rely on data from other species or captive populations to project the performance of the reintroduced population in the wild. We compared the reproductive success and survival of a Persian fallow deer (*Dama mesopotamica*) population reintroduced in Israel over the first 5 years of the project with the survival and reproduction parameters estimated while planning the reintroduction. In addition, we compared the actual growth of the wild population with the growth originally projected by a computer model in the original reintroduction program. We monitored 74 radio-collared individuals (57 females and 17 males) released semiannually 1996–2001. Survival during the first year after release was lower than later years (0.90 and 0.82 versus 0.95 and 0.88, for females and males, respectively). Such an impact was not anticipated in the original plan, but overall survival was higher than originally projected. As assumed in the reintroduction program, reproductive success improved significantly with time since release and overall, was higher than expected. The mean number of animals released annually was lower than planned. Overall, the growth of the reintroduced population was slower than projected, but the deviation was close to confidence limits and the pattern similar. After 5 years it appears that the original time frame of 8–10 years for project completion can be met or at worst will cause a 1-year delay. Over the short term of 5 years, projection models in reintroduction programs are useful tools for assessing the sustained use of the breeding core, depicting the dynamics of the population in the wild, providing a relatively accurate time frame for the successful completion of the project, and assessing project success.*

Key Words: *Dama mesopotamica*, radiotelemetry, reproductive success, survival rate

Modelos Demográficos y la Realidad en Reintroducciones: Ciervo Dama de Persia en Israel

Resumen: *Debido a que la mayoría de especies reintroducidas son raras, son escasos los datos sobre su dinámica. En consecuencia, los programas de reintroducción frecuentemente confían en datos de otras especies o poblaciones cautivas para proyectar el desempeño de la población reintroducida a su medio. Comparamos el éxito reproductivo y la supervivencia de una población de ciervo dama de Persia (*Dama mesopotamica*) reintroducida en Israel durante los primeros 5 años del proyecto con los parámetros de supervivencia y reproducción estimados durante la planificación de la reintroducción. Adicionalmente, comparamos el crecimiento de la población real con el crecimiento originalmente proyectado por un modelo de computadora en el proyecto de reintroducción original. Monitoreamos a 74 individuos con radio-collares (57 hembras, 17 machos) liberados semianualmente entre 1996 y 2001. La supervivencia durante el primer año después de la liberación fue menor a la de años posteriores (0.90 y 0.82 vs. 0.95 y 0.88, para hembras y machos, respectivamente). Tal impacto no fue anticipado en el plan original, pero la supervivencia total fue mayor a la proyectada originalmente. Como se asumió en el programa de reintroducción, el éxito reproductivo mejoró significativamente*

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con el tiempo posterior a la liberación y en general fue mayor a la esperada. El número promedio de animales liberados anualmente fue menor al planeado. En general, el crecimiento de la población reintroducida fue menor a la proyectada, pero la desviación fue cercana a los límites de confianza y el patrón fue similar. Después de 5 años, parece que se podrá cumplir el marco de tiempo original de 8-10 años para la finalización del proyecto, o en el peor de los casos habrá retraso de un año. En el corto plazo de 5 años, los modelos de proyección en programas de reintroducción son herramientas útiles para evaluar el uso sostenido del grupo reproductor, para esbozar la dinámica de la población en el medio natural, para proporcionar un marco de tiempo relativamente preciso para la finalización exitosa del proyecto y para evaluar el éxito del proyecto.

Palabras Clave: *Dama mesopotamica*, éxito reproductivo, tasa de supervivencia, radiotelemetría

Introduction

Our inability to project the outcome, and more specifically the actual performance, of reintroduced populations is due, in part, to limited knowledge (Caughley 1994). All factors that contribute to the success or failure of reintroductions are expressed through the dynamics of the population (survival, reproductive success, and dispersal). Most reintroduced species, however, are either endangered or even extinct in the wild and empirical data on their dynamics in the wild are absent. Furthermore, because in many cases reintroduced individuals are born in captivity, the release to an unfamiliar environment may affect their demographic performance. Specifically, reproductive success and survival rate may be reduced in the early years following release (Saltz & Rubenstein 1995; Novellie et al. 1996), at a time when their impacts on the population's viability are the most pronounced.

Our current knowledge on how the dynamics of reintroduced populations are affected by the procedure itself is also limited. In the absence of other options, we revert to other data sources, such as the studbook or data on the dynamics and behavior of similar species, as a basis for planning reintroductions and developing life-table-based models for projecting the reintroduced population's performance in the wild (Miller et al. 1994; Saltz & Rubenstein 1995; Cramer & Portier 2001). To date, though, no attempt has been made to evaluate the accuracy and therefore the utility of such projections. In this respect, postrelease monitoring becomes essential for assessing reintroduction methods and the validity of the original assumptions (Sarrazin & Barbault 1996).

We compared field data collected over 5 years on a Persian fallow deer (*Dama mesopotamica;* Randi et al. 2001) reintroduction in Israel with the population dynamics projected by a simulation model developed as part of the reintroduction program. The model and its output are described in detail in Saltz (1996) and in Saltz (1998).

The Persian fallow deer reintroduction in Israel was based on a long-term program of repeated releases from a permanent captive-breeding core population (Hai-Bar Carmel Nature Reserve, near the city of Haifa) that was established for the purpose of the reintroduction. The initial stock of deer was received in 1976, and by the end of 1995 the breeding core numbered more than 150 animals (Saltz 1998). Using a Leslie-matrix-based model incorporating demographic stochasticity, Saltz (1998) first determined the percentage of animals to be withdrawn from the breeding core for release based on a maximum sustained yield approach. Next, the performance of the population in the wild was simulated and the project duration was estimated.

The Persian fallow deer, previously abundant throughout western Asia, is currently listed as endangered (World Conservation Union [IUCN] 1996). It was extirpated from the Middle East during the nineteenth century as a result of hunting and habitat loss, and by the 1940s it was thought to be extinct throughout its range (Chapman & Chapman 1980). In 1956 two small, remnant populations were discovered in Iran. These populations served as the basis for the world captive herd (Jantschke 1991). A report from 1994 estimated the wild population in Iran at no more than 15 animals (Heidemann 1994).

When the reintroduction of the species in Israel was planned, there were no data on its ecology and specifically its dynamics in the wild. When designing the reintroduction program for the species in Israel, age-dependent reproductive success and survival rate values (i.e., a life table) were derived from the studbook with several modifications (Saltz 1996). Based on empirical data from the reintroduction of other species (Saltz & Rubenstein 1995; Novellie et al. 1996), Saltz (1998) assumed a postrelease effect in the form of reduced reproductive success during the animals' first years in the wild but assumed no such impact on survival rate. Survival of the older age groups in the wild (irrespective of the release), however, was considered to be lower than that of animals in captivity (Saltz 1996). Environmental stochasticity was not incorporated into these models because relevant data did not exist.

Postrelease monitoring of this reintroduced population examined the effects of the repeated-releases approach on space-use patterns (Dolev et al. 2002) and the behavioral adjustments of the reintroduced deer to their new environment (Perelberg et al. 2003). We examined the dynamics (reproductive success and survival rate) and growth of the reintroduced population over 5 years after the start of the reintroduction and tested how well they agree with the assumed dynamics and the growth projected in the original program (e.g., Saltz 1996; Saltz 1998).

Methods

The Reintroduction Procedure

The deer were reintroduced in the upper west Galilee region in northern Israel, in and around the Nahal Kziv Nature Reserve (35°23′S, 33°33′E). The reserve is a 15-km-long and 1-km-wide gorge along the Kziv Stream. The research area was defined according to the spatial distribution of the reintroduced population over time. As of August 2001, the borders of the distribution of the radio-collared individuals were 32°95′–33°05′N and 35°08′–35°19′E and the gorge and surrounding heights were included. A detailed description of the study area can be found in Dolev et al. (2002) and Perelberg et al. (2003).

The reintroduction project followed the IUCN guidelines for reintroduction (Kleiman 1989; IUCN 1996) and is based on a long-term approach developed by Saltz (1998). The program consisted of transfers from a permanent breeding core, Hai-Bar Carmel, to a habituation enclosure at the release site and “soft” releases into the wild. At the beginning of the project the herd in the breeding core stood at approximately 120 adult deer (62 adult females) and was considered large enough to support a reintroduction (Saltz 1996). Twice a year, during spring and autumn, about 12 deer were selected from the breeding core according to their age and sex (average of 6 males and 6 females each time; Table 1) and transferred to the habituation enclosure in Nahal Kziv Nature Reserve. The enclosure covered 11 ha with a natural habitat representative of the study area. In spring we attempted to transport only young (ages 1–2) females that were assumed not to be pregnant. In fall we transported females that were >4 years old.

After approximately 3 months in the habituation enclosure, the deer were released to the wild. Release was carried out by enticing the animals, with bait and water, into a fenced paddock along the outer fence of the enclosure. Once several animals were sighted in the paddock, the gates to the enclosure were closed, a section of the outer fence was removed, and the animals exited of their own accord. Using this technique, releasing all animals in the enclosure took from a few days up to more than a month.

We monitored introduced animals for 5 years after first release with radio telemetry, video equipment, and direct observations (Dolev et al. 2002; Perelberg et al. 2003). Radio collars were equipped with mortality sensors (Telonics, Mesa, Arizona) and were color-coded to enable individual identification by direct observations after the radio failed. Females from the first two releases were fitted with a unit weighing 1 kg (5-year life expectancy) and thereafter with 0.5-kg units (2- to 3-year life expectancy). Males were fitted with 1-kg units. We used video cameras (Sentinel Surveillance System, Sandpiper Technologies, Manteca, California) to overcome the limitations of observations of deer in the dense Mediterranean woodland. The equipment was used in two seasons: winter (October 1999–April 2000) and fawning season (May–June 2001). The cameras and data-logger telemetry receivers (Lotek Engineering, Ontario, Canada) were located in areas with known deer activity.

Survival

We examined survivorship for radio-collared deer for the time their radio collars were transmitting. Mortalities were sorted by their cause (when it was identified) and by the individual’s age, sex, and time since release. We analyzed survivorship for females (sample size for males was

Table 1. The released deer during the first 5 years of the project.

	No. of released deer		*No. radio collared*		*No. of mortalities of radio-collared deer*[a]	
Release	*females*	*males*	*females*	*males*	*females*	*males*
1996 spring	5	6	5	2	0	1
1996 autumn	8	9	8	1	2	0
1997 spring	10	9	10	2	4[b]	1
1997 autumn	6	8	6	2	0	0
1998 spring	0	3	0	0	0	0
1998 autumn	6	6	6	2	2	1
1999 spring	5	8	5	4	0	1
1999 autumn	4	5	4	0	0	0
2000 spring	6	4	6	0	2	0
2000 autumn	7	9	7	4	0	0

[a] *Mortalities were examined for radio-collared deer only for the time their radio collars were transmitting.*
[b] *Four cases of predation on females by domestic dogs occurred during the first month after release near the habituation enclosure. These cases were not included in the calculation of the survival rate in the wild because we considered this a singular event.*

too small) with the "known-fates" module in the program MARK (White & Burnham 1999). We were specifically interested in determining whether there was a reintroduction effect. We divided the study period into 1-month occasions, 61 occasions in all. Because in some release events, animals exited the habituation enclosure over a period that covered more than 1 month, we could not separate the data out into groups according to the release. Thus, we set up an individual covariate that described the number of months since release in each occasion. We tested the effect of the reintroduction by assuming an effect up to 12 months and comparing it with survival beyond that time. We then compared the following among four models: (1) the null, constant survival; (2) time since release; (3) age; and (4) the combination of time since release and age, with the quasi-Akaike information criterion (QAIC; Burnham & Anderson 2002). We also compared the likelihood of the best model relative to the other three. In addition, we calculated annual survival rate for individuals (males and females) with radio collars that were transmitting as the number of individuals that survived throughout the year relative to the number of individuals known to be alive at the beginning of the year. The annual mean survival rate for the study period was calculated as the average annual mortality rate weighted according to the annual sample size.

Reproduction

Direct observations, video recordings, pellets, and tracks determined the presence of fawns. We summarized data separately for each year (from 1997 to 1999) from the beginning of one fawning season (April) to the next because in the European fallow deer (*Dama dama*), all female fawns and most male fawns remain with their mother throughout their first year (Chapman & Chapman 1997). Because wild-born fawns were not radio tagged, we could not estimate fawn survival and differentiate it from reproductive success. Our observations on reproduction, then, reflect a combination of reproduction and recruitment, which we term "reproductive success" in this paper.

To associate reproductive success with the time that mothers were in the wild, we used only direct sightings of identified mature females (at least 3 years old). If an adult female was sighted with a fawn at least once between two fawning seasons, we considered that it had reproduced. Sighted females that were never accompanied by a fawn were considered barren. We classified the observed reintroduced females according to age and time in the wild (in years; i.e., 1, 2, and 3 years after release) and used a logistic regression to assess the impact of these two factors on reproductive success.

By considering the number of females in the wild during the breeding season and accounting for the time each female was in the wild since her release, we calculated the expected annual reproductive success at the population level. We cross-validated this value with the ratio of all sightings of adult females accompanied by fawns in a specific year.

Finally, we assessed the total number of adult females in the wild at the end of each year as the number of females with active transmitters + (the number of females with failed collars × their expected survival rate) + the number of wild-born adult females based on the calculated reproductive success and survival rate. We then compared these values with those projected by the original simulation model used in the reintroduction program described in Saltz (1998).

Results

In 10 release cycles between 1996 and 2001, 124 deer were released, and 74 deer (57 females and 17 males) were radio collared. Of the 50 uncollared deer, 49 were males. The age of 54 of the 57 released females was known. The original plan (Saltz 1998) called for the release of young females (<6 years old). The age distribution of the released animals deviated from the original plan: 26 (50%) were 5 years old or younger, 20 (40%) were 6–10 years old, and 4 (10%) were 11 or older.

Survival

Most mortalities of radio-collared deer (12 out of 14) occurred during the first year after release (Fig. 1). Even with mortality in the first year after release included, however, the annual survival rate of males and females in the wild was high (0.88 and 0.95, respectively; weighted average) in comparison with the captive population (Saltz 1998).

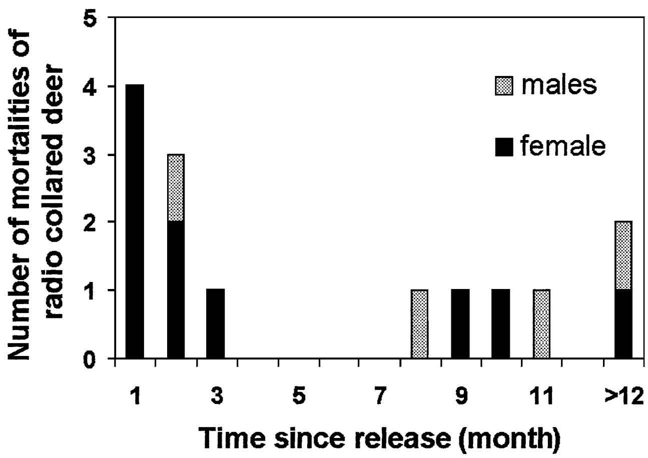

Figure 1. Number of mortalities in the wild of reintroduced Persian fallow deer with actively transmitting radio collars (17 males and 57 females) as a function of time since release.

Table 2. Comparison of the four models for survival of reintroduced Persian fallow deer based on MARK results: constant survival, a reintroduction effect, and an age effect, and their likelihood relative to the best model.

Model	*QAIC**	*No. of parameters*	*Deviance*	*Delta QAIC**	*Weight*	*Relative likelihood*
Constant survival	79.644	1	77.642	2.4516	0.189	0.29
Reintroduction effect	77.193	2	73.185	0.0000	0.644	1
Age effect	79.889	2	75.881	2.6963	0.167	0.26
Age × reintroduction	78.258	4	70.230	1.0648	0.274	0.59

**Quasi-likelihood Akaiki's information criteria recommended for count data (Burnham & Anderson 2002).*

Of the 17 radio-collared males, 4 died during the period in which the transmitter operated: a 5-year-old died as a result of being gored by an antler (2 months after release), a 6-year-old died of chronic disease (11 months after release), a 13-year-old was poached (3.5 years after release), and a 3-year-old died of unknown causes (8 months after release). Of the 57 radio-collared females, 6 died during the period in which the transmitter operated: an 11-year-old died of an infection (9 months after release), a 5-year-old died from a dislocated femur (2 months after release), and a 13-year-old was poached (2.5 years after release). The cause of mortality of the other females (ages 3, 6, and 6 years and 2, 3, and 10 months after release, respectively) was not determined. Domestic dogs killed 4, 1-year-old females in the first few weeks after they were released in the summer of 1997. These cases were not included in the calculation of the survival rate in the wild because we consider them a singular event. There are no data on other predation cases that occurred after release and all the other 11 females released at the age of 1 year survived to the following year.

Of the four models tested with MARK, the reintroduction-effect model was the most likely, with a relative likelihood 3 to 4 times greater than the null model or the age model (Table 2). The combined reintroduction effect × age model was a close second but had a likelihood of ~0.6 relative to the reintroduction effect alone. The MARK monthly survival estimates for the first 12 months increased from 0.983 to 0.998, whereas overall survival from the thirteenth month and after was 0.998. Thus, the 12-month cutoff was supported over a later cutoff. Female age at the time of release did not affect survival ($p = 0.26$) when tested with a Fisher's exact test (with females categorized according to the three age groups mentioned above), and female survival in the wild did not decline with age ($p > 0.6$, Fisher's exact test; Table 3).

During the study period, four males and one female dropped their collars and nine female transmitters failed (30–41 months after deployment). About half the females were observed after the transmitter failed. One female was found dead about 1 year after the collar failed. For the other individuals the survival beyond the transmitting period was unknown.

We obtained some anecdotal observations on fawn survival. During the study three dead fawns were found: a male fawn approximately 4–5 months old that apparently died of dehydration was found during the first fawning season, a young female was found dead in a water hole into which she presumably fell, and another fawn was killed by a predator.

Reproduction

Fawning season is between April and July and peaks during May. Fawns were observed in the wild since the first fawning season. In the first fawning season fawns were born to females that conceived in the breeding core and were transferred while they were pregnant. The full logistic regression of mother's age and time in the wild on reproductive success was not significant, but the reduced model of time in the wild on reproductive success was significant ($p = 0.0266$, scale $= 0.52$, logistic regression), with reproductive success increasing from 0.28 fawns/female/year for the females that were 1 year in the wild ($n = 18$) to 0.33 and 0.67 fawns/female/year for the females that were 2 ($n = 18$) and 3 years ($n = 3$) in the wild, respectively (Table 4). Fallow deer females may conceive at 16 months and then give birth when they are 2 years old (Chapman & Chapman 1997). The reproductive success of 2-year-olds, however, is low relative to prime-aged females (Saltz 1996). Because we could not have 2-year-olds that were more than a year in the wild (the youngest animals released were 1 year old), the effect of age and time in the wild, in this specific case, could not be separated. Consequently, we did not include the first year's data for 11 females released at the age of 1

Table 3. Annual survival rate of Persian fallow deer females in the wild.

Age (x)	*No. of females at age (x)*	*No. of females that survived to age (x + 1)*	*Survival rate**
1–2	19	19	1.00
3–9	51	48	0.94
10–13	12	11	0.92

**Survival rate was calculated by the total number of females that survived to age x + 1 out of the females at age x. Death cases that occurred under the effect of the reintroduction (during the transfer, habituation, and release) were not included in the calculation.*

Table 4. Reproductive success of Persian fallow deer in the wild in the first three fawning seasons as a function of time after release.

Time after release (years)	*No. of radio-collared females observed*	*Total no. of radio-collared females observed with fawns*	*Fawning season*	*Reproductive success*
up to 1	9	3	1997	
up to 1	2	0	1998	
up to 1	7	2	1999	
up to 1	18	5	total	0.28
1–2	5	2	1998	
1–2	1	0	1999	
1–2	6	2	total	0.33
2–3	3	2	1999	0.67

year. As expected, only 1 of these 11 females (0.09) was observed with a fawn at the age of 2 years. This value is lower than the already low reproductive success (0.5) recorded in the breeding core for this age group (Saltz 1998). It seems, then, that the reproductive success of young females and the survival of their fawns were low.

Factoring in the time that each female was in the wild, the expected reproductive success at the population level increased during the first 3 years of the reintroduction from 0.28–0.47 fawns/female/year (Table 5). This increase is expected because of the higher proportion of veteran females in the wild. The observed reproductive success (i.e., the proportion of females sighted with fawns out of all females sighted), however, did not fit this pattern, possibly because of differences in observability as a function of time since release.

Projected and Actual Growth

The actual growth of the adult female population (as calculated from the known number of surviving females, estimated reproductive success, and survival rate from the field data) followed the general growth pattern projected by the model but along the lower 2 SD boundary (Fig. 2). The sample size of 53 radio-collared females and the limitations of the monitoring period did not allow for the construction of a detailed life table similar to the one used in the original model presented by Saltz (1998). Overall, though, reproductive success in the wild was higher than originally projected by the model (Table 5). Similarly, excluding survival data from the first year after release, annual survival of females in the wild exceeded 95% for age groups 1 through 10, whereas in the model maximum annual survival rate for these age groups was 0.93. The reintroduced population did not reach the population size predicted because the original projection was based on annual releases of approximately 12–13 females (Saltz 1998), and 58 females were released over 5 years (e.g., 11.6 females/year). Furthermore, the average age of the released animals was older than planned. This produced a smaller population in the wild and resulted in a larger breeding core (Fig. 2). Rerunning the original model with an average of 11.6 females released every year increased the similarity to the field data.

Discussion

Although reintroductions are an important component of conservation biology (Beck et al. 1994; Wolf et al. 1996; Griffin et al. 2000), our ability to project their outcome and plan accordingly is still limited (Kleiman 1989; Griffith et al. 1989; Beck et al. 1994). One of the reasons is the limited knowledge we often have of the demography of target species—knowledge that is crucial for assessing reintroduction success (Sarrazin & Barbault 1996). Consequently, reintroductions are, rightly, perceived as risky.

The phenomenon of reduced reproductive success in the early years following the release of the deer relative to later years has been documented in other reintroduced species (Saltz & Rubinstein 1995; Novellie et al. 1996; Jiang et al. 2000). The actual impact of the release on reproductive success of the reintroduced Persian fallow

Table 5. Estimations of annual mean reproductive success of introduced Persian fallow deer.

Reproductive success	*Fawning season* 1997	1998	1999
Observed annual mean[a]	0.33 (3/9)	0.28 (2/7)	0.39 (4/11)
Expected annual mean based on the reproductive values in the wild[b]	0.29	0.39	0.47
Expected annual mean based on model estimations in the original reintroduction program[c]	0.15 (0.1)	0.23 (0.16)	0.32 (0.21)

[a] *Sightings of mature females accompanied by fawns divided by total observed mature females (number of sightings in parentheses).*
[b] *The values of reproductive success as a function of time after release in the wild (Table 4) multiplied by the relative fraction of individual females in the wild at each time step.*
[c] *The values of reproductive success as a function of time after release as predicted by the model (Salz 1998) multiplied by the relative fraction of individual females in the wild at each time step (model projections for recruitment at each year in parentheses).*

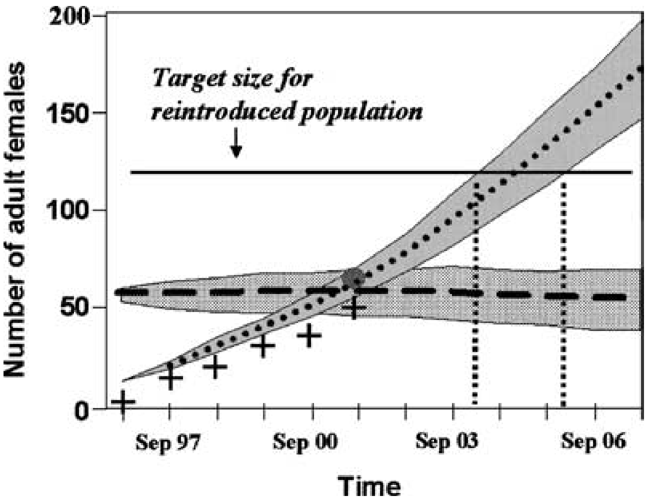

Figure 2. Projected and actual population growth of Persian fallow deer as a function of time from first release. Projected growth of females in the reintroduced population (dotted line) and breeding core (dashed line) of Persian fallow deer with an initial release of 13 adult females followed by annual releases of 12.9 ± 2.6 (mean ± SD) adult females from the breeding core. Shaded areas are ± 2 SD (Saltz 1998). The plus signs indicate the number of adult females in the wild, the filled circle is the number of adult females in the breeding core after 5 years, and the vertical dotted lines highlight the time range within which the target population is reached.

deer, however, was less than that assumed in the original projection model of Saltz (1998).

The reduced reproductive success in the years following release might be caused by a myriad of factors that are not mutually exclusive, including (1) stress induced by transfer and release protocols, unfamiliar area, and the breaking of social ties; (2) susceptibility to postnatal predation because of the mother's lack of experience; and (3) an Allee effect that results from low densities (Larkin et al. 2002), such as rapid dispersal of individuals to areas unoccupied by the other gender (Hartman 1995). In our study we ruled out the latter because telemetry data suggested that the males' range exceeded that of the females, and that males were present in locations where females were found. The individual reproductive success of the females increased significantly with time after release, as Saltz (1996) suggested. A longer monitoring period is needed to assess the duration of this "improvement" process. For reintroduced Père David's deer (*Elaphurus davidianus;* Jiang et al. 2000) and Asiatic wild ass (*Equus hemionus;* Saltz & Rubenstein 1995), it took 4 years for the release effect to fade.

The decline we detected in survival rate appeared to be restricted to the first year. Postrelease declines in survival were documented in other reintroductions and translocations of deer (O'Bryan & McCullough 1985; Jones & Witham 1990; Compton et al. 1995; Jones et al. 1997) and a wide range of other taxa (Sarrazin & Legendre 2000). This is most probably caused by the same factors listed for the decline in reproductive success. This possibility of reduced survival rate following release was not considered in the original plan (Saltz 1998) and was not included in the model used for the long-range planning of the reintroduction. Furthermore, the original plan did not foresee logistic and administrative problems occurring along the way, resulting in the release of fewer animals than the breeding core could actually supply and in an overall older age structure than intended.

In reintroductions, the age structure of the animals released is an important factor in the outcome (Sarrazin & Legendre 2000). This issue was evaluated in the planning of the Persian fallow deer reintroduction, with the release of age groups of 2–5 years found to be the most effective (Saltz 1996). This finding was then incorporated into the long-range program (Saltz 1998) but was not implemented.

On the other hand, overall survival rates in the wild and reproductive success following release were higher than the estimates used in the original model. By and large, however, reality did not differ considerably from the model's projection, and the main goals of the model have been successfully realized: (1) sustained use of the captive breeding core over the long term and maximization of the number of animals released; (2) general projection of the dynamics of the population in the wild; and (3) a relatively accurate ballpark time frame for the successful completion of the project (8–10 years), which at this point appears feasible. Success criteria for reintroduction are not yet clearly defined or widely accepted (Sarrazin & Barbault 1996). Several criteria have been offered for evaluating the success of a reintroduction—achieving a minimum viable population (Beck et al. 1994), the successful breeding of the first wild-born animal (Kleiman et al. 1991), a recruitment rate higher than adult death rate over 3 years (as cited in Cade & Temple 1995), and behavior patterns that are consistent with wild populations of the same or similar species (Perelberg et al. 2003). Predictive models can also be used to assess the success of reintroductions. Here, rather than looking at one specific parameter, the overall performance is evaluated and success is determined relative to predetermined criteria (i.e., the projected growth).

Conclusions

Demographic models are a useful tool for planning and evaluating the success of reintroduction programs. Because the reintroduction process has an effect on the survival and reproduction of the individuals, the demographic characteristics of reintroduced populations during the first few years of the reintroduction project cannot be considered normal and must be taken into account.

Models used for conservation should, as a rule, tend toward the conservative. This allows for errors such that, if reality deviates from expected, deviations would be positive or at least not devastating. The evaluation of the accuracy of a priori modeling of reintroduced population dynamics could be an opportunity to advocate for more feedback between fieldwork and modeling in a truly adaptive management approach.

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