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Survival of Marbled Teal (*Marmaronetta angustirostris*) released back into the wild

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Abstract

Reintroduction or re-enforcement programmes are major tools in species conservation, but there is a need for more studies that assess the influence of different husbandry and release methods on the survival of released animals. We investigated the survival of globally threatened Marbled Teal (*Marmaronetta angustirostris*) taken into captivity as ducklings when they became trapped in an irrigation channel, then released again after fledging. We used wing tags and mark–recapture models to estimate the survival of released teal. Ducklings rescued in 1996 ($n = 53$) were released soon after fledging in September and their survival was modelled for seven months until April 1997. Their apparent monthly survival rate (lower than true survival owing to loss of wing tags) was 0.85 ± 0.12 (±s.e.). Ducklings rescued in 1997 ($n = 44$) were released together in February 1998 over five months after fledging, and their survival was modelled for six months from February until August. Their apparent monthly survival rate was 0.54 ± 0.06. Ducklings rescued in 1998 ($n = 159$) were released in August–September soon after fledging and their survival was modelled for 10 months from August until June. Their apparent monthly survival rate was 0.83 ± 0.07. Monthly survival was significantly higher for the 1996 and 1998 cohort, suggesting that retaining birds in captivity after fledging had a negative impact on post-release survival. When birds were released in February, a lower proportion survived until the breeding season three months later than when they were released five months earlier in September.

Keywords: Mortality; Re-enforcement; Reintroduction; Release methods; Wing tags

1. Introduction

Reintroduction or translocation programmes are a fundamental tool in species conservation, and increasing their success is a major challenge for conservation biologists (Sarrazin and Barbault, 1996). Only a small proportion of such programmes have been shown to lead to the successful establishment of viable populations. Beck et al. (1994) considered that only 11% of bird reintroduction programmes had been successful. Many issues that contribute to failure have been identified, largely by trial and error, leading to the formulation of detailed guidelines for reintroductions with the aim of increasing their success (Black, 1991; Kleiman et al., 1994; IUCN, 1998). Comparative studies of diverse projects have identified factors such as good habitat quality and release of larger groups that tend to increase success (Wolf et al., 1996).

For individual projects, monitoring of released animals is vital to identify the causes of failure and potential solutions. However, Wolf et al. (1996) found that, in most programs in North America, Australia and New

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Zealand, birds or mammals were not even marked to enable post-release monitoring. There is a clear bias towards marking larger, longer-lived species that are easier to tag or radio-track.

Here we use mark-recapture models to estimate post-release survival of short-lived Marbled Teal *Marmaronetta angustirostris* in Spain. This species is globally threatened ((IUCN Vulnerable, BirdLife International, 2000) and its population in Spain (which holds the majority of the European population) and elsewhere has suffered major declines in recent decades (Green, 1996a; Green and Navarro, 1997). It begins breeding in its first year and undergoes major fluctuations in population size, with between 30 and 200 pairs breeding in Spain between 1992 and 2002.

During a re-enforcement programme of individuals rescued from the wild as ducklings, we estimate the survival of birds released into the wild over a three year period and examine factors that might affect survival among years. Different authors have used different terminology and from hereon we use "reintroduction" to refer to the release of animals to areas of the natural range where the species is extinct, and "re-enforcement" to refer to the addition of individuals to an existing population (IUCN, 1998; Hodder and Bullock, 1997).

2. Study area and methods

Marbled Teal used in this study were rescued as ducklings in 1996 to 1998 after they became trapped in a concrete irrigation channel (Navarro et al., 1995; Green et al., 1999) in the El Hondo wetlands (38°11′ N, 00°45′ W; 1650 ha). The wetlands are located in the south of Alicante province within the autonomous community of Valencia in Eastern Spain. El Hondo holds the majority of the European population of Marbled Teal (Green, 1996a; Green and Navarro, 1997) and is protected as a Natural Park and Ramsar site (Bernués, 1998). Large *Phragmites* reedbeds and problems of access to several of the privately owned wetlands complicate bird surveys. Marbled Teal are recorded all year round, but are more abundant from April to November inclusive (Navarro and Robledano, 1995). Broods hatch from early May to mid July (Green et al., 1999). Approximately a third of all broods became trapped in the irrigation channel until it was modified in autumn 1998 by incorporating ramps.

After rescue, ducklings were taken to the Recovery Centre for Fauna in Valencia (Centro de Recuperación de Fauna de El Saler, CRFES). The teal were then reared in groups in the absence of adults, and were later marked, and sexed by cloacal inspection shortly before release in El Hondo. Survival between arrival at the CRFES and release was high. In 1996, 94 of 117 ducklings survived to fledging (i.e. when growth of flight feathers is completed). In 1997, 46 of 60 ducklings survived to fledging. In 1998, 162 of 184 ducklings survived. They were released either shortly after fledging (the 1996 and 1998 cohorts) or after the winter hunting season (the 1997 cohort), thought to cause significant mortality of Marbled Teal at these wetlands (Navarro and Robledano, 1995). There were no differences in rearing methods between years. Between fledging and release, teal were kept in large pens to encourage flight. In order to allow detailed monitoring after release, we marked the teal with wing tags (patagial tags) prior to release. Although wing tags can have deleterious effects on the survival or reproductive success of birds (Calvo and Furness, 1992; Gaunt and Oring, 1997), they are widely used to study ducks (e.g. Pöysä and Virtanen, 1994; Guillemain et al., 2002).

Details of our marking methods and trials conducted to improve their design and durability are described elsewhere (Green et al., 2004). The tags were made from heavy duty but flexible white plastic with a two digit alpha-numeric code attached to the tag using black sticky plastic digits. They were attached by punching a hole in the patagium, passing a nylon wire through the patagium and the tag, and fixing nylon disks at either side. In 1996 we used wider tags (55 mm long and 28 mm wide at the base). In 1997–1998 we used narrower tags (15 mm wide) that were harder to read in the field but which caused less feather wear (Green et al., 2004). Trials in captivity and field observations showed that many tags became unreadable within weeks or months of marking because part or all of the digits were lost, due to the weathering effect of preening, wing-flapping etc. We suspect that more than half of the tags became illegible within 90–180 days of release, but we lack a reliable estimate of loss rates (Green et al., 2004). Large and small tags were made with the same materials and attached in the same way. We have no evidence to suggest that loss rate of legible tags varied with tag size.

A total of 53 marked birds (25 males) were released on 12 September 1996 shortly after fledging. A total of 44 teal (18 males) rescued as ducklings in 1997 were released on 18 February 1998 after the hunting season was over. A total of 159 teal (84 males) rescued as ducklings in 1998 were released shortly after fledging on 31 July ($n = 8$), 6 August ($n = 38$), 13 August ($n = 12$) and 16 September 1998 ($n = 101$). Following release, teal were monitored by us and other ornithologists in ad hoc surveys, reading the codes on tags using telescopes. Released teal mixed with the wild teal and showed no obvious behavioural differences. Teal sometimes visited other wetlands in the autonomous community of Valencia (Green et al., 2004), but only two marked birds were observed there and those observations were excluded from this study. The observer effort was not constant over seasons, principally because ornithological activity was more restricted during the winter hunting season.

Neither was it constant between years, owing to changes in the numbers of active observers and their access to different parts of the study site. The methods we used to model survival are designed to cope with such variation in observer effort. There was no evidence to suggest that long-distance dispersal movements affected our results. No tagged birds were reported from outside the community of Valencia, despite efforts to search for them in neighbouring regions and countries.

The observations of marked birds (from hereon referred to as "resightings") were used to analyse the survival and the resighting probabilities of teal separately for each cohort using program MARK based on capture–recapture data (White and Burnham, 1999; Cooch and White, 2002). Owing to high rates of loss of legible wing tags in the field after releasing teal, our survival models do not measure bird mortality, but rather a combination of tag loss and bird mortality, i.e. our "survival rates" exaggerate bird mortality rates (see also Nichols and Hines, 1993; Bradshaw et al., 2003).

Observations of marked teal were grouped in 30 day periods starting on the day of release in the wild. Thus our survival and resighting parameters refer to the probability that an individual was alive or observed (respectively) after one month. For the 1998 cohort, data for birds released on 31 July and 13 August were pooled and all the birds were considered as released on 13 August ($n = 20$), due to the small sample size. Birds released on 16 September were excluded from the first period and entered into the population of marked birds during the second time period. Survival was modelled for a different number of 30 day periods for the three cohorts studied due to the difference in the number of birds released, which affected the options for analysis. Thus, seven time periods were considered for the 1996 cohort (September 1996 to April 1997), six time periods were considered for the 1997 cohort (February to August 1998) and 10 for the 1998 cohort (August 1998 to June 1999).

The notation used for survival models follows Lebreton et al. (1992). For our initial model we used ($\phi_{s*t}p_{s*t}$) where both survival (ϕ) and resighting probability (p) were dependent on the factors sex (s) and time (t). All models were constructed using the sin-link function. Survival and capture probabilities were either considered to be constant for all individuals or to change as a function of sex and/or time. This produced a total of 16 different models (Table 1). Model selection was based on Akaike's information criterion adjusted for sample size (AICc, Burnham and Anderson, 2002). The model with the lowest AICc represents the best balance between loss of precision (due to overfitting) and bias of the estimates (due to underfitting, Burnham and Anderson, 2002).

Other models were considered equally supported when differences with the lowest AICc were less than 2 (Burnham and Anderson, 2002). The relative support for these models was estimated from the comparison of Akaike weights (w) for the final model in relation to other models with similar AICc scores. A bootstrap goodness of fit test was used to test the fit of the global models to the capture histories. A distribution of expected deviances was generated based on 1000 random simulations of the capture histories under the assumptions of the Cormark–Jolly–Seber model. No significant deviation of these assumptions was found for 1996 ($p = 0.40$), 1997 ($p = 0.12$) or 1998 ($p = 0.23$) datasets. Given the structure of the models, in some cases not all parameters could be estimated and these were not

Table 1
Summary of MARK survivorship models of wing-tagged Marbled Teal following release into the wild (after rescue as ducklings in 1996, 1997 and 1998), listing number of parameters (n) and the small sample size-adjusted Akaike's Information Criteria of the model (AICc)

Model structure	1996 cohort		1997 cohort		1998 cohort	
	n	AICc	n	AICc	n	AICc
$\phi_{s*t}p_{s*t}$	22	147.58	14	82.75	31	262.19
$\phi_{s*t}p_{t}$	17	135.63	11	73.15	27	259.36
$\phi_{s*t}p_{s}$	14	134.79	12	85.48	20	249.82
$\phi_{s*t}p_{.}$	13	131.50	11	82.13	19	254.33
$\phi_{t}p_{s*t}$	17	127.00	12	76.90	26	251.53
$\phi_{t}p_{t}$	10	120.24	**7**	**63.00**	17	237.89
$\phi_{t}p_{s}$	8	119.85	7	70.62	11	234.54
$\phi_{t}p_{.}$	7	118.16	6	68.10	10	235.75
$\phi_{s}p_{s*t}$	14	118.81	12	77.58	20	238.18
$\phi_{s}p_{t}$	8	116.48	7	63.85	11	228.37
$\phi_{s}p_{s}$	4	115.24	4	70.03	**4**	**223.35**
$\phi_{s}p_{.}$	**3**	**113.12**	3	67.75	3	225.40
$\phi_{.}p_{s*t}$	13	116.34	11	74.83	19	238.14
$\phi_{.}p_{t}$	7	114.41	**6**	**61.65**	10	226.40
$\phi_{.}p_{s}$	**3**	**112.98**	3	67.70	**3**	**223.16**
$\phi_{.}p_{.}$	**2**	**111.53**	2	66.19	**2**	**223.59**

Model structure was defined by survival probability (ϕ) and probability of resighting (p). Model subscripts indicate if sex (s) or time (t) were assumed to affect the parameters, $*$ indicates factorial models. All models converged on parameter estimates. Final models (see methods) are shown in bold.

included in the total number of parameters estimated for the model (after Cooch and White, 2002). The estimated survival rates for different cohorts were compared using the CONTRAST program (Sauer and Williams, 1989).

3. Results

3.1. Birds rescued in 1996

Of 53 birds marked and released, 12 were observed at least one month after release. The model most supported by the data was one with constant survival and resighting probabilities during the 7 months following release ($\phi.p.$). Monthly survival was estimated as 0.85 ± 0.12 and monthly resighting rate as 0.07 ± 0.03. Two other models had an AICc that was not significantly different from the most supported model (Table 1). These models differed from the most supported model in that survival or resighting parameters differed for males and females ($\phi.p_s$, $p_{\text{males}} = 0.08 \pm 0.04$, $p_{\text{females}} = 0.05 \pm 0.03$; $\phi_s p.$, $\phi_{\text{males}} = 0.90 \pm 0.13$, $\phi_{\text{females}} = 0.79 \pm 0.14$). However, the model without sexual differences in both parameters was supported 2.1 times more by Akaike weights than either of the other two models.

3.2. Birds rescued in 1997

Only five of 44 marked individuals were observed more than one month after release. The best approximating model for birds rescued in 1997 assumed constant survival between time periods and sexes but had time dependent resighting probabilities ($\phi.p_t$). Monthly survival was estimated as 0.54 ± 0.06 and resighting rate ranged between 0.00 ± 0.00 and 0.28 ± 0.18. Another model ($\phi_t\, p_t$) presented a similar AICc value to the most supported model (Table 1), but 30% of the parameters in this model were not estimable due to overfitting caused by the low number of observations. Furthermore, the best approximating model was supported 2.0 times more by Akaike weights than this alternative model.

3.3. Birds rescued in 1998

Of the 159 released birds, 17 were observed more than a month following release. For this cohort, similar support was found for three different models (Table 1, Akaike weight ratio 1.10–1.24). The model with the lowest AICc assumed constant survival between time periods and sexes, but sex dependent resighting probabilities ($\phi.p_s$) estimated at 0.83 ± 0.07 (ϕ), 0.02 ± 0.01 (p_{males}) and 0.05 ± 0.02 (p_{females}). However, two other models had very similar levels of support. One model corresponds to sex dependent survival and resighting probabilities ($\phi_s p_s$, $\phi_{\text{males}} = 0.98 \pm 0.14$, $\phi_{\text{females}} = 0.76 \pm 0.08$, $p_{\text{males}} = 0.01 \pm 0.01$, $p_{\text{females}} = 0.07 \pm 0.03$). The other one corresponds to time and sex independent survival and resighting parameters ($\phi.p.$, $\phi = 0.84 \pm 0.07$, $P = 0.04 \pm 0.01$).

3.4. Differences in survival between 1996, 1997 and 1998 cohorts

The survival of birds from the 1997 cohort was significantly lower than for 1996 and 1998 cohorts ($\chi^2 = 10.68$, 1 df, $P = 0.001$). Cohorts from 1996 and 1998 had similar survival estimates ($\chi^2 = 0.02$, 1 df, $P = 0.89$). These differences were consistent when using estimates derived from the alternative models with similar AICc scores to the best approximating models.

Our estimates of apparent survival rate for different cohorts suggest that, whatever the rate for the retention of legible tags, a lower proportion of the birds from the 1997 cohort released in February survived to breed than those from the other cohorts released months earlier (Appendix A).

4. Discussion

Marking small ducks such as Marbled Teal is complicated as leg rings are rarely visible and wing tags and nasal markers suffer high loss rates (Green et al., 2004). Given the likely deleterious effects of marking with wing tags (Green et al., 2004), we do not recommend them for general use with Marbled Teal or similar threatened duck species. However, our study illustrates how their limited use in a re-enforcement programme combined with mark–recapture models can provide vital information about factors that can determine their success.

Our results suggest low survival rates of released teal, but these rates are considerably underestimated owing to frequent loss of marks. The Marbled Teal appears to be adapted to fluctuating habitats and has a particularly high fecundity (Green, 1998, 2000). Thus, relatively high mortality rates are to be expected and are likely to be increased at El Hondo by lead poisoning (Mateo et al., 2001) and illegal hunting (Navarro and Robledano, 1995), although we have no precise estimates for the wild population. We found apparent monthly survival to be 54–57% higher in the 1996 and 1998 cohorts released after fledging in August–September than in the 1997 cohort released the previous February. It is unlikely that this reflects a seasonal difference in mortality in the wild teal population, with greater mortality from February to August (modelled for the 1997 cohort) than from August to the following June (1996 and 1998 cohorts). Mortality rates in duck populations are generally assumed to be higher over the post-breeding and wintering periods than over the breeding periods (Baldassare and Bolen, 1994), not lower as our results indicate. This is particularly true in areas exposed to hunting during winter, as in our study site. Furthermore, we found no evidence for seasonal variation in mortality rates within each cohort. We do not believe that a higher rate of tag loss in the 1997 cohort could explain our results, as we used similar tags for the 1997 and 1998 cohorts. If the tags themselves influenced mortality rates, we would have expected survival rates to be lowest for the 1996 cohort, which suffered more feather wear from the larger tags (Green et al., 2004).

Our results are probably due to teal released shortly after fledging (the 1996 and 1998 cohorts) being better prepared for survival in the wild than those that remain in captivity for several months between fledging and release (the 1997 cohort). An artificial, low-fibre diet and reduced flight activity in captivity cause changes in the physiology and morphology of birds that reduce the ability to digest natural food or to escape predators (Kehoe et al., 1988; Liukkonen-Anttila et al., 2000). An extended time in captivity after the fledging period may also cause behavioural problems that may reduce their capacity to adapt to conditions in the wild (Wallace, 2000). Numerous studies have found that captive-bred birds have behavioural problems in the wild (Kleiman et al., 1994), and translocations of wild-caught animals tend to be more successful than releases of captive-bred ones (Griffith et al., 1989). The post-fledging period when a teal becomes independent of its mother and siblings and undergoes a shift to a higher fibre diet (Fuentes et al., 2004) is likely to be a critical time for successful adaptation to the wild environment.

There is no evidence to suggest that the differences in mortality rates between cohorts were caused by the differences in numbers of Marbled Teal released together in each case. The birds were released into a wetland holding several hundred wild Marbled Teal and several thousand other ducks, and we do not expect that changes in the numbers released would directly affect mortality by competition or a dilution effect on predation. Furthermore, numbers released were very similar for the 1996 and 1997 cohorts, yet mortality rates differed. Neither do we have any evidence that the mortality rates of wild ducks changed between our study periods, e.g. due to changes in the ecological conditions in El Hondo. However, we can not rule out the possibility that the cohort effect we have observed was somehow caused by environmental variation between years.

Our results are similar to those of Sjöåsen (1996), who found that increasing the period that captive-bred otters spent between removal from their mother and their release into the wild reduced their subsequent survival rates. Likewise, the earlier that young Black-footed Ferrets were introduced to outdoor pens simulating wild habitat, the higher their survival when they were released into the wild (Biggins et al., 1998). The ideal comparison of the effects of our different release methods on survival rates would have been to randomly assign individuals from the same cohort to early and late release dates. Our comparison between cohorts is potentially influenced by other differences between them, although there were no differences in their handling in captivity. However, during release programmes for threatened species, it is rarely possible to carry out a perfect experiment given the many factors and stakeholders involved, and there are similar problems with the design of other studies (Bright and Morris, 1994; Sjöåsen, 1996).

The 1997 cohort was released in February with the intention of increasing survival rates by avoiding the hunting season, yet we have observed the opposite effect. However, since mortality in captivity is very low (see methods), the total number of teal surviving between fledging and February was increased by retaining them in captivity. Nevertheless, our results suggest that the number of teal breeding successfully the following spring are likely to have been reduced by such a late release owing to their high mortality rates (Appendix A).

If for example 50% of tags became illegible within six months (i.e. the monthly tag retention rate was 0.891), an apparent monthly survival rate of 0.83 estimated for the 1998 cohort (equivalent to an annual survival rate of only 0.107) would indicate a true monthly survival rate of 0.932. This would translate into a true annual survival rate of 0.43, which is about what would be predicted for wild Marbled Teal given their small size and large clutch size (Krementz et al., 1997). On the other hand, the apparent monthly survival rate of 0.54 estimated for the 1997 cohort would then represent a true monthly survival rate of 0.61 and a true annual survival rate of 0.002. Under these circumstances, 57% of teal released in September 1998 would have survived to the beginning of their first breeding season (in May 1999), compared to only 22% of teal released in February 1998.

High mortality during the initial period after release has been observed in most reintroductions of other species (Sarrazin and Legendre, 2000). In long-lived birds that take several years to commence breeding, mortality rates are naturally lower for adults than for juveniles, but adults tend to show greater release effects (i.e. greater increase in mortality or reduction in fecundity following release, Sarrazin et al., 1994). Nevertheless, owing to the loss of a high proportion of juveniles before they commence breeding, overall it is more efficient to release adults in many long-lived species (Black et al., 1997; Sarrazin and Legendre, 2000). In contrast, our results suggest that for short-lived birds such as Marbled Teal which breed in their first year, it may be more efficient to release juveniles at fledging than adults.

Our study illustrates the value of using modern mark–recapture methods to test the implications of different husbandry and release methods on the survival of animals in reintroduction and re-enforcement programmes. Such analyses can provide unique and revealing insights into the processes that influence the success of such programmes, and thus generate recommendations for improving their effectiveness (Black et al., 1997; Ostermann et al., 2001). Without such studies, release programmes are likely to be designed on the basis of speculation (e.g. Franzreb, 1990), so reducing the chances of success.

Our results suggest that delaying release of threatened birds after the fledging period (e.g. to reduce mortality from hunting) can be counter productive (as well as expensive). The post-fledging phase may be a crucial development window when experience in the wild is critical to future survival. Reintroduction programmes for threatened waterfowl have usually failed (Green, 1996b), probably for a variety of reasons. Our results suggest that the most effective strategy to increase post-release survival may be to release waterfowl straight after fledging. Further research is needed to assess the effects of different release strategies on waterfowl behaviour and breeding success.

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Appendix A. Demonstration that a higher proportion of teal survived to the breeding season in the 1998 than in the 1997 cohort

The apparent monthly survival rate of Marbled Teal observed in our analysis ϕ was a product of the true survival rate S and the monthly rate of retention of legible wing tags τ (which we assume to be constant between the 1997 and 1998 cohorts marked with an identical tag design), i.e. $\phi = S \times \tau$ (see also Bradshaw et al., 2003). According to our estimates of ϕ, $0.83 < \tau < 1$ (because τ must be less than unity but must exceed the maximum value of ϕ since S must also be less than unity). Thus $S_{1998} = \phi_{1998}/\tau = 0.83/\tau$, and $S_{1997} = \phi_{1997}/\tau = 0.54/\tau$.

The 1998 cohort were released eight months before the breeding season started, hence the survival rate between release and breeding = $(S_{1998})^8 = 0.83^8/\tau^8 = 0.225/\tau^8$.

The 1997 cohort were released three months before the breeding season started, hence the survival rate between release and breeding = $(S_{1997})^3 = 0.54^3/\tau^3 = 0.157/\tau^3$.

Thus the ratio between birds surviving from release to breeding in the 1998 cohort and those in the 1997 cohort = $(0.225/\tau^8)/(0.157/\tau^3) = 1.43/\tau^5$.

Therefore, for all possible values of τ, a higher proportion of birds survived to the breeding season in the 1998 cohort released five months earlier.

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