Biological Conservation 145 (2012) 95–101

Contents lists available at SciVerse ScienceDirect

Biological Conservation

journal homepage: www.elsevier.com/locate/biocon

Conservation cornerstones: Capitalising on the endeavours of long-term monitoring projects

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ARTICLE INFO

Article history:
Received 24 May 2011
Received in revised form 29 August 2011
Accepted 11 October 2011
Available online 25 November 2011

Keywords:
Ecological monitoring
Population decline
Conservation management
Threatening processes
Brush-tailed phascogale
Australia

ABSTRACT

Ecological monitoring is widely used to measure change through time in ecosystems. The current extinction crisis has resulted in a wealth of monitoring programs focussed on tracking the status of threatened species, and the perceived importance of monitoring has seen it become the cornerstone of many biodiversity conservation programs. However, many monitoring programs fail to produce useful outcomes due to inherent flaws. Here we use a monitoring program from south-eastern Australia as a case study to illustrate the potential of such endeavours. The threatened carnivorous marsupial, the brush-tailed phascogale (*Phascogale tapoatafa*), has been monitored at various locations between 2000 and 2010. We present strong evidence for a decline in relative abundance during this period, and also describe relationships with environmental variables. These results provide insights likely to be valuable in guiding future management of the species. In the absence of the monitoring program, informed management would not be possible. While early detection of population declines is important, knowledge of the processes driving such declines is required for effective intervention. We argue that monitoring programs will be most effective as a tool for enhanced conservation management if they test specific hypotheses relating to changes in population trajectories. Greater emphasis should be placed on rigorous statistical analysis of monitoring datasets in order to capitalise on the resources devoted to monitoring activities. Many datasets are likely to exist for which careful analysis of results would have benefits for determining management directions.

1. Introduction

Long-term monitoring is commonly employed to improve understanding of ecological systems and is the cornerstone of many conservation endeavours (Lindenmayer and Likens, 2010; Lovett et al., 2007). The goal of all monitoring programs is to detect change through time in an entity of interest, often in response to environmental change, anthropogenic disturbance, or targeted management actions (Legg and Nagy, 2006; Lindenmayer and Likens, 2009). As the number of species threatened with extinction continues to grow worldwide, long-term monitoring and research is becoming increasingly important for tracking the status of species (e.g. Hawkins et al., 2006; Mac Nally et al., 2009), with the ultimate objective being to document population declines and guide management to facilitate population persistence.

Despite their prevalence, the usefulness of monitoring programs is often equivocal. Projects frequently suffer from deficiencies including vague goals and objectives, inadequate study design, and lack of rigorous data analyses and self-assessment (Field et al., 2007; Lovett et al., 2007). Consequently, monitoring programs may fail to report any findings, or worse still, management actions may be based on anecdotal observations that lack quantitative support. Monitoring activities can also be limited by a focus on pattern but not process: documenting a decline may be of limited value if the underlying cause of the decline is not also identified. This has led to greater scrutiny of monitoring projects and a more critical evaluation of how scarce conservation resources are utilised (McDonald-Madden et al., 2010). Various authors (e.g. Lindenmayer and Likens, 2009; Nichols and Williams, 2006) have advocated an 'adaptive monitoring' framework whereby objectives are clearly defined, data collection is governed by careful study design, and findings are fed back into the framework to guide future efforts. A key component of this framework is

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0006-3207/$ - see front matter © 2011 Elsevier Ltd. All rights reserved.
doi:10.1016/j.biocon.2011.10.016

rigorous statistical analysis of collected data. Such analyses permit conclusions to be drawn with confidence and can be used to refine and improve monitoring protocols (Field et al., 2007; Lindenmayer and Likens, 2009). Despite the unquestionable value of this step, formal analysis is a much neglected aspect of monitoring programs (Field et al., 2007; Lovett et al., 2007).

We believe that long-term monitoring and research has an important role to play in conservation biology and that, when conducted appropriately, can be a valuable tool for informing management. In this paper we illustrate this point using a monitoring program for a threatened carnivorous marsupial, the brush-tailed phascogale (*Phascogale tapoatafa*), as a case study. Populations of *P. tapoatafa* have been monitored in south-eastern Australia since 2000, providing an opportunity to rigorously analyse the dataset and identify trends in relative abundance through time. Population trajectories are also analysed in relation to broad environmental variables. Through this case study we demonstrate the value of long-term monitoring and the importance of critical assessment of monitoring outcomes. We also argue that studies of mechanistic processes should be more routinely incorporated into monitoring programs to facilitate understanding of population trajectories and to allow informed management interventions.

2. Methods

2.1. Study species

P. tapoatafa is a small carnivorous marsupial that feeds mostly on arthropods. It is largely arboreal and requires tree hollows for nesting (van der Ree et al., 2006). Large area requirements typically result in low densities. All males die shortly after the annual breeding season, thereby obtaining a maximum age of 1 year. After weaning their first litter, most females do not survive to breed a second time (Rhind and Bradley, 2002; Soderquist, 1993).

In the south-eastern Australian state of Victoria, *P. tapoatafa* is currently distributed across north-eastern, central and western regions (Fig. 1) where it occurs in Dry Forests containing box, ironbark and stringybark eucalypt species with an open understorey and little ground cover (Menkhorst, 1995). Much of the original forest in this region has been cleared for agricultural development: remaining forest typically occurs on areas less suitable for agricultural production, is highly fragmented, and is often disturbed by grazing, mining and firewood collection (Environment Conservation Council, 2001). As a result of this habitat loss and degradation, *P. tapoatafa* has declined in both distribution and abundance (Menkhorst, 1995) and is classified as threatened in Victoria.

An Action Statement for *P. tapoatafa* prepared under the Victorian *Flora and Fauna Guarantee Act 1988* identifies long-term monitoring at key habitat sites as an important management action to identify changes in population size and vulnerability (Department of Sustainability and Environment, 2003). In response, the Brush-tailed Phascogale Coordinating Group has been conducting population surveys at various sites across Victoria since 2000.

2.2. Study area

P. tapoatafa populations have been monitored at 17 sites (Fig. 1), located in large forest blocks (⩾2200 ha) within the known distribution of the species. All sites contain Dry Forests and woodlands characterised by stringybark, box and ironbark eucalypts. Midstorey vegetation generally consists of a moderate to sparse shrub layer, over a moderate to sparse ground layer of grasses and herbs. Long-term mean annual rainfall across the monitoring sites ranges from 464 mm at Dalyenong to 788 mm at Kinglake (mean = 594 mm).

2.3. Monitoring protocol

Surveys for *P. tapoatafa* were always performed between February and May. Sites were surveyed a maximum of once per year, but only a subset of sites was surveyed in any given year (Fig. 2). Six sites (Bolangum, Paddys Ranges, Rushworth, Wellsford, Ararat and Pyrenees) were surveyed on just one occasion between 2000 and 2010, while Reef Hills and Mt. Pilot were surveyed on ten and nine occasions, respectively (Fig. 2).

A single trapping transect was employed at each monitoring site. Forty trap stations (39 at Hepburn, Ararat, Pyrenees and Mt. Cole) were established per transect with a spacing of 300 m between each trap station (total transect length = 11.7 km). Two large Elliott aluminium traps (48 cm × 15 cm × 16 cm) were deployed at each trap station. Traps were fixed to T-shaped wooden brackets, nailed to the bole of trees at a height of approximately 2 m. Large, rough-barked eucalypts were chosen for trap deployment. A bait mixture of peanut butter, rolled oats and honey was squeezed into

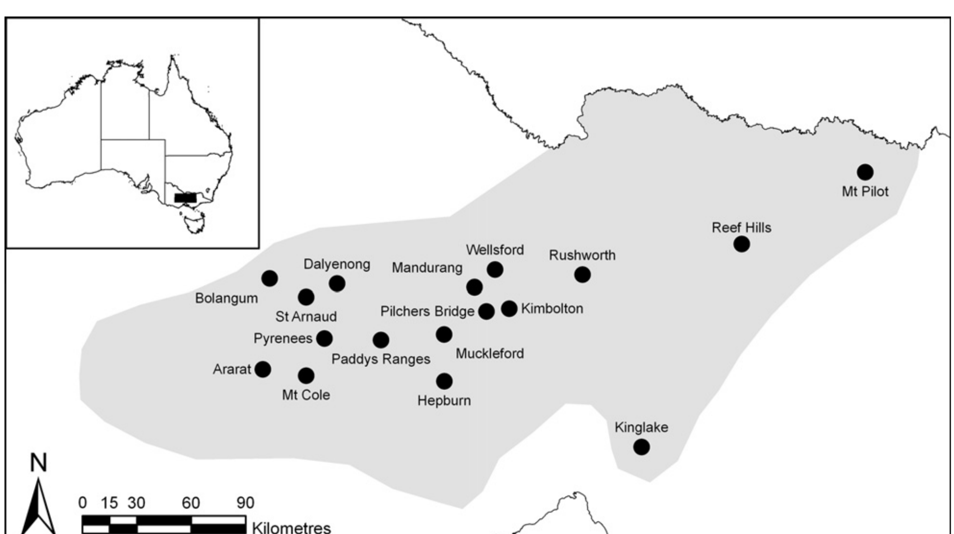

Fig. 1. Location of 17 monitoring sites for the brush-tailed phascogale (*Phascogale tapoatafa*) in Victoria, south-eastern Australia. Grey shading indicates the approximate current distribution of *P. tapoatafa* in Victoria.

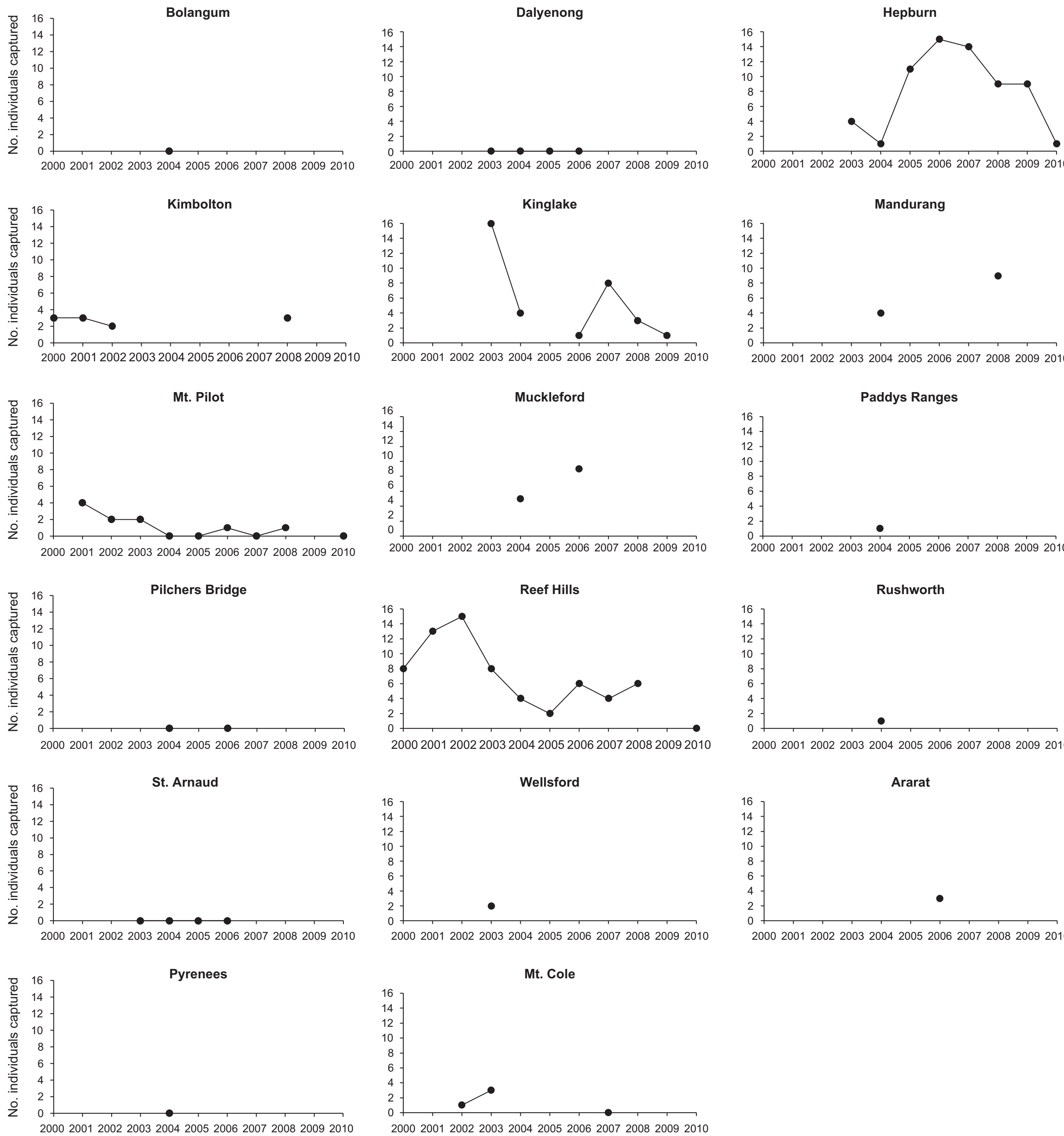

Fig. 2. Plots of the number of individual brush-tailed phascogales (*Phascogale tapoatafa*) captured in each year in which a monitoring site was surveyed. Lines connect surveys in consecutive years at a site.

one end of a cardboard cylinder (100 mm × 40 mm) and placed in traps. This cylinder provided insulation material to captured animals. A mixture of honey and water was sprayed on tree boles from the trap to ground-level to act as an attractant.

Traps were set for four consecutive nights, resulting in 320 trap nights of survey effort per site per survey (312 trap nights at Hepburn, Ararat, Pyrenees and Mt. Cole). Captured animals were identified, weighed, assessed for reproductive condition, and marked using a combination of ear notching and application of a small amount of non-toxic paint to the ear/tail before release.

2.4. Explanatory variables

A small number of environmental explanatory variables were developed to account for *P. tapoatafa* capture rates. A GIS layer of Ecological Vegetation Classes (Department of Natural Resources and Environment, 2002) was used in ArcMap 9.2 to describe the forest at each site. The dominant Native Vegetation Group was that which had the greatest total extent within a 5 km radius of the middle of each monitoring transect. A radius of 5 km was chosen as it: (1) encompassed the majority of the survey transect at each

Table 1
Output from generalised additive mixed models used to investigate: (a) changes in the standardised number of individual brush-tailed phascogales (*Phascogale tapoatafa*) captured through time; and (b) the relationship between the number of individuals captured and environmental explanatory variables.

Model	D^2	Explanatory variable	e.d.f.	F	P-value
(a) Change through time	0.28	*s*(Year)	1.00	12.23	0.001
(b) Environmental variables	0.41	NVG: Dry Forests[a]	n.a.[b]	−2.32[c]	0.024
		s(Mean annual rain)	1.00	10.97	0.002
		s(CPMR, by = BIF)	1.41	6.62	0.007
		s(CPMR, by = DF)	1.00	1.77	0.190

s(...) Indicates that an explanatory variable was fitted as a non-parametric smoothing term; e.d.f. represents the estimated degrees of freedom for each smoothing term; NVG = Native Vegetation Group; BIF = Box Ironbark Forests; DF = Dry Forests; CPMR = cumulative proportion of long-term mean annual rainfall received in previous 3 years.
[a] The reference category for the categorical variable Native Vegetation Group (NVG) was Box Ironbark Forests.
[b] Estimated degrees of freedom not applicable for parametric terms (i.e. explanatory variables not fitted with smoothing terms).
[c] *T*-values are used to test the significance of parametric terms.

site; and (2) defined an area relevant to the likely movements of captured individuals. The 17 monitoring sites were categorised by two distinct Native Vegetation Groups: (1) Box Ironbark Forest (Bolangum, Dalyenong, Kimbolton, Mandurang, Muckleford, Paddys Ranges, Pilchers Bridge, Reef Hills, Rushworth and Wellsford); and (2) Dry Forest (Hepburn, Kinglake, Mt. Pilot, St. Arnaud, Ararat, Pyrenees and Mt. Cole).

Rainfall data were obtained from the Bureau of Meteorology. The long-term mean annual rainfall for each site was obtained for the nearest weather station for which rainfall data were available. The total annual rainfall during specific survey years was obtained from the nearest weather station for which complete data were available for the relevant year. Due to incomplete records, different weather stations were used for some sites across years.

Numbers of *P. tapoatafa* may be more strongly influenced by rainfall in preceding years rather than that in the actual year of survey. Thus, an additional variable was developed that represents the cumulative proportion of the long-term mean annual rainfall (CPMR) received at a site in the preceding 3 years. For example, if a site was surveyed in 2003, this variable is the cumulative proportional difference between the actual and long-term mean annual rainfall for the site for the years 2000–2002 (values <1 indicate rainfall deficit, values >1 indicate rainfall surplus). Similar variables were created for the preceding 1 and 5 years. However, correlations between these variables were high ($r > 0.5$). The 3 year interval was chosen as it represents the best compromise between accounting for delayed response to rainfall events (Rhind and Bradley, 2002) and the life-history of the study species (maximum longevity = 2 years).

2.5. Data analyses

Analyses of *P. tapoatafa* captures occurred in two stages. First, captures were modelled as a function of survey year to allow change through time to be investigated. Second, captures were modelled in relation to broad environmental variables. Small sample sizes and unequal survey effort across sites precluded the use of a single model to address these questions.

Generalised additive mixed models (GAMMs) were used for all analyses. Additive models facilitate the modelling of non-linear relationships (Wood, 2006); inspection of raw data suggested that trends in *P. tapoatafa* captures through time may not have been linear. GAMMs provide further flexibility via the inclusion of random effects to account for non-independent error structures (Zuur et al., 2009). Site was included as a random effect in all models to control for repeated measures at individual monitoring sites.

To analyse change through time in the number of *P. tapoatafa* captured, it was necessary to only include sites where: (1) *P. tapoatafa* was detected; (2) $\geqslant 3$ surveys had been conducted; and (3) surveys were mostly conducted in consecutive years (i.e. there were no gaps >1 year between surveys). This resulted in four sites being considered for this analysis (Hepburn, Kinglake, Mt. Pilot, and Reef Hills). Year of survey was included as the sole explanatory variable in the GAMM (fitted with a smoothing function to allow for the possibility of a non-linear change in the number of individuals captured over time). The total number of individuals captured in each survey of each site (using a Poisson error distribution) formed the response variable. However, model validation revealed unacceptable heterogeneity in residuals caused by discrepancies in the number of individuals captured across sites (capture rates were comparatively low at Mt. Pilot; Fig. 2). To remove this heterogeneity, each time series was standardised by subtracting the mean and dividing by its standard deviation (Zuur et al., 2009). This scaled all data in the same range without removing trends through time. The standardised capture data formed a continuous variable that was approximately normally distributed, allowing the use of a Gaussian error distribution.

To investigate relationships between *P. tapoatafa* captures and environmental variables, data from all 17 monitoring sites were included. Standardising the dependent variable was not appropriate for this analysis since the aim was to account for heterogeneity across sites using explanatory variables. Thus, the response variable was a simple count of the total number of individual *P. tapoatafa* captured in each survey of each site. A Poisson error distribution was employed. The three environmental explanatory variables (Native Vegetation Group, long-term mean annual rainfall, and CPMR) were included in a single model. Smoothing terms were fitted for the two continuous variables (long-term mean annual rainfall and CPMR). In addition to being included as an explanatory variable, Native Vegetation Group was specified as a by-variable of long-term mean annual rainfall and CPMR to test the prediction that rainfall variables influence capture rates differently, depending on vegetation type (Zuur et al., 2009). Since year of survey was not included in this model, the fit of a model with and without a temporal correlation structure was assessed (Zuur et al., 2009). This is important since, in addition to repeated measures at a site creating a non-independent error structure, temporal error correlation may also exist whereby surveys closer in time are more correlated than those further apart. All GAMMs were performed using the R statistical program (R Development Core Team, 2009) and the “mgcv” package ver. 1.5–6 (Wood, 2006). Additional source code was used to calculate model deviance (Elith et al., 2008).

3. Results

From a trapping effort of 19,088 trap nights, a total of 222 individuals of *P. tapoatafa* was captured at 12 of the 17 monitoring sites. The number of individuals captured in any given year was highly variable across sites, ranging from a minimum of zero to a maximum of 16 (Fig. 2). Further, with the exception of sites where

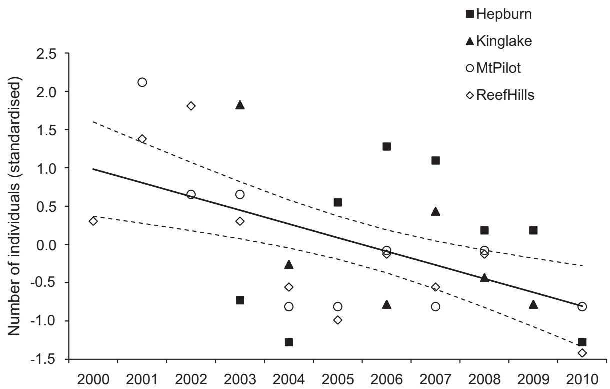

Fig. 3. The standardised number of individual brush-tailed phascogales (*Phascogale tapoatafa*) predicted to be captured between 2000 and 2010 by a generalised additive mixed model based on capture results from four monitoring sites (Hepburn, Kinglake, Mt. Pilot, and Reef Hills). Dashed lines represent the 95% confidence interval for predicted values. Actual standardised capture data for each monitoring site included in the model are also shown.

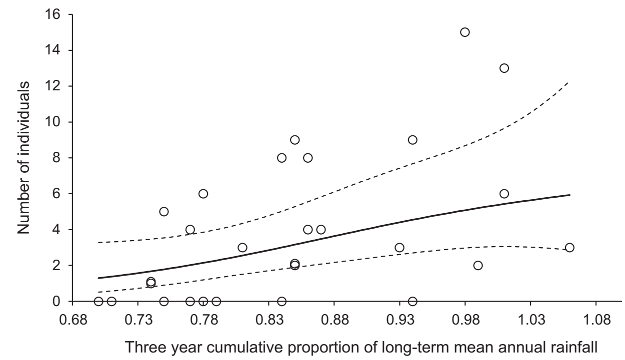

Fig. 4. The number of individual brush-tailed phascogales (*Phascogale tapoatafa*) predicted to be captured as a function of the cumulative proportion of the long-term mean annual rainfall recorded in the preceding 3 years at sites dominated by Box Ironbark Forests. Dashed lines represent the 95% confidence interval for predicted values. Predictions were generated using a generalised additive mixed model. Open circles represent actual data points (stacked symbols are used where data points overlap).

no individuals were encountered, capture rates were highly variable within sites, indicating that the size of local populations may vary markedly over time (Fig. 2).

3.1. Relative abundance through time

Plots of the number of individuals captured in each survey of each site show that capture rates were not consistent over time (Fig. 2). Year of survey was significantly related to the standardised number of individuals captured at the four sites (Hepburn, Kinglake, Mt. Pilot, and Reef Hills) for which suitable data were available (Table 1a). This suggests that: (1) the relative abundance of *P. tapoatafa* populations did indeed change over time during the monitoring period; and (2) a common pattern of change was evident across the four sites. Predicted values from the GAMM provide evidence that captures of *P. tapoatafa* have declined between the years 2000–2010 (Fig. 3). Despite having the freedom to fit a non-linear response, a linear decline in relative abundance provided the best fit (e.d.f. = 1; Fig. 3). This model explained 28% of variance in the data (D^2 = 0.28; Table 1a), a reasonable figure given the absence of other explanatory variables and the use of a single slope to describe trends across four geographically separated monitoring sites. Site-to-site variation is clearly present in the data; captures at Hepburn in particular show some deviation from model predictions. Nevertheless, evidence for a downward trend in relative abundance across multiple locations is strong (Fig. 3).

3.2. Influence of environmental explanatory variables

Inclusion of a temporal correlation structure (auto-regressive correlation; AR-1) to the GAMM relating capture rates to environmental variables substantially improved the fit of the model (AIC without correlation structure = 191.4; AIC with correlation structure = 181.0). This indicates that, within a monitoring site, the closer two surveys were in time, the higher their correlation (Zuur et al., 2009). Thus, all results are based on the model including this correlation structure.

The dominant Native Vegetation Group at a site influenced the number of *P. tapoatafa* captured (Table 1b); controlling for long-term mean annual rainfall and CPMR, capture rates were higher at Box Ironbark Forest sites than Dry Forest sites. Long-term mean annual rainfall at a site was also identified as an influential parameter: in both vegetation groups (by-variable removed due to common response) there was a positive, linear (e.d.f. = 1) association between captures and mean annual rainfall (Table 1b).

Values for the cumulative proportion of long-term mean annual rainfall (CPMR) ranged from 0.71 to 1.12, with a mean of 0.86 ± 0.01 (SE). Across all surveys at all sites (n = 60), 87% of years had CPMR values <1. This indicates that the study region experienced below average rainfall throughout much of the study period.

The number of animals captured was significantly related to CPMR at sites dominated by Box Ironbark Forests, but not at those dominated by Dry Forests (Table 1b). At Box Ironbark Forest sites, capture rates increased with CPMR (Fig. 4). This indicates that rainfall deficit over a 3 year period results in fewer *P. tapoatafa* being captured. Overall, the model relating individuals captured to environmental variables accounted for 41% of variance in the data (Table 1b).

4. Discussion

This case study highlights the fundamental importance of long-term monitoring data. A decline in the relative abundance of *P. tapoatafa* populations across four study locations over a 10-year period was identified. If species extinctions are to be avoided, it is critical to identify such declines as early as possible. This can only be achieved if monitoring data is at hand that allows population trajectories to be rigorously quantified. Relationships between the relative abundance of *P. tapoatafa* populations and broad environmental variables were also revealed. With careful consideration of relevant parameters, monitoring data has the potential to provide important insights that otherwise may go unnoticed. Such insights will often be valuable for understanding population trajectories.

There have been no previous formal analyses of long-term monitoring data for *P. tapoatafa* populations. Therefore, although recognised as being threatened with extinction in Victoria, contemporary population trajectories have until now been unknown. It was possible to analyse trends in relative abundance at four monitoring sites in this study. A common pattern of decreasing captures over time was identified, suggesting a decline in abundance during the monitoring period. The analysis identified a linear decline, although patterns for each site varied around model predictions. This variation is likely caused by heterogeneity across both space and time in habitat quality and local environmental conditions (see below). Despite this, evidence for an overall decline in relative abundance during the monitoring period is apparent and suggests that the status of *P. tapoatafa* populations in Victoria is not secure.

In Victoria, *P. tapoatafa* occurs in drier forests that receive moderate rainfall (Menkhorst, 1995). Long-term mean annual rainfall across the 17 monitoring sites ranged from 464 to 788 mm. Within this range, the number of animals detected increased with increasing mean annual rainfall. The most plausible explanation is that rainfall has a positive influence on food availability. *P. tapoatafa* feeds primarily on invertebrate taxa (Scarff et al., 1998; Traill and Coates, 1993). While factors influencing the abundance of invertebrates are complex (Recher et al., 1996; Woinarski and Cullen, 1984), there is evidence that biomass is positively associated with moisture levels (Scarff and Bradley, 2006; Taylor, 2008). If sites that receive higher rainfall do support a more abundant invertebrate fauna then they are also likely to support a greater number of *P. tapoatafa*. Other studies have concluded that aspects of *P. tapoatafa* life history are positively influenced by higher site productivity and associated food availability (Rhind and Bradley, 2002; van der Ree et al., 2001).

If rainfall does influence food availability then annual variation in rainfall could account for temporal variation in relative population size. This study provides some evidence for this, with the cumulative proportion of the long-term mean annual rainfall received at a site in the 3 years prior to survey influencing *P. tapoatafa* numbers at Box Ironbark sites. Below-average rainfall was experienced for much of the monitoring period and as the proportion of the mean rainfall declined (i.e. rainfall deficit), captures of *P. tapoatafa* also declined. Rainfall deficit was found to be an extremely limiting factor for *P. tapoatafa* in south-western Australia. Drought conditions not only resulted in lower population abundance, but were also correlated with a reduction in body size and sexual dimorphism (Rhind and Bradley, 2002), late timing of births and altered sex ratios of litters (Rhind, 2002), and an increase in communal nesting (Rhind, 2003). Reduced availability of invertebrates as a result of below-average rainfall was considered to be a key process driving these findings. Across central and northern Victoria, recent declines in insectivorous bird populations have been linked with reduced invertebrate biomass due to dry conditions (Mac Nally et al., 2009).

Female *P. tapoatafa* experience elevated mortality rates when raising young due to unusually high maternal investment (Soderquist, 1993). Additional stress placed on animals may increase mortality further. Therefore, any reduction to food supply associated with prolonged rainfall deficit may reduce population size via both direct adult mortality and associated failed reproduction (Rhind and Bradley, 2002). The highly synchronous, annual post-mating death of all adult males amplifies the vulnerability of the species to such factors (Traill and Coates, 1993). Population persistence depends on successful reproduction and natal recruitment each year; delaying reproduction until conditions improve is not an option for *P. tapoatafa*. Ongoing climate change is therefore of great concern for the future survival of this species.

While rainfall deficit influenced *P. tapoatafa* abundance at Box Ironbark sites, it was not influential at sites dominated by Dry Forests. Differences in long-term mean annual rainfall between the two groups of sites provide a plausible explanation. The Dry Forest sites generally occur at higher elevation and receive higher rainfall than the Box Ironbark sites (long-term mean annual rainfall: Box Ironbark mean = 557.6 ± 22.2 mm (SE); Dry Forest mean = 646.3 ± 35.8 mm (SE)). This higher rainfall may equate to greater invertebrate biomass and therefore, food supply (Scarff and Bradley, 2006; Taylor, 2008). At higher rainfall sites, periods of rainfall deficit may be of less consequence for invertebrate fauna. Alternatively, higher base levels of invertebrates may allow *P. tapoatafa* to better cope with reductions in food supply. While populations at Dry Forest sites did not decline in response to rainfall deficit, modelling results indicated that capture rates were actually lower at such sites compared to Box Ironbark sites, when rainfall variables were controlled. It is likely that invertebrate biomass, and habitat quality more generally, are governed by complex interactions between broad vegetation type, local site characteristics and variation in rainfall.

The relationship between rainfall deficit and dominant vegetation group could have important implications for management of *P. tapoatafa*. Rainfall deficit (drought) is a phenomenon that influences large spatial areas simultaneously, and may result in the synchronous decline of multiple populations (Koenig, 1999; Ranta et al., 1999). Populations in Dry Forest sites may be of particular conservation value since they are less likely to decline synchronously with populations in Box Ironbark sites during periods of rainfall deficit. However, population declines occurred at both site types. Moreover, *P. tapoatafa* was not detected at two Dry Forest sites (St. Arnaud and Pyrenees). Thus, factors operating at local scales (e.g. site-level measures of habitat quality, predation) clearly also are important.

The work of the Brush-tailed Phascogale Coordinating Group serves as an example of a monitoring program providing valuable long-term ecological data. Factors responsible for the success of the program include: (1) scientifically sound and appropriate project design and consistent methodology; (2) sustained effort over an extended period of time, and (3) ongoing commitment from a team of enthusiastic and knowledgeable people. To ensure ongoing success, the Coordinating Group has also undertaken a critical step that forms the basis of this paper: formal data analyses to identify (1) the status of *P. tapoatafa* populations, and (2) whether future efforts can be modified to maximise the program's effectiveness. From these analyses, we make three key recommendations to guide future monitoring:

1. *Reduce the number of sites monitored annually to a smaller number (e.g. eight) of key sites spread throughout the species' range*. Sporadic monitoring and associated small sample sizes resulted in population trajectories being analysed at four sites only. A more efficient approach would be to focus on a smaller set of key study sites and ensure that each site is monitored annually. This will likely produce useful information in the shortest time and for the least effort and expense. The subset of sites should be located throughout the species' range and be representative of its distribution (e.g. incorporating both Box Ironbark and Dry Forest vegetation types and populations displaying genetic differentiation).
2. *Systematically survey additional sites at greater time intervals (e.g. every 5 years) using labour-efficient techniques*. Surveying additional sites less frequently would allow the status of *P. tapoatafa* to be monitored at a larger number of locations to complement annual surveys of key sites. Labour- and cost-efficient techniques such as hairtubing and camera trapping could be employed if live-trapping is not feasible. Data provided from such techniques would be sufficient to monitor general population trends and to identify relationships with relevant explanatory variables.
3. *Collect site-level data relating to habitat characteristics and resource availability*. From limited research, the occurrence and life-history of *P. tapoatafa* has been shown to be influenced by local site characteristics. These include site productivity and the density of large trees with hollows (van der Ree et al., 2006, 2001), tree species composition (Traill and Coates, 1993), and food availability (Scarff and Bradley, 2006; Scarff et al., 1998). Other factors such as predation (Soderquist, 1993) and changes caused by wildfire and fuel reduction burning are also no doubt important. These site-level factors may interact with parameters operating at larger scales (e.g. rainfall deficit) to influence

P. tapoatafa populations. Thus, site-level data should be collected to allow investigation of specific hypotheses relating to population trajectories. Experimental manipulations (e.g. installation of nest boxes, predator control) may also provide valuable insights into processes driving population trends.

This case study illustrates the importance of long-term monitoring programs for documenting trends in population trajectories. For rare and threatened species, early detection of population declines is vital to allow time for appropriate intervention. In the absence of regular monitoring such early detection may not occur, particularly when dealing with cryptic species such as *P. tapoatafa*. However, knowledge of population decline is of limited value without an associated understanding of the causal mechanism(s) of decline (McDonald-Madden et al., 2010). While identifying such mechanisms can be challenging (e.g. Lindenmayer et al., 2011), monitoring programs must strive to test hypotheses related to population trajectories in addition to monitoring the trajectories themselves. Such targeted monitoring is more likely to produce results capable of guiding effective management (Caughley, 1994; Wintle et al., 2010).

Long-term monitoring is the cornerstone of many conservation activities around the globe. It is an unfortunate reality, however, that many datasets produced by monitoring activities are never rigorously analysed. Failure to conduct this step makes it difficult, if not impossible, to draw sound conclusions (Wintle et al., 2010). To capitalise on the resources invested in monitoring activities it is essential that data collected is subjected to appropriate analyses. Results can then be used for critical evaluation of the effectiveness of monitoring activities and to guide future endeavours to maximise both efficiency and the likelihood of positive biodiversity outcomes.

Acknowledgements

This project was funded by the Department of Sustainability and Environment, facilitated by Teigan Allen and Natasha McLean. The success of the Brush-tailed Phascogale Coordinating Group is due to the passionate support and input of many individuals (predominantly Department of Sustainability and Environment and Parks Victoria staff). We particularly acknowledge the contributions of the late John Seebeck and Steve Craig. Todd Soderquist designed the monitoring protocol that has been systematically applied across all sites. The efforts of many volunteers have allowed this program to operate on the smell of an oily rag. Particular thanks to Julie Flack, Geoff Barrow, David Tyson and Brian Pritchard (Parks Victoria). The comments of three anonymous reviewers improved the quality of the manuscript.

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