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Challenges of monitoring reintroduction outcomes: Insights from the conservation breeding program of an endangered turtle in Italy

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ABSTRACT

Captive breeding and reintroduction programs remain a powerful but divisive tool for management of threatened species, with a proven potential to avoid extinction, but low long-term success rates and high resource requirements. Monitoring the results of reintroductions is critical to be able to assess short- and long-term success, adjusting management decisions as new information becomes available. In this study, we assessed the first 15 years of the captive breeding and restocking program for the European pond turtle *Emys orbicularis* in Liguria, northern Italy. We estimated survival of released turtles by modelling mark-recapture monitoring data. We then used those estimates to update our prior expectations about long-term outcomes, and to adjust management decisions about the age of individuals to release. Modelling results suggest released turtles had sufficiently high survival, matching prior expectations, such that local extinction has been averted in the short-term. Survival was similar among candidate age classes for releases, suggesting the release of younger individuals can provide positive outcomes while reducing management costs. On the other hand, survival varied among sites, indicating the need for ongoing in-situ habitat management to ensure long-term persistence. Moreover, the late onset of sexual maturity in the species means reproduction of released animals cannot yet be determined with certainty. Captive breeding and reintroduction programs normally require long-term efforts; therefore, focused monitoring that is clearly linked to decision-making is necessary to continually refine and adjust management strategies.

1. Introduction

The conservation of endangered species requires diverse and often complex conservation strategies of variable intensity (Byers et al., 2013). Captive breeding programs for reintroduction and reinforcement of populations (hereafter "conservation breeding") are at the most management-intensive end of this spectrum. Since the first modern conservation breeding programs started in the late 1980s (see review in Seddon et al., 2007), reviews have highlighted low overall success rates (Dodd and Seigel, 1991; Fischer and Lindenmayer, 2000; Griffith et al., 1989; Wolf et al., 1996), compounded by a general difficulty in assessing long-term success (at least partly due to often inadequate monitoring; Ewen and Armstrong, 2007). To date, the few reviews of conservation breeding programs for reptiles have highlighted patterns similar to those of other taxa, where the potential for avoiding extinction in the short term is often challenged by unclear or negative outcomes in the long term (Dodd and Seigel, 1991; Ettling and Schmidt, 2015; Ewen et al., 2014; Germano and Bishop, 2009).

At the population level, the aim of reintroductions is to ensure population establishment and persistence, which in turn are ultimately determined by the vital rates (survival and fecundity) of released and wild individuals (Armstrong and Seddon, 2008). Those vital rates may be influenced by management decisions such as the sites and methods of release (such as "soft" or "hard" releases: Batson et al., 2015) and individual traits at release, such as age or body condition (e.g. Bremner-Harrison et al., 2004; Hardman and Moro, 2006). However, managers often have little information about whether and how their decisions affect vital rates and ultimately success. Monitoring of reintroduction outcomes generally aims to provide such information, in the expectation this will reduce uncertainty, facilitate decisions and improve management outcomes (e.g. Armstrong and Ewen, 2002; Bertolero et al., 2007; Steury and Murray, 2004). However, by itself the collection of collecting information does not automatically translate into better

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management decisions. Learning can be a slow process, for example in species with long generation times; imperfect detection, small sample sizes or confounding factors may limit the inference that can be made from monitoring data (Nichols and Armstrong, 2012). Even when sufficient information is collected, adequately responding to it by adjusting management actions can be challenging (McCarthy et al., 2012).

In this contribution, we illustrate how the information obtained by monitoring reintroduction outcomes can be used to update knowledge and adjust management decisions. We use as a case study the conservation breeding program for the European pond turtle *Emys orbicularis* in Liguria, northern Italy. We analyze the empirical data collected during the first seven years of turtle releases to estimate the vital rates of individuals, the management decisions that affect them and their influence on short- and long-term outcomes (respectively, avoiding extinction and ensuring population persistence). We then use these results to update prior expectations about management outcomes, assessing how monitoring has reduced uncertainty and modified management decisions.

2. Methods

2.1. Case study

The European pond turtle *E. orbicularis* is widely distributed throughout the European continent, north Africa and east Asia, and is therefore listed as Lower Risk/Near Threatened in the IUCN Red List (Tortoise and Freshwater Turtle Specialist Group, 1996). However, the listing requires updating since the species has become locally rare in several countries (Fritz and Chiari, 2013), following habitat destruction and fragmentation (Ficetola et al., 2004) and introduction of allochthonous species such as American sliders (*Trachemys* spp.) that have been linked to competition (Cadi and Joly, 2004) and to the spread of alien pathogens and parasites (Iglesias et al., 2015). Conservation programs for *E. orbicularis* are underway in several European countries, often involving conservation breeding and translocation actions (Fritz and Chiari, 2013). In the north-western Italian region of Liguria, the species was thought to be extinct following habitat destruction (Doria and Salvidio, 1994), until the rediscovery in the early 1990s of a few individuals of what was later identified as a separate subspecies endemic to Liguria (*E. orbicularis ingauna*; Jesu et al., 2004). The small number and old age of the captured individuals, and the lack of evidence of breeding in the wild, suggested an impending risk of extinction.

A program for in situ and ex situ conservation and restocking program for *E. orbicularis* was initiated in 1999, and an outdoor breeding facility ("Centro Emys") was built in 2000 on public land at Leca di Albenga, <2 km from the nearest known site of occurrence of the species. As of 2015, the breeding center hosts a total of 22 adult turtles (15 females and 7 males). From June until the end of July, a small opening connects the nesting site to the adult tank allowing females to lay egg clutches in clay-sandy soil. Eggs are left in the nest and hatchlings are collected after their emergence from the soil. Turtles are active from mid-March until October and overwintering takes place in mud on the bottom of the tanks. Newborns usually hatch in September, but sometimes eggs overwinter and hatch the following spring. Newborns are always transferred to the Aquarium of Genova and reared in a dedicated indoor facility for about two years before being returned to the outdoor facility for a period of acclimatization before release into the wild. Since 2008, turtles are released yearly in June or July at five different sites across the Centa river plain, after a screening for blood and gastrointestinal pathogens in accordance with veterinary protocols. Before release, each animal is individually marked following the methods of Cagle (1939) and by the subcutaneous implantation of a pit tag. To estimate survival patterns, released turtles at all sites are monitored annually, using baited funnel traps during three sessions (between May and August), each consisting of three consecutive trapping days.

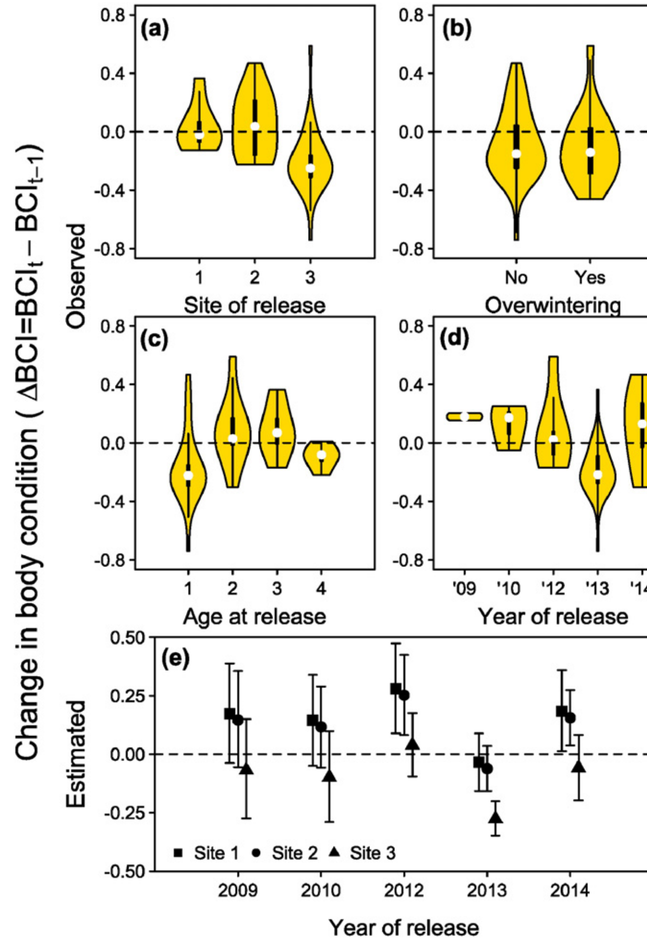

Fig. 1. Post-release body condition of turtles. The y-axis indicates the observed change in body condition index (BCI), calculated as the residual of a regression between body weight and carapace length, between the time of release and recapture in the first year. Plots (a) to (d) summarise the observed data, grouped by time of release, wintering type, age at release and year of release. Plot (e) indicates the estimated change in body condition for each year of the program, at each release site, as predicted by the model with the highest DIC support. The dashed line indicates no change, negative and positive values indicate condition loss and gain respectively.

The release program seeks to maximize the probability of persistence of *E. orbicularis* in the region; however, budget constraints influence management decisions, particularly in regard to the age of individuals to release. Among candidate age classes for release (3-, 4- and 5-year-olds), releasing older individuals may allow bypassing high-mortality juvenile stages and increase the chances of successful establishment, but the longer captive period would increase management costs. The relative benefits of releasing older turtles may also be offset by age-specific post-release mortality, for example if older turtles were more likely to disperse away from the site. In a previous study, Canessa et al. (2015b) used a stage-structured population model to predict the viability of a reintroduced population, focusing on management decisions about the age of individuals to release. Most of the information used to parameterize the model presented in Canessa et al. (2015b) was sourced from studies about other subspecies of *E. orbicularis*, across a wide range of geographic and environmental conditions (from the Iberian peninsula to Poland), potentially different from *E. o. ingauna*. This additional uncertainty was incorporated in model predictions by formalizing uncertainty in expert judgment as probability distributions for four uncertain parameters (survival of hatchlings, survival of turtles 3 to 5 years old, fecundity of subadult and adult turtles).

In this study, we sought to investigate how the monitoring data collected during the first seven years of turtle releases have assisted in reducing uncertainty surrounding the decision problem presented by Canessa et al., 2015b. To achieve this end, we followed a two-step analysis: first, we used mark-recapture models to analyze the data collected by monitoring released individuals. We then updated the survival estimates and predictions of viability described by Canessa et al. (2015b), and assessed how management decisions would be adjusted in response to the new state of knowledge.

2.2. Survival of released individuals

To estimate the survival of reintroduced individuals, we fitted a closed population (Cormack–Jolly–Seber) model to the mark-recapture data collected between 2008 and 2015. The choice of a closed-population model was justified by the fact that to date all recapture histories involve released animals; although mortality and permanent emigration are indistinguishable in this class of models, individuals that leave the release sites can be considered functionally lost for the purpose of the reintroduction. Moreover, radiotracked individuals in a preliminary study (2014–15) showed high site fidelity during the first two months following release, normally the period with the highest probability of homing (Cadi and Miquet, 2004), suggesting permanent emigration may represent only a small component of post-release mortality. We limited the analysis to the three release sites for which multiple years of recapture are available, excluding two sites where releases have begun only recently.

We built a set of ten candidate models to assess our hypotheses about survival. First, we assessed whether survival of released turtles was constant (model 1), or whether it varied by year (2), by the age of individuals at release (3) or by site of release (4). We also assessed whether survival was reduced in the first year after release, by a constant amount (model 4), or as a function of site of release (5), age at release (6), size at release, expressed as straight carapace length in mm (7), body mass (g) at release (8), or depending on whether the individuals released overwinter in the nest chamber (model 9; Mitrus and Zemanek, 2003). Finally, we also built a null model (model 10), reflecting a situation in which survival is constant and is not reduced in the post-release year (Table 1). Furthermore, we repeated all models with two different formulations, in which the probability of recapture was respectively constant and different among sites. This choice reflected the expectation that sites can be more or less difficult to survey. In total, we analyzed 20 models (10 formulations of survival × 2 formulations of recapture probability).

We fitted all models in JAGS (Plummer, 2005), with 100,000 iterations (including a 50,000 burn-in and applying a thinning rate of 10) for each of three separate Markov chains. We assessed convergence by visual inspection of the chain histories. We compared models by assessing the posterior distributions of estimated parameters and by using the Deviance Information Criterion (DIC; Spiegelhalter et al., 2002). The model with the lowest DIC score was considered the most parsimonious balance of fit and complexity, and models separated by 2 DIC points or less were considered to receive equal support (Spiegelhalter et al., 2002). Cormack–Jolly–Seber models condition on the first capture and therefore do not normally allow the estimation of population size (Lebreton et al., 1992); however, in this case the status of animals prior to the first capture is known, since to date only released individuals have been caught and no new individuals have entered the population. Therefore, we were able to use the model to estimate population size by counting all individuals that were estimated to be alive at each time step.

To better assess the existence of a post-release effect, we also assessed the post-release trend in body condition. We calculated a body condition index (BCI) following the method proposed by Lecq et al. (2014), where BCI corresponds to the residual of a linear regression between the logarithms of body mass (g) and straight carapace length (SCL, mm). We calculated the difference in body condition between release and the first year as $\Delta BCI = BCI_{t+1} - BCI_t$, where t is the year of release. We then modelled ΔBCI as a function of different aspects of the release program. We built and compared six candidate models including the following predictors of ΔBCI: (1) null model, with a constant rate of change; (2) age at release; (3) year of release; (4) site of release; (5) individuals from overwintering eggs; and (6) both site and year of release. We fitted all models in JAGS, using the same MCMC settings and diagnostics described above.

Table 1
Results of model comparison for mark-recapture of *E. orbicularis* 2008–2015. φ indicates annual survival, *p* the probability of recapture. Words in brackets indicate the covariates included in a given model. Site = release site; winter = released individual has spent pre-release winter in captivity (yes/no); age = age in a given year; size = SCL at release; weight = weight at release; rel = first year after release (i.e. model incorporates a post-release effect on survival).

Model	Deviance	DIC	pD	ΔDIC
φ (site) p (site)	292.5	447.5	155.0	–
φ (.) p (site)	312.2	454.0	141.8	6.5
φ (winter) p (site)	296.3	471.1	174.8	23.6
φ (rel) p (site)	308.2	484.3	176.1	36.8
φ (rel ∗ winter) p (site)	316.0	485.6	169.6	38.1
φ (year) p (site)	319.3	493.5	174.2	46.0
φ (age) p (site)	326.4	508.9	182.4	61.4
φ (rel ∗ size) p(site)	334.6	569.6	234.9	122.1
φ (rel ∗ age) p (site)	341.9	603.2	261.4	155.7
φ(rel ∗ weight) p (site)	322.0	636.3	314.2	188.8

2.3. Updating predictions and management decisions

We then used the estimates of survival obtained from the CJS models to assess how the monitoring data have modified our expectations about the outcomes of the release program. Because the simulation code used by Canessa et al. (2015b) has been modified since, we did not compare predictions directly between the two studies. Instead, we obtained new predictions of population viability using a newly written program in R (Appendix A1), repeating the simulation for two sets of parameters.

For the first set of parameters, we used the values originally presented by Canessa et al. (2015b) to represent our expectations in the absence of empirical data, based only on available information and expert judgment. For the second set, we replaced the estimates of survival for turtles between 2 and 7 years of age with the values estimated by the CJS model which received the highest DIC support. Where estimates differed by site, we repeated the simulation for each site, using the corresponding survival probabilities. In this second set of parameters, we left unchanged all other parameters from the original model, including fecundity and the corresponding uncertainty, for which no additional information has yet been collected.

For each set of parameters, we simulated the population trajectory over a 50-year period, following the annual release of 15 individuals at each site for ten years, and setting the initial population size equal to that estimated by the best CJS model (including uncertainty) for each site. We obtained predictions of viability under releases of 3-, 4- or 5-yr-old turtles, reflecting the original management question by Canessa et al. (2015b). We evaluated the differences between the two sets of predictions, and assessed whether the information collected has resolved the uncertainty in management decisions.

3. Results

3.1. Rearing and releases

As of September 2015, over 500 turtles have hatched in the outdoor facility. Survival in captivity during the first two years of age is relatively high compared to that of wild turtles (Mitrus, 2005; Paul, 2004), and about 60% of the hatchlings become available for restocking at an age of 3 or 4. Releases in the wild of turtles born in the breeding facility began in 2008 with 10 individuals and have regularly continued since, with a marked increase since 2013 made possible by an expansion of the rearing facilities. In 2011, an outbreak of septicemic cutaneous ulcerative disease (SCUD) infected several turtles in the rearing facilities, with only 4 individuals meeting the DRA standards for release. The remainder were retained in captivity and treated; the majority healed and was released the following year. Overall, 200 turtles have been released in five different sites.

3.2. Survival of released individuals

Overall, the CJS models suggested survival differed among release sites, but did not suggest yearly variation or differences related to age or size at release, and did not highlight significant reductions in survival during the first year post-release. We retained the set of models including a site-specific probability of recapture, since each of these performed better (ΔDIC > 10) than the corresponding model with constant recapture. Of this set, the model including site-specific survival received the highest DIC support (Table 1). Annual apparent survival (including the indistinguishable emigration component) differed considerably among sites (Site 1: $\varphi = 0.754 \pm 0.065$ s.d.; Site 2: $\varphi = 0.906 \pm 0.04$ s.d.; Site 3: $\varphi = 0.674 \pm 0.07$ s.d.). The null model with constant survival received marginal DIC support (ΔDIC = 6.5). The model with age-specific survival received no support (ΔDIC = 61.4), with confidence intervals for age-specific estimates overlapping. Similarly, all models including a post-release effect received no support, suggesting no evidence of a post-release reduction in survival. For the three sites modelled, the total number of individuals in the wild is now estimated at about 80 individuals (95% credible intervals: 75–91), not including more recent releases at other sites and any remaining wild-born turtles.

Models of the change in body condition index (BCI) in the first year after release suggested similar dynamics. The average change was a marginal loss of BCI (-0.009 ± 0.024 s.d., over a range of -0.76 to 0.59). The model that best explained the overall variation included site- and year-specific effects (Table 2). Mean changes ranged from an overall increase in BCI at Site 2 to almost no change at Site 1 and an average decrease at Site 3 (Fig. 1). The temporal trend was similar for all sites, with a visibly higher BCI loss in 2013 (Fig. 1). All other models received effectively no DIC support (ΔDIC > 10); however, for some models low precision for regression coefficients suggested small sample sizes may have limited power, particularly for the age-specific model (where visual observation of the data suggested a greater loss of BCI for 2-year old individuals; Fig. 1).

3.3. Updating predictions for management decisions

The lack of support for age-specific survival confirms expectations by Canessa et al. (2015b) that survival is largely similar between all age classes considered for release (between 2 and 6 years of age). For the best site (Site 2), estimates of survival provided by the best CJS model corresponded almost exactly with the parameterization used by Canessa et al. (2015b), which was assumed to describe “ideal” habitat conditions (Table 3). The estimated survival for turtles of age classes considered for release (3–5 years old) was slightly higher than expected for Site 2 and lower for sites 1 and 3.

These differences were reflected in the comparison of predicted viability under the original and updated parameter values. At site 2 the population was predicted to remain approximately stable once releases ended, largely matching predictions under the original parameterization (Fig. 2c). Conversely, populations at sites 1 and 3 were predicted to slowly decline following the end of releases (Fig. 2b, d). In regard to the uncertainty about the age of individuals to release, for Site 2 the estimated differences between releases of 3-, 4- and 5-yr-old turtles were smaller than expected (Fig. 2c). For sites 1 and 3, releasing older turtles improved population size during the release process, but still had little effect on the long-term viability, reflecting the lower survival (Fig. 2b, d).

4. Discussion

Upon the rediscovery of *E. orbicularis* in Liguria, its extinction at the regional level appeared imminent. After 15 years of conservation management and eight years of releases, the estimated number of turtles in the wild and their high survival suggests the program has been successful in averting local extinction in the short term. Survival in captivity between birth and release is markedly higher than the expected survival of wild turtles of the same age; bypassing high-mortality juvenile stages has rapidly increased the size of the captive population. Success in the establishment phase appears to be related to environmental conditions at release sites rather than to specific aspects of the release strategy such as release age. We found no evidence of a post-release reduction in survival compared to that expected in wild-born individuals, confirming the findings of previous reintroductions for *E. orbicularis* across Europe (Cadi and Miquet, 2004; Gariboldi and Zuffi, 1994; Meeske and Poggenburg, 2014). In contrast, Bertolero and Oro (2009) found that

Table 2
Results of model comparison for the change in body condition index (BCI, calculated as the residuals of a linear regression between the logarithms of body mass and length of individuals) during the first year after release. Words in brackets indicate the covariates included in a given model. Site = release site; winter = released individual has spent pre-release winter in captivity (yes/no); year = year of release; age = age at release (modelled as a quadratic effect).

Model	Deviance	DIC	pD	ΔDIC
ΔBCI (year ∗ site)	−37.6	−28.1	9.4	–
ΔBCI (year)	−19.2	−13.1	6.0	15.0
ΔBCI (site)	−12.0	−7.1	4.9	21.0
ΔBCI (age)	−10.8	−6.6	4.2	21.5
ΔBCI (.)	1.7	3.8	2.1	31.9
ΔBCI (winter)	2.6	5.8	3.2	33.9

Table 3
Vital rates for stage-structured model for *Emys orbicularis* (φ: survival, *f*: fecundity). For each parameter, “original” refers to the values used in Canessa et al. (2015b) on the basis of expert judgment and published information; “updated” indicates the values derived from the best Cormack–Jolly–Seber model fitted to the mark-recapture data for *E. o.* in Liguria. Minimum, mode and maximum indicate the parameters used to define a probability (beta-PERT) distribution for the respective mean parameters in the simulation; where min and max are missing, no parametric uncertainty was modelled. Environmental stochasticity values indicate the standard deviations used to define year-to-year variation in the simulation.

		Min	Mode	Max	Environmental stochasticity
			Hatchlings		
φ, original		0.02	0.08	0.2	0.03
1-yr-old					
φ, original		–	0.525	–	0.03
2-yr-old					
φ, original		–	0.8	–	0.14
3-, 4-, 5-, 6-yr-old					
φ, original		0.45	0.9	0.99	0.01
φ, updated	Site 1	0.62	0.75	0.89	0.01
	Site 2	0.82	0.91	0.99	0.01
	Site 3	0.53	0.67	0.82	0.01
7–11 yr-old					
φ, original		–	0.96	–	0.01
f, original		0.55	0.64	1.15	0.43
12+ yr-old					
φ, original		–	0.96	–	0.01
f, original		0.68	1.11	1.69	0.15

reintroduced individuals of *Mauremys leprosa* in the Ebre Delta (NW Spain) experienced high emigration and lower survival than estimated for wild individuals in suitable areas, suggesting short-term establishment failure driven by unsuitable habitat.

In spite of these short-term outcomes, the recovery plan for *E. orbicularis* in Liguria also follows the global pattern of translocations in that an assessment of long-term success remains difficult. Perhaps the greatest source of uncertainty in this regard is the insufficient information about successful reproduction of released individuals. This matches observations by Gariboldi and Zuffi (1994) and Cadi and Miquet (2004) for *E. orbicularis* reintroductions in Italy and France respectively, where survival of released individuals was high, but reproduction and juvenile stages remained absent or uncertain. At our release sites, mating behaviors have been repeatedly seen in the wild; in 2015, inguinal palpation of recaptured individuals suggested that reintroduced females were producing calcified eggs (D. Ottonello, *pers. obs.*). In spite of such positive clues, inference about recruitment to date has been complicated by the late onset of breeding in this species, its low fecundity rate, and the challenges of monitoring breeding events. For example, turtles released as 3-yr-olds may take up to about five years to breed in the wild; those wild-born juveniles may then take an additional 2–3 years to grow to a size that allows their capture during surveys. This would imply an 8-year time lag between the implementation of releases and the earliest opportunity to assess the adequacy of management (for example, whether reproduction is influenced by captive protocols or environmental conditions of the release sites). Such time lags between learning and application may occur in several long-lived reptile species (Towns et al., 2016), and may challenge even rigorously structured adaptive management protocols.

The difficulty of assessing long-term survival for *E. orbicularis* reinforces the importance of monitoring in conservation breeding and reintroductions (Ewen and Armstrong, 2007). Formally defining prior expectations is the first step to then be able to focus monitoring efforts and incorporate new information in decision making (Canessa et al., 2015a). This step may be sometimes challenged by the lack of baseline data for the target program. However, our results showed how through careful evaluation of available information and best-practice methods for defining expert judgment and associated uncertainty, Canessa et al. (2015b) had produced estimates of survival that were remarkably similar to those we later derived from empirical data.

The formal treatment of uncertainty also allowed us to directly link the results of our analyses to management decisions. In this case, the initial focus was on uncertainty about age-specific survival, particularly in the year after release. The analysis of monitoring results helped reduce this uncertainty and justified a decision to release younger turtles, with similar persistence outcomes but significant resource savings. In a suitable site, the predicted long-term persistence was sufficiently high not to warrant the release of older individuals. In less suitable sites, predicted viability was mostly determined by lower estimated survival, such that releasing older turtles would only provide limited benefits for persistence in the long term, while increasing management costs. Assuming most of the mortality in captivity is incurred in the early hatchling stages, the 60% of hatchlings that become available for release at 3 years of age would need to be maintained for another one or two years if releases occurred later. This would require a captive population 20% to 40% larger than for 3-yr-old releases; although the exact quantification of fixed and variable costs is difficult, it is reasonable to expect that expenditures for most items, such as food, maintenance and work hours, would increase accordingly. Additionally, the increased risk of overcrowding and disease would need to be addressed by either expanding infrastructure or by reducing the overall number of turtles produced. The same resources might be applied more effectively to additional in-situ management to improve site suitability.

Demographic projections suggest that population growth would be slow even in optimal conditions, and at considerable risk of failure in sub-optimal environments. Similarly, in a quantitative assessment of

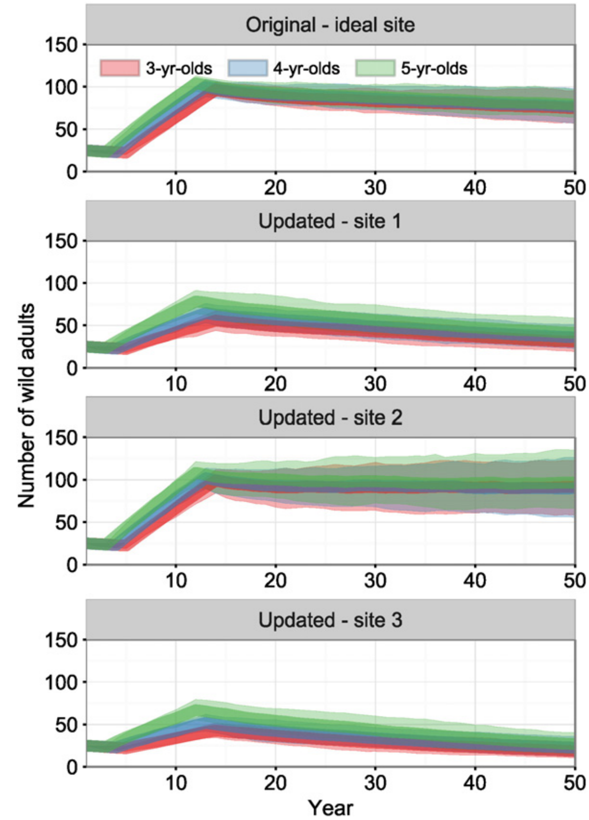

Fig. 2. Comparison of expected outcomes of the recovery plan under different states of knowledge. Values correspond to the number of sexually mature females predicted by a stage-structured population model (Canessa et al., 2015b). Panels refer to the different parameter values presented in Table 3. Dashed areas indicate 95% confidence intervals over 1000 simulations for parametric uncertainty.

headstarting for *E. orbicularis* in Germany, Paul (2004) also pointed at the requirement for long-term (>20 years) conservation actions to ensure population stability. Maintaining suitable habitats is particularly challenging given that the natural ponds occupied by *E. orbicularis* in Liguria tend to disappear naturally, and natural turnover is prevented in the highly human-modified landscape. However, long-term commitments such as the maintenance of suitable habitats and predator-free areas are challenges common to species conservation programs beyond conservation breeding and translocations. Both critics and advocates of conservation breeding agree that it should not be seen as a quick, long-lasting fix for species persistence; rather, its real potential is to provide time to develop effective long-term conservation actions in the wild (Byers et al., 2013; Rahbek, 1993; Redford et al., 2012; Snyder et al., 1996). As a result, if extinction can be avoided in the short-term, the real potential of captive programs and initial releases may lie in providing time for learning and acting effectively in-situ. Throughout this period, monitoring the outcomes of releases allows managers to address knowledge gaps, reduce uncertainty and keep improving management actions.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.biocon.2016.05.003.

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