This is a peer reviewed version of the following article: Griffiths, A. D., Rankmore, B., Brennan, K., & Woinarski, J. C. Z. (2017). Demographic evaluation of translocating the threatened northern quoll to two Australian islands. *Wildlife Research*, 44(3), 238-247; which has been published in final form at https://doi.org/10.1071/WR16165

Escape from the toads: evaluating translocation success of the threatened northern quoll to two Australian islands

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Running title: Island translocation of the northern quoll

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Abstract

Many threats causing decline in threatened species are difficult to control effectively *in situ*. For such species, translocation (to an area with reduced or no incidence of such threats) is increasingly used as a main conservation management approach. However, assessment of success in translocation programs may be difficult, because population trends (and their conservation implications) may vary markedly across time, and between comparable populations at different translocation sites. Here we describe a translocation case study that assesses and compares a range of demographic (and related) parameters simultaneously at two release sites, and across a long enough period to encompass establishment, growth and regulation phases. . The subject species is the northern quoll *Dasyurus hallucatus*, which has suffered very rapid and severe declines on the Australian mainland due to the uncontrolled and ongoing spread of the introduced cane toad *Rhinella marina*. We translocated 64 northern quolls to two islands (Astell and Pobassoo) in 2003, with translocation sites selected based on a comprehensive site selection process and consultation with Aboriginal landowners. Monitoring based on capture-mark-recapture methods occurred at regular intervals until 2009 followed by a one-off survey in 2014 to estimate abundance, apparent survival, recruitment and body condition. Broadly, demographic trends were similar across the two islands. Relative abundance (trap success) increased exponentially in the first three years, declined and then stabilised in subsequent years. The population of female northern quolls on the 12.7 km^2 Astell Island peaked in 2006 with an estimate of 3,640 (95% CI 3022 – 4257) and in 2014 was estimated to be 2193 (95% CI 1920 – 2467). On the 3.9 km^2 Pobassoo Island, the population peaked in 2007 with 617 (95% CI 531 – 703) females and in 2014 was estimated to be 451 (95% CI 359 – 543). Apparent survival and body condition decreased significantly following the population peak, possibly because the islands' carrying capacity was exceeded. Compared to mainland populations, both apparent survival and recruitment were higher in the translocated populations, possibly due to the absence of predators and presence of high quality habitat. We assessed the success of the program against established criteria and discuss the applicability of this study to translocations in other systems.

Key words

density-dependence, success criteria, capture-mark-recapture, recruitment, survival, abundance

Introduction

Translocation is the intentional movement of organisms from one area to another, in an attempt to establish or re-establish viable, free-ranging populations of imperilled species (IUCN, 1998) and it is an important and increasingly applied tool to conserve threatened species (Armstrong *et al.*, 2015). However, many translocations have failed or been characterised by poor conception (Griffith *et al.*, 1989; Wolf, Garland & Griffith, 1998). From analyses of the fate of many translocations, several factors are recognised to contribute to the success of these programs: the number of animals released, habitat quality, the location of the release area and the threat environment in the release sites(s) (Fischer & Lindenmayer, 2000; Griffith *et al.*, 1989; Perez *et al.*, 2012; Sheean, Manning & Lindenmayer, 2012). From such reviews, , there have been attempts to adopt a more integrated, standardised and theoretically sound approach to translocations (Armstrong & Seddon, 2008; Seddon, Armstrong & Maloney, 2007), including defining and evaluating the success of programs (Perez *et al.*, 2012; Robert *et al.*, 2015).

There are no general and broadly accepted success criteria for translocations and this limits our understanding of the factors that contribute to success or failure of translocations (Robert *et al.*, 2015). In part, the lack of universal success criteria is due to the diverse range of translocated species facing different threats in different environments. The establishment of viable population is commonly used to define success (e.g.Morris *et al.*, 2015), but it has been argued that a viability criterion that measures future potential rather than current state should be used to define success (Mace *et al.*, 2008). Recent research using IUCN Red List criteria, a globally accepted system for classification of extinction risk (Butchart *et al.*, 2005), indicate that two criteria (i.e., Criteria D - Population Size and E – Risk of Extinction) can be used to define long-term reintroduction success (Robert *et al.*, 2015), but that modifications will need to be made to the assessment criteria (Shier, 2015).

Australian mammals have had an extraordinary extinction rate over the last 200 years (Short & Smith, 1994; Woinarski, Burbidge & Harrison, 2014): at least 30 Australian terrestrial mammal species have become extinct over this period, representing well over a third of the world's recent mammal extinctions (Sattler & Creighton, 2002). Many threatening processes have been linked to this decline, including predation by introduced vertebrates, habitat loss, habitat modification from the introduction of exotic herbivores, disease and changed fire regimes (Burbidge & McKenzie, 1989; Johnson, 2006; Morton, 1990; Short & Smith, 1994; Smith & Quin, 1996; Woinarski & Braithwaite, 1990). Translocations have been used extensively to help conserve Australia's mammal fauna and success has typically focussed on relatively simple measures of population persistence or trends (Clayton *et al.*, 2014; Fischer & Lindenmayer, 2000; Morris *et al.*, 2015).

Here, we evaluate the success of an island translocation project for the northern quoll *Dasyurus hallucatus*. This carnivorous marsupial is extremely susceptible to toxins ingested during predation on the exotic cane toad *Rhinella marina*. Consequently, populations of northern quolls have been extirpated in many areas of mainland Australia following the invasion of those areas by cane toads (Burnett, 1997; Shine, 2010). Cane toads are spreading rapidly across mainland northern Australia and their eventual range is likely to encompass almost entirely that of the northern quoll (Kearney *et al.*, 2008; Sutherst, Floyd & Maywald, 1996). Cane toads have also naturally colonised (and may be inadvertently introduced to) some islands, where they have also caused substantial local losses of some predatory animals, including northern quoll (Woinarski *et al.*, 2011). In the medium term, it is unlikely that there will be any mechanism available to effectively slow the spread or reduce the population of cane toads in northern Australia. Because of these factors, the northern quoll is listed as endangered under Australia's *Environment Protection and Biodiversity Conservation Act*.

The broad objective of this translocation program was to establish at least one secure island population of northern quolls that would be likely to be viable for many decades. Importantly, translocations to islands provide a unique opportunity to undertake natural experiments, as the constellation of threats and resources may vary markedly between islands used as translocation sites. In this case, given some risk of catastrophe (notably invasion of the translocated site by cane toads), we use two separate translocation destinations, and hence can compare population trends across these two sites. For both island sites, we simultaneously monitored (and hence compare) demographic (and related) parameters over a long enough period to encompass the establishment, growth and regulation phases of the translocation program (sensu Sarazin 2007). We also compare these parameters with those available for this species in source (mainland) populations.

Materials and methods

Island selection

Broadly following established IUCN criteria for translocations (IUCN/SSC, 2013), we developed a candidate set of potentially suitable Northern Territory islands. The selection criteria included adequate size for persistence for at least 30 years (> 1 km^2, but preferably > 10 km^2); occurrence of suitable habitat (areas of rugged sandstone); absence of human habitation; relatively low risk of cane toad colonisation (limited visitation by humans, distant from mainland, not in the outflow area of mainland rivers); moderate accessibility; and absence of other conservation values susceptible to predation or competition from translocated quolls. These criteria restricted the candidate set to about ten islands. The final selection was made after a period of detailed consultation with the islands' Aboriginal landowners.

Two islands were selected as translocation sites: Astell (area =1268 ha, max. elevation = 74 m and distance to mainland = 5.4 km) and Pobassoo (area =392 ha, max. elevation = 78 m and distance to mainland = 2.3 km), both in the English Company group off north-eastern Arnhem Land (Fig. 1). Both islands are rugged and dominated by eucalypt (particularly *Eucalyptus tetrodonta*) woodlands, with small fringing areas of coastal vine thicket and mangroves (Woinarski *et al.*, 2000).

Translocations may have impacts on other species present at the destination site, and these impacts may be particularly severe when the translocated species is a predator such as the northern quoll. Previous detailed surveys had indicated that these islands did not support any plant (Woinarski *et al.*, 2000), ant (Woinarski, Reichel & Andersen, 1998), herpetofauna (Woinarski *et al.*, 1999a), bird (Woinarski *et al.*, 2001) or mammal (Woinarski *et al.*, 1999b) species of conservation significance likely to be affected by a quoll translocation. Indeed, there were no marsupials and only one species of rodent (*Hydromys chrysogaster*) recorded from these two islands (Woinarski *et al.*, 1999b). Furthermore, unlike many islands in this group they also lacked significant nesting sites for marine turtles (Chatto & Baker, 2008) and shorebirds (Chatto, 2003).

Founder population and translocation procedure

Founder stock was drawn from a range of sites across mainland Northern Territory, particularly from lowland areas on the Darwin rural fringe and Kakadu National Park. Collections were timed to immediately precede the cane toad invasion front, and coincided with the time when juvenile quolls became independent (February-March 2003). This cohort were considered most likely to adapt to translocation, especially since adults (especially males) have a short life expectancy. Animals were collected using Elliott and cage traps and held in purpose-built enclosures for 1 to 9 days before being transported to translocation sites. Nineteen animals (8 males; 11 females) were released at Pobassoo Island in February 2003, and then 45 animals (11 males; 34 females) were released at Astell Island in March 2003.

Monitoring

The translocated quoll populations were monitored on ten occasions following release, with all surveys conducted in collaboration with the islands' Aboriginal owners. The earlier surveys (2003 to 2005) occurred in the early to mid-Dry season (April to July). Subsequent surveys (2006 to 2009) took place in October or December, when adult males were largely absent and the weaned young of the year were entering the trappable population. We also conducted a survey in October 2014 to confirm the persistence of the quoll populations.

In the earlier monitoring surveys, sampling used either (or both) grids (an array of 7 ×10 traps, spaced 20 m apart) or transects (a line of 10 traps spaced 20 m apart) over 3 or 5 nights with multiple surveys occurring in some years. In later surveys, permanent trapping grids (an array of 5 × 5 traps, spaced 20 m apart) were established and sampled over five nights, with ten grids used on Astell and eight on Pobassoo. The survey in 2014 used half the permanent grids on each island. All traps used were cage traps (65 cm ×15 cm ×15 cm), baited with a mixture of peanut butter, honey and oats. Traps were set and baited in the late afternoon and checked (and then closed) in the early morning. For every quoll caught, we recorded its sex, body mass (g) and head length (mm). We marked all individuals by microchip (Destron PIT tags) except in the 2007 survey where eartags (Model 1005-1 self-piercing ear tag, National Band and Tag Co.) were used. All quolls were released at the grid immediately after processing.

Statistical analysis

We used two methods to assess changes in abundance. The first used the rate of trap success (captures per 100 trap nights) of all individuals (males and females) over all surveys. We used generalised linear regression to estimate long-term linear trend in northern quoll trap success for both island populations. The response variable was log transformed trap success and island and year were fixed-effects. In years when two surveys occurred we pooled the data into a single value for each island. To account for temporal correlation we used a first order autocorrelation term and a log link (Chaloupka & Limpus, 2001), and models were fitted using maximum likelihood estimation to allow comparison between models with different fixed effects (Pinheiro & Bates, 2000).

The second method used capture-mark-recapture data from the later surveys (2006 to 2009 and 2014) to estimate the density of female northern quolls. We used the closed-captures component within the Pradel Robust Design model for the 2006 to 2009 and a single closed-capture model for the 2014 survey using Program MARK v8.0 (White & Burnham, 1999). Using the full likelihood model we estimated the probability of initial capture (*p*) and the probability of recapture (*c*) of female quolls over five nights (Williams, Conroy & Nichols, 2002). We constructed a candidate set of models that included parameters representing no variation (null), linear trend, year and island for both *p* and *c*, which were combined with constant survival and recruitment models. Model selection was based on Akaike's Information Criterion, corrected for small sample size (AIC_c: (Burnham & Anderson, 2002). The likelihood of each model, relative to others in the candidate set, was estimated with AIC_c

weights (*w*) and models were ranked according to this measure (Burnham & Anderson, 2002). To estimate density we calculated the effective trapping area of the 0.64 ha trapping grid by adding a boundary area around its perimeter of half of the average home range size of an individual (Williams *et al.*, 2002). This resulted in an effective trapping area of 5.63 ha (based on a home range of 2.3 ha for females in rocky habitat (Schmitt *et al.*, 1989). We then divided the population estimate by the effective trapping area to estimate density of female quolls on each island for each year.

Body condition for each individual was estimated using the scaled mass index (Peig & Green, 2009; Peig & Green, 2010). The index is the predicted body mass for individual *i* when the linear body measurement (head length, mm) is standardised to the mean value of the study population and scaled to the slope of standardised major axis regression of body mass and head length. We used linear regression model differences in scaled mass index among years (2005 to 2009 and 2014), between the two islands and sexes and compared them using AIC model selection.

To estimate apparent annual survival and recruitment between 2006 and 2009 we used Pradel temporal symmetry Robust Design models (Pradel, 1996) using Program MARK v8.0 (White & Burnham, 1999). By analysing the encounter history of all marked individuals in the population going backwards in time, it is possible to estimate the probability of an individual entering the population. Apparent survival (φ) is the probability that an animal that has not emigrated from the population is alive at time $i + 1$ given it was alive at time *i* (Williams *et al.*, 2002). Recruitment (*f*) is defined as a per capita recruitment probability (i.e., net new animals per animal alive at occasion *i* entering the marked population between occasions *i* and $i + 1$). The link function was logit for survival and log for recruitment. The temporal symmetry model assumes the area sampled does not change during the study and all animals have some probability of being captured, there is no response to being trapped and there is little difference among animals in being captured. In addition, the Pradel model is an extension of the Cormack-Jolly-Seber (CJS) model that assumes that every marked animal has the same probability of survival, tags are not lost or misidentified, emigration is permanent and the fate of each animal is independent of other animals (Williams *et al.*, 2002).

We analysed the effect of the two islands, annual rainfall, density dependence, temporal variation and body size on apparent survival and recruitment based on the method of linear modelling of explanatory covariates originally proposed by Lebreton *et al* (1992) (see Appendix A). We modelled rainfall (mm) as a time-specific covariate over the interval between two primary periods. Temporal variations were represented by year and linear trend, and were expressed as time-specific covariates. Density dependence was modelled as a time-specific covariate and we used the total number of quolls captured on each island from the previous year. Body mass was modelled as an individual covariate. We constructed a priori candidate sets of models from these variables based on known biology and the published ecological literature, comprising additive and, for some models, interactive combinations. Each temporal covariate was scaled to range between zero and positive and negative one. The best capture-recapture model was used when comparing different apparent survival and recruitment models. If the 95% confidence interval for the slope of the logit- or log-explanatory covariate (β) did not include zero, the relationship was considered statistically significant (Williams *et al.*, 2002). There is no goodness-of-fit test for the robust-design model, therefore we used separate tests for the open and closed parts of the model. For the open model we collapsed each primary period and performed a median goodness-of-fit on a model containing all temporal covariates with the CJS model in Program MARK (Cooch & White, 2014).

Results

Survey effort and goodness-of-fit

In total, we recorded 2,327 northern quolls captures from 13,507 trap nights across all sampling periods. The capture rate on Astell Island (1523 captures from 7,776 trap nights: 19.6% trap success) was higher than for Pobassoo (804 captures from 6,431 trap nights: 12.5% trap success). For the capture-mark-recapture (CMR) data, there was no evidence of over-dispersion: the median $\hat{c}$ test estimated a $\hat{c}$ of 1.16 and therefore we made no adjustments to $\hat{c}$ in the CMR modelling.

Population trend

From the initial release of 64 northern quolls in 2003, there was a rapid increase in trap success for three years. On Astell Island, trap success peaked in 2005 whereas on Pobassoo Island it remained high from 2005 to 2007 and then decreased (Figure 2a). At both islands, trap success stabilised at a reduced level in later years. Trap success was significantly higher on Astell than Pobassoo Island (Table 1). The best-supported model for variation in trap success contained the factor Island and a quadratic linear trend, representing non-linear change in trap success over the seven years (Table 1).

The density estimates of female northern quolls followed a similar pattern to trap success: a very rapid increase in density of female northern quolls for a few years after the introductions and then a decline and finally stability (Figure 2b). Density was higher on Astell than on Pobassoo Island and densities decreased on both islands after 2007 (Figure 2b). Initial capture (p) and recapture probabilities (c) differed over time and between islands (Appendix Table A1). For the one-off survey in 2014 the density on Astell Island was similar to that in 2009 but on Pobassoo the 2014 density was higher than in 2009. Extrapolation of the density estimates showed that the population of female northern quolls on Astell Island peaked in

2006 with an estimate of 3640 (95% CI 3022 – 4257) and in 2014 it was 2193 (95% CI 1920 – 2467). On Pobassoo Island the population peaked in 2007 with 617 (95% CI 531 – 703) and in 2014 was estimated to be 451 (95% CI 359 – 543).

Body condition (represented by scaled mass index) varied considerably across the six years and also differed between the two islands. The best-supported model for variation in body condition contained only the parameter year and the next best model contained the interaction between parameters year and island suggesting a different pattern in body condition of quolls over time between Astell and Pobassoo Islands (Table 2). Inspection of model coefficients showed a significant decrease in body condition on Astell Island for the years 2006 to 2008 compared to 2005 (Fig 3a). A similar but less pronounced pattern was observed on Pobassoo Island with body condition being significantly lower in 2006 and 2007 but not 2008 (Fig 3b).

Apparent survival and recruitment

Apparent survival of female quolls varied over the period 2006-2009 and corresponded to the population peak and subsequent decline. The best-supported model contained parameters representing density dependence and the interaction of year and body mass (Table 3). Inspection of beta coefficients for the top ranked model showed that higher number of northern quolls in the previous year was negatively related to apparent survival ($\beta = -1.17$, 95% CI: -1.50 to -0.84). In addition, body mass influenced apparent survival. In 2006-2007 there was a negative but non-significant relationship ($\beta = -0.12$, 95% CI: -0.45 to 0.21) and in 2007-2008 there was a significant negative relationship ($\beta = -0.84$, 95% CI: -1.21 to -0.46). However, in 2008-2009 the relationship was positive and non-significant ($\beta = 0.24$, 95% CI: -13 to 0.62). Model-averaged estimates of apparent survival on Astell and Pobassoo were 0.42 (95%CI: 0.34-0.50) and 0.28 (95%CI: 0.22-0.34) in 2006-2007, 0.10 (95%CI: 0.06-0.15) and 0.12 (95%CI: 0.08-0.18) in 2007-2008 and 0.63 (95%CI: 0.51-0.73) and 0.57 (95%CI: 0.46-0.67) in 2008-2009.

Recruitment of female northern quolls varied over the four years of monitoring. The best-supported model included terms that related to rainfall over the previous 12 months. Inspection of beta coefficients showed that increasing rainfall had a negative but non-significant effect on recruitment in (β = -0.36, 95% CI: -0.7 to 0.4). There was no evidence that recruitment differed between the two island populations (β = 0.15, 95% CI: -0.08 to 0.40). Model-averaged estimates of recruitment on Astell and Pobassoo were 0.56 (95%CI: 0.46-0.65) and 0.55 (95%CI: 0.43-0.73) in 2006-2007, 0.61 (95%CI: 0.49-0.73) and 0.59 (95%CI: 0.43-0.73) in 2007-2008, and 0.41 (95%CI: 0.24-0.61) and 0.40 (95%CI: 0.25-0.58) in 2008-2009.

Discussion

The quoll translocation program successfully met the criteria for translocations recommended in a recent global review of translocation protocols (Perez *et al.*, 2012) (Table 4). As described in this paper, the program to date has been successful in establishing two independent introduced populations, with those populations increasing and persisting to a level markedly higher than the founder stock. This can be viewed as the populations reaching their regulation phase after going through establishment and growth phases (Sarrazin & Barbault, 1996). Using IUCN Criterion D (total number of mature individuals), the Astell Island population would be classed as Least Concern while the Pobassoo Island population would be classed as Vulnerable as the abundance estimate in 2014 was below 1000 individuals (Robert *et al.*, 2015). However, given the smaller area of Pobassoo Island it would be unlikely that the long-term carrying capacity of the island could support a population of adult females >1000 individuals. Therefore, the direct application of Criterion D in this situation is not appropriate.

The results from monitoring provide a number of important insights into population dynamics of northern quoll in a predator and threat-free environment. The translocated island populations exhibited extraordinary rates of increase in the initial years followed by a decline and stabilisation of the population. There was evidence suggesting negative density-dependence within five years of the initial translocation with apparent survival decreasing with increasing abundance of northern quolls. In age-structured populations of large herbivores, recruitment (juvenile survival, proportion of females breeding) is considered more sensitive to density dependence than adult survival (Gaillard, Festa-Bianchet & Yoccoz, 1998). In this study, there was little variation in recruitment and lower survival of females with larger body mass during the years with high density. Female northern quolls are relatively short-lived (maximum life expectancy 4 years), have high fecundity (average litter size of six young per year) and the majority of females reproduce each year (Braithwaite & Griffiths, 1994; Oakwood, 2000). It is plausible that the population increased initially in response to lack of predation and an unexploited abundant food resource, but then declined to stabilise at a lower level due to the impacts of the quoll-induced food depletion. Variation among years in body condition supports this hypothesis as the scaled mass index was lowest when relative abundance (i.e., trap success) was highest in 2007 then increased when relative abundance levelled off in 2009 and 2014. The observed stabilisation of the populations from 2006 to 2014 suggests there has been no significant and sustained habitat degradation. Availability of food resources has probably declined (unsurprisingly given high densities of a novel predator) but now stabilised.

There were differences between the two island populations and mainland populations that provide further insight into the population dynamics of the species. Both apparent survival and recruitment were higher on both islands than for (pre-cane toad) mainland sites. Comparable estimates of maximum apparent annual survival for female quolls on the mainland (Kakadu National Park) was 0.49 (Griffiths & Brook, 2015) compared to 0.63 in this study. Per capita recruitment rates were higher in the translocated populations on both islands compared to the mainland. Recruitment rates on both islands ranged from 0.40 to 0.60 compared to 0.25 to 0.35 on the mainland (at Kakadu) (Griffiths & Brook, 2015). This suggests that the absence of predators and possibly higher habitat quality led to increased survival and recruitment.

The current study raises some important issues for future management of the translocated populations The translocation was timely as alternative conservation management options such as building cane toad-proof enclosures were either too expensive or risky (Brook & Whitehead, 2005) or had not been developed (O'Donnell, Webb & Shine, 2010). As evident in the fate of important populations of other mammal species on some other Northern Territory islands following the spread of introduced species (Woinarski *et al.*, 2011), the most critical factor relates to biosecurity, particularly ensuring that cane toads do not colonise or are introduced to these two islands. To some extent, this will require the ongoing involvement and interest of the islands' Aboriginal owners. A feature of this program to date has been the full involvement of these landowners, and these landowners have developed a strong sense of responsibility for these translocated populations. Reintroduction of some individuals from the translocated island populations to the mainland is another management option, although this is unlikely to be successful while the principal mainland threat remains unabated.

Translocation and reintroduction are becoming more common and documenting the success or failure of these programs using universal criteria will lead to better outcomes in the future (e.g., Germano *et al.*, 2014). Given the relatively small size of the founder populations, it is likely that the high intrinsic growth rate of the species coupled with predator- and toad-free translocation sites contributed to both populations reaching the regulation phase. Our results also highlight the importance of islands in translocation and Morris *et al.* (2015) showed that island translocations have been more successful than translocations on the mainland. Furthermore, comprehensive monitoring allowed for a thorough evaluation of the translocation program.

Acknowledgements

Funding was provided by the Natural Heritage Trust, Australian Geographic, Threatened Species Network of Worldwide Fund for Nature, and the Tropical Savannas Cooperative Research Centre. The Territory Wildlife Park provided temporary housing of the founder population. The endorsement, interest and cooperation of traditional owners of the areas from which the quoll's founder populations as well as the destination islands was vital to the success of this project, and we acknowledge particularly the Gumarr Marthakal and Dhimurru ranger groups, Terry Yumbulul and Bruce Lirrwa Ganambarr. The Gumarr Marthakal ranger group has maintained an enduring involvement in this program. In addition, the Northern Land Council, Kakadu National Park, and Arafura Pearls provided much needed support, and many people contributed to fieldwork over the course of the project. All research was conducted under Charles Darwin University Animal Ethics Permits: A03017, A13026 and A01001.

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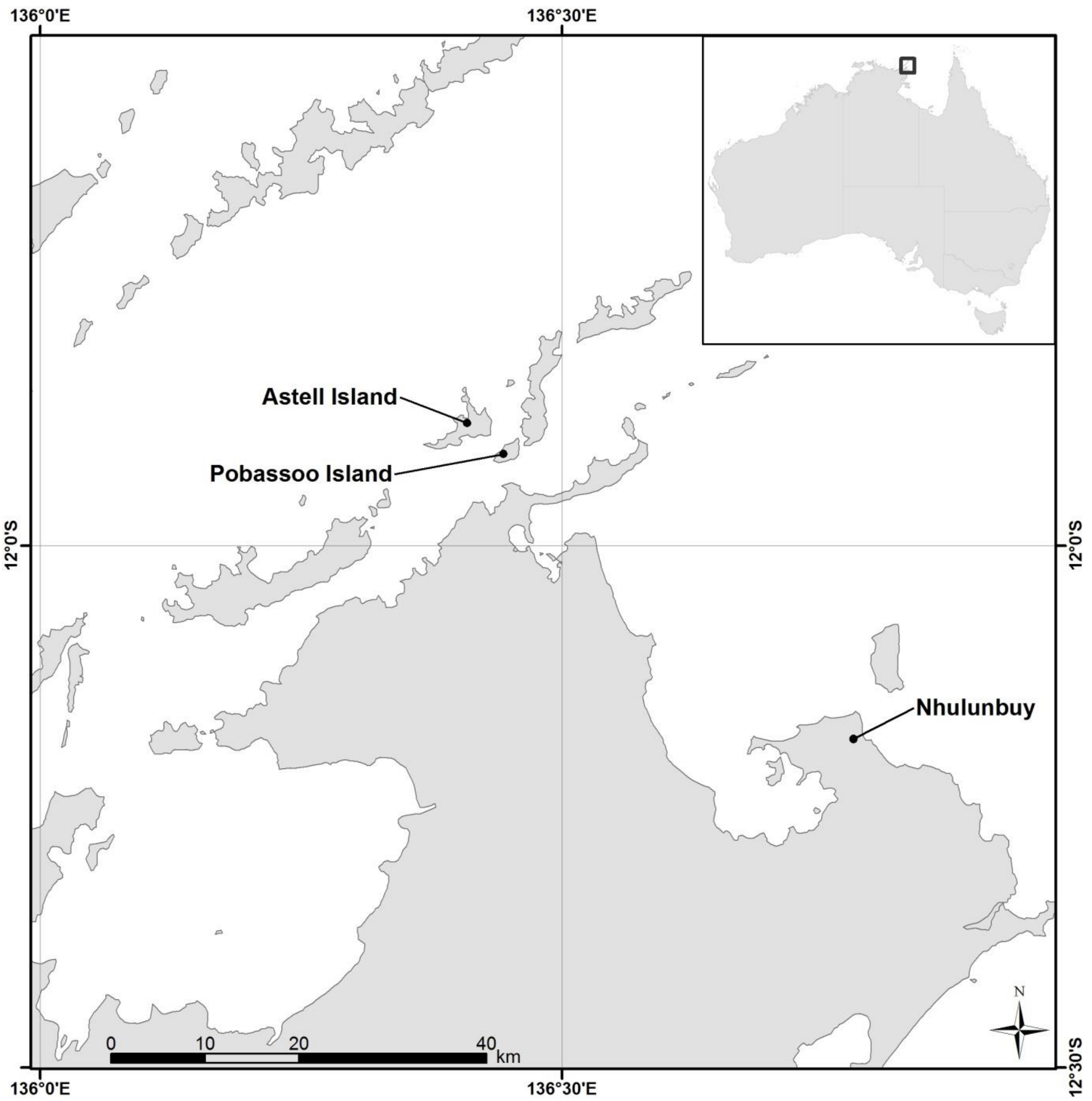

Fig 1 Location of Astell and Pobassoo Islands, Northern Territory.

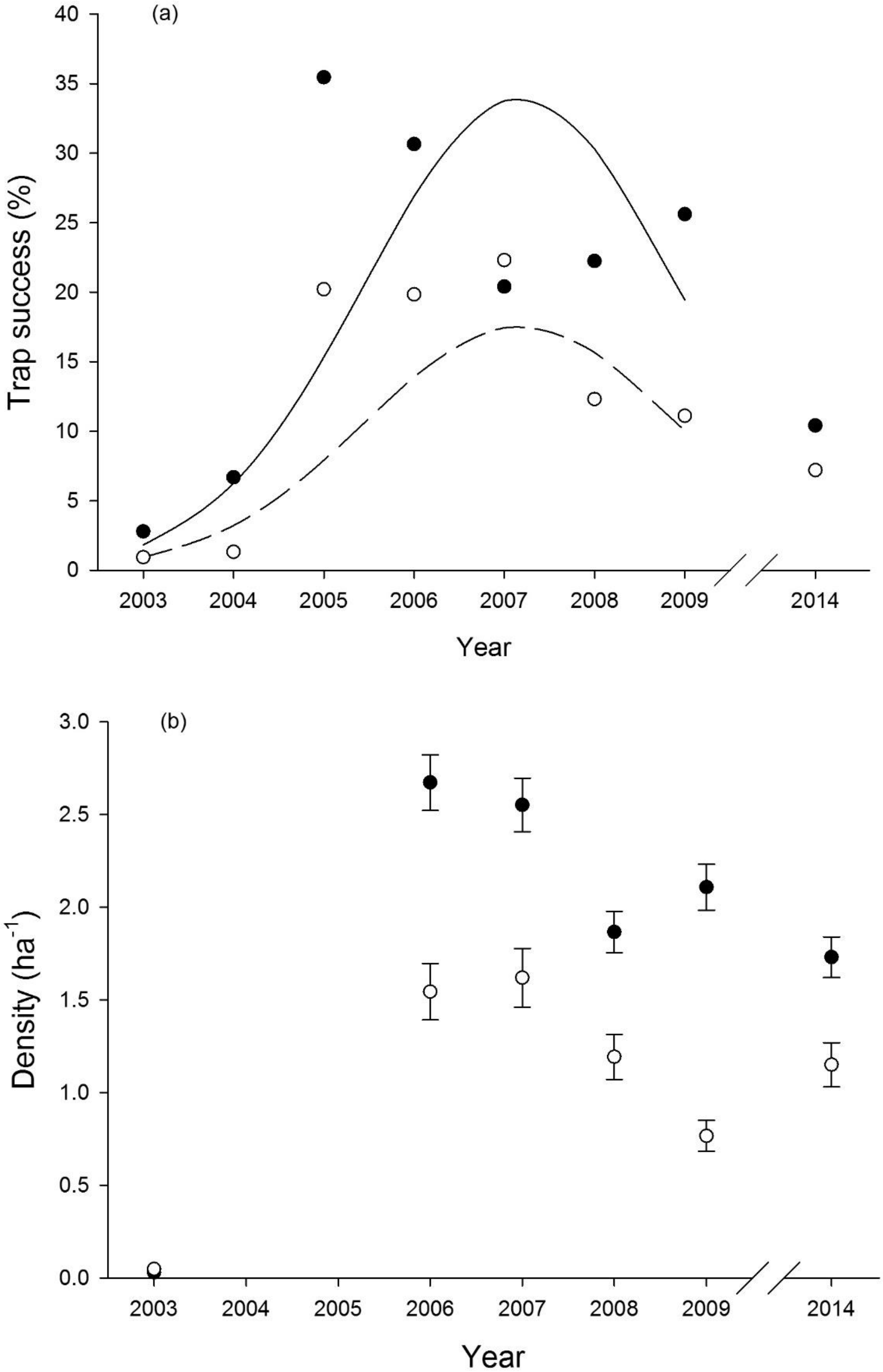

Fig 2 Time series of (a) trap success of all northern quolls and (b) density of female northern quolls on the two islands (open circles – Pobassoo, closed circles – Astell). Error bars are one standard error.

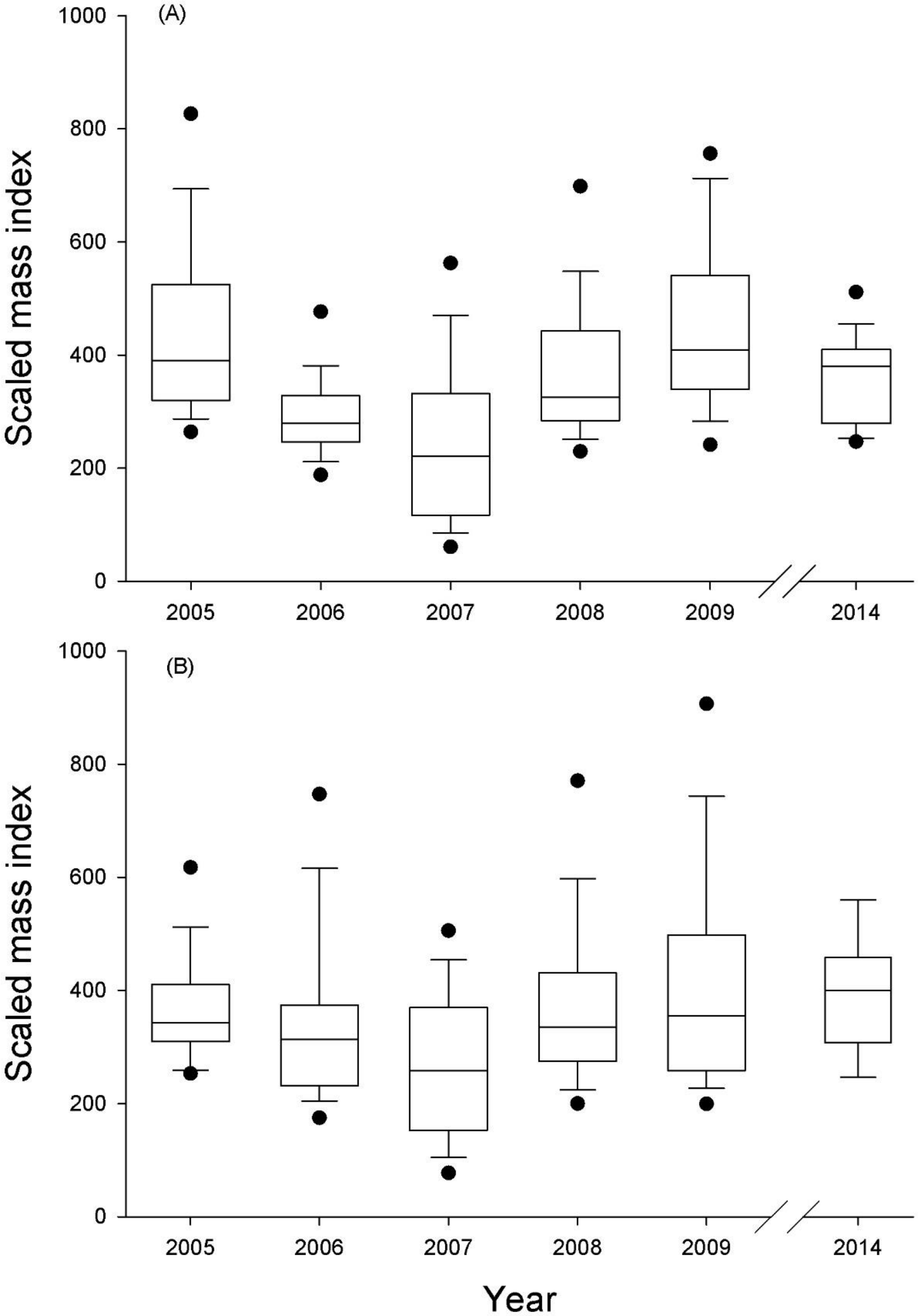

Fig 3 Boxplots of body condition (scaled mass index) of northern quolls over time on (a) Astell and (b) Pobassoo Islands. Outliers (filled circles) represented by 5/95th percentiles.

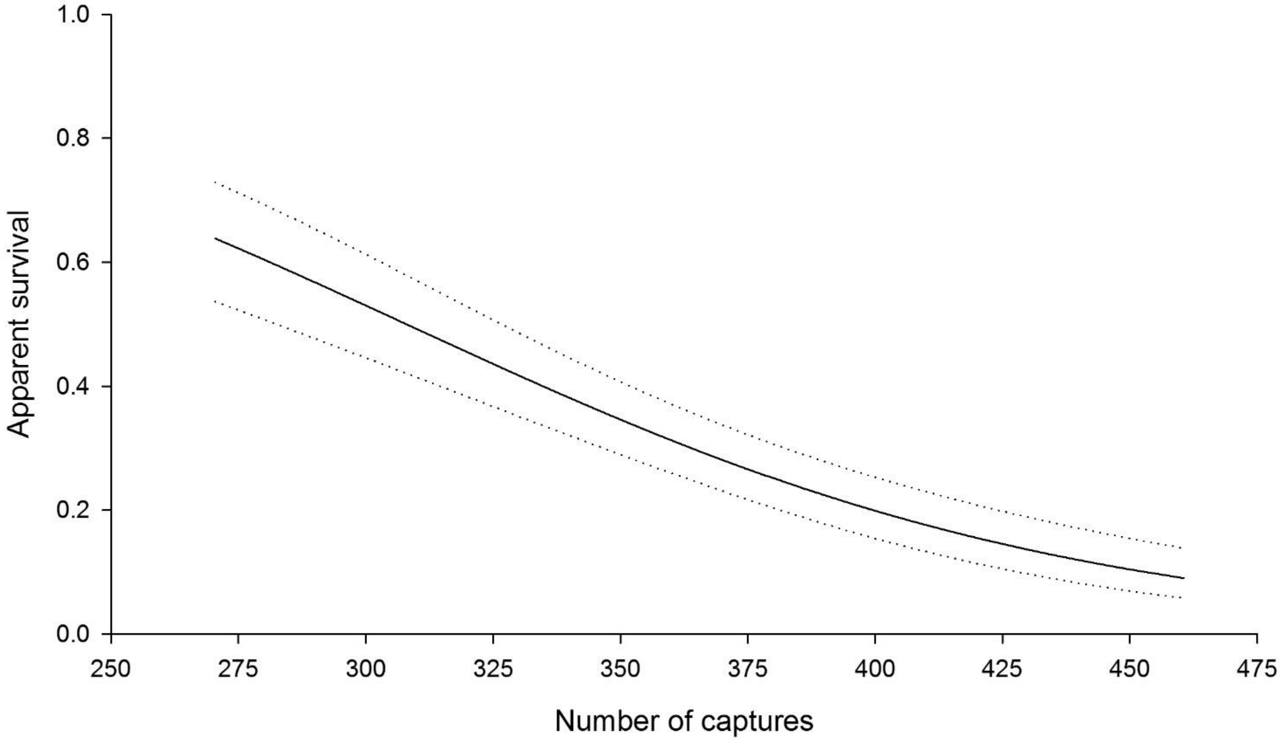

Fig 4 Predicted relationship between apparent survival of female northern quolls and the number of captures in the previous year based on the top-ranked model. Dashed lines represent 95% confidence intervals.

Tables

Table 1. Summary of model-selection results for northern quoll trap success on Astell and Pobassoo Islands from 2003 to 2009. All models contain a temporal autocorrelation parameter representing captures in the previous year on each island. *K* is the number of parameters. AIC_c is Akaike's Information Criterion, corrected for small sample size. ΔAIC_c shows the difference between the model AIC_c and the lowest AIC_c out of the set of models. AIC_c weights (w_i) are the relative likelihood of model *i* (normalised to sum to 1). The bigger the delta the smaller the weight and the less plausible model *i*.

Models	*K*	AIC_c	ΔAIC_c	w_i	Model likelihood
~ Island + Year + $Year^2$	5	30.20	0	0.77	1
~ Year + $Year^2$	4	32.85	2.65	0.20	0.27
~ Year	3	37.97	7.77	0.16	0.02
~ Island + Year	4	39.80	9.60	0.00	0.01
~ Null	2	40.58	10.38	0.00	0.01
~ Island	3	42.77	12.57	0.00	0

Table 2. Summary of model-selection results for northern quoll body conditon linear models (scaled mass index) on Astell and Pobassoo Islands over six years of monitoring. See Table 1 for explaination of table column headings.

Models	*K*	AIC_c	ΔAIC_c	w_i	Model likelihood
~ Year	7	9232.99	0	0.51	1
~ Year * Island	13	9234.05	1.06	0.30	0.59
~ Year + Island	8	9235.02	2.03	0.18	0.36
~ Null	2	9364.58	131.59	0	0
~ Island	3	9365.62	132.63	0	0
~ Sex	4	9360.82	127.83	0	0

Table 3 Summary of model-selection results for female northern quoll apparent survival and recruitment (Pradel Robust Design). All models fitted with p(Island * 4=5) c(Island * 3=4) N (t) parameterisation. See Table 1 for explaination of table column headings.

Model	K	AIC_c	ΔAIC_c	w_i	Model likelihood
Phi(density + body mass * year) *f*(rain)	29	1647.32	0.00	0.55	1.00
Phi(island + density + body mass * year) *f*(rain)	30	1649.39	2.07	0.19	0.36
Phi(density + body mass * year) *f*(island)	29	1649.98	2.66	0.14	0.26
Phi(density + body mass * year) *f*(rain + density + trend)	32	1651.37	4.06	0.07	0.13
Phi(density + body mass * year) *f*(island + year)	32	1652.47	5.15	0.04	0.08
Phi(density) *f*(density)	26	1664.09	16.77	0.00	0.00
Phi(island *t) *f*(.)	29	1664.68	17.36	0.00	0.00
Phi(rain + density) *f*(.)	26	1664.80	17.48	0.00	0.00
Phi(density + body mass * year) *f*(island * t)	30	1665.66	18.34	0.00	0.00
Phi(island + rain + density + body mass + trend) *f*(density)	30	1665.66	18.34	0.00	0.00
Phi(island + rain + density + body mass + trend) *f*(year)	31	1665.74	18.42	0.00	0.00

Phi(Island *t) *f*(Island * t)	34	1666.42	19.10	0.00	0.00
Phi(island + rain + density + body mass + trend) *f*(island + rain + density + trend)	33	1667.86	20.54	0.00	0.00
Phi(t) *f*(t)	28	1668.69	21.37	0.00	0.00
Phi(rain + body mass * year) *f*(island)	29	1672.01	24.69	0.00	0.00
Phi(rain) *f*(rain)	26	1674.78	27.46	0.00	0.00
Phi(rain) *f*(.)	25	1676.74	29.42	0.00	0.00
Phi(island + rain) *f*(island + rain)	28	1676.74	29.42	0.00	0.00
Phi(.) *f*(Island * t)	29	1713.35	66.03	0.00	0.00
Phi(.) *f*(.)	24	1715.26	67.94	0.00	0.00
Phi(.) *f*(island)	25	1715.50	68.18	0.00	0.00
Phi(body mass * year) *f*(island + rain + density + trend)	31	1716.67	69.35	0.00	0.00
Phi(body mass * year) *f*(.)	27	1718.98	71.66	0.00	0.00

Table 4 Evaluation of the northern quoll translocation program against criteria in Perez *et al* (2012).

Level	Criteria	Outcome
Necessity of the translocation	Is the species or population under threat?	Listed as Endangered by IUCN and under Australian legislation from threat of cane toad *Rhinella marina*, a highly toxic introduced species preyed upon by northern quolls.
	Have the threatening factors been removed or controlled or were they absent in the release?	Both release sites were islands that were not inhabited by cane toads.
	Are translocations the best tool to mitigate conservation conflicts?	Unable to control cane toads on mainland Australia.
Risk evaluation	Are risks for the target species acceptable?	Relatively small number of animals taken from founder populations that were about to be impacted by cane toads.

	Are the risks to other species or the ecosystem acceptable?	Comprehensive surveys conducted at release sites to identify species at-risk of introducing a carnivorous northern quoll.
	Are the possible effects of the translocation acceptable to local people?	Extensive consultation with Aboriginal landowners from the founder population and release sites.
Technical and logistical suitability	Does the project maximise the likelihood of establishing a viable population?	The founder population was relatively small (n = 64) but species has high fecundity.
	Does the project include clear goals and monitoring?	Clear goals established at the start of the program and regular monitoring of the translocated populations was undertaken.
	Do enough economic and human resources exist?	Long-term commitment by NT Government.
	Do scientific, governmental and stakeholder groups support the translocation?	Strong support from government, private and Aboriginal communities.

Appendix A

Parameters used and their biological significance to construct capture-mark-recapture models of apparent survival (ϕ), recruitment rate (*f*), derived from binomial likelihood-based models for female northern quolls on the translocation to Astell and Pobassoo islands. Category relates to how each parameter was used in the design matrix.

Name	**Category**	**Used in**	**Apparent survival, or recruitment is …**	**Notes**
Null		ϕ, *f*	constant	
Island	Category	ϕ, *f*	different among the two islands	Area of each island: Astell (1292 ha) and Pobassoo (392 ha)
Body mass	Individual covariate	ϕ	related to body mass of individual	Body mass (g) at first capture
Rainfall	Time - specific covariate	ϕ, *f*	influenced by total rainfall in the previous 12 months	Previous 12 monthly total rainfall (mm) in primary trapping occasion (taken from Gove Airport, 40 km south)

Year	Time - specific covariate	ϕ, *f*	different among each year of sampling	Period between the four years (2006 to 2007, 2007 to 2008, 2008 to 2009)
Linear trend	Time - specific covariate	ϕ, *f*	constrained by either a positive or negative linear trend over the study	Capture intervals numbered from 1 to 4
Density	Time - specific	ϕ, *f*	Influenced by the number of northern quolls in the previous year	Number of captures of all quolls on each island in the previous year

Table A1 Summary of model selection results for initial capture (*p*) and recapture (*c*) probability as part of the Pradel Robust Design modelling. All models contained the parameters Phi(Island * t) f(Island * t) and N(t).

Model	K	AICc	ΔAICc	w_i	Model likelihood
p(Island * 4=5) c(Island * 3=4)	34	1666.42	0	0.99	1
p(Island) c(Island)	24	1677.15	10.73	0.01	0.005
p(4=5) c(3=4)	27	1688.37	21.95	0	0
p(4=5) c(3=4)	27	1691.23	24.81	0	0
p(.) c(.)	22	1704.97	38.55	0	0
p(t) c(=p)	25	1799.07	132.65	0	0
p(t) c(=p)	24	1820.51	154.09	0	0
p(Island) c(=p)	22	1838.31	171.89	0	0
p(.) c(=p)	21	1851.12	184.70	0	0