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Spatial bias in implementation of recovery actions has not improved survival of Orange-bellied Parrots *Neophema chrysogaster*

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Running Head: Survival of Orange-bellied Parrots

Abstract

Not all conservation interventions are successful at correcting threatening processes and the odds of failure increase with uncertainty concerning the true threats to a population. Failure of conservation actions to improve demographic rates might be evidence of their ineffectiveness, or that other unaddressed threats nullify the potential benefits of interventions. Knowledge of key threatening processes that afflict Orange-bellied Parrots *Neophema chrysogaster* is lacking, but population modelling predicts that actions in the breeding range are unlikely to correct decline unless mortality during migration/wintering is addressed. Despite this, there has been a spatial bias in recovery effort towards the breeding range in recent decades. We model annual survival data spanning 1995 – 2017 for the last known wild population to evaluate whether the predictions about the efficacy of recovery efforts are accurate. Based on our best-supported model, probability of adult survival was constant at 0.58, but juvenile survival declined from 0.51 to 0.20. Survival did not improve when we considered the effects of recovery actions in the breeding grounds (which only aimed to correct local scale threats anyway). This result supports predictions that conservation interventions in the breeding ground alone are not sufficient to recover this species. We conclude that although interventions in the breeding ground may have corrected local threats, birds succumbed to other threats during migration/winter. It is crucial that new targeted interventions be identified and implemented to reduce mortality of Orange-bellied Parrots in their migration/winter habitats to prevent extinction.

Key Words

Survival, mark recapture, Cormack Jolly Seber models, conservation, threatened species, population viability model

Introduction

Effective conservation relies on detailed understanding of species biology and clear diagnosis of threats (Caughley 1994). Gathering this information is not always straightforward, and knowledge gaps are a major hindrance for effective management of many species (Scheele *et al.* 2018). As a result, when knowledge of threats is incomplete, conservation managers may implement actions that they presume will be effective (Wintle *et al.* 2010). Such 'educated guesses' may pay off when managers can make reasonable assumptions about the types of threats a species faces (e.g. protecting nesting birds from predation is likely to prove beneficial on rat-infested islands). However, interventions often fail (Scheele *et al.* 2018), and the odds of failure increase with uncertainty concerning the true threats to a population (Caughley 1994; Doherty and Ritchie 2017). It is not always possible to diagnose threats confidently, and this is further complicated if the original causes of population decline are superseded by new threats that arise at small population sizes (e.g. Allee effects – inverse density dependence) (Crates *et al.* 2017). In such scenarios, conservation managers could use adaptive management to trial different actions that might improve metrics of population health. However, this requires careful evaluation of how population vital rates respond to the intervention (Gerber and Kendall 2018). Furthermore, it is important to evaluate conservation actions in the context of life history. For example, targeted action at one time/place may mitigate a local threatening process (Crates *et al.* 2018b), but this benefit may not support population recovery if individuals succumb to different threats at other times/places (Crates *et al.* 2018a; Crates *et al.* 2019). Thus, the failure of conservation actions to improve demographic rates might be evidence of their ineffectiveness, or that other unaddressed threats nullify their potential benefits.

The Orange-bellied Parrot *Neophema chrysogaster* may be the rarest parrot in the world and now breeds only at Melaleuca in southwestern Tasmania, Australia (Lat: 43°25' S, Long: 146° 9' E) (Department of Environment, Land, Water and Planning 2016; Stojanovic *et al.* 2018). In 2016/17 the wild population declined to only two breeding females and 12 males (Stojanovic *et al.* 2018). Unusually for a parrot, the species is an obligate migrant, wintering in coastal habitats along southeastern mainland Australia (Loyn *et al.* 1986). Their key threatening processes are not clearly diagnosed, but habitat loss and degradation, disease and small population size have been implicated in their decline (Department of

Environment, Land, Water and Planning 2016). This uncertainty hinders the species management because recovery strategies that directly target the most important threats are difficult to develop and prioritize (Department of Environment, Land, Water and Planning 2016). Most direct management of Orange-bellied Parrots is implemented at Melaleuca, and includes provision of nest boxes, supplementary food, predator control and release of captive-born birds to increase the number of breeding pairs, correct adult sex ratio bias and maximize reproductive success (Troy and Hehn 2019). In contrast, during migration/winter the species can occur at multiple locations along about 1200 km of coast, and conservation efforts are mostly indirect. The main conservation actions in the migration/winter range involve studies of habitat use (Loyn *et al.* 1986), ecological modeling (White *et al.* 2017), population (Starks *et al.* 1992) and habitat monitoring (Tolsma *et al.* 2014), plus reservation and removal of livestock from habitat, rehabilitation of hydrological processes, control of predators, weeds and human access (Department of Environment, Land, Water and Planning 2016). Aggregation of the entire parrot population at Melaleuca makes management of threats to the breeding population feasible (Troy and Hehn 2019), which partially explains the spatial bias in recovery actions.

Previous studies have suggested that conservation actions at the breeding grounds are likely to be ineffective at reducing mortality over migration and winter (Drechsler *et al.* 1998), which is a severe threat to the species (Department of Environment, Land, Water and Planning 2016). Given the spatial bias in recovery efforts for this species, we evaluate the predictions of Drechsler *et al.* (1998) by modeling annual survival. Survival is a useful demographic trait to study because it is the outcome of multiple, cumulative, discrete threats over the full annual cycle. If the predictions of Drechsler *et al.* (1998) are correct, we would expect that survival of Orange-bellied Parrots has not improved over two decades, despite an increase in conservation attention and effort directed at their protection in breeding habitat.

Methods

Study context

A citizen science monitoring program has been implemented by the Tasmanian Government at Melaleuca since 1979 (Department of Environment, Land, Water and Planning 2016). Monitoring consists of observation of individually colour banded birds at feed tables during the summer breeding season by volunteers (Department of Environment, Land, Water and Planning 2016; Troy and Hehn 2019). We collated survival data from the monitoring program between 1995 and 2017 for this study. The Tasmanian Government implemented most recovery actions annually, including provision of nest boxes, supplementary feeding, predator and competitor control, and health management, so these activities were generally consistent over time. In 2010, due to a steep decline in the population size of Orange-bellied Parrots, 21 juveniles were collected from the wild as new founders for the captive population (Martin *et al.* 2012). Orange-bellied Parrots may be unusually vulnerable to Allee effects (Crates *et al.* 2017), and after 2010, the collection of juveniles for captive breeding reduced the wild population size. Later recovery actions including release of captive-bred parrots (Troy and Hehn 2019) further altered wild population size. Hence, we subset our data into two time periods (i) 1995 – 2010, i.e. natural demographic rates, and (ii) 2011 – 2017 i.e. demographic rates potentially influenced by recent management actions.

Survival data

We accounted for potential misidentification errors in the Tasmanian Government's citizen science sightings data set which could affect our models by filtering the data (Isaac *et al.* 2014). Parrots seen fewer than five times needed either to be (i) verified by >1 observer, or (ii) seen by the same person >3 times to be considered alive (Troy and Kuechler 2018). We assumed that if a parrot was incorrectly categorised as dead using our criteria, such infrequently detected individuals were unlikely to have successfully bred, and thus did not contribute to the population growth. Since the species now probably only breeds at Melaleuca (Stojanovic *et al.* 2018) we assumed this was a closed population, and that loss of individuals was due to death, not dispersal to other breeding locations. We constructed capture histories from 1995 to 2017. During this period, banding nestlings in nest boxes was the main way marked birds entered the population. The first occasion in capture histories represented nestlings banded in boxes, and subsequent occasions represented observations at feeders over successive breeding seasons. We classified individuals in the first time-step of capture histories as juvenile, and all subsequent time-steps as adult. This approach does not differentiate between mortality in the breeding season and mortality during migration/winter, but based on recent evidence we assumed that most juveniles died during migration/winter (DPIPWE. unpublished data).

We used Cormack Jolly Seber models to estimate annual survival rates of Orange-bellied Parrots, and explored whether the survival component was constant (i.e., ϕ(.)), or varied with age class (i.e., $\phi_j(.)$ ϕ_a(.), for juveniles and adults, respectively), year (as a linear trend, i.e., ϕ(Year)) and time period (using a dummy variable corresponding to 1995 – 2010, and 2011 – 2017, i.e., ϕ(Period)). We did not fit year as a factor (i.e. to estimate annual survival) because data were too sparse in some years and age classes. In addition to these main effects, we also fitted two age class × year interaction models, where adult survival was either held constant (i.e., ϕ_j (Year) ϕ_a(.)) or allowed to vary as a linear trend with year (i.e., ϕ_j(Year) ϕ_a(Year)). We also fitted year × time period (i.e., ϕ(Year × Period)), age class × time period (i.e., ϕ_j (Period) ϕ_a(Period)), and year × age class × time period (i.e., ϕ_j(Year × Period) ϕ_a(Year × Period)). Recapture probability (*p*) was held constant because of high detection likelihood at feeders (Stojanovic *et al.* 2018) except in the global model. Survival analyses were conducted using RMark (Laake 2013) in R version 3.6.3 (R Development Core Team 2020) as an interface to Program MARK (White and Burnham 1999) and model selection was based on ΔQAICc <2 (Buckland *et al.* 1997). All models were fitted in RMark, but program MARK was used to calculate median ĉ for the global model. We corrected corresponding model selection in RMark for the estimate of median ĉ. Code and data are presented in supplementary materials.

Results

We present data for 797 Orange-bellied Parrots hatched between 1995 and 2017. Five hundred and twenty-two birds (65 %) died in their first year of life and 275 (35 %) died in their second year of life or later, corresponding to a mean lifespan of 1.76 years (range: 0 – 11).

Based on the unconstrained global model, we corrected model selection in RMark by the median ĉ (4.77). We present a full list of survival models ranked by AICc in Table 1. We found no evidence that annual survival rates varied between the two time-periods we considered. The most parsimonious model (based on lowest AICc and fewest parameters) included constant adult survival and a juvenile survival trend over time, with constant recapture probability. The next best model was within two ΔAICc and included juvenile and adult survival trends over time. The best-supported model showed that probability of adult survival was constant at 0.58 over time, but juvenile survival declined from 0.51 to 0.20 over the study. We report effect sizes and standard errors in Figure 1, and based on this model, recapture probability (p) was 0.94 ± 0.01 se.

Table 1. List of nine models fitted to Orange-bellied Parrot survival data from the last known breeding ground between 1995 and 2017. Model notations are as follows: '.' is a constant effect, 'Year' is a linear trend over year, 'ϕ_j' refers to juveniles and 'ϕ_a' is adults, 'time period' relates to 1995 – 2010 and 2011 – 2017 (before/after the time when the wild population size was reduced due to collection of 21 birds for captive breeding). * indicates the preferred models.

Model parameters	**N.Par.**	**QAICc**	**ΔAICc**	**Weight**	**Qdeviance**
ϕ_j(Year) ϕ_a(.)p(.)*	4	419.29	0.00	0.50	67.91
ϕ_j(Year) ϕ_a (Year)p(.)	5	421.23	1.95	0.19	67.84
ϕ_j(Year × Time Period) ϕ_a (Year × Period)p(.)	6	422.30	3.02	0.11	66.89
ϕ_j(Year + Time Period) ϕ_a (Year + Time Period)p(.)	6	422.54	3.25	0.10	67.13
ϕ_j(.) ϕ_a(.)p(.)	3	423.86	4.57	0.05	74.49
ϕ_j(Time Period) ϕ_a (Time Period)p(.)	5	424.03	4.74	0.05	70.64
ϕ(Year)p(.)	3	433.15	13.87	0.00	83.78

ϕ(.)p(.)	2	434.24	14.96	0.00	86.89
ϕ(Time Period)p(.)	3	435.18	15.89	0.00	85.81
ϕ(Year × Time Period)p(.)	5	435.82	16.53	0.00	82.42

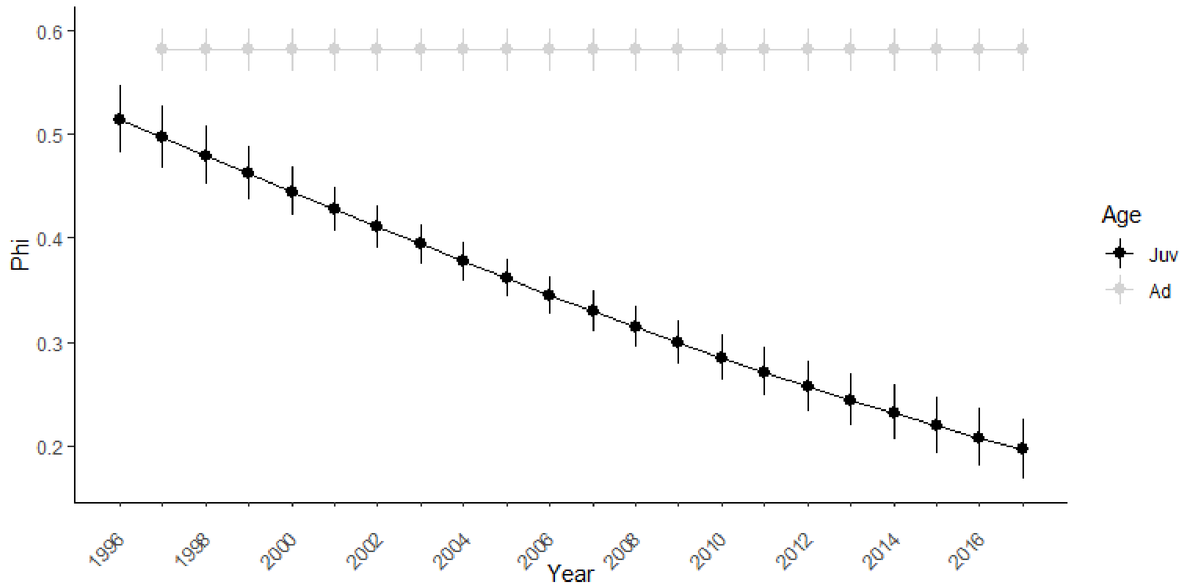

Figure 1. Modeled estimates of survival probabilities (Phi(ϕ) mean ± se) of Orange-bellied Parrots at their last known breeding ground in Tasmania, Australia. Over the entire study, conservation interventions were implemented at the breeding ground, but these actions did not improve the declining survival of juveniles.

Discussion

Survival was age-related in Orange-bellied Parrots, and juvenile survival more than halved over the study period. We found no support for the models that included effects of different survival probabilities in the period before/after collection of juveniles for captive breeding in 2010 artificially reduced wild population size. Instead, the best-supported models only contained effects of age and time. These results reveal a chronic decline of annual survival rates for juveniles, despite intensive conservation efforts at the breeding ground, and suggest that targeted efforts to protect Orange-bellied Parrots in their migration/winter habitats are needed. Our results also contrast with those of an earlier study that found no temporal trend in juvenile survival of this species (Holdsworth *et al.* 2011). Our study supports the prediction of Drechsler *et al*. (1998) that interventions in the breeding ground alone are not enough to recover this species unless threats during migration/winter are concurrently addressed.

There are two possible explanations for our results: (i) the interventions undertaken in Tasmania do not address the primary threats in the breeding ground, or (ii) the interventions do mitigate threats in the breeding ground, but mortality during migration/winter nullifies the benefits. We consider the first explanation less likely given that the aims of the Tasmanian interventions are not intended to improve inter-annual survival but instead focus on local threats to survival and reproductive success within the breeding season (Troy and Hehn 2019). For example, releasing captive-born female parrots has corrected male-biased adult sex ratios and increased reproductive output at Melaleuca (Troy and Hehn 2019), meaning that more nests are initiated and more juveniles enter the population than would have occurred without intervention (which may be evidence that these interventions mitigate the local threats they target). However, survival during migration and winter are likely to be at least partly or entirely independent of interventions in breeding habitat. If the potential second explanation is true, then our results provide empirical support for modeled predictions that ongoing recovery actions in the breeding ground will not improve the conservation status of Orange-bellied Parrots without addressing mortality during migration/winter (Drechsler *et al.* 1998). Testing these hypotheses is crucial because this information will clarify the aspects of life history (breeding, migration, wintering) that should be targeted with new interventions.

Unfortunately, the initial causes of population decline in Orange-bellied Parrots may have been usurped as the principal threats to the species by multiple component Allee effects (Crates *et al.* 2017). For example, a migration component Allee effect may have major implications for a juvenile's first full migration if they rely on flocking for safety in numbers or for the experience afforded by uncommon adults to survive and maintain population-level migration culture (Codling *et al.* 2007). Juveniles may also select poor-quality winter habitat (Crates *et al.* 2017) if they depend on the few remaining adults for habitat selection (Couzin *et al.* 2005; Schmidt *et al.* 2015). Given the low contemporary population size of the species and the sparse but extensive geographic area of their contemporary migration/winter distribution (Department of Environment, Land, Water and Planning 2016), it is unlikely that the survival impacts of historical threats (e.g. deteriorating habitat quality) can be disentangled from the potential recent emergence of Allee effects. Recent activities like releasing captive born parrots in areas of high quality wintering habitat may be effective at overcoming some habitat selection Allee effects in winter, but the survival impacts of these efforts are currently unclear. Other threats like genetic component Allee effects, which may be signaled by low contemporary hatching success and heightened disease vulnerability (Morrison *et al.* 2020; Stojanovic *et al.* 2018), are common in small populations (Heber and Briskie 2010; Whiteman *et al.* 2006) and can affect juvenile survival (Keller *et al.* 2007; Olson *et al.* 2011; Purwandana *et al.* 2015). We suggest that in context of our results and uncertainty about important threats and intervention options (Drechsler 2000; Drechsler *et al.* 1998), increased focus on reducing mortality during migration and in winter should be a conservation priority. However, given that translocation of captive birds to the wild population has demonstrably mitigated some localized threatening processes at

Melaleuca, we suggest these actions should continue while more targeted interventions are trialed in the migration/winter range.

Our study reaffirms that when faced with uncertainty about the factors driving the decline of small populations, it is important to identify and implement management actions that can improve vital demographic rates. Before translocations are implemented it is typically necessary to ensure that the factors driving a species decline are identified and can be corrected (IUCN/SSC 2013), but this has not been achieved for the Orange-bellied Parrot. The migration/winter life history phases pose substantial logistic challenges and identifying where and when to act is difficult. Locations where the species aggregates (e.g. key staging sites for migration and recently used wintering areas) are good starting points for management to maximize habitat availability (e.g. selective weed control and revegetation). However, we caution that sites that are important for Orange-bellied Parrots today may not reflect the utilization of historically important sites identified in earlier work (Loyn *et al.* 1986), both in terms of location and the food plants available/utilized. Prioritizing interventions at places that achieve both short-term goals (e.g. food availability immediately) and long-term habitat restoration goals may be a good starting point. Furthermore, reducing mortality outside the breeding season may provide at least a temporary reprieve from unidentified and unresolved threats during migration/winter. ‘Head starting’ of wild juveniles (holding them in captivity for their first winter before release at Melaleuca the following spring) and ‘ranching’ of captive-born mothers (by releasing them to breed in spring/summer and recapturing them to winter in captivity) have been implemented since 2017 (Troy and Hehn 2019). How these efforts have affected demographic parameters (e.g. annual survival, per capita population growth) is not yet clear.

Our study provides empirical support for the predictions of Drechsler *et al*. (1998) and highlights the need for conservation managers to find new ways of overcoming the challenges of working on small populations that disperse over large areas. Focusing conservation effort where Orange-bellied Parrots aggregate to breed has corrected some threats, but has not improved a key demographic rate. Unfortunately, important alternative actions in migration/wintering habitats, including winter releases of captive-bred birds aimed at attracting migrating individuals to high quality habitat, are still in the trial phase. We hope that our study is a warning to other conservation practitioners to (i) model demographic responses to management actions when uncertainty is high, (ii) act upon those results early and (iii) regularly evaluate the impact of actions on population vital rates.

Ethics

The research utilised data collected by the Tasmanian Department of Primary Industries, Parks, Water and Environment (DPIPWE) during their implementation of the Orange-bellied Parrot Tasmanian Program.

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Survival data were collected by volunteers from DPIPWE and Wildcare Friends of the Orange-bellied Parrot.

Author contribution statement

DS conceived the study, analysed the data and drafted the manuscript. JP analysed the data and reviewed the manuscript. ST conceived the study and reviewed the manuscript. PM, RL and RH reviewed the manuscript.

Competing Interests

The authors have no competing interests.

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