

# Demographic evaluation of translocating the threatened northern quoll to two Australian islands

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## Abstract

**Context.** Translocation is widely used to help avoid extinction of species from threatening processes. A fundamental objective of translocation is to establish self-sustaining populations; estimating demographic parameters is critical to assessing success of these programs and can also be used to support future management actions.

**Aims.** We estimated demographic parameters to evaluate the success of translocating the northern quoll *Dasyurus hallucatus* to two islands (Astell and Pobassoo), in response to the threat posed by the introduced cane toad *Rhinella marina* on the Australian mainland.

**Methods.** We used capture–mark–recapture methods to monitor both populations at regular intervals from initial release in 2003 until 2009 and a one-off survey in 2014.

**Key results.** Relative abundance (trap success) increased exponentially in the first 4 years, declined, and then stabilised in subsequent years. The population of female northern quolls on Astell Island peaked in 2006 with an estimate of 3640 (95% CI 3022–4257), and on Pobassoo Island the peak was 2007 with 617 (95% CI 531–703) females. In 2014 the population had decreased to 2193 (95% CI 1920–2467) on Astell and 451 (95% CI 359–543) on Pobassoo. Apparent survival and body condition decreased significantly following the population peak, possibly related to density dependence.

**Conclusion.** Both populations of northern quolls reached their regulation phase after going through establishment and growth phases, which included exceeding carrying capacity. The pattern was similar between the populations.

**Implications.** Increased survival and recruitment at threat-free translocation sites contributes to successful establishment of self-sustaining populations.

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## 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/SSC 2013); it is an important and increasingly applied tool to conserve threatened species (Armstrong *et al.* 2015). Unfortunately, many translocations have failed or been characterised by poor conception (Griffith *et al.* 1989; Magdalena Wolf *et al.* 1998). From analyses of the fate of many translocations, it is clear that several factors 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) (Griffith *et al.* 1989; Fischer and Lindenmayer 2000; Pérez *et al.* 2012; Sheean *et al.* 2012). From such reviews, there have been attempts to adopt a more integrated, standardised and theoretically sound approach to translocations (Seddon *et al.* 2007; Armstrong and Seddon 2008), including population

monitoring, modelling and structured decision making (Armstrong and Reynolds 2012; Converse *et al.* 2013).

The establishment of a self-sustaining population is a fundamental objective of any translocation. While there is no single definition of how to determine whether this objective has been met (Robert *et al.* 2015; Shier 2015), it is necessary to obtain accurate estimates of demographic parameters for the translocated population to be able to evaluate success. These parameters can be used to assess the establishment of self-sustaining populations using population viability analysis (e.g. Lindenmayer *et al.* 1993) and development of models to guide management of populations (e.g. McCarthy *et al.* 2012). In addition, they can provide important information on the dynamics of the translocated population and help identify problems where management intervention may be needed.

Translocations have been used extensively to help conserve Australia's mammal fauna, and success has typically focused on measures of population persistence or trends (Fischer and

Lindenmayer 2000; Clayton *et al.* 2014; Morris *et al.* 2015). Australian mammals have had an extraordinary extinction rate over the last 200 years (Short and Smith 1994; Woinarski *et al.* 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 and 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 and McKenzie 1989; Woinarski and Braithwaite 1990; Short and Smith 1994; Smith and Quin 1996; Johnson 2006).

Here we estimate demographic parameters using capture–mark–recapture methods for an island translocation for the northern quoll *Dasyurus hallucatus*. This omnivorous marsupial is extremely susceptible to toxins ingested during predation on the exotic cane toad *Rhinella marina*. Consequently, populations of northern quolls have drastically declined 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 (Sutherst *et al.* 1996; Kearney *et al.* 2008). 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. In addition, the northern quoll has experienced population declines and range contraction in parts of its distribution before the arrival of cane toads (Braithwaite and Griffiths 1994). Because of these factors, the northern quoll is listed as endangered under Australia's *Environment Protection and Biodiversity Conservation Act*.

The objective of this paper is to evaluate, using demographic parameters, whether the translocation program established at least one secure island population of northern quolls. Importantly, translocations to islands provide a unique opportunity to undertake natural experiments, as they act as populations closed to threats, and resources may vary between islands used as translocation sites. In this case, given some risk of catastrophe (notably invasion of the translocated sites by cane toads and tropical cyclones), we used two separate translocation destinations, and hence were able to compare population trends across these two sites. For both island sites, we simultaneously monitored demographic parameters to examine difference between sites, evidence of density dependence (Armstrong *et al.* 2005) and variation-caused by environmental covariates such as rainfall. This information will inform future management of the populations and provide insight into future translocations of other mammal species.

## 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 the following: (1) adequate size for persistence for at least 30 years ( $>1 \text{ km}^2$ , but preferably  $>10 \text{ km}^2$ ); (2) occurrence of suitable habitat (areas of rugged sandstone); (3) absence of human habitation; (4) relatively low risk of cane toad colonisation (i.e. limited visitation by humans, distant from mainland, not in the outflow area of mainland rivers); (5) moderate accessibility; and (6) absence of other conservation values susceptible to predation or competition from translocated quolls. These criteria restricted the candidate set to about 10 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 Co. group off north-eastern Arnhem Land (Fig. 1). Both islands are rugged and dominated by eucalypt (particularly *Eucalyptus tetradonta*) woodlands, with small fringing areas of coastal vine thicket and mangroves (Woinarski *et al.* 2000).

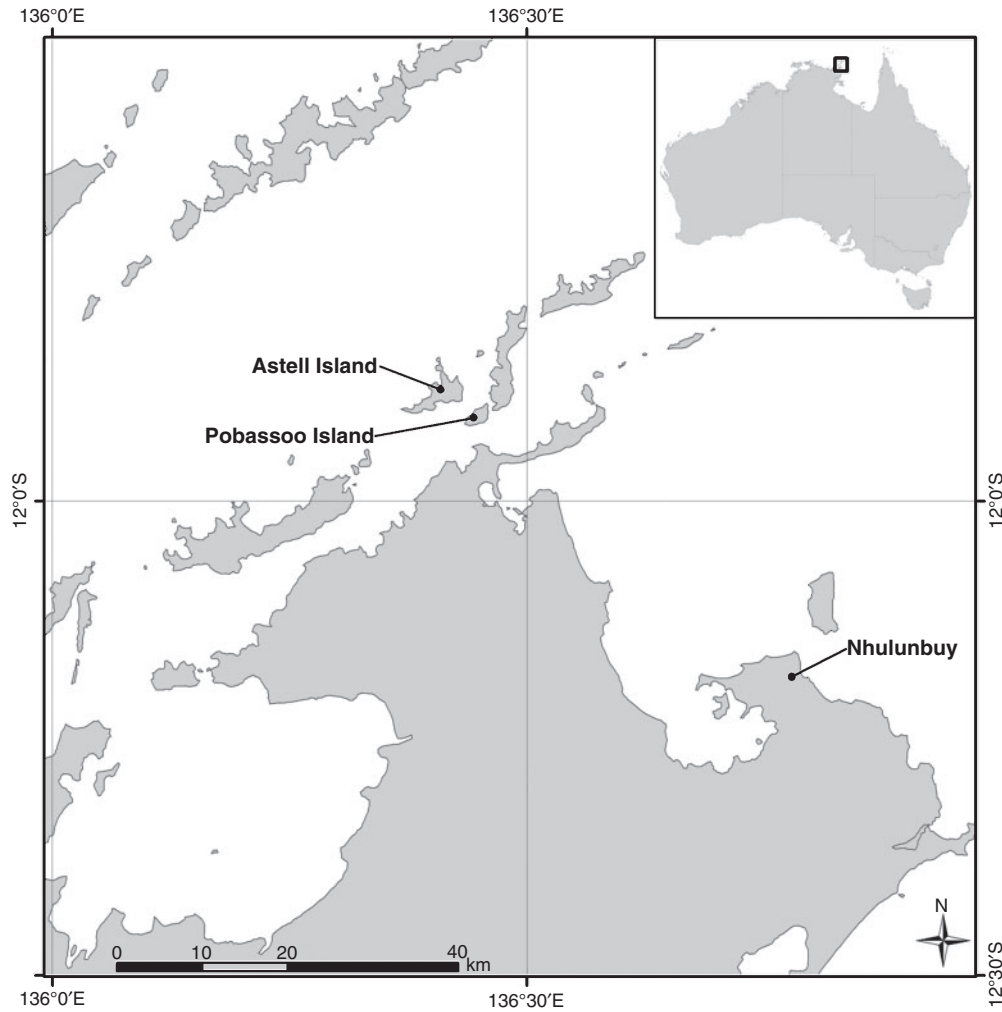
Translocations may have impacts on other species present at the destination site, and these impacts may be particularly severe when the translocated species is a predator such as the northern quoll. Previous detailed surveys had indicated that these islands did not support any plant (Woinarski *et al.* 2000), ant (Woinarski *et al.* 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 and 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 age class was considered most likely to adapt to translocation, since adults (especially males) have a short life expectancy. Animals were collected using Elliott (10 cm × 10 cm × 33 cm) and cage (15 cm × 15 cm × 60 cm) traps and held in purpose-built enclosures for 1–9 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. The sizes of founder populations were set to account for some initial post-release mortality and what we believed to be the minimum number of animals for successful establishment. No further translocations were made during the program.

### Monitoring

The translocated quoll populations were monitored on 10 occasions following release, with all surveys conducted in



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

collaboration with the islands' Aboriginal owners. The earlier surveys (2003–05) occurred in the early to mid-dry season (April to July). Subsequent surveys (2006–09) 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  $7 \times 10$  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  $5 \times 5$  traps, spaced 20 m apart) were established and sampled over 5 nights, with 10 grids used on Astell and eight on Pobassoo. The survey in 2014 used half the permanent grids on each island (due to funding constraints). The change in survey methodology post-2006 was due to a need to collect more robust capture–mark–recapture data once the population had become established. All traps used were cage traps ( $13 \text{ cm} \times 13 \text{ cm} \times 40 \text{ cm}$ ), baited with a mixture of peanut butter, honey and oats. In total, there were 7776 trap-nights on Astell Island and 6431 trap-nights on

Pobassoo Island over the entire monitoring program. 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 measured 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 and Limpus

2001), and models were fitted using maximum likelihood estimation to allow comparison between models with different fixed effects (Pinheiro and Bates 2000).

The second method used capture–mark–recapture data from the later surveys (2006–09 and 2014) to estimate the density of female northern quolls. We used the closed-captures component within the Pradel Robust Design model for the 2006–09 and a single closed-capture model for the 2014 survey using Program MARK v8.0 (White and 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 5 nights (Williams *et al.* 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 and 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 and 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). To estimate the population size of female quolls we divided the density estimate on each island for each year by the effective trapping area. There was considerable homogeneity across all trapping grids on both islands – therefore the extrapolations are considered accurate.

Body condition for each individual was estimated using the scaled mass index (Peig and Green 2009; Peig and 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 (2005–09 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 and 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 ( $\phi$ ) 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 that the area sampled does not change during the study and all animals have some probability of being captured, that there is no response to being trapped and that 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 every marked animal has the same probability of survival, tags are not lost or misidentified, emigration is permanent and that 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 1). 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 +1 and –1. 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 ( $\beta$ ) 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 and White 2014).

## Results

### *Survey effort and goodness-of-fit*

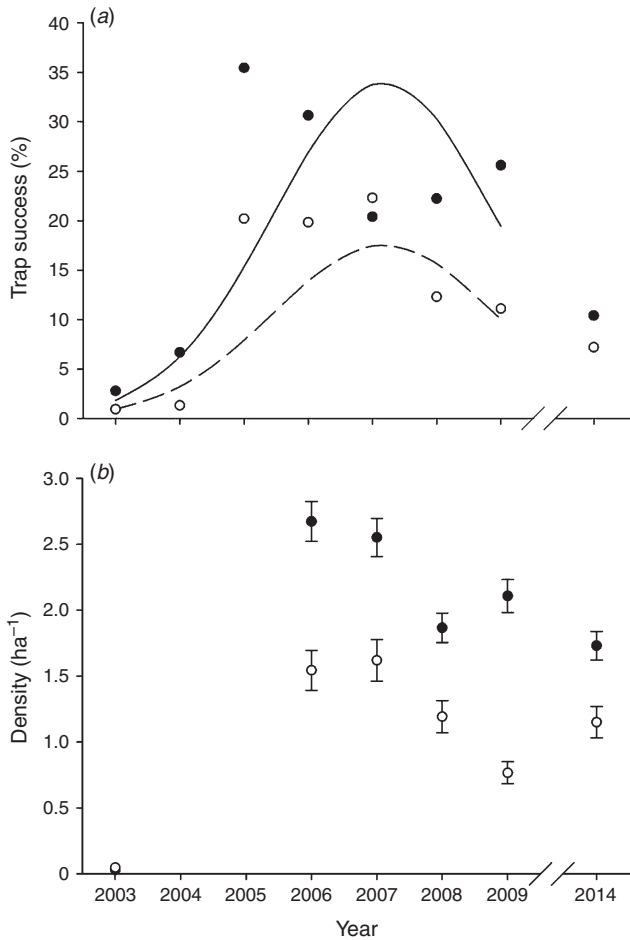
In total, we recorded 2327 northern quolls captures from 13 507 trap-nights across all sampling periods (2003–09 and 2014). The capture rate on Astell Island (1523 captures from 7776 trap-nights: 19.6% trap success) was higher than for Pobassoo (804 captures from 6431 trap-nights: 12.5% trap success). 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 3 years. On Astell Island, trap success peaked in 2005, whereas on Pobassoo Island it remained high from 2005 to 2007 and then decreased (Fig. 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 7 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 (Fig. 2*b*). Density was higher on Astell than on Pobassoo Island and densities decreased on both islands after 2007 (Fig. 2*b*). Initial capture (*p*) and



**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.

**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.  $\Delta AIC_c$  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	<i>K</i>	$AIC_c$	$\Delta AIC_c$	$w_i$	Model likelihood
Island + Year + Year <sup>2</sup>	5	30.20	0.00	0.77	1.00
Year + Year <sup>2</sup>	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	0.00

recapture probabilities (*c*) differed over time and between islands (Appendix 2). 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). We note that the use of different home-range estimates will change these results by increasing or decreasing the effective trapping area.

Body condition (represented by scaled mass index) varied considerably across the 6 years and also differed between the two islands. The best-supported model for variation in body condition contained only the parameter year. 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 2006–08 compared with 2005 (Fig. 3*a*). 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. 3*b*).

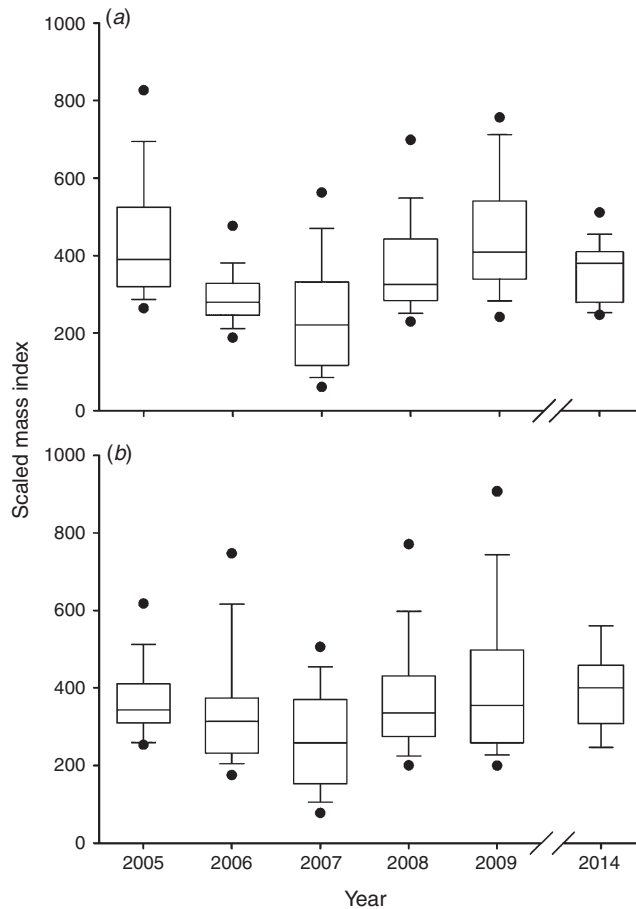
*Apparent survival and recruitment*

Apparent survival of female quolls varied over the period 2006–09 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 of female quolls (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  $-0.84$ ) (Fig. 4). In addition, body mass influenced apparent survival. In 2006–07 there was a negative but non-significant relationship ( $\beta = -0.12$ , 95% CI  $-0.45$  to  $0.21$ ) and in 2007–08 there was a significant negative relationship ( $\beta = -0.84$ , 95% CI  $-1.21$  to  $-0.46$ ). However, in 2008–09 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 2006–07, 0.10 (95% CI 0.06–0.15) and 0.12

**Table 2. Summary of model-selection results for northern quoll body condition linear models (scaled mass index) on Astell and Pobassoo Islands over 6 years of monitoring**

See Table 1 for explanation of table column headings

Models	<i>K</i>	$AIC_c$	$\Delta AIC_c$	$w_i$	Model likelihood
Year	7	9232.99	0.00	0.51	1.00
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	0.00	0.00
Island	3	9365.62	132.63	0.00	0.00
Sex	4	9360.82	127.83	0.00	0.00



**Fig. 3.** Boxplots of body condition (scaled mass index) of northern quolls over time on (a) Astell and (b) Pobassoo Islands. Band inside the box represents the median value, box boundaries the 25th and 75th percentiles, whiskers the 10th and 90th percentiles and filled circles are the 5th and 95th percentiles outliers.

(95% CI 0.08–0.18) in 2007–08 and 0.63 (95% CI 0.51–0.73) and 0.57 (95% CI 0.46–0.67) in 2008–09.

Recruitment of female northern quolls varied over the 4 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 ( $\beta = -0.36$ , 95% CI  $-0.7$  to  $0.4$ ). There was no evidence that recruitment differed between the two island populations ( $\beta = 0.15$ , 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 2006–07, 0.61 (95% CI 0.49–0.73) and 0.59 (95% CI 0.43–0.73) in 2007–08, and 0.41 (95% CI 0.24–0.61) and 0.40 (95% CI 0.25–0.58) in 2008–09.

## Discussion

Based on the demographic parameters presented here, there is clear evidence that the translocation program to date has been able to establish two independent northern quoll populations. The translocated island populations exhibited extraordinary rates of increase in the first 4 years, followed by a decline and

stabilisation of the population. Both populations appear to have reached their regulation phase within 6 years after going through establishment and growth phases (Sarrazin and Barbault 1996). The pattern was similar for both islands though there was evidence that the pattern was more pronounced on the larger island (Astell).

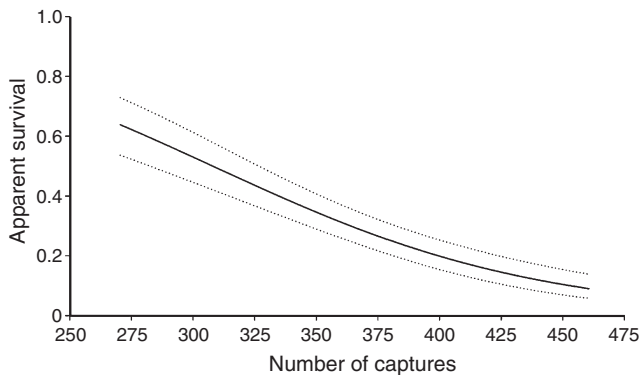
Several demographic factors are likely to have contributed to the successful establishment of the northern quoll populations. The species has a high maximum annual population growth rate of 1.35 (95% CI 0.11–5.46) and short generation length (Hone *et al.* 2010) and this resulted in very high rates of increase in the first 4 years following the initial release on both islands. This was despite there being several catastrophic events within the first 4 years: ~70% of Astell Island was burnt in 2003 and two Category 5 tropical cyclones passed directly over both islands on 2005 and 2006, with wind gusts to  $325 \text{ km h}^{-1}$  and  $>300 \text{ mm}$  of rainfall (Gove Airport: Bureau of Meteorology). Furthermore, 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 and Brook 2015) compared with 0.63 in this study. Per capita recruitment rates were higher in the translocated populations on both islands compared with the mainland. Recruitment rates on both islands ranged from 0.40 to 0.60 compared with 0.25 to 0.35 on the mainland (Griffiths and Brook 2015). This indicates that the absence of predators and presence of high quality habitat (rocky woodland) contributed to the high deterministic growth rate. Based on this information, both populations appear to have avoided early reintroduction failure due to the high population growth rate in a predator- and threat (i.e. cane toad)-free environment, despite the founder populations being relatively small.

There was evidence suggesting negative density dependence within 5 years of the initial translocation, with female (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 *et al.* 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 mostly reproduce each year (Braithwaite and 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 pattern was similar between the two islands but more pronounced on the larger island (Astell), possibly caused by the higher density and consequent competition for food. 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

**Table 3. Summary of model-selection results for female northern quoll apparent survival and recruitment (Pradel Robust Design)**

All models fitted with  $p(\text{Island} \times 4 = 5)c(\text{Island} \times 3 = 4)N(t)$  parameterisation. See Table 1 for explanation of table column headings

Model	$K$	$AIC_c$	$\Delta AIC_c$	$w_i$	Model likelihood
$\varphi(\text{density} + \text{body mass} \times \text{year})/f(\text{rain})$	29	1647.32	0.00	0.55	1.00
$\varphi(\text{island} + \text{density} + \text{body mass} \times \text{year})/f(\text{rain})$	30	1649.39	2.07	0.19	0.36
$\varphi(\text{density} + \text{body mass} \times \text{year})/f(\text{island})$	29	1649.98	2.66	0.14	0.26
$\varphi(\text{density} + \text{body mass} \times \text{year})/f(\text{rain} + \text{density} + \text{trend})$	32	1651.37	4.06	0.07	0.13
$\varphi(\text{density} + \text{body mass} \times \text{year})/f(\text{island} + \text{year})$	32	1652.47	5.15	0.04	0.08
$\varphi(\text{density})/f(\text{density})$	26	1664.09	16.77	0.00	0.00
$\varphi(\text{island} \times t)/f(\cdot)$	29	1664.68	17.36	0.00	0.00
$\varphi(\text{rain} + \text{density})/f(\cdot)$	26	1664.80	17.48	0.00	0.00
$\varphi(\text{density} + \text{body mass} \times \text{year})/f(\text{island} \times t)$	30	1665.66	18.34	0.00	0.00
$\varphi(\text{island} + \text{rain} + \text{density} + \text{body mass} + \text{trend})/f(\text{density})$	30	1665.66	18.34	0.00	0.00
$\varphi(\text{island} + \text{rain} + \text{density} + \text{body mass} + \text{trend})/f(\text{year})$	31	1665.74	18.42	0.00	0.00
$\varphi(\text{Island} \times t)/f(\text{Island} \times t)$	34	1666.42	19.10	0.00	0.00
$\varphi(\text{island} + \text{rain} + \text{density} + \text{body mass} + \text{trend})/f(\text{island} + \text{rain} + \text{density} + \text{trend})$	33	1667.86	20.54	0.00	0.00
$\varphi(t)/f(t)$	28	1668.69	21.37	0.00	0.00
$\varphi(\text{rain} + \text{body mass} \times \text{year})/f(\text{island})$	29	1672.01	24.69	0.00	0.00
$\varphi(\text{rain})/f(\text{rain})$	26	1674.78	27.46	0.00	0.00
$\varphi(\text{rain})/f(\cdot)$	25	1676.74	29.42	0.00	0.00
$\varphi(\text{island} + \text{rain})/f(\text{island} + \text{rain})$	28	1676.74	29.42	0.00	0.00
$\varphi(\cdot)/f(\text{Island} \times t)$	29	1713.35	66.03	0.00	0.00
$\varphi(\cdot)/f(\cdot)$	24	1715.26	67.94	0.00	0.00
$\varphi(\cdot)/f(\text{island})$	25	1715.50	68.18	0.00	0.00
$\varphi(\text{body mass} \times \text{year})/f(\text{island} + \text{rain} + \text{density} + \text{trend})$	31	1716.67	69.35	0.00	0.00
$\varphi(\text{body mass} \times \text{year})/f(\cdot)$	27	1718.98	71.66	0.00	0.00



**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.

declined (unsurprisingly, given high densities of a novel predator) but now stabilised. It is unclear why larger females had lower apparent survival during periods of high population density. Body size may be related to age but this was not examined in this study.

The study raises some important issues for future management of the northern quoll. The translocation was timely for the northern quoll, as alternative conservation management options such as building cane toad-proof enclosures were either too expensive or risky (Brook and Whitehead 2005) or

had not been developed (O'Donnell *et al.* 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 or other islands containing at-risk species. 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 individuals from the translocated island populations to the mainland is another management option. Based on demographic data from mainland populations, a carrying capacity of  $\sim 1.0 \text{ ha}^{-1}$  female northern quolls is required for long-term population viability in mainland populations (Griffiths *et al.* 2015). Prior to release, this demographic information could be used to model the number of animals to be released and the potential impact of predators. Conversely, reintroduction of northern quolls to mainland predator exclosures could result in overabundance, and demographic parameters presented in this study could be used to develop appropriate release strategies to minimise risk to other species. Future translocation of northern quolls to islands should target threat-free islands that do not contain threatened species that may be preyed upon by quolls. We note that a reduction in genetic diversity of the translocated populations occurred after only three generations but was

higher than endemic island populations (Cardoso *et al.* 2009). Future management actions should incorporate maintaining and monitoring genetic diversity of the translocated populations.

## Conclusions

We have demonstrated the need to collect detailed demographic information for translocated populations in order to prove the fundamental objective of establishing self-sustaining populations is achieved. Given the relatively small size of the founder populations in this study, it is likely that the high intrinsic growth rate of the species due to high survival and recruitment in a predator- and toad-free translocation sites contributed to both populations reaching their regulation phase and persisting into the future. The results may also contribute to a more unified approach in the use of demographic parameters, such as carrying capacity, in developing universal criteria for defining reintroduction success (Robert *et al.* 2015; Shier 2015).

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**Appendix 1. Parameters used and their biological significance to construct capture-mark–recapture models of apparent survival ( $\varphi$ ), 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		$\varphi, f$	constant	
Island	Category	$\varphi, f$	different among the two islands	Area of each island: Astell (1268 ha) and Pobassoo (392 ha)
Body mass	Individual covariate	$\varphi$	related to body mass of individual	Body mass (g) at first capture
Rainfall	Time-specific covariate	$\varphi, f$	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	$\varphi, f$	different among each year of sampling	Period between the 4 years (2006–07, 2007–08, 2008–09)
Linear trend	Time-specific covariate	$\varphi, f$	constrained by either a positive or negative linear trend over the study	Capture intervals numbered from 1 to 4
Density	Time-specific covariate	$\varphi, f$	Influenced by the number of northern quolls in the previous year	Number of captures of all quolls on each island in the previous year

**Appendix 2. 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  $\text{Phi}(\text{Island} \times t)$ ,  $f(\text{Island} \times t)$  and  $N(t)$

Model	$K$	AICc	$\Delta\text{AICc}$	$w_i$	Model likelihood
$p(\text{Island} \times 4 = 5)c(\text{Island} \times 3 = 4)$	34	1666.42	0.00	0.99	1.000
$p(\text{Island})c(\text{Island})$	24	1677.15	10.73	0.01	0.005
$p(4 = 5)c(3 = 4)$	27	1688.37	21.95	0.00	0.000
$p(4 = 5)c(3 = 4)$	27	1691.23	24.81	0.00	0.000
$p(\cdot)c(\cdot)$	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
$p(\text{Island})c(= p)$	22	1838.31	171.89	0	0
$p(\cdot)c(= p)$	21	1851.12	184.70	0	0