Monitoring programs to assess reintroduction efforts: a critical component in recovery

E. Muths & V. Dreitz

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Abstract

Monitoring programs to assess reintroduction efforts: a critical component in recovery.— Reintroduction is a powerful tool in our conservation toolbox. However, the necessary follow–up, i.e. long–term monitoring, is not commonplace and if instituted may lack rigor. We contend that valid monitoring is possible, even with sparse data. We present a means to monitor based on demographic data and a projection model using the Wyoming toad (*Bufo baxteri*) as an example. Using an iterative process, existing data is built upon gradually such that demographic estimates and subsequent inferences increase in reliability. Reintroduction and defensible monitoring may become increasingly relevant as the outlook for amphibians, especially in tropical regions, continues to deteriorate and emergency collection, captive breeding, and reintroduction become necessary. Rigorous use of appropriate modeling and an adaptive approach can validate the use of reintroduction and substantially increase its value to recovery programs.

Key words**:** Reintroduction, Monitoring, Adaptive processes, Amphibians, *Bufo baxteri.*

Resumen

Programas de seguimiento para evaluar los esfuerzos de reintroducción: un componente crítico en la recuperación.— La reintroducción es un utensilio muy potente en nuestra caja de herramientas conservacionista. No obstante, el seguimiento necesario, es decir, el seguimiento a largo plazo, no es un hecho común, y si se da, puede ser poco rigurosa. Sostenemos que el seguimiento válido es posible, incluso cuando los datos son escasos o están dispersos. Presentamos aquí un medio de seguimiento basado en datos demográficos y un modelo de proyección utilizando al sapo de Wyoming (*Bufo baxteri*) como ejemplo. Usando un proceso repetitivo, se trabajan gradualmente los datos existentes de tal forma que aumente la fiabilidad de las estimas demográficas y sus subsecuentes deducciones. La reintroducción y el seguimiento defendible pueden hacerse cada vez más importantes, dada la problemática de los anfibios, especialmente en las regiones tropicales, donde continua deteriorándose, y se hacen necesarias la captura y la cría en cautividad para la reintroducción posterior. Un uso riguroso de la construcción de modelos apropiada y un punto de vista adaptativo pueden hacer válido el uso de la reintroducción y aumentar sustancialmente su valor en los programas de recuperación.

Palabras clave: Reintroducción, Seguimiento, Procesos adaptativos, Anfibios, *Bufo baxteri.*

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Introduction

Species reintroductions have become an increasingly popular tool in wildlife management (Wolf et al., 1996; Stanley–Price, 1991; Griffith et al., 1989; Kleiman, 1989). Reintroductions were used initially to resolve human–animal conflicts, to augment game populations, and to establish populations of non–native species but as more species have become imperiled and required more intensive management, this tool has become an integral part of many endangered species programs (Fischer & Lindemayer, 2000; Griffith et al., 1989). As habitat fragmentation increases (Noss et al., 2006) and the effects of global climate change become more evident (e.g. Knowles et al., 2006), reintroductions are likely to become an increasingly important tool for maintenance of demographically and genetically viable populations (Bright & Morris, 1994; Griffith et al., 1989). This may be increasingly true for amphibian species given the current outlook especially in the tropics (e.g., Stuart et al., 2004).

Importantly, long–term monitoring, which is rarely implemented, is a necessary follow–up to such programs (Dodd, 2005). We contend that monitoring is possible, even with sparse data. Using an iterative process, a data–poor project can evolve, such that each iteration produces more reliable data. Rigorous use of sound field methods, appropriate modeling, parameter estimation, and an adaptive approach can validate the use of reintroduction and substantially increase its value to recovery programs.

Background

Reintroduction programs for threatened and endangered species have various goals, including augmentation of population numbers, introduction of satellite populations to reduce risk of species extirpation, movement from a negatively impacted site to a mitigation site, or repatriation following extirpation by anthropogenic or natural causes. The overarching goal is to have a self–sustaining population of the target species at the site in perpetuity. There are a number of terms used for the movement of animals (by humans) from one place to another including translocation, introduction, supplementation, relocation, repatriation, and reintroduction; we use reintroduction throughout in the broadest sense.

Some reintroduction programs have been successful, such as those for natterjack toad (*Bufo calamita*), black–footed ferret (*Mustela nigripes*) and peregrine falcon (*Falco peregrinus anatum*) (Denton et al., 1997; Stanley–Price, 1991; Cade & Weaver, 1983), but many reintroduction programs fail (Seigel & Dodd, 2002; Griffith et al., 1989). Reintroductions are fraught with challenge; reasons for failure vary and are attributable to a range of factors (Snyder et al., 1996; Short & Smith, 1994; Kleiman, 1989). Monitoring is often the most challenging portion of a reintroduction program because of the perceived costs, but it is arguably the most critical. Boersma, et al. (2001) state, "one cannot possibly know whether management is working and whether it needs to be adaptively altered unless its effects are monitored".

Gauging success

Based on the goal of a viable population after reintroduction (Caughley & Gunn, 1996), the success of a program should be measured not only by the successful release of individuals, but by the ability of those animals to reproduce successfully and create a self–sustaining population (Dodd, 2005). Monitoring efforts can provide an assessment of program efficacy (Semlitsch, 2002) as well as a feedback mechanism among all aspects of recovery (e.g. captive breeding, habitat restoration) in an adaptive management framework (Bar–David et al., 2005). In some cases gauging success must be done in small increments. Adequate data may not be available in the short term to evaluate the entire program or make completely informed decisions. In spite of this, an iterative, yet quantitative approach will yield a more reliable assessment of the program in the long run.

Monitoring–considerations

Factors that contribute to the success or failure of reintroductions are estimated through the dynamics of the population (e.g. reproduction, dispersal, survival) but these data usually do not exist (Bar–David et al., 2005). In many cases where reintroduction is considered it is nearly impossible to collect these data because the population of interest has very few adult animals left, is restricted to a single location, is infected by disease, or is otherwise compromised (Dodd, 2005). For example, long–term data from amphibian populations are rare (but see, e.g., Daszak et al., 2005, Whitfield et al., 2007) and amphibian species, about which very little is known, are being lost at an unprecedented rate.

In spite of these obstacles, simulations or traditional prospective power analyses can be conducted to produce a target sample size; that is, the number of reintroduced individuals needed to reliably estimate parameters of interest. Reasonable sample size targets can be based on an array of data: studies of natural populations of the species, empirical data on a similar species, biological insight from experts, or captive colonies. *A priori* sample size calculations are used in other types of biological studies (Eng, 2004), and should not be overlooked when implementing reintroduction programs. Traditional power analyses are often used to calculate sample sizes for experiments, but efforts to relocate are seldom purely experimental and changes in study design can invalidate power analyses (Eng, 2004). One alternative to power analyses is simulations.

Another critical issue in monitoring is spatial variation and detectability (Pollock et al., 2002).

For example, few, if any, species are so conspicuous that they are always detected during field surveys when present (MacKenzie et al., 2004). Some monitored reintroductions of birds and mammals include an estimation of detection rate (e.g., Ostermann et al., 2001; Bar–David et al., 2005), but we know of no monitored reintroduction programs for amphibians that estimate detection rate or attempt to remove the effects of incomplete detectability. Assuming that count data represents population size in order to extract information on other demographic parameters such as survival and reproduction can lead to erroneous conclusions (Williams et al., 2002). Given these concerns, more rigorous attention to the adequacy and appropriateness of monitoring and sufficient documentation of the process is necessary (Mazerolle, 2006; Maunder, 1992; Oldham et al., 1991).

Material and methods

The Wyoming toad

The Wyoming toad was first recorded in Wyoming in 1946 as the Canadian toad, *Bufo hemiophrys* (Baxter, 1947). Porter (1968) recognized the Wyoming populations as a distinct subspecies, (*B. h. baxteri*), and Smith et al. (1998) elevated these populations to the species level as *B. baxteri*. From their discovery to about 1970, Wyoming toads were considered common and abundant within their restricted range (Baxter & Stone, 1985). Rapid declines in the 1970s presaged the extinction of Wyoming toads in the wild. The Wyoming toad was listed as an endangered species in 1984 (USFWS, 1984) and is suggested to be one of the most endangered amphibians in North America (Odum & Corn, 2005). The proximate cause of decline in Wyoming toads is likely infection by the fungus *Batrachochytrium dendrobatidis* (Bd) with the resulting chytridiomycosis causing unsustainable mortality of adult toads (Odum & Corn, 2005). Other factors, such as pesticides, predation, or habitat alteration, have been proposed to contribute to the decline of this species, but little evidence supports these hypotheses (Odum & Corn, 2005).

Currently, the Wyoming toad population is not self–sustaining and relies on annual supplementation with captive–reared animals (Odum & Corn, 2005). Between 1995 and 1999, over 9,500 Wyoming toads, mainly post–metamorphs (< 4 mos.) were reintroduced at Mortenson Lake National Wildlife Refuge (MLNWR, Albany County, Wyoming) where Wyoming toads were last known to occur in the wild (Odum &Corn, 2005). MLNWR is the site of recent reintroduction efforts (Jennings et al., 2001) and is described elsewhere (Parker & Anderson, 2003).

Except for photographic capture–recapture work from 1990 to 1992 (Odum & Corn, 2005) and the release and monitoring study in 2002 (this study), monitoring of reintroduction efforts are limited to visual encounter surveys (i.e., individual counts) during early spring and/or fall in a given year (Jennings et al., 2001). The individuals conducting the survey are mostly volunteers with varying experience in locating Wyoming toads. The bi–annual survey entails workers walking around the lakeshore in the putative preferred habitat (saturated soils) of Wyoming toads and counting the number of individuals encountered. These individual counts enumerate toads observed by life history stage; young–of–year, juveniles (1 yr old), and adults. Toads are not handled and no attempt is made to determine if a toad was previously counted during the survey (Dreitz, 2006).

Study design

The goal of this project was to determine whether or not a reintroduction and long–term monitoring program was feasible for the Wyoming toad. The project was financially constrained to a single field season. To address the goal, we needed to determine 1) the feasibility of releasing, recapturing and monitoring post–metamorphic toads and 2) the efficacy of sparse data in building a model that would yield useful information (e.g. how many individuals to release and survival estimates).

Captive propagation of Wyoming toads has been successful (Jennings et al., 2001) so that locating a source population was not an issue. *A priori* simulations were conducted using the robust design framework (Pollock, 1982) and information based on the biology of the Wyoming toad and other bufonids (e.g. Odum & Corn, 2005). We used a conservative scenario to set survival and capture probability.

Field sampling: marking and capture

All post–metamorphs released in 2002 were marked by clipping the 2nd digit on the left forefoot. Post–metamorphs were held in captivity at least one additional day after marking then released at MLNWR. Captive rearing facilities included Saratoga National Fish Hatchery, Wyoming Game and Fish Department's Sybille Wildlife Research Center and the Detroit Zoological Association. Post–metamorphic toads were staged and marked at the Saratoga Hatchery and the Sybille Research Center in Wyoming. The potential for disease was monitored at these facilities but individual animals were not tested prior to release. The release location was not tested for the presence of Bd because methods for testing water for this fungus were not yet available. We allowed at least one week acclimation period after release before initiating field sampling. An 82–section grid was established around Mortenson Lake. Each grid cell was approximately 25 m x 25 m, and extended from waters edge out towards upland habitat. Time–constrained (20 minute) visual encounter surveys (Crump & Scott, 1994) were conducted in every third cell around the lake (= 28 cells) by trained surveyors. All equipment, including waders, was disinfected with bleach daily.

The robust design (Pollock, 1982; Kendall et al., 1995, 1997) includes *k* primary sampling periods, each with l_i secondary sampling periods. Primary sampling periods are separated from each other by sufficient time to expect gains (birth and immigration) and losses (death and emigration), that is, the population is "open" to demographic and geographic changes. Further, each primary period includes l_i secondary periods separated from each other by sufficiently short time intervals for the population to be effectively "closed" to gains and losses (sensu Kendall et al., 1995). In our case, selected cells were sampled on 3 consecutive days (= secondary periods) in each of the summer months (June, July, and August = primary periods). Primary periods were approximately 4 weeks apart. For each survey occasion, a team of two observers was assigned to cells such that no team surveyed the same cells during the 3–day session. All toads observed were captured. At the conclusion of the 20 minute search, toads were inspected for marks. Additional toes were clipped to give each captured individual a unique number (Martof, 1953). Toads were released at the site of capture.

Analysis: robust design

We used the robust design to estimate apparent over–summer survival of post–metamorphic Wyoming toads. The robust design incorporates features of both the open and closed mark–release–recapture models (see above), with the major advantage of being able to estimate survival and population size in a single study. Information from secondary periods is used to estimate conditional capture (p_{ij}) and recapture (c_{ij}) probabilities and the number of animals in the population (N_i). Our ability to detect an individual was measured by capture and recapture probabilities. The pooled capture probabilities for each primary period are used to estimate apparent survival (the product of true survival and fidelity; [Φ_1,..., Φ_{1k-1}]). Recently metamorphosed individuals are unlikely to leave the sampling area until they hibernate for the winter (Odum & Corn, 2005). The assumptions of the robust design are summarized by Kendall et al. (1995) and are similar to assumptions of other capture–recapture models.

Over–summer survival (of released post–metamorphic Wyoming toads) rather than population size, was our primary objective. We used an extension of the robust design, the Huggins estimator, which removes the estimates of population size from the likelihood and allows capture and recapture probabilities to be modeled as functions of individual covariates (Huggins, 1991, 1989). Population size, if needed, can be derived.

Additional releases of captive bred post–metamorphs occurred between the primary periods. Our approach to modeling the demographic parameters followed Pollock et al. (1990), Lebreton et al. (1992), and Burnham & Anderson (2002). We first developed a list of covariates likely to influence one or more of the parameters, and developed a set of candidate models. We modeled over–summer survival as constant (Φ) or varying between the primary sampling periods (Φ_t). We assumed that there was no temporary emigration (i.e. $\gamma_i'' = \gamma_i' = 0$), and set initial capture probability equal to recapture probability ($p_{ij} = c_{ij}$, hereafter capture probability). We considered three different effects on capture probabilities: observers, micro–habitat within cells, and mean air temperature during secondary surveys compiled from data collected at the Laramie Regional Airport. The observers (*obs*) effect was based on probable variability in the abilities of survey teams to observe and capture post–metamorphs. The effect of cell in the survey grid (*cell*) was included because it is likely that the number of post–metamorphs in a cell varies due to micro–habitat differences among cells. The air temperature (*temp*) effect was based on amphibian physiology. We assumed that, to a point, post–metamorphs would be more active at warmer temperatures.

Model selection criteria and parameter estimation

Model selection and inference was based on information–theoretic methods using the small sample size correction to Akaike's Information Criterion, AIC_c (Hurvich & Tsai, 1989; Burnham & Anderson, 2002). We did not correct for extra binomial variation because there is currently no standard approach to estimate this in the robust design model (Williams et al., 2002). Once AIC_c values were computed for each model, we ranked the models based on the relative distances, $DAIC_c$, between the best approximating model and competing models. Normalized Akaike weights (w_i), which provide a strength of evidence for each model, were then computed (Burnham & Anderson, 2002). Instead of using parameter estimates from a single "best" model, we model averaged parameter estimates across all models (Burnham & Anderson, 2002).

Population projection model

The minimum number of animals to release to meet a recovery goal of a pre–defined number of breeding females is a common question for many recovery teams. To illustrate the potential of our approach in a reintroduction and monitoring program, we built a projection model based on a hypothetical target of 150 females. Using this value, the projection model provides the number of releases necessary over a 5 year period to reach that target. Projection models (e.g. Caswell, 2001) are flexible, such that a variety of parameters can be estimated or set to a target value.

The number of adult females at a given time *t*, N_{A_t}, is calculated as:

Table 1. Number of captures and air temperatures (from Laramie Regional Airport) during each secondary survey at MLNWR in 2002.

Tabla 1. Número de capturas y temperaturas del aire (del aeropuerto regional de Laramie) durante cada transecto secundario en el MLNWR en 2002.

	Primary periods					
	June	July	August	June	July	August
Secondary surveys	Captures			Air temperatures (ºC)		
Day 1	74	93	6	19.4	17.1	15.8
Day 2	44	62	6	19.9	19.8	15.4
Day 3	82	92	7	20.7	15.8	13.6

$$N_{A_t} = N_{A_{t-1}} S_A + (N_{J_{t-1}} \psi_{J \rightarrow A})\, r$$

where N_{J_t} is the number juveniles in the population at time *t;* S_A *is* the probability of an adult surviving from time *t* to *t*+1; $\psi_{J \rightarrow A}$ *is* the probability a juvenile becoming an adult from time *t* to *t*+1; and *r* is the sex ratio of males to females in the adult population.

And the number of juveniles in the population is:

$$N_{J_t} = N_{J_{t-1}} S_J (1 - \psi_{J \rightarrow A}) + N_A F_A S_E S_{Post} + I S_{Post}$$

where S_J is the probability of a juvenile surviving at time *t;* S_{Post} is the probability of a post–metamorph surviving; S_E is the probability of an egg surviving to metamorphosis; F_A is the fecundity of adult females in the population per year (defined as the number of reproducing females that a single female produces in one year); and *I* is the number of post–metamorphs released into the population per year.

We used an optimization routine to get the least–sum–of–squares estimate for *I*.

Any projection model needs information on population dynamics (i.e., survival, reproduction) and like many reintroduced species, information on the demography of Wyoming toads is limited (Jennings et al., 2001). We used values from a hypothetical life table (P. S. Corn, unpublished data) for our projection model including: S_A = 0.20, S_J = 0.57, S_E = 0.10, and *r* = 0.5. The values $\psi_{J \rightarrow A}$ = 0.19 and F_A = 2 were based on information from herpetologists who have worked on Wyoming toads over the last 20 years (P. S. Corn, E. Muths).

Results

Releases

Between June and August 2002, 8,124 post–metamorphic Wyoming toads were released with 74% released prior to the June sampling. We captured 459 post–metamorphs during field sampling; most captures occurred in July with the fewest in August (table 1). None of the captured animals showed signs of disease and none were found dead. Air temperatures ranged from 13.6 to 20.7ºC (18.4 ± 1.9ºC, mean ± SD), with June the warmest and August noticeably cooler (table 1).

Model results

The data were best explained by the model with constant over–summer survival and time–varying capture probabilities. Time variation in over–summer survival and capture probabilities was also a competitive model (table 2). The model–averaged estimate of the over–summer survival of post–metaphoric Wyoming toads was 0. 21 (table 3). The model–averaged estimate of the capture probabilities for the August primary period was low, 0.01, while June and July were somewhat higher, 0.09 and 0.07, respectively (table 3). Estimates of the number of post–metamorphs in the study area ranged from 594 to 1,304. Estimates for August were imprecise as a result of the low number of individuals captured.

Population projection model

The projection model predicted that a minimum of 5,000 post–metamorph releases each year are necessary to achieve our hypothetical goal of 150 adult females in the population after 5 years of releases.

Discussion

We determined that relocating post–metamorphic Wyoming toads is feasible. Our over–summer survival rate (0.21) was greater than our worst–case scenario expectation (0.10), but our capture rate (0.08) was substantially lower than our worst–case scenario expectation (0.15). While the capture probability during the last session was likely compro-

Table 2. Summary of model selection results for released post–metamorphic Wyoming toads at MLNWR in 2002 with models ranked by ascending $DAIC_{c.}$

Tabla 2. Resumen de los resultados de la selección de modelos para las sueltas post–metamórficas de los sapos de Wyoming en el MLNWR en 2002, con los modelos ordenados según una ΔCIA_c ascendente.

Model	Deviance	K	AIC_c	ΔAIC_c	w_i
$\Phi_. \gamma'' = \gamma'' = 0 \quad p_t = c_t$	1235.9364	9	1254.3311	0.0000	0.7038
$\Phi_t \gamma'' = \gamma'' = 0 \quad p_t = c_t$	1235.5861	10	1256.0696	1.7385	0.2951
$\Phi_t \gamma'' = \gamma'' = 0 \quad p_{temp*t} = c_{temp*t}$	1261.4109	3	1267.4628	13.1317	0.0010
$\Phi_. \gamma'' = \gamma'' = 0 \quad p_{temp*t} = c_{temp*t}$	1267.7440	2	1271.7699	17.4388	0.0001
$\Phi_. \gamma'' = \gamma'' = 0 \quad p_{obs+t} = c_{obs+t}$	1223.7014	25	1276.6560	22.3249	0.0000
$\Phi_t \gamma^{3''} = \gamma^{3''} = 0 \quad p_{obs+t} = c_{obs+t}$	1223.6667	26	1278.8649	24.5338	0.0000
$\Phi_t \gamma'' = \gamma'' = 0 \quad p_{cell*t} = c_{cell*t}$	1396.7434	10	1417.2269	162.8958	0.0000
$\Phi_. \gamma'' = \gamma'' = 0 \quad p_{cell*t} = c_{cell*t}$	1407.1175	9	1425.5122	171.1811	0.0000

Table 3. Modeled average results for over-summer survival and capture probabilities: Pm. Parameter; SE. Standard error; CI. 95% confidence interval.

Tabla 3. Resultados promedio modelados para la supervivencia pasado el verano y para las probabilidades de captura: Pm. Parámetro; SE. Error estándar; CI: Intérvalo de confianza del 95%.

Pm	Estimate	SE	Lower CI	Upper CI
Φ	0.2095	0.0884	0.0852	0.4302
$p_{1.} = c_{1.}$	0.0880	0.0246	0.0503	0.1495
$p_{2.} = c_{2.}$	0.0716	0.0183	0.0430	0.1168
$p_{3.} = c_{3.}$	0.0080	0.0091	–0.0098	0.0257

mised by cool weather, animals should have been larger and therefore easier to see. We do not expect metamorphic toads to emigrate at this time of the year (before hibernation, Parker & Anderson, 2003), therefore, the very low number of captures suggests high mortality between July and August. We cannot attribute mortality to Bd. There were no adult animals or carcasses from released animals to test for Bd. At the time of this study assays to test the environment (e.g. water, Kirshtein et al., 2007) were unavailable. Carey et al. (2006) report that duration of exposure and dosage influence survival in boreal toads (*Bufo boreas*) and predict that there is a threshold level of infection that must be reached to cause death. Since our released animals came from Bd–free facilities and there was minimal opportunity for contact with other amphibians, it is unlikely that the threshold levels of Bd, if it was present, were met, at least within the short time–frame of this study.

Capture probability is important as it is tied closely to the precision of the population size estimate (White et al., 1982). It is critical to increase capture rate by increasing the number of secondary periods and / or by increasing the number of primary periods (likely to increase precision). Effort (number of observers or search time per cell) could also be increased. Based only on technician costs, the cost of one season of monitoring was minimal. Technicians were a combination of students paid at an hourly wage, volunteers, and staff from various participating agencies. Depending on the source of technicians, increasing the number of secondary surveys should not be prohibitive.

Projection models can evaluate an array of parameters, with a great deal of flexibility in the equations. These models can assist in evaluating the overall performance of a population and, importantly, recovery program success relative to predetermined criteria (e.g. Caswell, 2001). Such models (i.e. Population viability analyses) have been applied to Wyoming toads (Program VORTEX, Jennings et al., 2001). Our projection model has the small advantage of using additional data (this study) that was unavailable when Program VORTEX was applied to the Wyoming toad data, and illustrates the incremental nature of collecting information on critically endangered species. The demographic estimates we used were the only ones available; they are prelimi-

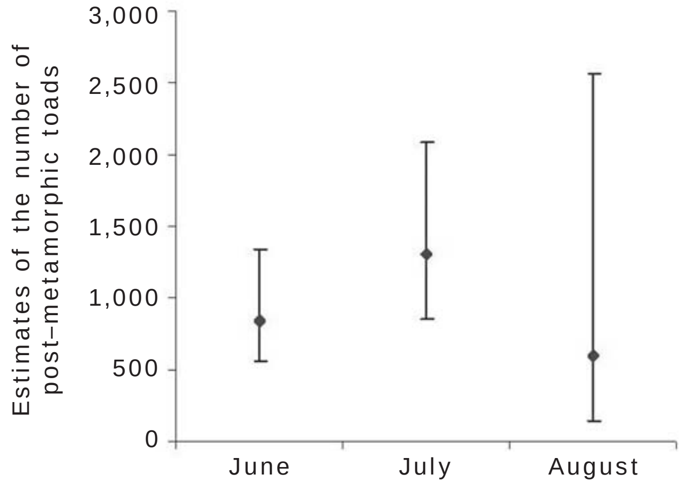

Fig. 1. Estimates of the number of post–metamorphic toads present at MLNWR.

Fig. 1. Estimas del número de sapos post–metamórficos presentes en el MLNWR.

nary at best, with some data based on estimates made when the population was likely stressed by disease. In addition, our simple projection model was based on an assumption of constancy over time due to the limited data available, which is most likely not the case in most amphibian reproductive scenarios. Stochatiscity and density dependence are important considerations that can be added to the model as more data accumulate. While the results of our projection model should be viewed with caution, they are based on biologically authentic information and illustrate the functionality of such a model in our iterative and adaptive framework.

There are a number of definitions of adaptive management. We use Salafsky et al. (2001) and Margoluis & Salafsky (1998) who define adaptive management as the incorporation of research into conservation action. We advocate such a process and submit that our preliminary monitoring lays the foundation for using such an approach on Wyoming toad reintroduction. Our estimates and projection model results are clearly the first iteration of what should be a long–term release and monitoring program. With each year, methods can be refined as the precision and accuracy of the data improve. For example, the over–summer survival rate can be used in the projection model to more reliably examine a suite of parameters that may be of interest to the project, specifically the number of animals to be released (as we calculated above), the number of adult females expected to survive and reproduce with a certain number of releases, or in sensitivity/elasticity analyses. Although our projections were based on over–summer survival rather than the more informative annual survival probability, it is still an improvement over guesses alone and, if the animals do not survive over–summer, it is clear that they will not survive until the next summer. As more data become available, a more detailed approach to adaptive resource management (e.g. Holling, 1978) could be applied where an explicit objective is defined, specific models are developed and assessed, and the results applied in determining the best conservation action to take.

Reintroduction is an important component of conservation biology (Wolf et al., 1996; Griffith et al., 1989) although our ability to project the outcome of reintroduction programs, and to plan accordingly, is still limited (Dodd, 2005; Kleiman, 1989). The point we make is not a new one: Reintroductions, to be of any long–term use, must be monitored. We have shown that rigorous monitoring is possible if defensible information is gathered, built upon, and used to monitor the release of post–metamorphic Wyoming toads. By using appropriate simulations for initial sample size decisions, modeling to estimate parameter values, an AIC–based decision criterion to evaluate competing models, and a projection model to provide information for the next iteration of releases and monitoring, the approach is straightforward and adaptive. Basing a program on defensible methods allows managers to respond relatively quickly to modeled data that provide valuable inferences about biological changes in the system.

Interestingly, more traditional metrics, such as indices that do not provide the opportunity to improve estimation efforts or to address changing circumstances, appear to be used more often in herpetology than for other taxa (Mazerolle, 2006, but see, for example; Scherer et al., 2005; Bailey et al., 2004a, 2004b). While our example is applicable to a broad range of taxa and endangered species programs, it may be especially pertinent to amphibians. The current outlook for amphibians, especially those in tropical regions, is grim (Mendelson et al., 2006; Stuart et al., 2004), and drastic measures, such as collecting the remaining animals from the wild and using captive breeding programs have been advocated (Mendelson & Rabb, 2005). If amphibian declines continue at their current alarming rate (e.g., Mendelson et al., 2006; Lannoo, 2005) and large scale "ark" projects (Mendelson & Rabb, 2005) are used, the implementation of reintroduction projects that are accountable and amenable to adaptation will be pivotal.

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