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Predicting post-release establishment using data from multiple reintroductions

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ABSTRACT

For any reintroduction it is important to maximise the probability of released individuals establishing in the target area (settling and surviving to breed). Factors influencing establishment have typically been studied at single sites, making it impossible to assess factors that vary at the site level (e.g. connectivity) or quantify unpredictable variation among sites. Using data from 14 reintroductions of the North Island robin (*Petroica longipes*) to native forest reserves, we show how Bayesian modelling can be used to identify general drivers of establishment and to account for site-to-site variation when making predictions for new sites. High landscape connectivity and high rat tracking rates (a density index) at reintroduction sites were key factors associated with lower individual establishment probabilities. Habitat similarity between source and release sites was also important, as robins sourced from native forest had higher establishment than those from exotic pine forest. Previous predator experience appeared to affect establishment in sites with mammalian predators, as founders sourced from sites with these predators had higher establishment than those from other sites. Our approach can be applied to a wide range of species that are being reintroduced to multiple sites, providing guidance on source and release site selection, efficacy of management interventions, and the numbers of individuals to release to achieve desired initial population sizes. The results are not only applicable to these particular species, but can be used to predict site suitability for reintroductions of species with similar dispersal behaviour or other ecological characteristics.

1. Introduction

Reintroduction is increasingly used to re-establish populations of threatened species within their historical ranges (Sarrazin and Barbault, 1996; Seddon et al., 2007). However, many reintroduction attempts are unsuccessful (Griffith et al., 1989; Sarrazin, 2007; Wolf et al., 1996) and the underlying causes of failure are rarely well understood (Dickens et al., 2010; Fischer and Lindenmayer, 2000; Letty et al., 2007). Analysis of factors influencing reintroduction outcomes is therefore important to improve the success of future reintroduction programmes (Ewen and Armstrong, 2007; Le Gouar et al., 2012; Sarrazin and Barbault, 1996; Sutherland et al., 2010).

The two key phases affecting the dynamics of reintroduced populations are establishment and persistence (Armstrong and Seddon, 2008). While the ultimate goal of any reintroduction is population persistence (Seddon, 1999), this is only achievable if the population survives the establishment phase. There is often elevated mortality (e.g. Calenge et al., 2005; Kreger et al., 2006) and dispersal (e.g. Moehrenschlager and Macdonald, 2003; Tweed et al., 2003) immediately after release, meaning that reintroductions can fail during the establishment phase even if conditions at the new site would enable persistence once established (Armstrong and Seddon, 2008). Dispersal and mortality can have similar costs because individuals who disperse and settle away from the reintroduction area will not contribute demographically or genetically to the population (Le Gouar et al., 2012).

Because individuals are lost soon after release, the effective initial population size, commonly defined as the number of individuals that survive to the breeding season, is often much lower than the number of individuals released (Armstrong and Seddon, 2008; Armstrong and Wittmer, 2011). This in turn can exacerbate problems faced by small populations, including demographic stochasticity, environmental stochasticity, Allee effects and loss of heterozygosity. Maximising initial population size is therefore an important consideration for any reintroduction.

The most obvious approach to increase the initial population size is to release more individuals. The benefit of larger release groups is widely cited in the literature (e.g. Deredec and Courchamp, 2007; Griffith et al., 1989; Wolf et al., 1998). However, releasing more individuals has a trade-off with impact on the source population (Armstrong and Wittmer, 2011) and can also have financial and logistical repercussions. There may also be a

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trade-off at an individual and ethical level, as larger founder groups can result in more individuals being lost due to post-release dispersal or mortality.

An alternative to releasing more individuals is taking measures to reduce post-release mortality or dispersal, thereby increasing the probability of founders settling in the reintroduction area. Population establishment is dependent on the probability of reintroduced individuals establishing at the new site, so understanding the key determinants of individual establishment is important for reintroduction success. Post-release survival and dispersal can be affected by various aspects of a reintroduction; including the translocation process (e.g. release strategy, Devineau et al., 2011), characteristics of the individuals involved (e.g. age or sex, Masuda and Jamieson, 2012; Moehrenschlager and Macdonald, 2003), conditions at the reintroduction site (e.g. predator levels, Moorhouse et al., 2009), similarity between release and source sites (Lawrence and Kaye, 2011; Roe et al., 2010; Stamps and Swaisgood, 2007), and the habitat matrix surrounding the reintroduction site (La Morgia et al., 2011). Establishment of reintroduced individuals can therefore be facilitated at various levels; although the most appropriate and effective measures will depend on the species in question. For example, riparian brush rabbits (*Sylvilagus bachmani riparius*) held longer in enclosures before release had higher post-release survival (Hamilton et al., 2010), whereas delayed release of stitchbirds (*Notiomystis cincta*) lowered survival compared to birds released immediately (Castro et al., 1995).

Analysis of data collected after reintroduction can provide crucial information about factors affecting establishment of individuals post-release. Importantly, modelled relationships can then be used to make predictions before new reintroductions take place, providing guidance to managers about site suitability and appropriate measures to improve reintroduction success. However, identification of factors influencing post-release establishment is often based on data from single sites (e.g. Bernardo et al., 2011; Jõgar and Moora, 2008; Roe et al., 2010; Tweed et al., 2003). While these studies can provide valuable insights for the site in question, factors influencing success throughout a species' range may not be apparent in results from a single site (Jachowski et al., 2011). Using data from reintroduction attempts at multiple sites provides more certainty that identified relationships are general (Johnson, 2002) and therefore applicable to other sites. Analyses of data from single sites are also limited to factors that can be manipulated within that site (for example, release techniques or supplementary feeding). Potentially more important factors, such as habitat quality or connectivity, only vary among sites so analysing data from multiple sites is necessary to evaluate their influence on reintroduction outcomes.

There are numerous examples where single species have been released into multiple sites for conservation purposes. In New Zealand and Australia, more than 40 vertebrate species have each been translocated to at least five different sites (e.g. http://rsg-oceania.squarespace.com/nz/; Short, 2009). In southern Africa, most large herbivores (e.g. Linklater et al., 2011; Van Houtan et al., 2009) and carnivores (e.g. Hayward et al., 2007) have been reintroduced to multiple sites. There are also examples from other parts of the world, including Griffon vultures (*Gyps fulvus*) in France (Le Gouar et al., 2008) and black-footed ferrets (*Mustela nigripes*) in North America (Jachowski et al., 2011). These multiple releases create a unique opportunity to integrate data among sites to identify the key influences on reintroduction outcomes, while also accounting for any unexplained site-to-site variation in population parameters. The results obtained would not only be applicable to the species that have already been reintroduced to multiple sites, but could be used to predict site suitability for reintroductions of species with similar dispersal behaviour or other ecological characteristics.

We present an approach whereby data from multiple reintroduced populations are integrated into a Bayesian hierarchical model to identify important factors influencing post-release establishment. We model establishment data for North Island robins (*Petroica longipes*) reintroduced to 14 sites, and show how the resulting model can be used to make predictions for a candidate reintroduction site under alternative management scenarios. The strength of our approach is the ability to model the general influences on establishment while accounting for site-to-site variation, thereby enhancing predictive capability and enabling targeted management to improve reintroduction success.

2. Methods

2.1. Species and reintroductions

The North Island robin is a small (26–32 g) insectivorous forest passerine endemic to New Zealand. The species was historically found over the entire North Island, but is now restricted to native forest remnants and exotic plantations in the central North Island, as well as some offshore islands (Higgins and Peter, 2002). Robins are susceptible to predation, primarily by exotic ship rats (*Rattus rattus*) (Brown, 1997; Powlesland et al., 1999), but also other exotic mammals such as stoats (*Mustela erminea*) and native avian predators such as morepork owls (*Ninox novaeseelandiae*). Their breeding season is generally from early September to February, and juveniles become sexually mature by the start of the breeding season after that in which they fledge.

North Island robins were reintroduced to 15 different sites (31–1100 ha forested area) between 1997 and 2007 and analysable data were available for 14 of these (Table 1). Thirteen of the sites were on the North Island and two (Glenfern, Windy Hill) were on Great Barrier Island, a ca. 28,500 ha island off the north-east of the North Island. Reintroductions always occurred between March and August. Pre-release monitoring was conducted at all sites prior to reintroduction and no robins were found. Birds were caught from the wild and were released immediately on arrival at the release site. Robins typically undergo a period of dispersal post-release, and become sedentary once pairs and territories are established in the breeding season. All sites, including the proposed site, were managed to control exotic mammalian predators. At the time of reintroduction, two sites were fenced to exclude mammalian predators, which were eradicated after fencing, hence those species were expected to be absent. Another site was fenced but had openings for vehicle access, so mammalian predators remained present. All reintroductions were to areas of native forest, and birds could potentially disperse into unmanaged forest in the surrounding landscape. One site also had an exotic pine forest plantation within its boundary.

2.2. Data collection

We compiled data to assess the probability of released individuals establishing at each reintroduction site, where "establishment" is defined as surviving and remaining at the site until the start of the breeding season (late August). We specifically modelled return rates, which are the proportions of released individuals that remain at the site and are detected (Cam et al., 2005; Martin et al., 1995), as it was impossible to separately estimate establishment and detection probabilities from the data available for some sites. We included data on return rates from initial reintroduction attempts only, so any supplementary translocations in subsequent years were excluded from our analysis. All birds were individually colour banded prior to release, and data on the number of birds released and post-release sightings of individuals were available