ANALYSIS OF CAPTURE DATA:

a case study using Program MARK for analysis of brushtail possum trapping data and its relevance to conservation management of the western ringtail possum

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I hereby declare that this is my own account of research completed while enrolled at Murdoch University, and has not previously been submitted for a degree at any other university.

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November, 2007

Cover photo: Common brushtail possum - *Trichosurus vulpecula* (top left and right), southern brown bandicoot - *Isoodon obesulus* (bottom left), chuditch – *Dasyurus geoffroii* (bottom right).

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Abstract

This study presents an example of the use of program MARK and the Information-Theoretic (I-T) approach in obtaining population estimates for common brushtail possums at Leschenault Peninsula Conservation Park and Yalgorup National Park in south-western Western Australia. These conservation reserves are translocation release sites for the western ringtail possum, *Pseudocheirus occidentalis*. The aim of this project was to quantify the population size of the common brushtail possum, *Trichosurus vulpecula*, at these sites, to act as baseline data for future work in assessing its availability as an alternative prey species for predators of the western ringtail possum. The study also served as a pilot study to evaluate the use of trapping webs and analysis using the I-T approach through program MARK.

Trapping was conducted using Sheffield wire cages and Elliott aluminium folding traps at six 10 ha trapping webs. Trapping was carried out over spring 2006 and summer 2007 for five consecutive nights for each season. Captured animals were individually marked and morphological data was recorded.

Data analysis involved obtaining population estimates using the Huggins Closed Captures model type in program MARK for data on common brushtail possums during the summer trapping session. The analysis incorporated use of covariates on dominant vegetation structure, presence or absence of 1080 fox baiting and the sex of each animal to derive estimates of population size. The data were analysed in a model selection framework to determine which model best described the data.

Over the spring and summer trapping periods a total of 311 individuals belonging to fifteen species were captured. The species most commonly captured was the common brushtail possum. Analysis of brushtail possum data indicated a general trend towards three categories of baiting/vegetation interactions. The highest population estimates were obtained at the two trapping webs which were 1080 baited and consisted of an open vegetation structure. The two baited webs with dense vegetation had lower population estimates but these were comparable to each other. The lowest population estimates were obtained at the unbaited site, with a marginally higher population estimate occurring at the densely vegetated trapping web.

These results are in accordance with the hypothesis which stipulated that where the risk of predation was low (in baited areas) brushtail possums would prefer vegetation which provides optimum habitat (open vegetation structure), and that where predation risk is higher (in unbaited areas), the possums would select dense vegetation which provides a predator refuge. The data indicated 1080 baiting for foxes is benefiting brushtail possum populations and highlights the value of open vegetation with a high availability of tree hollows.

Evaluation of the effectiveness of trapping web design revealed that the assumption that the probability of capture at the centre is 1 was supported. The study found trapping sessions could potentially be limited to four days in the future. Recommendations were also made to increase the distance between trapping webs and between trap points within the webs.

The study highlighted the benefits of using multi-model inference and I-T approach in quantifying trapping data. This method proved more informative to that of using simple counts to provide an index of abundance. By modelling variables such as vegetation type, baiting regime, sex and by accounting for capture heterogeneity, it was possible to derive a large amount of information from a relatively sparse dataset.

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Chapter 1 Introduction

A question frequently asked in ecology is “how many are there?” Trapping is commonly undertaken to sample a population in order to estimate the population size of a target species in a particular area. This provides some very useful information on both species abundance and population demographics, which may aid in the management and conservation of the species.

There are two broad categories of methods which may be used to infer the size of a population from trapping data. The first involves Capture-Mark-Recapture (CMR) data, which allows the number of animals observed to be corrected for the detection probability, providing estimates of the actual population (Sutherland, 2006; White, 2005). The second involves obtaining an index of the population, in the case of which marking of animals is preferred, but not compulsory.

Indices provide a measure related to abundance without estimating the actual population size (Sutherland, 2006). Such indices allow comparisons between seasons or sites provided it can reasonably be assumed that the probability of capture does not vary. Population indices therefore assume that variation in the index is proportional to variation within the population (White, 2005). Commonly applied abundance indices include capture success per unit of effort or known to be alive methods.

Known to be alive methods are based on the minimum number of individuals known to be alive (MNA) within the study area. This includes all animals observed during a sampling period. If studies extend over several sampling periods, an animal may be added to the MNA even if it was not observed during a particular occasion. An animal is deemed ‘alive’ if it was observed before and after a capture occasion, based on the logical assumption that it was alive but not detected (Sutherland, 2006).

Capture success involves calculating the number of captures, usually per 100 trap nights. A trap night is equivalent to one trap set for one night (Hafner, 1977; Krebs, 2001). There are two methods for obtaining capture success; by using all captures or by excluding recaptures from calculations (i.e. capture rates for individual animals).

Neither method for calculating capture success is ideal. Including all captures in capture success rates may result in overestimation of the actual population if the targeted species shows trap happiness (Sutherland, 2006). Using only individuals to calculate capture success may be more representative of the actual population but does not consider recaptures. This may result in underestimation of the population as this approach does not consider how many traps may already be occupied by recaptured animals, potentially preventing the capture of new individuals. The decision of which approach to adopt should be based on the biology of the targeted species.

While capture success and known to be alive methods do allow some comparison of population size, they may often be unreliable and biased (White, 2005). It may not be reasonable to assume that capture probability will not change between seasons or between study sites. Capture success based on all captures may be particularly unreliable as this method does not consider trap heterogeneity or the trapping history (naïve vs. experienced) of an individual.

Quantifying trapping data using population indices may often be sufficient for comparisons within a study, where variables are kept relatively constant. However, it is often impossible to reliably compare trapping data between studies. Where practical, it is preferable to use CMR methods to determine absolute population size. Such methods often allow population densities to be derived, given that the effective trapping area is known, and generally provide more information than simple abundance indices (Sutherland, 2006).

A review, undertaken during this study, of terrestrial vertebrate trapping studies published in 2002 - 2006 in three of Australia's leading journals – Wildlife Research, Australian Journal of Zoology and Austral Ecology (formerly Australian Journal of Ecology) revealed 58 articles reported on population size, density or abundance from trapping data.

The majority of studies (72.4%) used some form of capture rate as an abundance index. Capture-Mark-Recapture methods were employed in 17.2% of studies, other methods were used for 5.2%, 10.3% of studies did not specify their methods of analysis, while 3.4% did not quantify abundance/population size at all (Note: percentages sum to over 100% as some studies incorporated more than one method). Of the studies publishing data in the form of abundance indices, 24.1% of studies presented the total number of captures, 19.0% used capture success, and 5.2% published the mean number of captures. A further 24.1% of journal articles used known to be alive methods.

The review of journals revealed a frequent lack of adequate description of the analyses used. Of the 19.0% of studies presenting capture success, only 18.2% specified that these capture rates were based on individual captures and 9.1% involved all captures. The remaining studies did not specify how capture success was derived. Given the large differences which may exist between the two methods, it is easy to see why there are difficulties in making comparisons of abundance between studies. Of the 58 journal articles examined 10.3% did not even describe how population/abundance estimates were obtained.

The majority of the studies examined published trapping data in the form of abundance indices. Only 17.2% of studies used CMR methods to obtain population estimates or densities. The occasional study, which compared abundance indices, explained these were used because the data was too sparse (contained too few captures) to enable more detailed analyses. There is a clear need for the evaluation and adoption of methods for analysis of sparse data in a way which allows meaningful information to be inferred and which enables reliable comparisons to be made between studies.

Software packages such as program MARK (White, 2001), which utilise the Information-Theoretic (I-T) approach, were designed specifically for this purpose. This study presents an example of the use of program MARK (White, 2001) and the I-T approach in obtaining population estimates for common brushtail possums at Leschenault Peninsula Conservation Park (LPCP) and Yalgorup National Park (YNP) in south-western Western Australia (WA). These conservation reserves are translocation release sites for the western ringtail possum, *Pseudocheirus occidentalis*. One of the aims of this project was to quantify the population size of the common brushtail possum, *Trichosurus vulpecula*, at these sites, to act as baseline data for future work in assessing its availability as an alternative prey species for predators of the western ringtail possum.

In this introductory chapter, the basic principles of the I-T approach will be explained, followed by a description of the capabilities of program MARK (White, 2001). The program was used to assess the availability of common brushtail possums in the context of conservation management of the western ringtail possum. As such, a brief background on the decline and on translocations of ringtail possums is provided, followed by information on the biology of brushtail possums. Introduced predators have been implicated in the decline of western ringtail possum populations. A brief background on introduced predators in Australia is provided, including an evaluation of the impacts introduced predators may have on native fauna, both directly through predation as well as indirectly (e.g. through mesopredator release). Finally, the aims of the project are outlined.

1.1 Information-Theoretic

Increasingly, scientists are moving away from the traditional null-hypothesis testing when analysing data. The major drawback of null-hypothesis testing is that it is relatively uninformative and does not utilise the full potential of the data (Burnham & Anderson, 2001). Information-Theoretic (I-T) approaches, on the other hand, allow a multitude of inferences to be made from limited data. I-T methods allow the selection of the 'best' model from an *a priori* set and enable ranking of these models.

Selection of the 'best' model involves determining the model that optimally approximates reality given the data (Burnham & Anderson, 2001). In other words, the best model is the model that is closest to the truth. From an ecological perspective, a model which accurately describes the truth is unlikely to exist. Models are only estimations of the truth and selecting the best model is a trade-off between bias and variance (the principle of parsimony). Models containing too few parameters (variables) are subject to bias, whereas models with too many parameters have high variance (i.e. poor precision) (Burnham & Anderson, 2001).

The difference between reality and the approximating model may be estimated using Kullback-Leibler (K-L) information which is defined as

$$I(f, g) = \int f(x) \log_e \left(\frac{f(x)}{g(x \mid \theta)} \right) dx$$

where $I(f, g)$ (K-L information) is the information lost when model g is used to approximate reality, f (Buckland *et al.*, 1993; Burnham & Anderson, 2001).

Akaike (1973; 1974) developed Akaike's Information Criterion (AIC) which allows model selection using the relationship between K-L information and maximum likelihood (Akaike, 1974; Burnham & Anderson, 2001). This is defined as

$$\mathrm{AIC} = -2\log_e(L(\theta\text{-hat} \mid data)) + 2K,$$

where $\log_e(L(\theta\text{-hat} \mid data))$ is the value of the maximised log-likelihood over the unknown parameters (θ) given the data and the model, and K is the number of estimatable parameters in the approximating model (Akaike, 1974).

Akaike's Information Criterion may perform poorly if too many parameters are modeled in relation to the sample size (Burnham & Anderson, 2002; Sugiura, 1978). To account for this, Sugiura (1978) and Hurvich and Tsai (1989) built upon AIC by adding a bias-correction term. The result was AIC_c which is defined as

$$\mathrm{AIC}_c = \mathrm{AIC} + \frac{2K(K+1)}{n - K - 1},$$

where n is the sample size. As a general rule of thumb, use of AIC_c is recommended when the ratio n/K is small (<40) (Burnham & Anderson, 2002).

Akaike's procedures are now referred to as Information-Theoretic. This approach requires selecting an *a priori* candidate set of models. The best model is then selected by assigning an AIC (or AIC_c) value for each model within this set. The model with the lowest AIC value is considered the best fit for the data (Akaike, 1974; Buckland *et al.*, 1993; Burnham & Anderson, 2001). Models are ranked by rescaling the AIC values so the model with the minimum AIC has a value of 0 (Burnham & Anderson, 2001; White, 2005). This is achieved using ΔAIC values which are calculated using the formula

$$\Delta_i = \mathrm{AIC}_i - \mathrm{minAIC}.$$

This transformation allows easy comparison of the likelihood of the candidate models. Models with the lowest ΔAIC values are considered best. As a general rule of thumb, models with $\Delta\mathrm{AIC} \leq 2$ are strongly supported by the data, models with $2 \leq \Delta\mathrm{AIC} \leq 7$ have some, but considerably less support, whereas models with $\Delta\mathrm{AIC} > 10$ have essentially no support (Burnham & Anderson, 2001; Burnham & Anderson, 2002). Akaike weights (w_i) are the 'weight of evidence' in favour of model *i* as being the K-L best model in the candidate set and may be interpreted as the probability that the model is in fact the best model (Burnham & Anderson, 2001).

Where several models are strongly supported by the data, it is possible to model average, i.e. to combine estimates from multiple models (Burnham & Anderson, 2002). Model averaging incorporates model selection uncertainty into the estimates by combining information from all models proportional to the model's Akaike weight (Burnham & Anderson, 2001; White, 2005). This allows for greater precision of estimates as a larger amount of information may be used (Burnham & Anderson, 2002; White, 2005).

When examining trapping data, overdispersion of the data is to be expected. Overdispersion occurs when there is more variation than predicted by Poisson or multinomial probability distributions, or when $\hat{c} > 1$ (where $\hat{c}$ is the overdispersion coefficient). When no overdispersion exists $\hat{c} = 1$ (Burnham &

Anderson, 2001). Where overdispersion occurs, the quasi-likelihood modification to AIC (QAIC) is used and is denoted as

$$QAIC = \frac{-2\log(L(\hat{\theta}))}{\hat{c}} + 2K \quad .$$

There are a number of software applications (e.g. SURVIV, JOLLY, JOLLYAGE, SURGE, SURPH) available for analysis of Capture-Mark-Recapture data according to the I-T approach (Cooch, 2001). The software of interest which will be discussed further is program MARK (White, 2001).

1.2 Program MARK

Program MARK was developed by Gary White (2001) to facilitate analysis of data from marked individuals. The software provides parameter estimates based on model selection and likelihood theory (all estimates are maximum likelihood estimates). Program MARK (White, 2001) allows the user to define the parameters, or variables, to be modeled. These include group-specific, time-specific or individually specific covariates (White, 2005). This allows multiple sparse datasets to be combined to provide additional information on the parameters (White, 2005). Models are defined by placing constraints on parameters, which reduces the number of separate parameters to be modeled and so improves the precision of estimates. However, this comes at the cost of introducing bias by making additional assumptions (Sutherland, 2006).

For all data analysis, MARK adopts a linear approach to model fitting (Cooch, 2001; White, 2001). The linear model may be expressed as follows:

$$\mathbf{y} = \mathbf{X}\boldsymbol{\beta} + \boldsymbol{\varepsilon}$$

Where y is a vector of the response variables, X is a matrix with independent variable ("1" or "0") elements, β is a vector of parameters and ε is a vector of random error terms (Cooch, 2001; Cooch & White, 2007). Beta (β_x) values correspond to each of the parameters, including the intercept, and are equivalent to the slope of the linear model. Therefore, a positive β value refers to a positive slope (i.e. upward trend) and a negative β value refers to a negative slope (i.e. a downward trend).

Program MARK (White, 2001) allows analysis of a variety of data types including CMR, dead recovery and telemetry data. It also allows analysis of data for 17 classes of model types (White *et al.*, 2001). These include Closed Capture models, which assume the targeted population is closed for the duration of the trapping period. "Closed" is defined as having no births, no deaths, no immigration and no emigration (Kendall, 1999).

Closed Capture models (Otis *et al.*, 1978; White *et al.*, 1982) allow the estimation of population size (N) by modeling the initial capture probability (p) and the recapture probability (c). Capture and recapture probabilities may be modeled by attribute groups or as a function of time, but not as a function of individual-specific covariates. This is a result of incorporating animals which were never captured into the likelihood in order to estimate population size. Clearly, individual covariates can not be measured for animals which were never captured.

Huggins Closed Capture models (Huggins, 1989; Huggins, 1991) do not condition population estimates (*N-hat*) into likelihood and so allow individual-specific covariates to be modeled. The trade-off is that population estimates are more biased (i.e. not as efficient) than with the Otis Closed Captures data type.

However, Huggins population estimates perform reasonably well when the proportion of the population trapped is at least 60% (White, 2002). As it is likely that at least some degree of capture heterogeneity will exist among brushtail possums requiring individual covariates to be modeled, this project focuses on analysis of CMR data using the Huggins Closed Captures model type.

The ideal opportunity to test the applicability of program MARK (White, 2001) and the I-T approach on sparse datasets arose in the context of a case study on western ringtail possum declines. Population estimates were obtained for common brushtail possums at translocation release sites for the western ringtail possum.

1.3 Western ringtail possum background and translocations

The western ringtail possum is a threatened marsupial species endemic to the south-west of Australia (Maxwell *et al.*, 1996). The species has undergone a considerable reduction in distribution since European settlement (de Tores *et al.*, 2005; Jones *et al.*, 2004; Maxwell *et al.*, 1996; Wayne *et al.*, 2005b). As with most species, its decline and failure to recover to stable levels despite conservation efforts may be attributed to a number of factors. Management of the western ringtail possum must be viewed from a holistic perspective which considers all factors present in the environment such as habitat quality, climatic conditions, human disturbances, disease and other species in the community, including both introduced and native predators.

The western ringtail possum is one of the many species to have undergone population decline following human colonisation. Ringtail possums historically occurred over much of the south-west of Australia from north of Perth to east of Albany, although fossil records suggest their distribution may have been even larger (de Tores *et al.*, 2005; Jones *et al.*, 2004; Maxwell *et al.*, 1996). Since European settlement their range has contracted by 50 – 90% (Maxwell *et al.*, 1996; Wayne *et al.*, 2005b). The species is now absent from much of its former northern and inland range and is concentrated in three areas around Busselton and Bunbury, Manjimup and Albany (de Tores *et al.*, 2005; Maxwell *et al.*, 1996; Wayne *et al.*, 2005b). This has resulted in a listing of the species as "threatened" by the World Conservation Union (IUCN) in 1996 (Australasian Marsupial & Monotreme Specialist Group, 1996). Presently, the highest densities of western ringtail possums are thought to occur in near-coastal areas in the Busselton region, located at the southern point of the Swan Coastal Plain (Wayne *et al.*, 2005b).

Reasons for the decline of the western ringtail possum include land clearing, logging, changes in fire regimes, predation by feral cats (*Felis catus*) and foxes (*Vulpes vulpes*) and competition with the common brushtail possum (Jones *et al.*, 2004; Maxwell *et al.*, 1996; Wayne *et al.*, 2006). It is believed the effects of predators are magnified in areas with sparse vegetation structure.

Foxes are less likely to have a significant impact on possum species in areas with high continuity of tree canopy. This allows possums to limit the time spent on the ground foraging and therefore reduces their susceptibility to predation (How & Hillcox, 2000; Jones *et al.*, 2004; Wayne *et al.*, 2006).

The western ringtail possum is a specialised arboreal folivore with peppermint (*Agonis flexuosa*) foliage constituting up to 79 - 99% of their preferred diet (Jones *et al.*, 1994b). For this reason, the possums reside in areas where peppermint is the dominant or co-dominant tree species (Jones *et al.*, 2004). Tuart (*Eucalyptus gomphocephala*) is equally important for habitat use. Western ringtail possums rest in either dreys built in peppermint or in hollows in old tuarts (Jones *et al.*, 1994b). Although peppermint is locally common around Busselton, increasing pressures from residential and semi-rural developments have resulted in habitat destruction and fragmentation in the greater Busselton and Bunbury area. This has necessitated translocations of western ringtail possums (de Tores, 2005).

The Department of Environment and Conservation (DEC; previously CALM) has been conducting translocations of western ringtail possums since 1991. Between 1991 and 2001 a total of 381 possums were translocated to four release sites. These included Leschenault Peninsula Conservation Park (106 possums from 1991-1997), two sites within Yalgorup National Park (142 possums from 1995-2001) and Lane Poole Reserve / Keats Forest Block (133 possums from 1996-1999) (de Tores, 2005). All four sites were baited for fox control using dried meat baits containing the toxin sodium monofluoroacetate, also known as 1080 (de Tores, 2005; de Tores *et al.*, 2004).

Initial monitoring showed some level of success at each of the four sites (de Tores *et al.*, 2004; de Tores *et al.*, 1998). However, full criteria for a successful translocation had not been met at any of the sites. Further monitoring at LPCP revealed extensive surveying effort was required to detect western ringtail possums, indicating the population had declined severely (de Tores, 2005).

Additional translocations of western ringtail possums resumed in 2004 and are continuing (de Tores, 2005). Possums are being translocated to LPCP and YNP, including to a newly established unbaited control site within YNP (de Tores, 2005). Initial results have indicated a higher rate of predator-related deaths at the baited site at LPCP, mostly caused by cats and pythons, than at the unbaited site at YNP (de Tores, 2005).

Foxes were initially implicated in the failure of the original translocations. However, it is unlikely that a single factor, such as predation by foxes, can be responsible for fauna decline (de Tores *et al.*, 2004). Alternative hypotheses included predation by feral cats, chuditch (*Dasyurus geoffroii*) and south-western carpet pythons (*Morelia spilota imbricata*), competition with common brushtail possums, disease (e.g. toxoplasmosis), drought, unsuitable habitat, and prey switching (de Tores *et al.*, 2004). The higher rate of predation by feral cats and pythons at the baited LPCP translocation sites may also be indicative of mesopredator release (discussed further in section 1.5.3).

In addition, spatial and temporal variation in the availability of other prey species may be low, which would require predators to prey heavily upon western ringtail possums at particular times. Prey switching was considered a possibility at LPCP as the decline in western ringtail possum numbers may have coincided with the previously abundant rabbit population crashing following management practices (de Tores, 2007, pers comm.; de Tores *et al.*, 2004). Rabbits (*Oryctolagus cuniculus*), when abundant, have been found to be the dominant prey species for both foxes and feral cats (see below and: Brunner *et al.*, 1975; Catling, 1988; Molsher *et al.*, 1999; Risbey *et al.*, 1999). A severe reduction in this food resource may have caused exotic predators to increase their consumption of native species, such as the western ringtail possum. Previous studies in eastern Australia have found the common ringtail possum, *Pseudocheirus peregrinus*, to occur frequently in the diet of the feral cat and fox (Brunner *et al.*, 1975; Roberts *et al.*, 2006; Triggs *et al.*, 1984). This would suggest ringtail possums may be a preferred prey species in the absence of rabbits.

This study examines the availability of alternative prey species to western ringtails with a specific focus on brushtail possum populations. Both species are arboreal possums and may be expected to be subject to similar predation pressures, assuming predators do not actively select for either species.

1.4 Common brushtail possum

The common brushtail possum, or koomal, is a widely distributed species common throughout the Australian continent. However, the south-west Australian subspecies *T. vulpecula hypoleucus* has been in decline since the early 20th century as a result of habitat loss, disease, and thinning by trappers in the past (How & Hillcox, 2000). The subspecies has declined in over 50% of their former range in the south-west of WA since European settlement (How & Hillcox, 2000). This has resulted in a listing under the Lower Risk (near threatened) Category in the *1996 Action Plan for Australian Marsupials and Monotremes* (Maxwell *et al.*, 1996). *T. v. hypoleucus* is the only subspecies of brushtail possum to have a conservation listing. It is geographically isolated from other subspecies, resulting in morphological differences such as a smaller size and longer, dense fur (Wayne *et al.*, 2005c).

Generally, female brushtail possums reach sexual maturity between 1 – 4 years of age depending on the population (Wayne *et al.*, 2005c). A study of the Abba River (near Busselton, WA) brushtail possum population (How & Hillcox, 2000) revealed females may not reach sexual maturity until their third year. Only one breeding season was observed in the year. Births occurred from late autumn to early winter (April to June), with births peaking in May. A similar single autumn breeding season was also observed further inland, at Chariup (Wayne *et al.*, 2005c).

The peak population density at Abba River during 1991 and 1992 was estimated to be 1.77 - 2.84 ha^{-1} using both the Lincoln index and known to be alive methods (How & Hillcox, 2000). This was the highest population density recorded in any natural population in WA to date. The largest home range was estimated at 4.9 ha for a male and 4.3 ha for a female (How & Hillcox, 2000). Studies of introduced populations of brushtail possums in New Zealand have found males to be more mobile than females (Cowan &

Clout, 2000), leading to a higher turnover rate of males within a population (Efford, 2000). Young males tend to disperse from their mothers when establishing their home range and usually settle at least 2 – 3 home ranges away from their mother's (Clout & Efford, 1984; Cowan & Clout, 2000). Females show a higher degree of fidelity to the natal home range (Clout & Efford, 1984). Juveniles disperse at the age of about 8 months, when they have reached sexual maturity (Cowan & Clout, 2000). Until that time they accompany their mother continuously (Day *et al.*, 2000).

Brushtail possums usually have den sites in the trunks and branches of tuarts and also use peppermint trees frequently (Jones & Hillcox, 1995). They may also rest in hollow logs and rabbit warrens (How & Hillcox, 2000). Brushtail possums have been observed evicting ringtail possums from tree hollows without any resistance from the smaller ringtails (How & Hillcox, 2000). Although predominately an arboreal species, common brushtail possums spend 10 – 15% of their time feeding and moving about on the ground (Cowan & Clout, 2000).

The diet of brushtail possums does not vary with season significantly and consists mainly of peppermint, flooded gum (*Eucalyptus rudis*), tuart, native wisteria (*Hardenbergia comptoniana*) and swamp paperbark (*Melaleuca raphiophylla*) (How & Hillcox, 2000). They are generalist and opportunistic feeders and eat leaves, fruit, flowers, insects, bird's eggs and occasionally meat (Nugent *et al.*, 2000).

Factors such as predation risk from foxes may cause brushtail possums to come down to the ground less frequently. Pickett *et al.* (2005) observed such a behavioural response in stands of cypress-pine but not in eucalypt stands. They hypothesise feral cats were more likely to replace the predator pressures of foxes in eucalypt stands than in cypress-pine. Changes in foraging behaviour in response to differing predator pressures have also been observed in other studies (Banks, 2001; Lima & Dill, 1990).

1.5 The impact of introduced predators

In order to establish a framework against which future work on the impacts of predation in western ringtail possums can be interpreted, it is useful to explore the impacts of introduced predators and their interactions with prey populations. Introduced predators have been implicated in the decline of a number of Australian species and are considered to have altered the ecology of many parts of the country. Fox baiting (using 1080) is commonly utilised to aid in the conservation of declining populations. However, it is important to be informed about the predator-prey relationships and about the potential effects of predator control (such as mesopredator release) when undertaking any such actions.

Almost half of the world's mammalian extinctions in recent history have been Australian species (Johnson, 2006; Johnson *et al.*, 2007; Short & Smith, 1994). There are three primary factors which may be responsible for this; habitat alteration, diversion of natural resources to humans and introduced animals (e.g. agriculture), and direct threat to populations by introduced predators and by habitat fragmentation (Burbidge & McKenzie, 1989).

The significance of the impact introduced species have on populations of native fauna has been recognised in the Commonwealth *Endangered Species Protection Act 1992* with the listing of predation by feral cats and foxes as key threatening processes. Introduced species have also been assigned the highest priority ranking for environmental issues in the latest West Australian State of Environment Report (EPA, 2007). There are now three exotic cursorial predators in Australia and these have become widely distributed; the European red fox, the domestic cat and the dingo (*Canis familiaris dingo*) (Burbidge & McKenzie, 1989).

Although the exact time and mode of arrival of cats in Australia is not certain, Abbott (2002) concluded that cats were introduced following European settlement in a number of locations, having first arrived on the east coast of Australia in 1788 (Abbott, 2002; Jones & Coman, 1981; Rolls, 1969). In addition to accidental dispersal, domestic cats were intentionally released into the wild during the 19th century (Rolls, 1969). Cats were quick to colonise; they were recorded in European settlements in the south-west of Australia by 1826 and were well established in the wild by the 1850s. They have since dispersed over the entire continent becoming Australia's most widespread carnivore (Burbidge & McKenzie, 1989).

The European fox was first successfully introduced to Australia in 1871 (although previous efforts date back as early as 1845) when it was deliberately released in Victoria for recreational hunting (Burbidge & McKenzie, 1989; Jarman, 1986; Rolls, 1969; Saunders *et al.*, 1995). Foxes were abundant in Victoria by 1885 (Jarman, 1986). Within 50 years of introduction foxes had spread to WA with the first record in 1912 (Kinnear *et al.*, 1988; Saunders *et al.*, 1995). Foxes follow a similar distribution pattern to that of rabbits, a major component of their diet, and both species spread throughout Australia at similar rates (Jarman, 1986).

1.5.1 Impact of introduced predators on native fauna

There is some debate regarding the extent of the impact introduced predators have on native fauna. The impact of predators is often difficult to quantify as there are usually other factors involved in a species' decline. Most impacts stem from direct predation. Unequivocal evidence of predation on native species comes from the large volume of dietary studies conducted on cats and foxes (Brunner *et al.*, 1975; Catling, 1988; Coman, 1973; Croft & Hone, 1978; Jones & Coman, 1981; Lunney *et al.*, 1990; Martensz, 1971; Paltridge, 2002; Paltridge *et al.*, 1997; Read & Bowen, 2001; Triggs *et al.*, 1984). Rabbits and rodents constitute a major component of diet for both feral cats and foxes (Coman, 1973; Croft & Hone, 1978; Liberg, 1984; Martensz, 1971). The diet of both species is supplemented by other small mammals, birds, reptiles, amphibians, invertebrates, carrion and vegetation (Green & Osborne, 1981; Jędrzejewski & Jędrzejewska, 1992; Jones & Coman, 1981; Molsher, 1999). Cats have generally been found to consume more native mammal species than foxes (Read & Bowen, 2001; Risbey *et al.*, 1999). Of the native fauna foxes do consume, the majority are reptiles and some birds (Martensz, 1971; Read & Bowen, 2001). However, these are generally a minor component of fox diet as the diurnal activity of reptiles and birds differs from the nocturnal habits of the fox.

Cats have been found to prey on a number of native mammals including vulnerable and endangered species (Horsup & Evans, 1993). Among the long list of the hunted are the lesser long-eared bat (*Nyctophilus geoffroyi*), common brushtail possum, southern bush rat (*Rattus fuscipes*), antechinus species (*Antechinus stuartii*, *A. swainsonii*), common ringtail possum, western ringtail possum, numbat (*Myrmecobius fasciatus*), rufous hare-wallaby (*Lagorchestes hirsutus*) and bandicoot species (*Perameles nasuta*, *P. gunnii*) (Friend & Thomas, 1994; Jones & Coman, 1981; Lundie-Jenkins *et al.*, 1993; Wayne *et al.*, 2005b).

Although both cats and foxes display seasonal changes in diet they are regarded as mostly opportunistic, and prey consumption generally reflects availability and abundance of the prey (Green & Osborne, 1981; Jędrzejewski & Jędrzejewska, 1992; Jones & Coman, 1981; Lever, 1959; Molsher, 1999). In some cases however, individuals may prey selectively on a particular species (Gibson *et al.*, 1994b; Green, 2002; Konecny, 1987). When this occurs disproportionately to the abundance of the preferred prey this can have severe ramifications for threatened species. For example, one individual cat was responsible for killing five juvenile allied rock-wallabies (*Petrogale assimilis*) out of a total of 11 juveniles in a population in Queensland. This contributed to the species' rapid decline over the next four years from 83 to 26 individuals (Molsher, 1999). There is no doubt fox predation played a role in the decline of the rufous hare-wallaby in the Northern Territory. A single fox was responsible for the extinction of one of the two last known wild populations of the species (Lundie-Jenkins *et al.*, 1993).

It is difficult to assign the cause of decline in many native species to predation by any single species. Species extinction and decline is most likely to occur as a result of the cumulative impact of factors such as habitat destruction and fragmentation, climate change, changes in fire regimes, disease, and the introduction of other feral species (Kinnear *et al.*, 1988). However, foxes and cats are likely to be significant contributors to the cumulative threatening processes affecting native fauna regardless of the reasons for the original decline of a population (Kinnear *et al.*, 1988).

Feral cat predation was the main reason for the failure of a reintroduction program for the rufous hare-wallaby in central Australia (Gibson *et al.*, 1994a; Gibson *et al.*, 1994b; Lundie-Jenkins *et al.*, 1993) and the brush-tailed bettong (*Bettongia penicillata*) in NSW (Priddel & Wheeler, 2004). Cats have also been a limiting factor in reintroduction attempts of the burrowing bettong (*Bettongia lesuer*) and the brushtailed phascogale (*Phascogale tapoatafa*) as well as numerous macropod species (Christensen & Burrows, 1994; Short *et al.*, 1992). Studies have also shown foxes to pose a significant threat to populations of numbats, malleefowl (*Leipoa ocellata*) and chuditch (Friend & Thomas, 1994; Friend & Thomas, 2003; Morris *et al.*, 2003; Priddel & Wheeler, 1997; Saunders *et al.*, 1995; Short, 2004). Populations of these species all increased following the application of intensive fox control using 1080 baiting.

1.5.2 Predator diets and prey availability

Despite an overwhelming volume of dietary studies conducted in the eastern states or in central Australia there is a surprising shortage of literature dealing with the diets of introduced predators on mainland WA. Although opportunities for dietary analyses in WA have often existed these haven't been fully exploited for various reasons. For example, Marlow *et al.* (2000) collected 204 foxes near Carnarvon and found most stomachs contained food; mostly remains of reptiles and mammals. However, the focus of the study was to determine demographic characteristics of a population of foxes with a reproductive focus and not to conduct a quantitative dietary analysis. Examination of stomach contents was therefore purely opportunistic due to the destructive nature of the sampling techniques employed.

In a notable exception, Risbey *et al.* (1999; 2000) examined the stomach contents of foxes on Heirisson Prong, WA. A comprehensive study was conducted manipulating and monitoring fox and feral cat abundance, which concurrently examined predator diet and monitored prey abundance. The geography of the Heirisson Prong peninsula allowed sectioning of the area into three zones of varying predator densities (1 - low cat, low fox; 2 – low cat, high fox; 3 – high cat, high fox). Dietary analysis was conducted by examining stomach and intestine contents of foxes and cats removed during routine control efforts. Mammals were found to be the most important prey group for both cats and foxes (with rabbits dominating their diets) followed by invertebrates, birds and reptiles. Cats were found to exhibit greater dietary diversity than foxes.

Surprisingly, there appears to be no published data on fox or cat diets in southwestern Australia, a region extensively controlled for introduced predators as part of the Western Shield program (Possinghman *et al.*, 2004).

Heirisson Prong lies considerably north of the area examined by this study and so care must be taken when making any predictions or comparisons between predator diets in the study area and elsewhere in WA.

Furthermore, it is important to note most dietary studies conducted thus far have involved analyses of scat or stomach contents of feral cats or foxes. These generally produce results in the form of frequency occurrence or weight composition of a particular species without relating this information to prey availability. Without this information it is difficult to judge what proportion of a prey species' population is being taken and what impact this has on the species. In order to accurately assess this prey availability must be determined concurrently with dietary analyses.

One study in which prey availability and diet analyses were conducted concurrently was conducted by Paltridge (2002). Scat analyses of foxes, cats and dingoes were conducted in the Tanami Desert (NT) concurrently with abundance monitoring of prey species. Prey species abundance was determined by a number of means. Bird abundance was assessed using aural and visual walked transects. Invertebrates, small reptiles and small mammals were trapped using two lines of 25 Elliott traps and two lines of five 25 L bucket pitfall traps. Sand plots were used to assess the activity/abundance of macropods, varanids, bilbies (*Macrotis lagotis*) and Australian bustards (*Ardeotis kori australis*). It must be noted that sandplots do not allow identification of individuals making it difficult to assess how many individual have visited the area. Sandplots only give an index of relative activity and are not the most effective means of determining abundance (de Tores & Berry, 2007). However, Paltridge (2002) employed a range of techniques in assessing relative abundance and did this sufficiently to allow comparison with predator diet.

The study was particularly interesting as it was the first study to simultaneously monitor cat, fox and dingo diet in relation to prey availability. It was also the first study to compare these factors in an area where rabbits do not occur – this was particularly significant considering most studies to date had found rabbits to be the staple prey species for both cats and foxes. Paltridge (2002), contrary to previous dietary studies, did not find mammals dominated diet and instead reptiles contributed more to the diets of cats, foxes and dingoes. Despite mammalian prey being of lesser importance than elsewhere, several vulnerable mammals were found to be consumed.

Though certainly an interesting and important study, it is difficult to extrapolate any findings to the situation in south-west Australia. The study occurred at a time when mammal availability was generally low, which may have amplified the value of reptiles in predator diets. The study confirmed the opportunistic nature of predation by foxes, cats and dingoes as the seasonal importance of prey items fluctuated in accordance to the abundance and activity of the prey species available. It would therefore be interesting to see if the results could be replicated under conditions with higher mammalian availability. Clearly, the climate, topography, vegetation, and therefore fauna assemblages present in the desert in the NT are markedly different to those of south-western Australia. The availability of reptiles would likely be much lower in the cooler, wetter climate in the south of WA. This highlights the gaps in knowledge in the region and the need for similar studies to be undertaken in various areas of the country.

1.5.3 Mesopredator release

Given the strong body of evidence indicating introduced predators do impact on native species at least to some extent, a common conservation management strategy has been to reduce introduced predator numbers. However, feral animal control also has the potential to reduce or even negate efforts to boost populations of target species of conservation concern. By manipulating one species the whole community may be affected.

Mesopredator release occurs when a dominant predator is reduced or removed, allowing population increase to occur in medium sized predators, also known as mesopredators (Courchamp *et al.*, 1999). By releasing mesopredators from predation or competition pressures by top predators, shared prey species are in turn subject to intensified predation pressures from mesopredators (Courchamp *et al.*, 1999; Johnson *et al.*, 2007). Removal of top predators has the potential to result in a severe reduction in populations of prey species and declines in biodiversity (Courchamp *et al.*, 1999; Johnson *et al.*, 2007; Palomares *et al.*, 1995).

The greatest potential for mesopredator release exists when interspecific competition or intraguild predation occurs (Robley *et al.*, 2004). Cats and foxes overlap in their distribution and share similar diets. They have also been shown to mutually avoid each other, resulting in spatial segregation (Molsher, 1999). Paltridge (2002) is one example of a study where cat remains were found in scats of other top predators (3.3% of fox scats and 9.1% of dingo scats). Cat abundance has been found to increase following fox control (Christensen & Burrows, 1994; Risbey *et al.*, 2000) suggesting foxes limit cat populations through interspecific competition. Additionally, Riseby, *et al.* (2000) observed a significant decline in small mammal captures where only fox control was carried out, whereas captures of small mammals increased where both cats and foxes were actively controlled. Therefore, there is evidence to suggest the effects of mesopredator release are of greatest concern when fox control is implemented without concurrent cat control – a common situation in Australia (Robley *et al.*, 2004).

The importance of simultaneous feral animal control programs is emphasised by island studies outlined by Courchamp *et al.* (1999). In many cases, the presence of rat species (*Rattus rattus, R. norvegicus* or *R. exulans*) may be detrimental to native island birds. Often, where cat control is implemented, rat populations may increase intensifying predation on bird species. On the other hand, if rodents are removed prior to feral cat control, bird species may face greater cat predation as a result of prey switching.

Further evidence of the effects of mesopredator release was provided by a statistical analysis of the distribution overlap of various native mammal species and dingoes with geographical information on species extinctions (Johnson *et al.*, 2007). The analysis unequivocally demonstrated that overlap with dingoes acts to protect ground dwelling mammals species from extinction and decline. Cats are a serious threat to small mammal species in the central and northwestern deserts of Australia from which dingoes are absent. This provides direct evidence that in the absence of top predators small mammal species are exposed to over predation by mesopredators.

1.6 Overview and aims of the study

There is a clear need for more research on the impacts of introduced predators and their diet specific to south-western WA. Further work is required relating the diets of introduced predators to the availability of prey species. The case study of the western ringtail possum provides an opportunity to investigate this area further.

The project aimed to determine the availability of the common brushtail possum as an alternative prey species to the western ringtail possum at sites where western ringtail possums have been translocated (at LPCP and YNP). The lack of unequivocal translocation success of the western ringtail possum has been attributed to a number of possible factors (de Tores *et al.*, 2005; de Tores *et al.*, 2004). Predator-prey relationships are hypothesised to be largely responsible and may involve mesopredator release and/or prey switching.

An assessment of availability of alternative prey species would allow future comparisons with the occurrence of prey items in the diet of foxes, feral cats and carpet pythons. A disproportionately high level of predation upon the western ringtail possum compared to its relative availability would suggest predators actively select for the western ringtail possum as prey. Any temporal differences in the preference for western ringtail possums might be informative for the timing of release of translocated possums. Furthermore, establishing and quantifying the impact of introduced predators on western ringtail possums will aid in effectively controlling these predators to assist in ringtail possum recovery. There is a need for future research on the role of cats and mesopredator release in fox baited areas. However, these are goals for the future, and the first objective is to determine effective ways of estimating alternative prey abundance. Thus the main questions this study poses are ‘is it possible to estimate population size of at least one of these alternative prey species, the common brushtail possum?’; and further, ‘can techniques such as use of trapping webs, which are theorised as appropriate when dealing with sparse data, be used routinely?’ A major component of the study was to assess whether trapping data from the study sites constituted sparse data and further, to assess the practicality of using trapping webs under these circumstances.

The assessment of prey availability is planned to continue beyond the life of this project. This project provides baseline data and the opportunity to generate hypotheses only. Any trends regarding the impacts of introduced predators and prey availability uncovered within the scope of this project may be further pursued and explored in the future. The study should also be viewed as a pilot study to evaluate the use of trapping webs and analysis using the I-T approach through program MARK (White, 2001). The analysis incorporated use of covariates on vegetation structure, fox baiting practices and sex of each animal to derive estimates of population size. The data were analysed in a model selection framework to determine which model best described the data.

1.6.1 Objectives

In summary, the objectives of this project can be grouped under three main points, namely;

1. To evaluate the usefulness of using program MARK (White, 2001) for analysis of trapping data with particular reference to the common brushtail possum populations at Leschenault Peninsula Conservation Park and Yalgorup National Park.

2. To obtain baseline data which may be used in the assessment of alternative prey species to the predators of the western ringtail possum at ringtail possum translocation sites, thereby contributing to the conservation management of this declining species.

3. To assess the feasibility of using trapping web design in obtaining abundance estimates for alternative prey species.

Chapter 2 Methodology

This chapter presents a description of the study sites located within Leschenault Peninsula Conservation Park (LPCP) and Yalgorup National Park (YNP) in south-west Western Australia, the timing of the study as well as the general trapping protocols employed. Methods of statistical analysis are also discussed.

2.1 Study sites

Vertebrate fauna were studied at each of three sites at which western ringtail possums were released as part of DEC's translocation program described earlier (Section 1.3). These included two sites which were baited for fox control using dried meat/sausage baits containing 3.0 mg of the toxin 1080; Leschenault (LPCP) and Preston Beach Road (YNP). The third site, Martin's Tank, was also within YNP and was unbaited. The location and relative position of the three sites is shown in Figure 1.

Each of the study sites incorporates two trapping webs. The location of each trapping web was selected based on the dominant vegetation type present in the area. At each study site, one trapping web was positioned in tuart dominated vegetation, and one web was located where peppermint dominated with few tuart trees present. The purpose of this was to provide some comparison of prey species abundance for the two major vegetation structural units commonly utilised by western ringtail possum (Jones *et al.*, 1994a; Jones *et al.*, 2004).

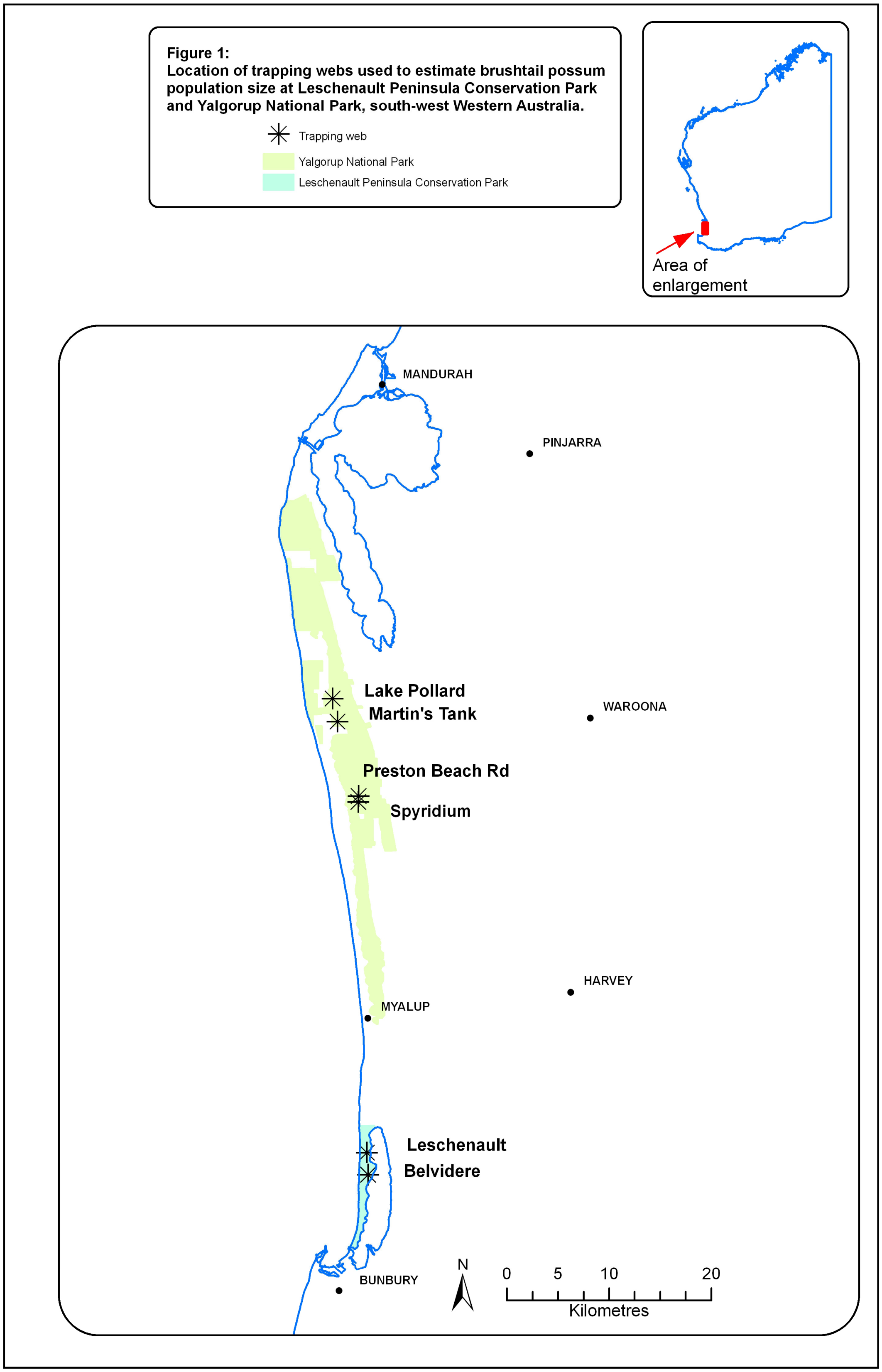


Figure 1:
Location of trapping webs used to estimate brushtail possum population size at Leschenault Peninsula Conservation Park and Yalgorup National Park, south-west Western Australia.

2.1.1 Leschenault Peninsula Conservation Park (LPCP)

Two trapping webs were located within LPCP; Leschenault and Belvidere (Figure 2). Leschenault Peninsula is located 150 km south of Perth and 22 km north of Bunbury at 115° 41′ E, 33°12′-18′ S (CALM, 1998). Leschenault Peninsula stretches for 11 km in a north – south direction separating the Indian Ocean from the Leschenault Inlet and covering 1071 ha (CALM, 1998). The Peninsula comprises a barrier dune system which is mostly composed of Quindalup dunes (also known as Safety Bay sands) and harbours 201 known plant species (CALM, 1998). There are no planned burns for LPCP (de Tores, 2007, pers comm.).

The area experiences a Mediterranean climate of hot, dry summers and mild, wet winters. The mean annual rainfall is 840 – 871 mm which occurs mainly between May and September during frequent winter storms (CALM, 1998). Temperatures are generally cooler than on the mainland due to the sea breeze with mean maximum temperatures ranging from 16.9 ˚C in July to 27.7 ˚C in February and temperatures rarely exceeding 35 ˚C (CALM, 1998).

Leschenault Peninsula is bordered on three sides by water and joins the mainland only by a 600 m wide isthmus. The park is therefore ideal for effective fox control as the isthmus provides the only means of re-invasion (CALM, 1998). Fox control of LPCP using 1080 baits commenced in 1991 and has been maintained at a monthly baiting regime (de Tores *et al.*, 2005; de Tores *et al.*, 2004). Baits are tethered to minimise take by non-target species. Tethering of baits was introduced in 2000 in response to the perceived risk of ravens taking baits and dropping these in areas of high public use (Brazell, 2007, pers comm.).

2.1.1.1 Leschenault trapping web (LES)

The Leschenault trapping web is located to the west of the main north – south public access track (Figure 2). The web is dominated by peppermint with thick understorey in some areas on grey sandy soils. The vegetation structure at the web was considered to be representative of dense peppermint habitat at LPCP. Set-up of the trapping web commenced shortly after a storm had passed through the area and knocked down a significant amount of vegetation. This resulted in a large amount of debris on the ground during both trapping sessions.

2.1.1.2 Belvidere trapping web (BEL)

The Belvidere trapping web (Figure 2) is located to the east of the major north – south running management access track near the Belvidere picnic area. The web is on grey sandy soils with frequent large areas of bare ground. The vegetation is dominated by tuart with peppermint throughout and has an open understorey. The vegetation structure is considered representative of open tuart woodland habitat at LPCP.

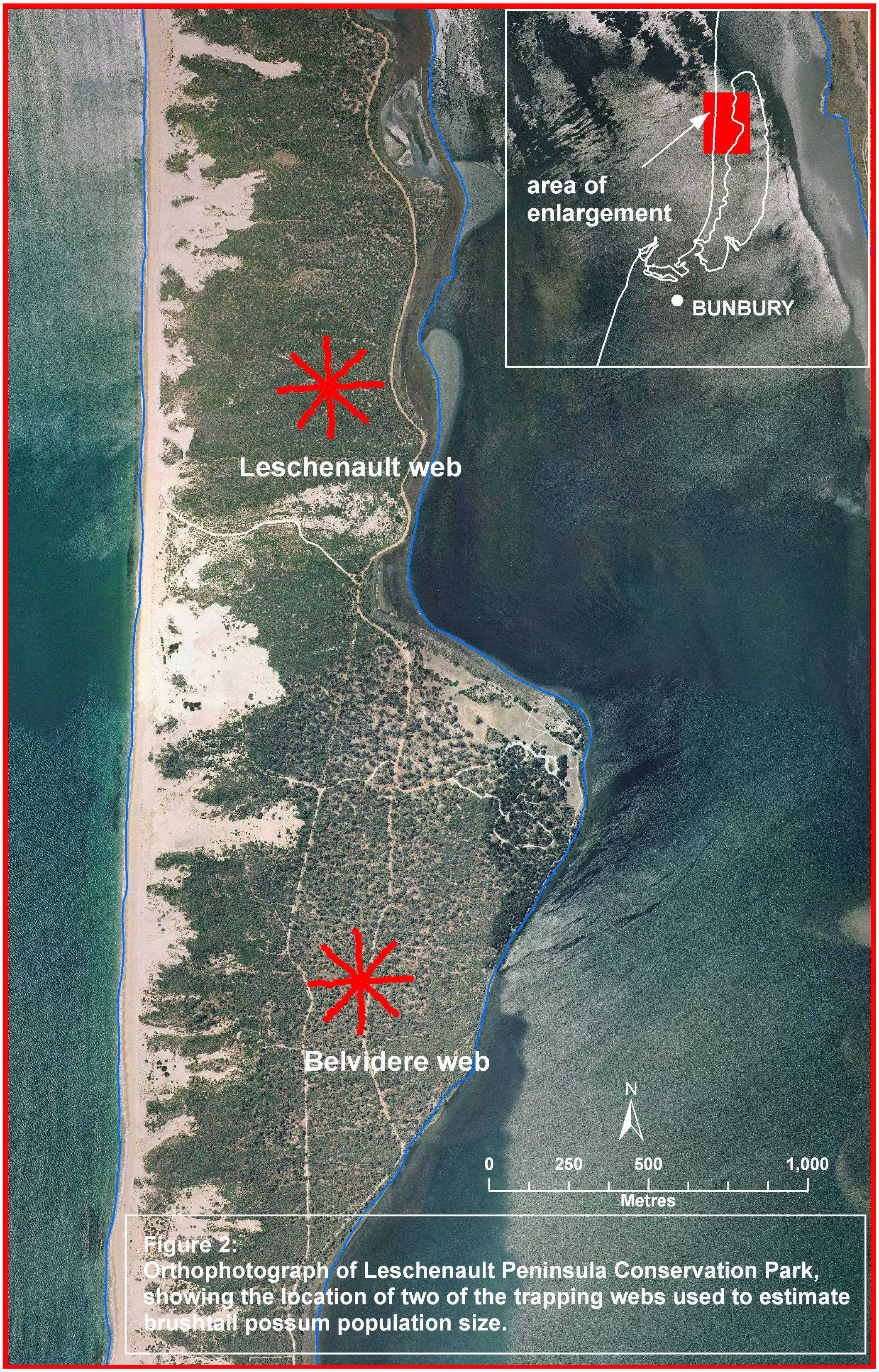


Figure 2:
Orthophotograph of Leschenault Peninsula Conservation Park, showing the location of two of the trapping webs used to estimate brushtail possum population size.

2.1.2 Yalgorup National Park (YNP)

Yalgorup National Park encompasses a chain of elongated lakes, which have been recognised as Wetlands of International Importance under the Ramsar Convention, and the surrounding bushland contained within several disconnected blocks of land (CALM, 1995). The park is located 105 km south of Perth at 32°51′26″S and 115°40′19″E (Anonymous, 2006) and consists of a long, thin stretch of land covering an area of 12,888 ha between the towns of Melros and Myalup (CALM, 1995).

YNP lies on sands and clays on underlying Tertiary and Quaternary limestone, with limestone outcrops occurring commonly. The sands are predominately part of the Spearwood Dune System and are composed of leached sand at the surface with creamy yellow sand below (CALM, 1995).

Due to its long and narrow nature, YNP varies widely in its vegetation structure and floristics. The park contains five broad vegetation complexes; Yoongarillup, Vasse, Cottesloe, Quindalup and Karrakatta (CALM, 1995). Vegetation specific to the trapping webs is described below. Parts of YNP were selected as western ringtail translocation sites due to the presence of peppermint, a primary food source, and tuart, which provides suitable tree hollows (de Tores *et al.*, 2004). There are no planned burns for the duration of the western ringtail possum translocation and monitoring program (de Tores, 2007, pers comm.).

Trapping webs were established at two sites within YNP; Preston Beach Road (Figure 3) and Martin's Tank (Figure 4). The Preston Beach Road site is

573 ha and consists of the area baited monthly for fox control (de Tores *et al.*, 2004). Baits are un-tethered and surface-laid. The site is bound by Lake Preston and areas of similar vegetation which do not undergo baiting. There is potential for re-invasion by foxes from the north, south and east.

The Martin's Tank site is unbaited and has an area of 516 ha. The site boundaries are nominal as the site is surrounded by adjoining unbaited vegetation. The site boundary was delineated by negotiation with DEC to ensure the site was managed as a fire exclusion zone for the duration of the western ringtail possum translocation research (de Tores, 2007, pers comm.).

2.1.2.1 Spyridium trapping web (SPY)

The Spyridium web is located to the east of the maintenance track leading south off Preston Beach Road immediately west of the Information Bay. The soil consists of grey sand with yellow sand a short distance below and with underlying limestone often within 10 – 30 cm of the surface. Small limestone outcrops are common. The web is highly heterogeneous with about 30% covered in stands of peppermint on the western portion of the web. The remainder is dominated by either basket bush (the trapping web's namesake; *Spyridium globulosum*) or melaleuca/parrot bush (*Dryandra sessilis*) heath resulting in dense vegetation throughout the web (CALM, 1995). Occasional tuart or jarrah is scattered throughout the web and grasstrees are present, mostly in patches. Swamp paperbark are also present closer to Lake Preston (i.e. on the western side).

2.1.2.2 Preston Beach Road trapping web (PBR)

The Preston Beach Road trapping web lies to the west of the maintenance track leading south off Preston Beach Road immediately west of the Information Bay. Sandy soils dominate the area. The web is relatively homogenous and consists of the Yoongarillup vegetation complex which is dominated by peppermint with tuart throughout and coastal heath (CALM, 1995). There is a noticeable lack of grasstrees (*Xanthorrhoea sp.*). The understorey is generally open and mostly consists of melaleuca.

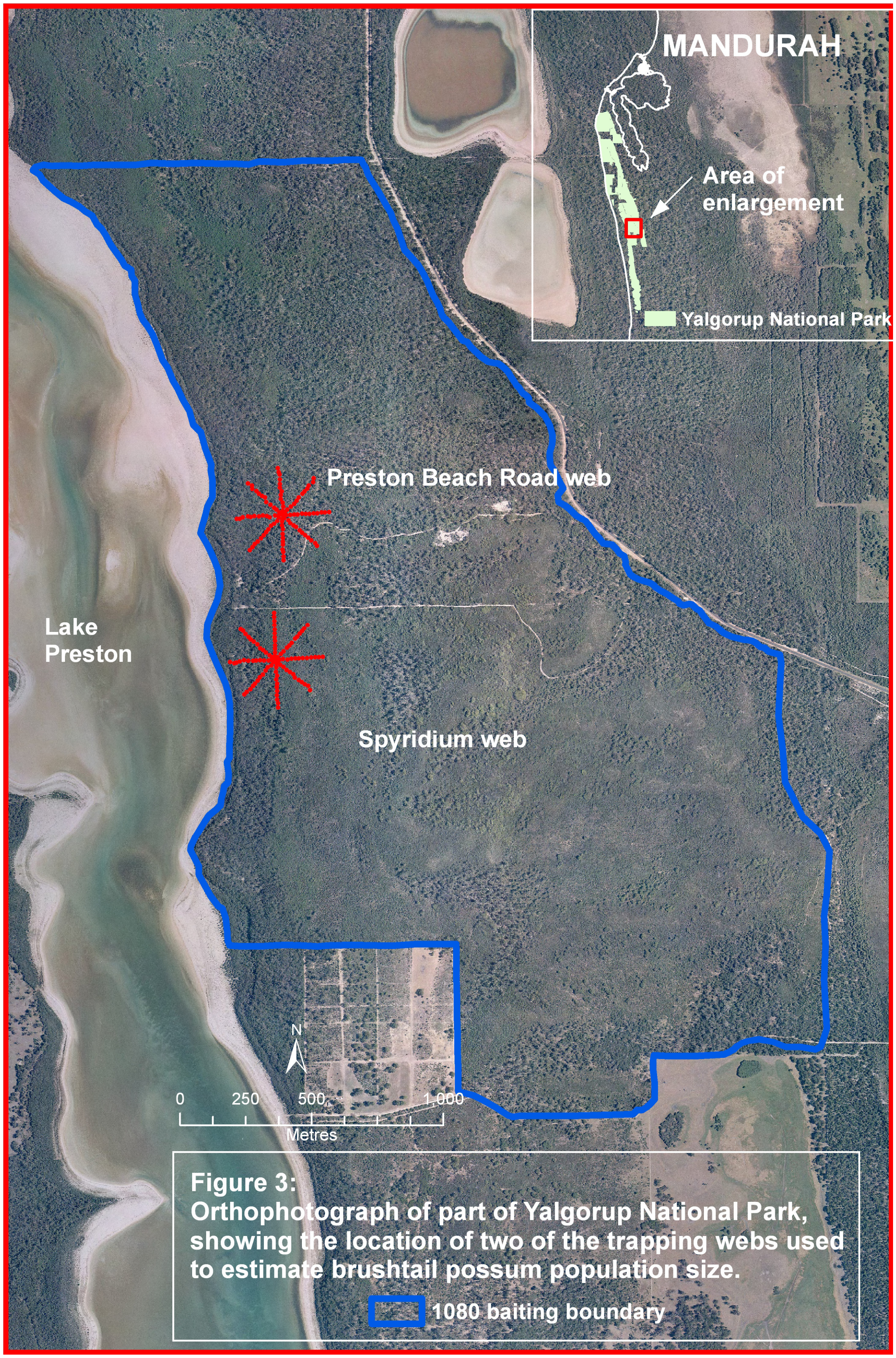


Figure 3:
Orthophotograph of part of Yalgorup National Park, showing the location of two of the trapping webs used to estimate brushtail possum population size.

2.1.2.3 Lake Pollard trapping web (LP)

The trapping web is located south of Lake Pollard crossing over the main access track. The web is dominated by grey sands with calcareous material and seashells occurring throughout, particularly close to the lake. The web comprises the Yoongarillup vegetation complex which is dominated by peppermint with tuart throughout and coastal heath. The second storey includes she-oak (*Allocasuarina fraseriana*) and banksia species (*Banksia grandis*, *B. attenuata*, *B. littoralis*). Understorey species include golden wattle (*Acacia saligna*), prickly moses (*A. pulchella*), green stinkwood (*Jacksonia sternbergiana*), coastal honeymyrtle (*Melaleuca acerosa*), buttercup (*Hibbertia hypercoides*) and basket bush as well as numerous grasstrees (CALM, 1995). The vegetation at Lake Pollard is generally heterogeneous with patches of particular plant species occurring in some areas.

2.1.2.4 Martin's Tank trapping web (MT)

The trapping web is situated between Preston Beach Road North and Martin's Tank campground to the west of Martin's Tank Lake. Sandy soils dominate the web. Martin's Tank comprises the Cottesloe vegetation complex which is typically dominated by mixed eucalypt forest (tuart, *Eucalyptus calophylla*, *E. marginata*) with a tall second storey of *Banksia attenuata*, peppermint and she-oak (CALM, 1995). The web has a generally sparse understorey and unburnt grasstrees with thick 'skirts' occur in patches throughout. The dominant canopy species is tuart, but of very poor condition with many trees dead or dying.

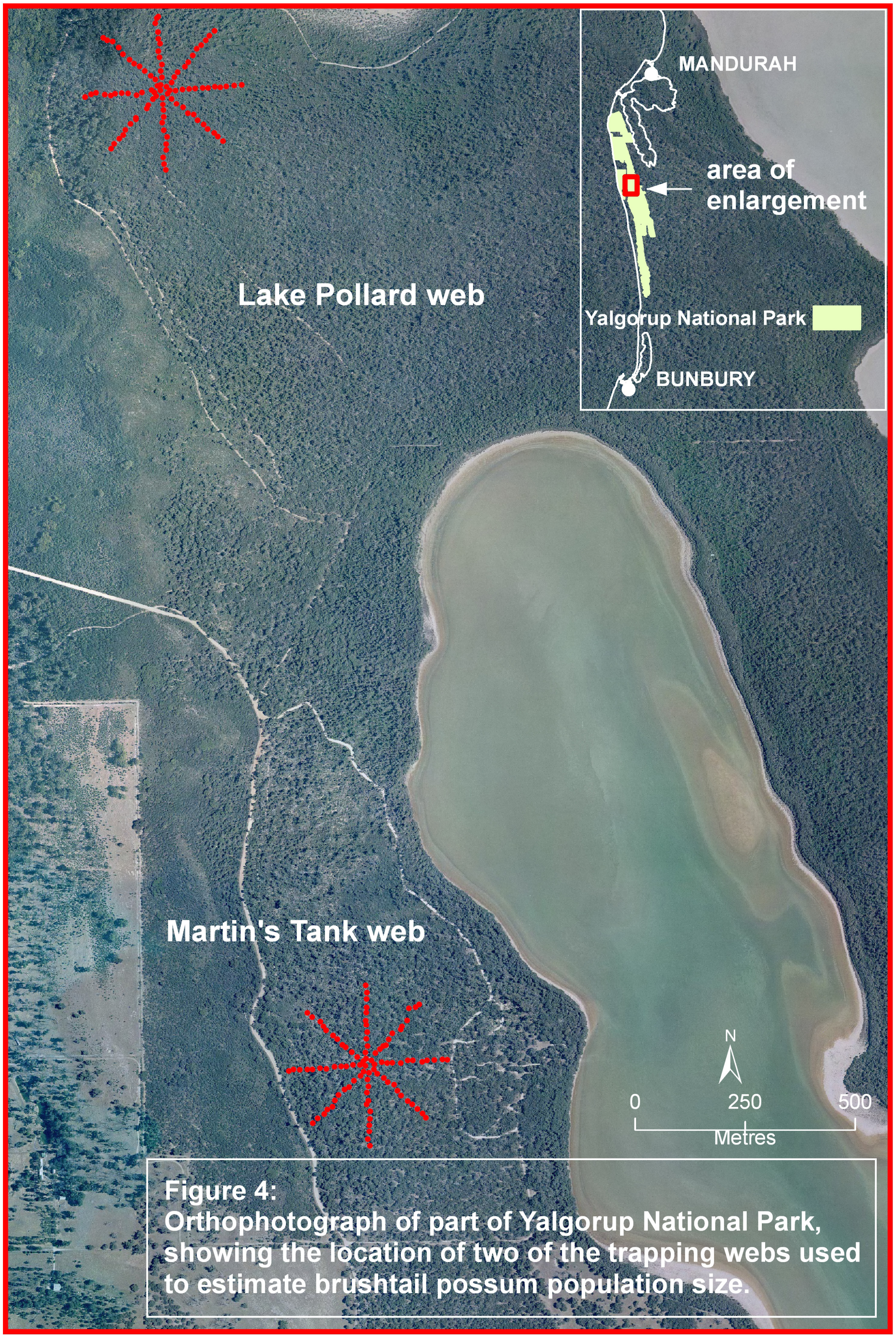


Figure 4:
Orthophotograph of part of Yalgorup National Park, showing the location of two of the trapping webs used to estimate brushtail possum population size.

2.1.3 Description of trapping web locations

The location of each trapping web was chosen to be representative of the main vegetation types occurring at each site; namely, dense peppermint dominated vegetation and the more open tuart/jarrah (*E. marginata*)/banksia woodland. A description of the vegetation structure of each trapping web is summarised in Table 1.

Table 1: Description of each trapping web used to determine the availability of alternative prey species at western ringtail possum translocation sites in Leschenault Peninsula Conservation Park and Yalgorup National Park.

Site	Trapping web	Main vegetation unit	Soil type	1080 baiting regime
Leschenault (LPCP)	Leschenault	Dominated by peppermint with generally thick understorey.	Grey sandy soils.	Tethered 1080 baits delivered monthly.
	Belvidere	Dominated by tuart with peppermint throughout and an open understorey.	Grey sandy soils with areas of bare ground.	Tethered 1080 baits delivered monthly.
Preston Beach Road (YNP)	Spyridium	Heterogeneous. Peppermint stands in 30% of the web. Remainder dominated by basket bush or melaleuca/parrot bush heath. Tuart and jarrah throughout. Grasstrees mostly in patches.	Grey sand with yellow sand below. Underlying limestone with outcrops common.	Un-tethered 1080 baits delivered monthly.
	Preston Beach Road	Dominated by peppermint with tuart throughout. Open understorey mostly consisting of melaleuca. Fairly homogenous. No grasstrees.	Sandy soils dominate.	Un-tethered 1080 baits delivered monthly.
Martin's Tank (YNP)	Lake Pollard	Peppermint dominant. Tuart occurring throughout with coastal heath.	Grey sands, calcareous material throughout.	Not baited.
	Martin's Tank	Mixed eucalypt forest. Second storey of banksia, peppermint and she-oak. Dominant canopy species is tuart of poor quality.	Sandy soils dominate.	Not baited.

2.2 Trapping

2.2.1 Trapping methods

Trapping webs were constructed as described by Lukacs, *et al.* (2005). Each trapping web consisted of a central point with eight 180 m ‘arms’ radiating outwards from this point. The arms were arranged in a north, northeast, east, southeast, south, southwest, west and northwest orientation. The total area of each trapping web was 10.17 ha. Each arm contained 12 trap points 15 m apart as well as one in the centre, forming a set of concentric circles around the centre point. The trap points were marked with jarrah stakes which were sequentially numbered with permanent marker on flagging tape in order to ensure consistent placement of traps. See Figure 5 for web layout.

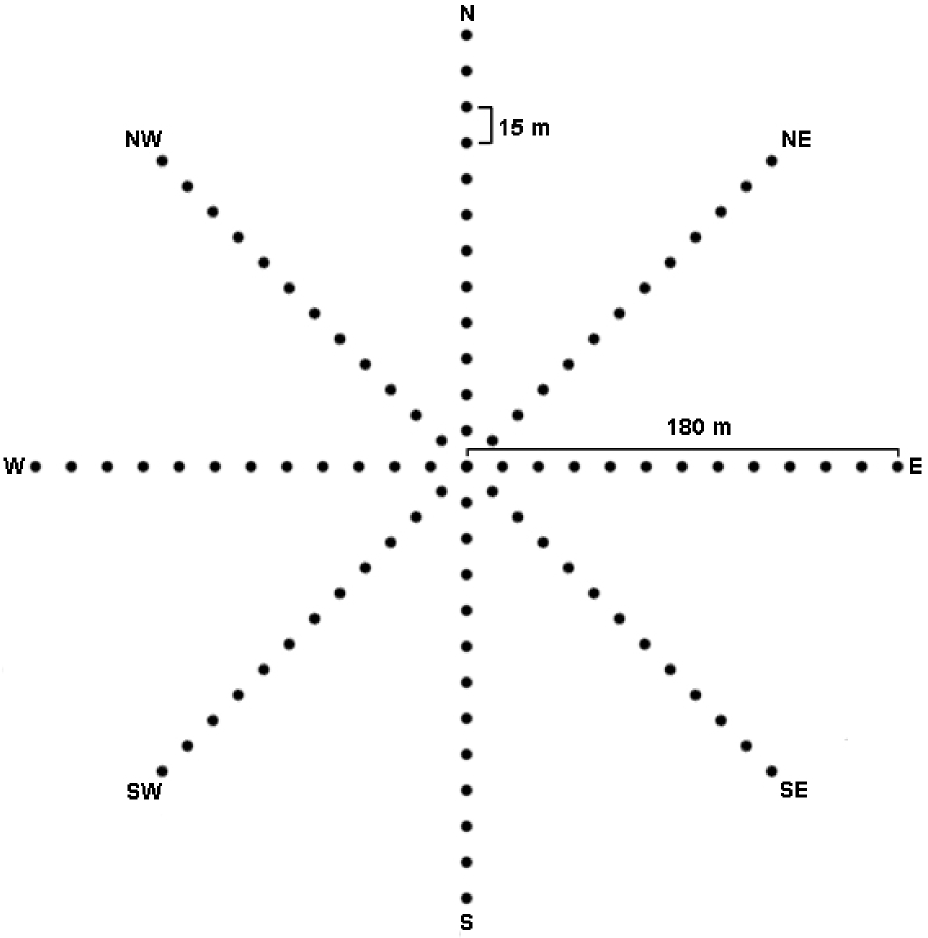


Figure 5: Layout of trapping web design used for each trapping web in Leschenault Peninsula Conservation Park and Yalgorup National Park where each dot represents a trap point.

The focus was on determining the availability of alternative prey to the western ringtail possum. Wire cages and Elliott traps were therefore used these were of appropriate size to target small to medium sized vertebrates. One Sheffield wire cage (570 x 200 x 200 mm) and one Elliott folding aluminium trap (327 x 100 x 90 mm) was placed at each trap point, with a total of 97 traps of each type at each trapping web. Animals were protected from the weather by placing Hessian coverings over the rear two thirds of the cage traps and over entire Elliott traps. Elliott traps were additionally supplied with bedding material (cotton wool or polar fleece cut to size) to protect animals from the weather. Traps were placed in shaded positions and under vegetation where possible to provide shelter from the sun (Figure 6 and Figure 7).

Figure 6: Demonstration of the typical set-up of a Sheffield wire cage three-quarters covered with Hessian. The trap is placed in a shady position under vegetation cover.

Figure 7: Demonstration of the typical set-up of an Elliott aluminium folding trap covered in a piece of Hessian. The trap is placed in a shady position under vegetation cover.

Traps were baited with a mixture of peanut butter, rolled oats, honey and jellied sardines (tinned cat food). The bait was replaced as needed on a daily basis. Trapping was carried out according to DEC Standard Operating Procedures (CALM, 2005). The traps were checked and cleared as early as practicable each morning and animals were released at the trap location immediately following processing. However, some brushtail possums were held for radio-collaring and were released in the evening of the day of capture. Radio-collaring was carried out to enable common brushtail possums to be included in the concurrent western ringtail possum translocation monitoring program which incorporates monitoring of *in situ* populations of brushtail possums.

Captured animals were transferred into appropriately sized Hessian or calico bags. They were weighed inside the bags to obtain a gross weight from which a bag weight was later subtracted. Mammals were then sexed and their reproductive condition was assessed by examining the pouch (in marsupials) or testes. Detailed measurements were taken for mammals at the beginning of the first trapping session. These included head-body, tail, right pes, and head lengths. However, trapping protocol was reassessed during the second week of spring trapping due to observed stress in the captured mammals. The trapping protocol was adjusted to include only weight, sex and reproductive condition data in order to reduce handling time and minimise stress to the animal. Sex was not identified in reptiles. Snout – vent and snout – tail measurements were taken for all reptiles excluding snakes. Once the appropriate measurements were taken the animal was marked (if possible) before being released at the point of capture. See Table 2 for marking techniques.

Table 2: Marking techniques employed for each species trapped at Leschenault Peninsula Conservation Park and Yalgorup National Park.

Species	Individual identification	External marking
Common brushtail possums	Trovan® passive implant transponder (PIT). Trovan® passive implant transponders measured 11 mm in length and weighed 0.1 g (Figure 8 and Figure 9).	Left ear in males or right ear in females marked with ear tags with reflective tape covered in transparent heat shrink, allowing greater visibility in future spotlighting transects conducted by DEC.
Chuditch	Trovan® passive implant transponder (PIT).	Left ear in males or right ear in females marked with non-reflective ear tags.
Quendas	Trovan® passive implant transponder (PIT).	Left ear in males or right ear in females marked with non-reflective ear tags.
House mice	Trovan® passive implant transponder (PIT).	Left ear in males or right ear in females marked using 1mm ear punch.
Varanids	Trovan® passive implant transponder (PIT) (N/A for juveniles which were too small).	Small dab of paint between shoulders.
All skinks	Toe-clipped using the 1, 2, 4, 7 numbering system (Figure 10).	Toe-clipped.
Snakes	Not marked	N/A
Frogs	Not marked	N/A
Birds	Not marked	N/A

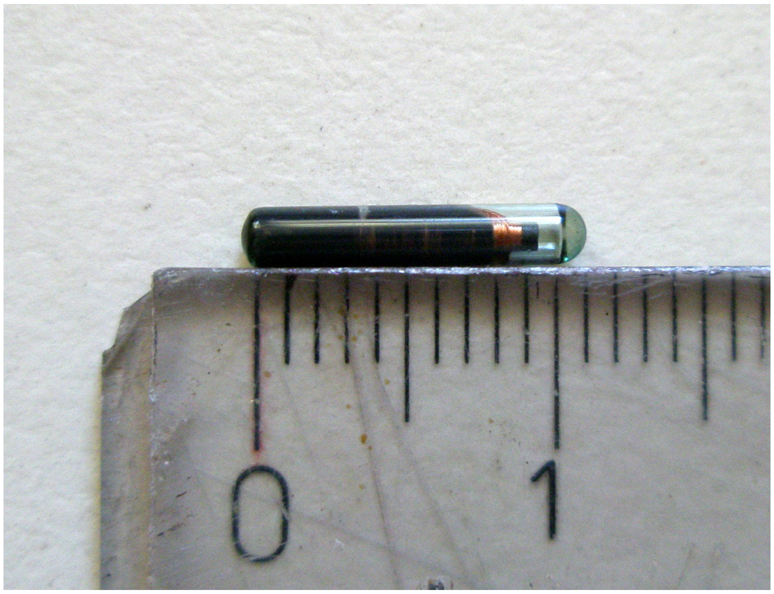


Figure 8: Trovan® passive implant transponder (PIT) used to individually mark mammals and adult varanids trapped at Leschenault Peninsula Conservation Park and Yalgorup National Park.

Figure 9: Trovan® passive implant transponder (PIT) and applicator needle used to individually mark mammals and adult varanids trapped at Leschenault Peninsula Conservation Park and Yalgorup National Park. The number is the unique code of the PIT.

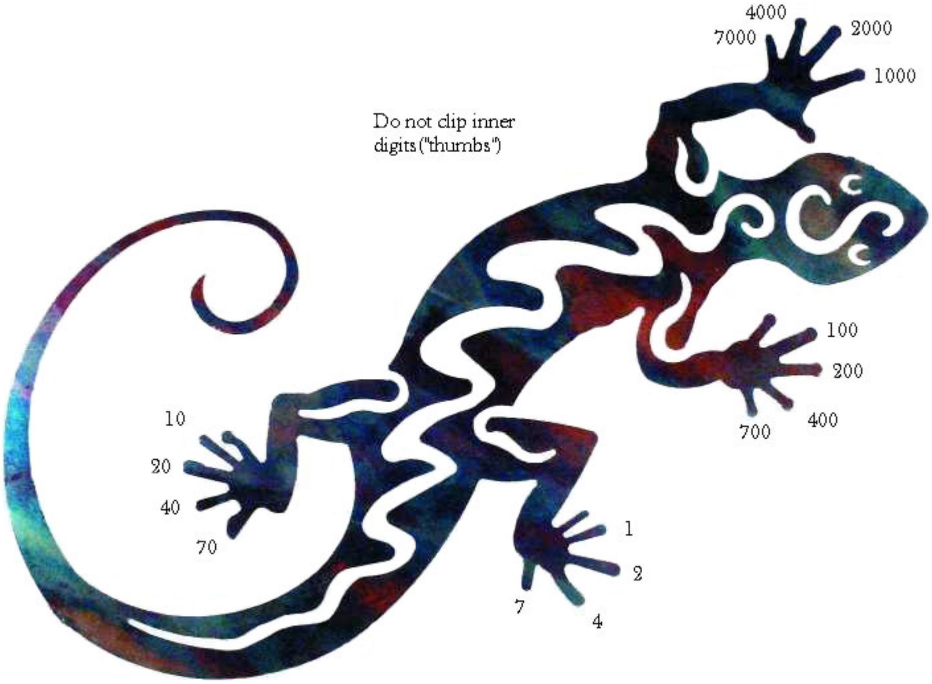


Figure 10: Toe-clipping method following the 1, 2, 4, 7 numbering system used to mark reptiles at Leschenault Peninsula Conservation Park and Yalgorup National Park.

2.2.2 Timing of trapping

Trapping was conducted in November/December 2006 and in February/March 2007 to provide an indication of seasonal variability in prey availability. The trapping sessions were considered representative of spring and summer respectively. Each trapping session consisted of five consecutive trap nights at each web. A list of specific dates during which trapping occurred at each of the trapping webs is provided in Table 3.

Table 3: Dates of trapping carried out at trapping webs at Leschenault Peninsula Conservation Park and Yalgorup National Park.

Trapping web	**Spring ‘06**	**Summer ‘07**
Leschenault	17/12/06 – 21/12/06	05/03/07 – 09/03/07
Belvidere	23/11/06 – 27/11/06	12/03/07 – 16/03/07
Spyridium	29/11/06 – 03/12/06	19/03/07 – 23/03/07
Preston Beach Road	30/11/06 – 04/12/06	26/03/07 – 30/03/07
Martin’s Tank	07/12/06 – 11/12/06	19/02/07 – 24/02/07
Lake Pollard	08/12/06 – 12/12/06	26/02/07 – 02/03/07

2.3 Data Analysis

2.3.1 Trap success

As mentioned previously (Chapter 1), there are two methods of calculating trapping success. One involves only counting new individuals for each trapping session. The other method involves all captures including initial captures of individuals and all subsequent recaptures.

Heterogeneity such as trap shyness or trap happiness may bias capture success rates which include recaptures. This may be expected particularly for brushtail possums as these have been observed to be trap happy in previous studies (Efford, 2004; Wayne *et al.*, 2005a). For this reason, trap success was calculated based on individual captures only (i.e. recaptures were excluded) to provide a more reliable index of population size.

Trap success was obtained for each species by measuring the number of individual animals caught per 100 trap nights per session. The number of trap nights for all species other than bobtail lizards (*Tiliqua rugosa*) was 485 (97 traps x 5 nights) as most species were only likely to be caught in one type of trap, either Elliott or cage.

Bobtails could potentially be trapped in both cages and Elliotts thereby doubling the number of potential traps. Hence, the number of trap nights used for calculation of capture success for this species was 970. However, this is only the maximum number of trap nights as some individuals may have been too large to fit into Elliott traps. It is impossible to determine the effective number of trap nights for bobtails without knowing what proportion of the population was too large enter both trap types. Caution must therefore be taken when comparing capture success for bobtail lizards to that of other species.

2.4 Population size and program MARK

Population estimates (*N*-hat) were derived using Huggins Closed Captures models in program MARK (see section 1.2; White, 2001). Analysis using MARK (White, 2001) was limited to the data gathered on common brushtail possums during the summer trapping session. Data was too sparse to run analyses for spring data for brushtail possums or to analyse data for other species. Doing so would have resulted in non-sense population estimates with unrealistically large standard errors. Therefore, the following methods pertain to the summer session for brushtail possums only.

A candidate set of *a priori* models was analysed to determine the preferred model, i.e. the model closest to the truth (as described previously – section 1.1). The models estimated the parameters *p* and *c*, where *p* is the probability of detecting (encountering) an individual for the first time given that it is alive and in the sample, and *c* is the probability of recapture conditional on having been captured at least once before.

2.4.1 Hypotheses

The primary differences between the six trapping webs were based on 1080 fox baiting regime and on the dominant vegetation structure. The locations of the trapping webs were chosen to be representative of these differences. Based on these two variables alone it becomes apparent that the six trapping webs may essentially be grouped into four broad categories as presented in Table 4.

Table 4: Trapping web groupings according to 1080 fox baiting regime and vegetation structure.

Baiting regime	Vegetation structure	Trapping webs (group number in parentheses)
Baited	Dense	Leschenault (1) Spyridium (3)
	Open	Belvidere (2) Preston Beach Road (4)
Unbaited	Dense	Lake Pollard (5)
	Open	Martin's Tank (6)

Assuming 1080 baiting is effective in reducing fox numbers and therefore minimising the effect of predators on prey species, common brushtail possum population estimates should to be highest in areas where baiting occurs. In areas that are not baited, brushtail possum population size could reasonably be expected to be highest in areas of dense vegetation, as this would provide refuge to avoid predation. Consistent with this, it is hypothesised that where baiting occurs the importance of vegetation cover and predator refuges is not as great. Possums are known to spend a large amount of time foraging on the ground and appear to prefer open areas (Cowan & Clout, 2000; Jones & Hillcox, 1995; Kerle, 1984). Brushtail possums are also known to select old tree hollows as rest sites, particularly tuart trees (How & Hillcox, 2000; Jones & Hillcox, 1995), which dominate the tree canopy at the ‘open’ trapping webs within the study sites. Therefore, it is hypothesised the highest estimate of population size will be within the preferred open vegetation at baited sites, where predator pressures are assumed to be minimal.

A number of variables may also influence the probability of capture (both initial and recapture). These are listed in Table 5.

Table 5: Variables expected to influence the parameters *p* and *c*, where *p* is the probability of initial capture and *c* is the probability of recapture, in analyses of trapping data for common brushtail possums in summer at Leschenault Peninsula Conservation Park and Yalgorup National Park. *Note*: Unless stated otherwise the variable is expected to influence both *p* and *c*.

Variable	**Effect**	**Influence on *p* and/or *c***
Time	Probabilities change over time (day 1-5).	- Trap wariness – may take a few days of exposure to a trap for individual to enter it (only applies to *p*).
Sex	Mobility differences between sexes - males have larger home ranges.	- Males travel further so more likely to encounter trap. - Males less likely to re-encounter trap due to higher mobility (only applies to *c*). - Females more cautious if with young so less likely to enter trap.
Vegetation	Open vs. dense vegetation. I.e. plant diversity, % groundcover, canopy connectivity, shelter, predator refuge, food resource, etc.	- If high canopy connectivity, may spend less time on ground. - Dense vegetation will allow individuals greater security (more shelter/predator refuges) allowing greater mobility.
Baiting	Baited vs. unbaited. May: reduce predator numbers; or not affect amount of predators but alter species of predators (mesopredator release).	- Where there are more predators, individuals may limit mobility - reduce foraging range, prefer dense vegetation, come down to ground less often - less likely to encounter trap.
Baiting + Vegetation	See above.	- In areas where vegetation is dense, providing predator refuges and high canopy connectivity (less time spent on ground), can expect effect of baiting to be insignificant (i.e. same *p* and *c* for webs with dense and open vegetation in baited areas.

2.4.2 Models tested

The above factors were incorporated into a candidate set of basic *a priori* models (Table 6). For the purposes of MARK (White, 2001), data was categorised into six groups which were equivalent to the six trapping webs as follows:

1. Leschenault
2. Belvidere
3. Spyridium
4. Preston Beach Road
5. Lake Pollard
6. Martin's Tank

In order to enable modeling the importance of the main variables tested (namely baiting regime, vegetation structure and sex), three individual covariates were specified as follows:

- Baiting regime:

  Baited = 1, Unbaited = 0

- Vegetation:

  Dense = 1, Open = 0

- Sex:

  Male = 1, Female = 0

These groups and covariates were incorporated into a set of candidate models (Table 6) which were compared using the Information-Theoretic (I-T) approach in program MARK (White, 2001). The models were formulated based on hypotheses arising from the previously mentioned (Table 5) variables that could affect capture probabilities of brushtail possums. Additional models were created by constraining the probability of recapture to be constant. The global model is also included in MARK analysis; however, it is not considered a biologically likely model and is used primarily to construct the remaining models.

Table 6: Candidate set of *a priori* models pertaining to trapping data for common brushtail possums in summer at Leschenault Peninsula Conservation Park and Yalgorup National Park. The symbol * indicates an interaction effect.

Model syntax	**Model description**	**Model hypothesis**
p(g)c(g)	The probabilities of capture and recapture are different for each group (i.e. different for each trapping web).	The estimate of population size (*N*-hat) is best modeled by the unique characteristics of each web – with each web having different probabilities for capture and recapture.
p(g)c(.)	The probability of initial capture is different for each group (i.e. different for each trapping web) but the probability of recapture remains constant.	The estimate of population size (*N*-hat) is best modeled by the unique characteristics of each web – with each web having different a probability of initial capture. The effect of capture heterogeneity is strong enough that probability of recapture is constant.
p(s)c(s)	The probabilities of capture and recapture are different for males and females.	The estimate of population size (*N*-hat) is best modeled by the effect of sex – with males and females having different probabilities for capture and recapture.
p(s)c(.)	The probability of initial capture is different for males and females but the probability of recapture remains constant.	The estimate of population size (*N*-hat) is best modeled by the effect of sex – with males and females having a different probability of initial capture. The effect of capture heterogeneity is strong enough that probability of recapture is constant.

Model syntax **(cont.)**	**Model description** **(cont.)**	**Model hypothesis** **(cont.)**
p(g+s)c(g+s)	The probabilities of capture and recapture are different for each group (i.e. different for each trapping web) and are different for males and females.	The estimate of population size (*N*-hat) is best modeled by both sex and the unique characteristics of each web – with each web and either sex within each web having different probabilities for capture and recapture.
p(g+s)c(.)	The probability of initial capture is different for each group (i.e. different for each trapping web) and is different for males and females but the probability of recapture remains constant.	The estimate of population size (*N*-hat) is best modeled by both sex and the unique characteristics of each web – with each web and either sex within each web having a different probability of initial capture. The effect of capture heterogeneity is strong enough that probability of recapture is constant.
p(g)c(g) g1=g3, g2=g4, g5=g6	The probabilities of capture and recapture are the same for group 1 and group 3 (Leschenault and Spyridium webs); the same for group 2 and group 4 (Belvidere and Preston Beach Road webs); and the same for group 5 and group 6 (Lake Pollard and Martin's Tank webs).	The estimate of population size (*N*-hat) is best modeled by the combination of vegetation and baiting. The webs which are baited and have dense vegetation (groups 1 and 3) will have different capture and recapture probabilities from webs which are baited and have open vegetation (groups 2 and 4). The unbaited webs (groups 5 and 6) will have different capture and recapture probabilities from the groupings above, but will have comparable capture and recapture probabilities to each other.

Model syntax **(cont.)**	**Model description** **(cont.)**	**Model hypothesis** **(cont.)**
p(g)c(.) g1=g3, g2=g4, g5=g6	The probability of initial capture is the same for group 1 and group 3 (Leschenault and Spyridium webs); the same for group 2 and group 4 (Belvidere and Preston Beach Road webs); and the same for group 5 and group 6 (Lake Pollard and Martin's Tank webs), but the probability of recapture remains constant.	The estimate of population size (*N*-hat) is best modeled by the combination of vegetation and baiting. The webs which are baited and have dense vegetation (groups 1 and 3) will have different initial capture probabilities from webs which are baited and have open vegetation (groups 2 and 4). The unbaited webs (groups 5 and 6) will have different initial capture probabilities from the groupings above, but will have comparable initial capture probabilities to each other. The effect of capture heterogeneity is strong enough that probability of recapture is constant.
p(g+s)c(g+s) g1=g3, g2=g4, g5=g6	The probabilities of capture and recapture are different for males and females and the same for group 1 and group 3 (Leschenault and Spyridium webs); the same for group 2 and group 4 (Belvidere and Preston Beach Road webs); and the same for group 5 and group 6 (Lake Pollard and Martin's Tank webs).	The estimate of population size (*N*-hat) is best modeled by the combination of vegetation and baiting as well by sex. The webs which are baited and have dense vegetation (groups 1 and 3) will have different capture and recapture probabilities from webs which are baited and have open vegetation (groups 2 and 4). The unbaited webs (groups 5 and 6) will have different capture and recapture probabilities from the groupings above, but will have comparable capture and recapture probabilities to each other. The capture and recapture probabilities will be different between males and females for each of the above groupings.

Model syntax (cont.)	**Model description (cont.)**	**Model hypothesis (cont.)**
p(g+s)c(.) g1=g3, g2=g4, g5=g6	The probability of initial capture is different for males and females and the same for group 1 and group 3 (Leschenault and Spyridium webs); the same for group 2 and group 4 (Belvidere and Preston Beach Road webs); and the same for group 5 and group 6 (Lake Pollard and Martin's Tank webs), but the probability of recapture remains constant.	The estimate of population size (*N*-hat) is best modeled by the combination of vegetation and baiting as well by sex. The webs which are baited and have dense vegetation (groups 1 and 3) will have different initial capture probabilities from webs which are baited and have open vegetation (groups 2 and 4). The unbaited webs (groups 5 and 6) will have different initial capture probabilities from the groupings above, but will have comparable initial capture probabilities to each other. The initial capture probabilities will be different between males and females for each of the above groupings. The effect of capture heterogeneity is strong enough that probability of recapture is constant.
p(v)c(v)	The probabilities of capture and recapture are different for dense vegetation and sparse vegetation.	The estimate of population size (*N*-hat) is best modeled by the effect of vegetation – with dense and open vegetation having different probabilities for capture and recapture.
p(v)c(.)	The probability of initial capture is different for dense vegetation and sparse vegetation, but the probability of recapture remains constant.	The estimate of population size (*N*-hat) is best modeled by the effect of vegetation – with dense and open vegetation having different probabilities for initial capture. The effect of capture heterogeneity is strong enough that probability of recapture is constant.
p(v+s)c(v+s)	The probabilities of capture and recapture are different for dense vegetation and sparse vegetation and are different for males and females.	The estimate of population size (*N*-hat) is best modeled by the effects of both sex and vegetation – with dense and open vegetation having different capture and recapture probabilities and males and females having different probabilities for capture and recapture within either vegetation type.

Model syntax **(cont.)**	**Model description** **(cont.)**	**Model hypothesis** **(cont.)**
p(v+s)c(.)	The probability of initial capture is different for dense vegetation and sparse vegetation and is different for males and females, but the probability of recapture remains constant.	The estimate of population size (*N*-hat) is best modeled by the effects of both vegetation and sex – with dense and open vegetation having different probabilities for initial capture and males and females having different probabilities for initial capture within either vegetation type. The effect of capture heterogeneity is strong enough that probability of recapture is constant.
p(b)c(b)	The probabilities of capture and recapture are different for baited and unbaited webs.	The estimate of population size (*N*-hat) is best modeled by the effect of baiting – with baited and unbaited webs having different probabilities for capture and recapture.
p(b)c(.)	The probability of initial capture is different for baited and unbaited webs, but the probability of recapture remains constant.	The estimate of population size (*N*-hat) is best modeled by the effect of baiting – with baited and unbaited webs having different probabilities for initial capture. The effect of capture heterogeneity is strong enough that probability of recapture is constant.
p(b+s)c(b+s)	The probabilities of capture and recapture are different for baited and unbaited webs and different for males and females.	The estimate of population size (*N*-hat) is best modeled by the effects of both baiting and sex – with baited and unbaited webs having different probabilities for capture and recapture and males and females having different probabilities for capture and recapture within either baiting regime.

Model syntax **(cont.)**	**Model description** **(cont.)**	**Model hypothesis** **(cont.)**
p(b+s)c(.)	The probability of initial capture is different for baited and unbaited webs and different for males and females, but the probability of recapture remains constant.	The estimate of population size (*N*-hat) is best modeled by the effects of both baiting and sex – with baited and unbaited webs having different initial capture probabilities and males and females having different initial capture probabilities within either baiting regime. The effect of capture heterogeneity is strong enough that probability of recapture is constant.
p(v*b)c(v*b)	There is an interactive effect of vegetation and baiting for the probabilities of capture and recapture. I.e. *p* and *c* are the same for group 1 and group 3 (Leschenault and Spyridium webs); the same for group 2 and group 4 (Belvidere and Preston Beach Road webs); but different for group 5 and group 6 (Lake Pollard and Martin's Tank webs).	The estimate of population size (*N*-hat) is best modeled by the combination of vegetation and baiting. The webs which are baited and have dense vegetation (groups 1 and 3) will have different capture and recapture probabilities from webs which are baited and have open vegetation (groups 2 and 4) and from webs which are unbaited and have dense vegetation (groups 5 and 6).
p(v*b)c(.)	There is an interactive effect of vegetation and baiting for initial capture probability. I.e. *p* is the same for group 1 and group 3 (Leschenault and Spyridium webs); the same for group 2 and group 4 (Belvidere and Preston Beach Road webs); but different for group 5 and group 6 (Lake Pollard and Martin's Tank webs). The probability of recapture remains constant.	The estimate of population size (*N*-hat) is best modeled by the combination of vegetation and baiting. The webs which are baited and have dense vegetation (groups 1 and 3) will have different initial capture probabilities from webs which are baited and have open vegetation (groups 2 and 4) and from webs which are unbaited and have dense vegetation (groups 5 and 6). The effect of capture heterogeneity is strong enough that probability of recapture is constant.

Model syntax **(cont.)**	**Model description** **(cont.)**	**Model hypothesis** **(cont.)**
p(v*b+s)c(v*b+s)	The probabilities of capture and recapture are different between sexes and there is an interactive effect of vegetation and baiting. I.e. *p* and *c* are the same for group 1 and group 3 (Leschenault and Spyridium webs); the same for group 2 and group 4 (Belvidere and Preston Beach Road webs); but different for group 5 and group 6 (Lake Pollard and Martin's Tank webs).	The estimate of population size (*N*-hat) is best modeled by the combination of vegetation and baiting as well as the effect of sex. The webs which are baited and have dense vegetation (groups 1 and 3) will have different capture and recapture probabilities from webs which are baited and have open vegetation (groups 2 and 4) and from webs which are unbaited and have dense vegetation (groups 5 and 6). The capture and recapture probabilities of males and females will be different from each other within each of the above groupings.
p(v*b+s)c(.)	The probability of initial capture is different between sexes and there is an interactive effect of vegetation and baiting. I.e. *p* is the same for group 1 and group 3 (Leschenault and Spyridium webs); the same for group 2 and group 4 (Belvidere and Preston Beach Road webs); but different for group 5 and group 6 (Lake Pollard and Martin's Tank webs). The probability of recapture remains constant.	The estimate of population size (*N*-hat) is best modeled by the combination of vegetation and baiting as well as the effect of sex. The webs which are baited and have dense vegetation (groups 1 and 3) will have different initial capture probabilities from webs which are baited and have open vegetation (groups 2 and 4) and from webs which are unbaited and have dense vegetation (groups 5 and 6). The initial capture probabilities of males and females will be different from each other within each of the above groupings. The effect of capture heterogeneity is strong enough that probability of recapture is constant.

The model which corresponds to the hypothesis of four groups described earlier (Section 2.4.1) corresponds to the (v*b) models which describe an interaction between vegetation and baiting. Modeling using vegetation and baiting covariates is equivalent to coding them by groups where group 1 = group 3, group 2 = group 4 and groups 5 and 6 are different. Both of these methods of coding result in the same AIC_c values and are representative of the same model.

Weight and time may also affect capture rates of brushtail possums. These variables were explored but it was decided not to include these in the analysis. The effect of time was expected to be most significant between trapping sessions. As only the summer session contained sufficient data for analysis using MARK (White, 2001), clearly this effect could not be modeled. Similarly, the effect of weight was expected to be greatest between trapping session. Variation in weight between individuals within one trapping session was mostly related to dimorphism of the sexes. In this case differences in capture probability were expected to be reflected in the models containing sex as a parameter.

2.4.3 Encounter histories

In order to analyse trapping data in program MARK (White, 2001), the first step was to create an input file using Microsoft Office Excel. The encounter history for each individual brushtail possum captured throughout the summer period was created by assigning a coding of "1" for captured and "0" for not captured for each of the five days within the trapping session. Groups were also coded for with a series of six numbers in a similar fashion where “1” represented the corresponding group (e.g. 1 0 0 0 0 0 = group 1; 0 1 0 0 0 0 = group 2; and so forth). Individual covariates were also coded for with “1” representing males, dense vegetation and baited sites. This resulted in a sequence of 1’s and 0’s which were then concatenated to a single entry for each individual. This list was then copied into a text file, the extension of which was changed to “.inp”. This transformed the file to a suitable format for use by MARK (White, 2001). An excerpt of the input file is provided below:

```
/* Trichosurus vulpecula summer trapping */

/* 6 groups (webs), 1 Leschenault, 2 Belvidere, 3 Spyridium,
4 Preston Beach Rd, 5 Lake Pollard, 6 Martin’s Tank */

/* 3 individual covariates, bait (baited=1 unbaited=0) veg
(dense=1 open=0) sex (male=1 female=0)*/

/* LAKE Tri vul F 001*/   11111   0 0 0 0 1 0     0 1 0 ;

/* LAKE Tri vul F 002*/   11011   0 0 0 0 1 0     0 1 0 ;

/* LAKE Tri vul M 009*/   00001   0 0 0 0 1 0     0 1 1 ;

/* MART Tri vul F 001*/   11111   0 0 0 0 0 1     0 0 0 ;
```

Note that each encounter history is delineated by a semi-colon. Also, any text entered between the forward slashes and asterisks (/* text */) is considered a comment by program MARK (White, 2001). Therefore, the individual names of the animals captured are not recognised by the program and these are included for the user’s reference only. Similarly, program MARK (White, 2001) does not recognise the group labels and covariate names entered as a comment at the start of the input file. To allow recognition of groups and covariates by MARK, these were entered separately in the program interface when creating a new file.

2.4.4 Design matrix

The design matrix is essentially a tool used for constructing models in program MARK (White, 2001). It is a matrix showing the structure of the dummy coding variables in the analysis (Cooch, 2001). The basic structure of a design matrix consists of a series of columns and rows. The columns represent the parameters being individually estimated (Cooch, 2001). The rows correspond to the number of levels of the main effect (Cooch, 2001), which in this case is equivalent to the capture occasions for each group (i.e. days 1-5 for group 1, days 1-5 for group 2, etc). Note that while the probability of initial capture is modeled using five capture occasions (corresponding to the number of trap nights) only 4 occasions are used when modeling the probability of recapture as it is impossible for an animal to be recaptured on the first day.

Program MARK (White, 2001) automatically designs the fully time-dependent or ‘global’ model p(g*t)c(g*t) upon creation of a new file. This model serves as a basis for creating all other models. The design matrix may be modified in various ways (e.g. by removing parameters, adding individual covariates, or by constraining variables) to examine the relative fit of the candidate models (Cooch, 2001). The global model for the brushtail possum summer data is shown in Figure 11 with Figure 12 displaying a close-up of the portion of the design matrix corresponding to probability of initial capture.

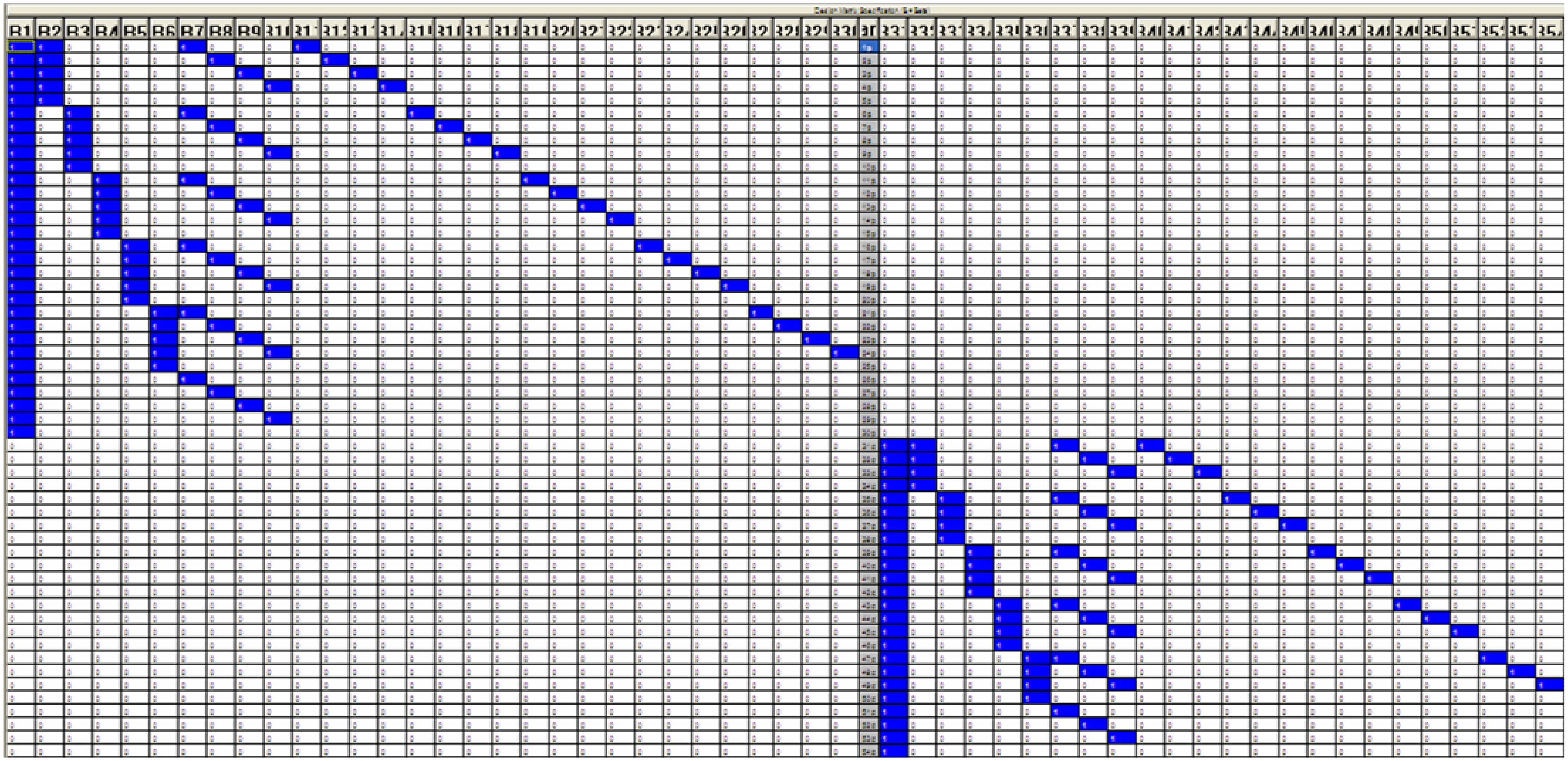
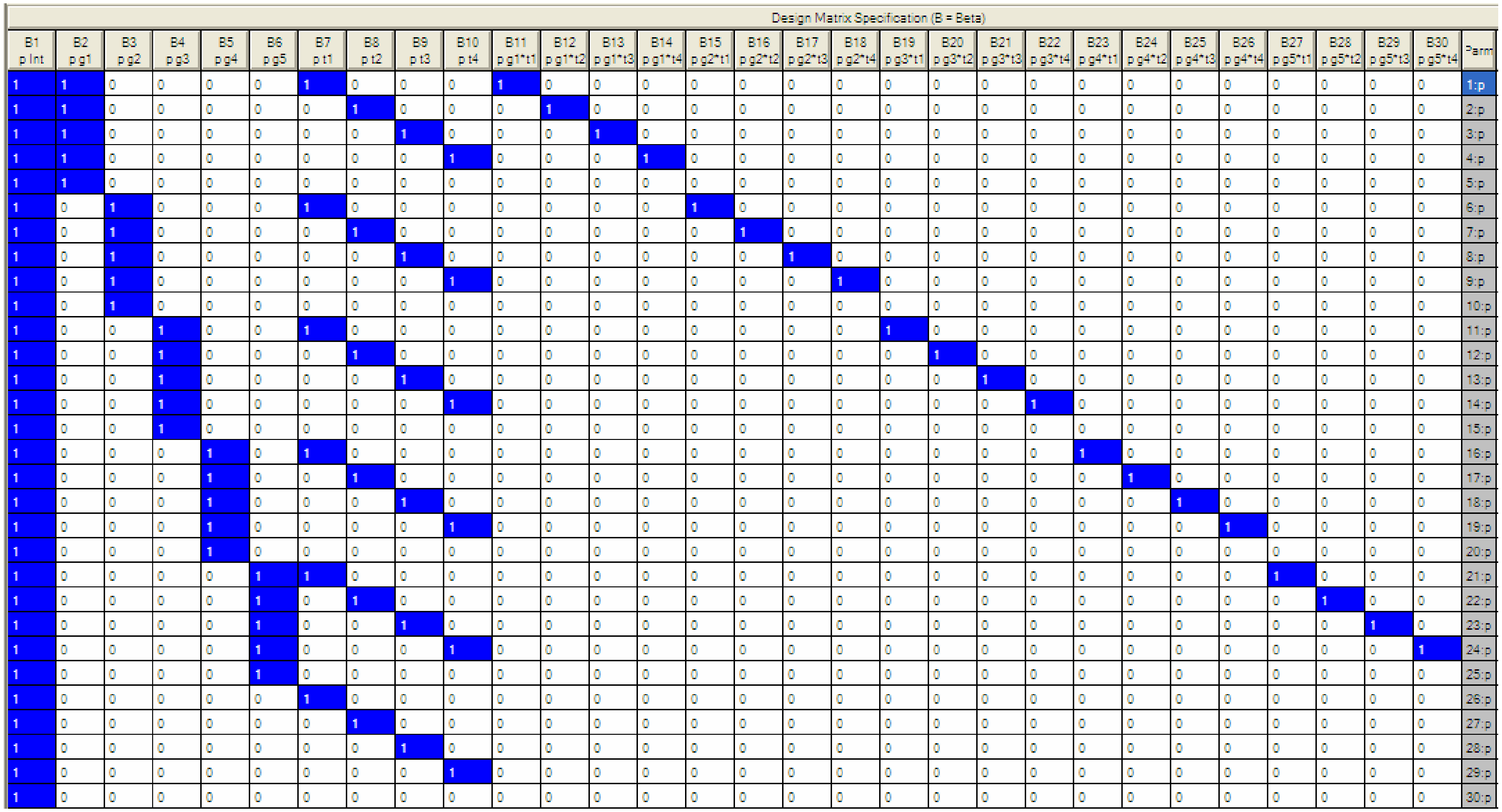


Figure 11: Design matrix in program MARK illustrating the coding used for the global model p(g*t)c(g*t) for common brushtail possums trapped in summer 2007 at Leschenault Peninsula Conservation Park and Yalgorup National Park.



Design Matrix Specification (B = Beta)

B1 p Int	B2 p g1	B3 p g2	B4 p g3	B5 p g4	B6 p g5	B7 p t1	B8 p t2	B9 p t3	B10 p t4	B11 p g1*t1	B12 p g1*t2	B13 p g1*t3	B14 p g1*t4	B15 p g2*t1	B16 p g2*t2	B17 p g2*t3	B18 p g2*t4	B19 p g3*t1	B20 p g3*t2	B21 p g3*t3	B22 p g3*t4	B23 p g4*t1	B24 p g4*t2	B25 p g4*t3	B26 p g4*t4	B27 p g5*t1	B28 p g5*t2	B29 p g5*t3	B30 p g5*t4	Parm
1	1	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1:p
1	1	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2:p
1	1	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3:p
1	1	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4:p
1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5:p
1	0	1	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6:p
1	0	1	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7:p
1	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	8:p
1	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	9:p
1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10:p
1	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	11:p
1	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	12:p
1	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	13:p
1	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	14:p
1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	15:p
1	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	16:p
1	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	17:p
1	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	18:p
1	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	19:p
1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	20:p
1	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	21:p
1	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	22:p
1	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	23:p
1	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	24:p
1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	25:p
1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	26:p
1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	27:p
1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	28:p
1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	29:p
1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	30:p

Figure 12: Excerpt of design matrix in program MARK illustrating the coding used to model the probability of initial capture using the global model p(g*t)c(g*t) for common brushtail possums trapped in summer at Leschenault Peninsula Conservation Park and Yalgorup National Park.

The global model assumes both *p* and *c* are different between all groups and are time-dependent, and that there is an interaction between the group effect and time. Figure 12 illustrates the portion of the design matrix relating to probability of initial capture, i.e. to p(g*t). The columns correspond to β parameter values and are numbered accordingly.

The first column refers to the c intercept which is always constant for all occasions/groups as a result of the linear structure of the model (see section 1.2). Columns 2 to 6 contain the variable coding for groups. The first five rows correspond to the five trapping occasions for group 1, which in this case is the Leschenault trapping web. Column 3 has the group effect for group 2 (Belvidere) and so on. Group 6 on the other hand, is coded for with all "0"s, which is sufficient to differentiate it from all other groups while reducing the number of unnecessary parameters (columns). Essentially, the variable coding scheme may be interpreted in the following manner; "1" if coding for the corresponding variable (e.g. male), "0" if other (e.g. female) (Cooch, 2001). In this case, the code "00000" would signify "not group 1, 2, 3, 4 or 5" therefore it codes for group 6.

Columns 7 to 10 refer to the time-dependency component of the model. The coding results in each trapping occasion (day) affecting *p* differently, but the effect of a particular occasion is equal for all groups. Again, day 5 is coded for using all 0's. The remaining columns (40 to 54) are representative of the interaction between group effect and time, and as such are simply a product of the two variables.

As the global model assumes the probabilities of initial capture and of recapture are both affected by the same variables, it is logical that both *p* and *c* will be coded for in the same way. The only exception is that, as mentioned earlier, there is one fewer occasion for recaptures (*c*) than for initial captures (*p*). As such, the quadrant in the design matrix corresponding to recaptures only has 24 rows (6 groups x 4 occasions), rather than 30 (6 groups x 5 occasions). Similarly, there are fewer parameters (columns) corresponding to the time effects as there are now only four time intervals rather than five.

2.4.5 C-hat ($\hat{c}$) adjustment

The variance inflation factor is a measure used to quantify the amount of overdispersion or underdispersion (see section 1.1). For the trapping data on common brushtail possums at LPCP and YNP, this overdispersion factor was accounted for by adjusting the $\hat{c}$ from 1.0 to 2.0 in increments of 0.25. The best model was obtained using an adjusted $\hat{c}$ of 1.5. This required the smallest increase (0.5 from 1) of ĉ after which the ranking of the preferred model did not change. A $\hat{c}$ adjustment of 1.5 was therefore applied to all further analyses.

Chapter 3 Results

Over the spring and summer trapping periods a total of 311 individuals belonging to fifteen species were captured (Table 7). Captures for only eight of these species consisted of more than one individual. These included common brushtail possums, quendas (or southern brown bandicoots – *Isoodon obesulus*), house mice (*Mus musculus*), bobtail lizards, south-western crevice skinks (*Egernia napoleonis*), Australian striped ctenotus (*Ctenotus australis*), black-headed monitors (*Varanus tristis*) and one non-target species – Australian magpies (*Gymnorhina tibicen*).

The species most commonly captured was the common brushtail possum with a total of 154 individuals having been trapped 586 times. Of these, 80 were female and 74 were male. The mean weight for males (1571.10 g) was higher than for females (1453.36 g) which is characteristic of the dimorphism of the species (How & Hillcox, 2000).

Captures of *C. australis* were limited to Leschenault Peninsula Conservation Park (LPCP). The species was not previously known to occur at LPCP (CALM, 1998). The capture of the species resulted in a new record of the species in the area and a range expansion of the species (Maryan, 2006, pers comm.). South-western crevice skinks captures were restricted to the Lake Pollard trapping web within Yalgorup National Park (YNP). Fewer crevice skinks were trapped in spring than in summer with capture success rates of 1.24 and 1.65 respectively.

Table 7: Species numbers trapped at Leschenault Peninsula Conservation Park and Yalgorup National Park (pooled across all webs) including numbers of individuals trapped and numbers of total captures (including recaptures).

Species name	**Spring**		**Summer**		**Total over both seasons**	
	Individuals	Captures	Individuals	Captures	Individuals	Captures
Acanthiza apicalis	0	0	1	1	**1**	**1**
Ctenotus australis	3	3	2	2	**5**	**5**
Dasyurus geoffroii	0	0	1	2	**1**	**2**
Egernia napoleonis	6	7	8	8	**13**	**15**
Gymnorhina tibicen	2	2	4	4	**6**	**6**
Isoodon obesulus	11	13	12	26	**13**	**39**
Litoria moorei	1	1	0	0	**1**	**1**
Morelia spilota	1	1	0	0	**1**	**1**
Morethia lineocellata	1	1	0	0	**1**	**1**
Mus musculus	21	25	4	5	**23**	**30**
Notechis ater occidentalis	0	0	1	1	**1**	**1**
Pseudonaja affinis	1	1	0	0	**1**	**1**
Tiliqua rugosa	75	93	27	32	**87**	**125**
Trichosurus vulpecula	90	177	123	409	**154**	**586**
Varanus tristis	2	2	1	1	**3**	**3**
Total	**214**	**326**	**184**	**491**	**311**	**817**

3.1 Trap success

3.1.1 Common brushtail possums

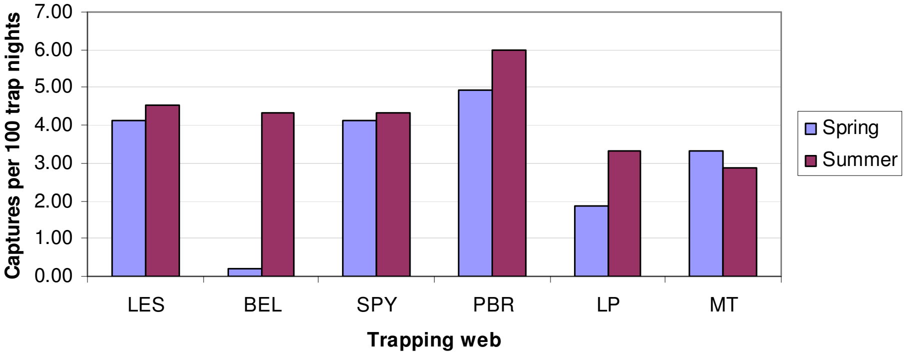


Figure 13: Trap success for common brushtail possums trapped at Leschenault Peninsula Conservation Park and Yalgorup National Park.

The common brushtail possum was the most commonly trapped species. The highest rate of captures occurred at the Preston Beach Road trapping web while the fewest captures occurred at the Martin's Tank site. Overall, capture success for brushtail possums ranged from 1.86 to 5.98 captures per 100 trap nights with the exclusion of one outlier at Belvidere in spring (Figure 13). Only one brushtail possum was captured over the spring session at Belvidere resulting in a capture success rate of 0.21. However, the summer capture success rate for the same trapping web was comparable to those at other webs at 4.33 captures per 100 trap nights.

To investigate reasons for the dramatic difference between seasons in capture rate of brushtail possums at Belvidere, the number of cage traps occupied by bobtail lizards, brushtail possums and other species was compared with the total number of traps occupied during the spring (Figure 14) and summer

(Figure 15) trapping sessions. Examination of this data revealed a definite dominance by bobtail lizards in spring and by brushtail possums in summer.

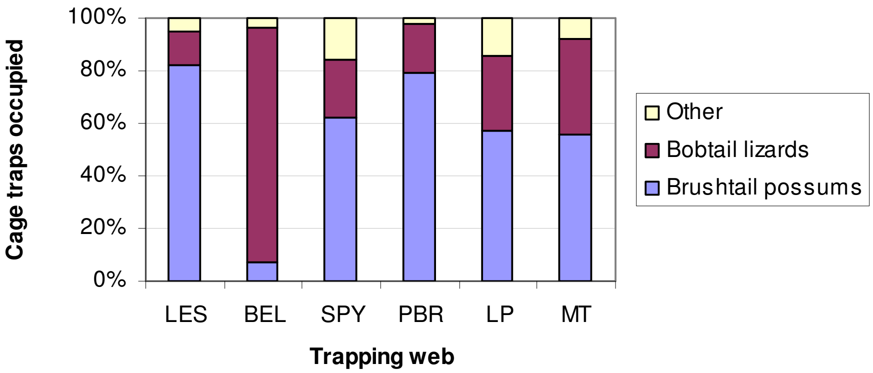


Figure 14: The proportion of cage traps occupied by common brushtail possums, bobtail lizards and other species in spring at Leschenault Peninsula Conservation Park and Yalgorup National Park.

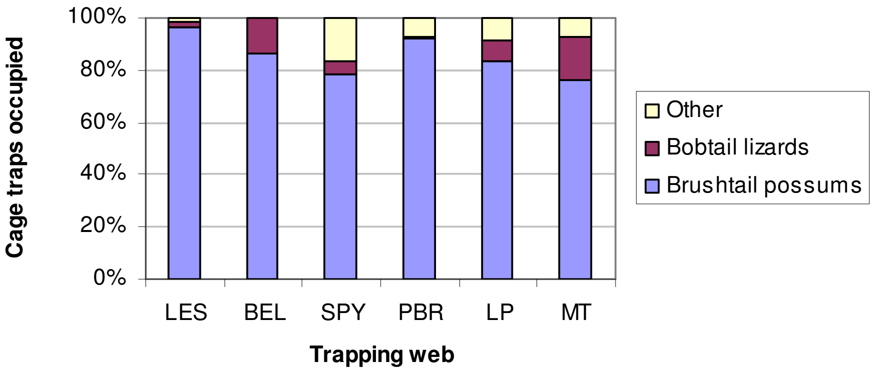


Figure 15: The proportion of cage traps occupied by common brushtail possums, bobtail lizards and other species in summer at Leschenault Peninsula Conservation Park and Yalgorup National Park.

3.1.2 Quendas

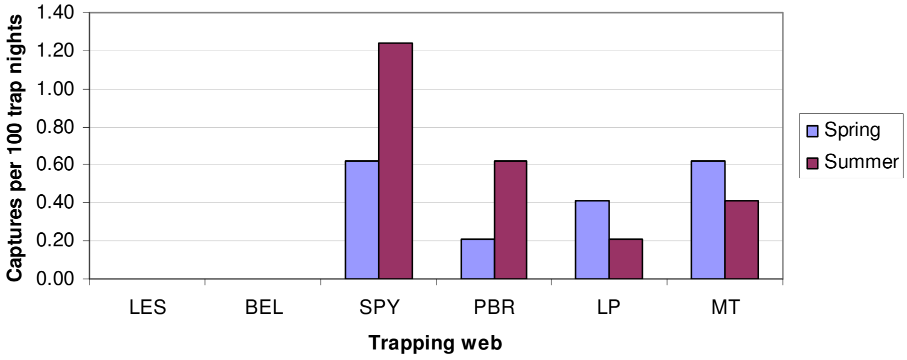


Figure 16: Trap success for quendas trapped at Leschenault Peninsula Conservation Park and Yalgorup National Park.

No quendas were captured at either trapping web at LPCP (Figure 16). At the two sites where quendas were present, capture success ranged from 0.21 at the Preston Beach Road web and Lake Pollard to 1.24 at Spyridium. Capture success varied between seasons. Interestingly, most captures occurred in summer at both Preston Beach Road trapping webs whereas more quendas were captured during spring at the two webs at the Martin’s Tank site. However, as only 1 – 3 quendas were trapped at most webs these differences may not be biologically significant.

3.1.3 House mice

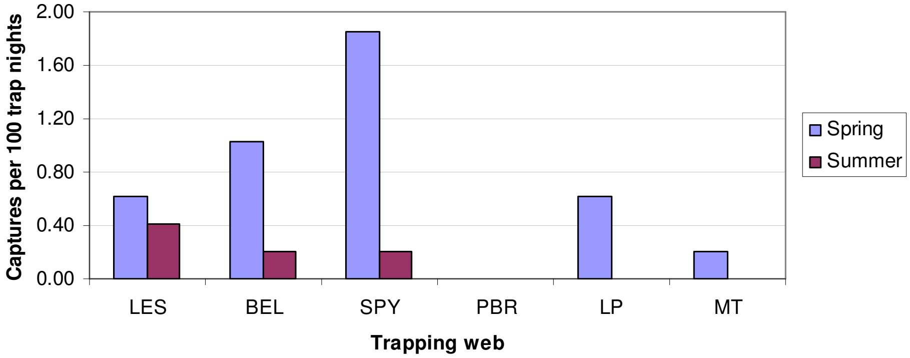


Figure 17: Trap success for house mice trapped at Leschenault Peninsula Conservation Park and Yalgorup National Park.

House mice were captured at all trapping webs with the exception of Preston Beach Road (Figure 17). There were considerably more house mice captured in spring than in summer with capture rates peaking at 1.86 captures per 100 trap nights at Spyridium. Numbers of house mice captured in summer fell to only one or two individuals per trapping web (capture rates of 0.21 – 0.41). No mice were trapped at the Martin's Tank site in summer.

3.1.4 Bobtail lizards

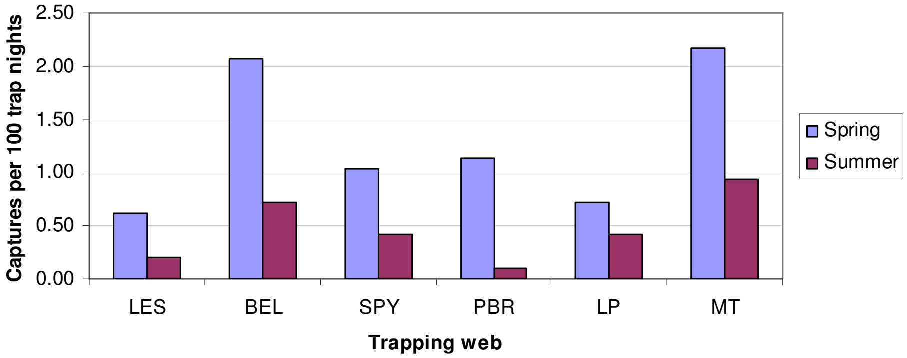


Figure 18: Trap success for bobtail lizards trapped at Leschenault Peninsula Conservation Park and Yalgorup National Park.

Bobtail lizards were the second most commonly trapped species during the study. Capture rates ranged from 0.21 at Preston Beach Road in summer, to 2.16 at Martin's Tank in spring (Figure 18). Capture success rates were markedly higher in spring than in summer across all trapping webs. The highest number of captures occurred at Martin's Tank and Belvidere. The largest differences between sessions are evident at the Belvidere, Preston Beach Road and Martin's Tank webs.

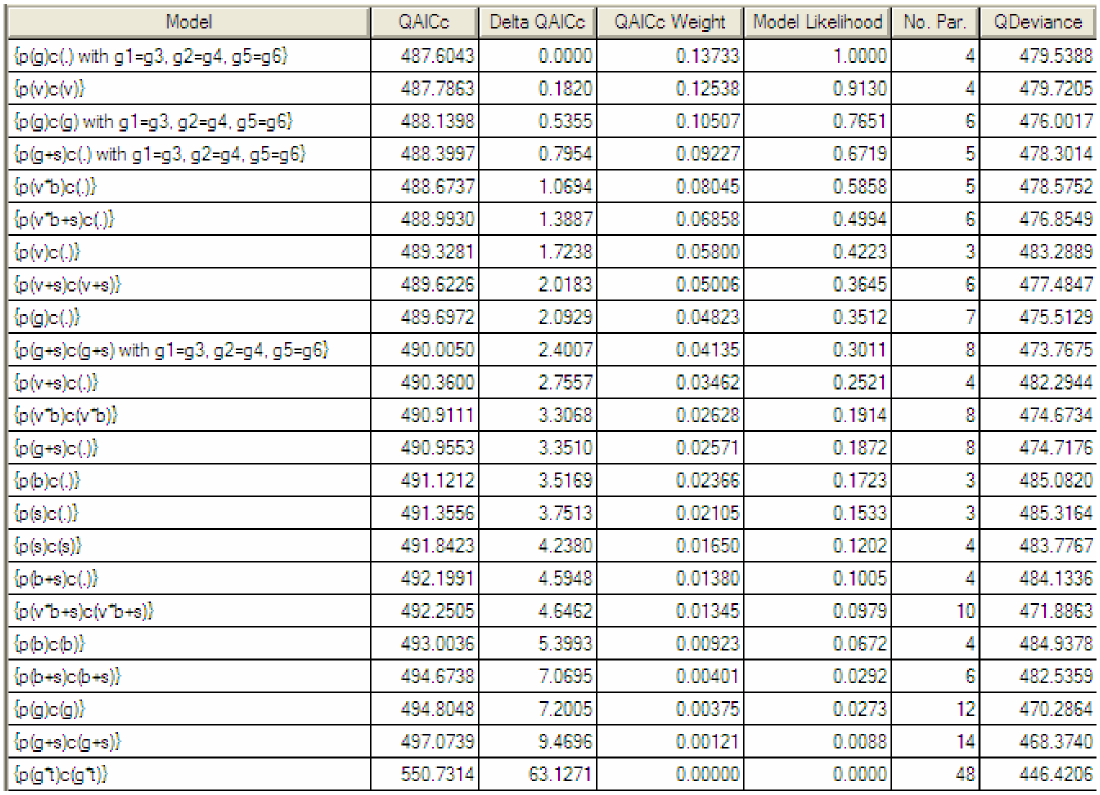
3.2 MARK analysis

A total of 23 candidate models were analysed in Program MARK (White, 2001) including the fully time-dependent global (p(g*t)c(g*t)) model. After a ĉ adjustment to 1.50, {p(g)c(.) where g1 = g3, g2 = g4, g5 = g6} was selected as the preferred model. Models are considered to fit the data well if they have a ΔAIC of 2 or less (Burnham & Anderson, 2001; Burnham & Anderson, 2002). Seven models fit this criterion (Table 8). Three of the four models tested containing the group effect where g1 = g3, g2 = g4, g5 = g6 were within these top 7 models. Two models contained a vegetation and bait interaction, which is equivalent to g1 = g3, g2 = g4, g5 ≠ g6. The remaining two models belonging to the top seven contained a vegetation effect on its own.

Clearly, vegetation is a factor of very high importance in determining the capture probabilities of brushtail possums at LPCP and YNP. Given that five of the top seven models also included the variable for baiting in some form, it becomes evident capture rate is regulated by some form of relationship between 1080 baiting and vegetation.

Also of note, five of the seven models within a ΔAIC_c of 2.0, including the preferred model, had *c* constrained as constant throughout all groups. This would suggest capture heterogeneity exists, e.g. trap happiness. Models containing sex as a coefficient ranged from being within the preferred models to being highly unlikely (ΔAIC_c of 9.47).

Table 8: Program MARK results browser showing the ranking of all models tested for common brushtail possum trapping data.



Model	QAICc	Delta QAICc	QAICc Weight	Model Likelihood	No. Par.	QDeviance
{p(g)c(.) with g1=g3, g2=g4, g5=g6}	487.6043	0.0000	0.13733	1.0000	4	479.5388
{p(v)c(v)}	487.7863	0.1820	0.12538	0.9130	4	479.7205
{p(g)c(g) with g1=g3, g2=g4, g5=g6}	488.1398	0.5355	0.10507	0.7651	6	476.0017
{p(g+s)c(.) with g1=g3, g2=g4, g5=g6}	488.3997	0.7954	0.09227	0.6719	5	478.3014
{p(v*b)c(.)}	488.6737	1.0694	0.08045	0.5858	5	478.5752
{p(v*b+s)c(.)}	488.9930	1.3887	0.06858	0.4994	6	476.8549
{p(v)c(.)}	489.3281	1.7238	0.05800	0.4223	3	483.2889
{p(v+s)c(v+s)}	489.6226	2.0183	0.05006	0.3645	6	477.4847
{p(g)c(.)}	489.6972	2.0929	0.04823	0.3512	7	475.5129
{p(g+s)c(g+s) with g1=g3, g2=g4, g5=g6}	490.0050	2.4007	0.04135	0.3011	8	473.7675
{p(v+s)c(.)}	490.3600	2.7557	0.03462	0.2521	4	482.2944
{p(v*b)c(v*b)}	490.9111	3.3068	0.02628	0.1914	8	474.6734
{p(g+s)c(.)}	490.9553	3.3510	0.02571	0.1872	8	474.7176
{p(b)c(.)}	491.1212	3.5169	0.02366	0.1723	3	485.0820
{p(s)c(.)}	491.3556	3.7513	0.02105	0.1533	3	485.3164
{p(s)c(s)}	491.8423	4.2380	0.01650	0.1202	4	483.7767
{p(b+s)c(.)}	492.1991	4.5948	0.01380	0.1005	4	484.1336
{p(v*b+s)c(v*b+s)}	492.2505	4.6462	0.01345	0.0979	10	471.8863
{p(b)c(b)}	493.0036	5.3993	0.00923	0.0672	4	484.9378
{p(b+s)c(b+s)}	494.6738	7.0695	0.00401	0.0292	6	482.5359
{p(g)c(g)}	494.8048	7.2005	0.00375	0.0273	12	470.2864
{p(g+s)c(g+s)}	497.0739	9.4696	0.00121	0.0088	14	468.3740
{p(g*t)c(g*t)}	550.7314	63.1271	0.00000	0.0000	48	446.4206

Due to the large number of models which are considered to fit the data well (are within a ΔAIC_c of 2) population estimates were obtained using model averaging. This approach factors in all the models within the candidate set as well as their relative importance (weight). The model averaged population estimates are provided in Figure 19 with more details available in Appendix 2.

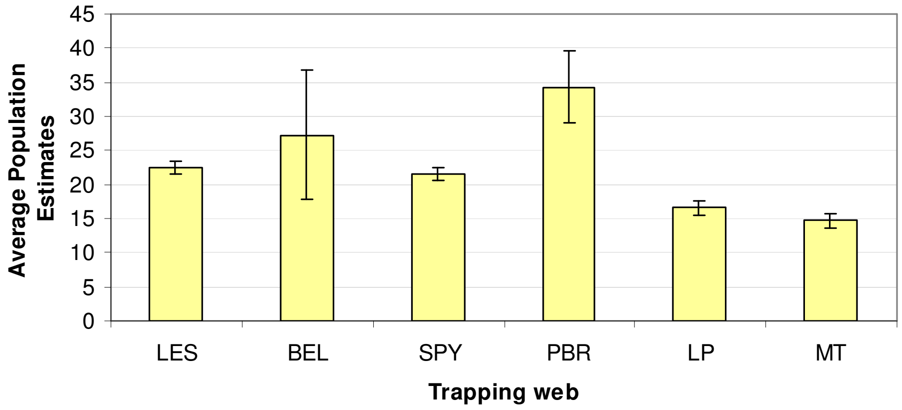


Figure 19: Model averaged derived population estimates using Huggins Closed Population Estimation and an adjusted ĉ of 1.50 for summer trapping of common brushtail possums at Leschenault Peninsula Conservation Park and Yalgorup National Park.

The highest populations were found at the Preston Beach Road (YNP) and Belvidere (LPCP) trapping webs with population size estimated to be 34.32 and 27.24 brushtail possums respectively. However, these two trapping webs also produced the highest standard errors (5.19 and 9.53 respectively). Belvidere had a particularly large standard error so any interpretation should be made with caution. The Preston Beach Road trapping web had the highest population size estimate which, even given the standard error, was markedly higher than at all other webs, except Belvidere.

There were some clear similarities between the Leschenault and Spyridium webs (with population estimates of 22.48 and 21.49 respectively) and the Lake Pollard and Martin's Tank webs (with population estimates of 16.53 and 14.69 respectively). The standard error for these for trapping webs was low, ranging from 0.91 (at Leschenault) to 1.10 (at Lake Pollard).

There appears to be a general trend towards three categories of baiting/vegetation interactions. The highest population estimates were obtained at the two trapping webs which were 1080 baited and consisted of an open vegetation structure. The two baited webs with dense vegetation had lower population estimates but these were comparable to each other. The lowest population estimates were obtained at the unbaited site, with a marginally higher population estimate occurring at Lake Pollard – the densely vegetated trapping web.

The estimated β parameters for all the models which contained sex as a coefficient had positive values for sex when modelling initial capture rates and negative values when modelling recapture rates (for those models where *c* was not constant). For example, the optimal model (with a ΔAIC_c value of 2.02) to contain the sex effect for both p and c {p(v+s)c(v+s)} produced a β value of 0.45 (standard error of 0.45) for '*p* sex' and -0.34 (standard error of 0.30) for '*c* sex'. This indicates males are more likely to be captured initially than females, but are less likely to be recaptured.

Table 9: Beta (β) parameter estimates for the model {p(g)c(.) with g1=g3, g2=g4, g5=g6} for common brushtail trapping data in summer.

Parameter	Beta (β)	Standard Error	95% Confidence Interval	
			Lower	Upper
1: p Intercept	0.1842225	0.4052005	-0.6099704	0.9784154
2: p g1 = g3	0.1076125	0.5270501	1.1406307	-0.9254057
3: p g2 = g4	-1.1453878	0.6227021	0.0751082	-2.3658838
4: c Intercept	1.2016448	0.1506207	0.9064282	1.4968614

As evident from Table 9, the β parameter estimates for the best model {p(g)c(.) with g1=g3, g2=g4, g5=g6} indicate a positive slope for groups 1 and 3 as well as groups 5 and 6 (which in this case are representative of the β estimate for the p intercept) and a negative slope for groups 2 and 4. This indicates brushtail possums are least likely to be captured at the Belvidere and Preston Beach Road trapping webs (groups 2 and 4). This is confirmed by examination of the real function parameter estimates (Table 10).

Table 10: Real function parameter estimates corrected for $\hat{c} = 1.50$ for the model {p(g)c(.) g1=g3, g2=g4, g5=g6} for summer trapping data on common brushtail possums at Leschenault Peninsula Conservation Park and Yalgorup National Park.

Parameter / Group	Estimate	Standard Error	95% Confidence Interval	
			Lower	Upper
p g1 and g3	0.5724453	0.0824907	0.4088406	0.7216037
p g2 and g4	0.2766449	0.0946196	0.1314822	0.4913972
p g5 and g6	0.5459258	0.1004455	0.3520659	0.7267937
c (all groups)	0.7688173	0.0267710	0.7122686	0.8171060

The lowest probability of initial capture occurred at Belvidere and Preston Beach Road (groups 1 and 3), i.e. at the 1080 baited trapping webs with open vegetation. The remaining four trapping webs had comparable probability of initial capture at approximately 0.57 (Leschenault and Spyridium) and 0.55 (Lake Pollard and Martin’s Tank), particularly when the standard error is considered (0.08 and 0.1 respectively). The probability of subsequent capture (0.77) was higher than that of initial capture. For the model {p(g)c(g) with g1=g3, g2=g4, g5=g6} the real function parameter estimates for recapture rates were as high as 0.82 (for Belvidere and Preston Beach Road) which fits within the 95% confidence interval for the model where probability of recapture is constrained as constant. This, coupled with the high ranking of models containing a constant rate of recapture, provides strong evidence for capture heterogeneity among brushtail possums.

3.3 Comparison of methods for quantifying data

For the data analysed, the minimum number known to be alive (MNA) is equivalent to the total number of individuals captured. The model averaged population estimates derived by program MARK (White, 2001) using Huggins Closed Captures were compared to the MNA for the summer trapping session for brushtail possums (Figure 20).

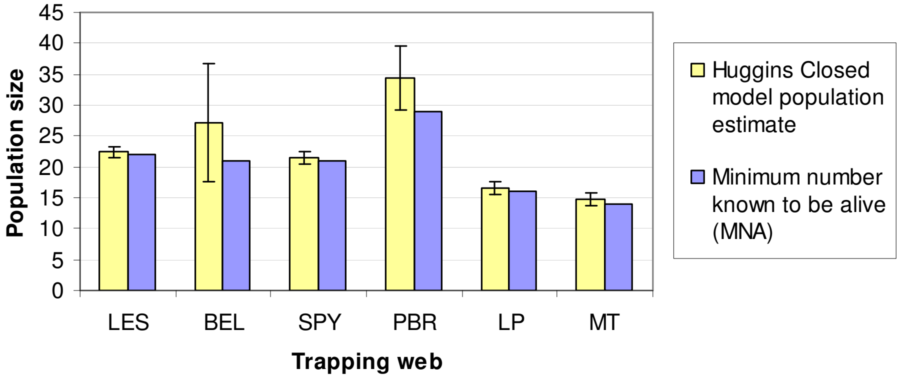


Figure 20: Comparison of population estimates derived through model averaging using program MARK and minimum number known to be alive (MNA) for trapping data on common brushtail possums at Leschenault Peninsula Conservation Park and Yalgorup National Park.

It is clear that, in this case, MNA is comparable to the population estimates. The MNA values were consistently lower than the population estimates derived from the Huggins closed model estimate. However, all MNA fell within the standard errors of the population estimates with the exception of the Preston Beach Road web. The MNA at Preston Beach Road was 0.13 below the lowest likely estimate, which equates to less than one individual.

Chapter 4 Discussion

4.1 Availability of alternative prey species

4.1.1 Quendas

No quendas were observed at either of the trapping webs at Leschenault Peninsula Conservation Park (LPCP). This is despite a previous reintroduction conducted by DEC (CALM at the time) as part of the Western Shield program (de Tores, 2007, pers comm.). Further trapping in other areas of LPCP would be required to confirm if quendas are absent from the entire Peninsula. Given the quenda release site was within a few hundred metres of the Belvidere trapping web (de Tores, 2007, pers comm.), the results suggest the reintroduction may not have been successful and the population did not persist. Alternatively, the population may have persisted at undetectable density and/or individuals may have dispersed to more suitable habitat not represented within the trapping webs at LPCP.

The highest capture success for quendas occurred at the Spyridium trapping web (Yalgorup National Park; YNP) (Figure 16). This may be related to the thick understorey occurring throughout the area. The preferred habitat of quendas is dense shrubland with considerable cover in the understorey (0 - 1 m height interval) and with adjacent forest and woodland (Friend, 1990; Maxwell *et al.*, 1996). Spyridium appears to have the thickest growth habit of all the trapping webs and so may have provided optimal habitat for quendas.

Interestingly, fewer quendas were captured at Lake Pollard than at Martin's Tank. This is despite Lake Pollard being roughly classified as dense and Martin's Tank as open vegetation. It is possible that, although the broad classifications are fitting, there are more detailed structural differences between the trapping webs which may account for this difference in capture success.

Personal observations of the trapping study indicated quendas may prefer areas of grasstrees with long skirts as quendas were frequently captured at trapping points in the vicinity of such vegetation. While there were many grasstrees occurring throughout the Lake Pollard web, these were generally very tall (over 1 m, but sometimes much taller) and often showed signs of a past fire event. Martin's Tank, on the other hand, comprised a significant area of grasstrees whose thick skirts reached the ground (roughly located in the area between the West and North-east arms of the trapping web). All quendas trapped at Martin's Tank were captured on the North-east, North and North-west arms. This may provide some evidence that quendas prefer habitat which contains thick grasstree skirts.

Confirmation of such a trend would require detailed vegetation descriptions to be obtained for each of the trapping webs, including observations of plant species present and vegetation structure (percentage ground cover, vegetation height, etc.). This information could then be related to the specific location of captures of quendas. There is also a need to increase the sample size as the maximum number of individuals captured at any given trapping web was six. This could be achieved by either continuing trapping over several sessions or by pooling data across all trapping webs. Alternatively, radio-tracking of quendas could be undertaken to assess habitat use by quendas. Confirmation of a relationship between dense ground cover and thick grasstrees and the abundance of quendas could provide information on the importance of long-unburnt vegetation in conservation management of the species.

4.1.2 House mice

Where house mice occurred (at all but the Preston Beach Road trapping web), markedly more mice were captured in spring than in summer. These differences were possibly attributed to changes in weather conditions and in food availability as well as to breeding patterns.

In Victoria, house mice were observed to peak in breeding activity in October with the number of breeding females falling by the end of February (Chambers *et al.*, 2000). A study in Queensland (Krebs *et al.*, 1995) found that breeding males had larger home ranges and were more active than breeding females. Following cessation of the breeding season home-ranges increased ten-fold and the majority of mice became nomadic. This was consistent with results from the Victorian study (Chambers *et al.*, 2000) which also found an increase in home-range size after the breeding season.

Assuming similar timing for the breeding season in WA, both trapping sessions at LPCP and YNP were undertaken during the breeding season, with summer trapping occurring at its final stages. The majority of house mice captures were male at 73.33% compared to 13.33% females (sex was not determined for the remainder). This difference most likely reflects the difference in activity between breeding males and females suggesting males were more likely to encounter a trap. The low capture success in summer may be attributed to a decline in activity and higher dispersion towards the end of the breeding period.

Generally, capture success for house mice was highest at trapping webs that were densely vegetated (Leschenault, Spyridium and Lake Pollard) with the exception of spring data for Belvidere. The highest numbers of house mice were captured at Spyridium, the trapping web with possibly the densest understorey. Assuming these differences in capture rate realistically reflect differences in the abundance of house mice, there appears to be a preference for dense vegetation.

This preference may relate to the availability of protection from predation. Animals utilise a variety of predator avoidance strategies in an attempt to extend their longevity. A common indicator (cue) of predation risk for small mammals is the intensity of illumination, such as moonlight (Clarke, 1983; Dickman, 1992; Wolfe & Summerlin, 1989). Dickman (1992) examined the response of house mice in WA to native and introduced predators and found mice increased their use of dense vegetation on moonlit nights when the probability of detection by predators was high (due to higher visibility). This may be the reason for capture success of house mice being higher at densely vegetated trapping webs, where predator protection is high, than at the more openly vegetated webs where the probability of detection by predators increases.

4.1.3 Bobtail lizards

Capture success for bobtail lizards decreased consistently across all trapping webs from spring to summer. These seasonal differences may be attributed to a number of variables including food availability, breeding patterns or climatic factors. The most likely explanation involves a combination of these.

Though remaining solitary for most of the year, bobtails are a monogamous species with partnerships reforming with each successive year for up to seven years (Bull, 1990). Pairs form in early September following which the lizards may spend up to 50% of their daily activity together (Bull *et al.*, 1991). In South Australia, mating occurs in late October or early November followed by live births in the summer months of March and April (Bull *et al.*, 1994; Bull *et al.*, 1993). Pairs are known to separate suddenly following mating (Bull *et al.*, 1991; Fergusson & Algar, 1986). After this time, the activity, and therefore encounter rate, of both sexes of bobtails declines significantly (Bull *et al.*, 1991). This is most likely related to the rising temperatures associated with summer and the resulting lower food resource.

Spring trapping occurred in late November and early December (Table 3). This is approximately one month following the end of the breeding season as predicted by the South Australian study. However, it is possible that the breeding season may be slightly later in the year in WA than in eastern states. During the spring trapping session bobtail lizards were frequently observed in pairs on roads. As few studies on bobtails have been undertaken in WA it is difficult to specify the exact time of the breeding season specific to the region. It is likely that the differences in capture rate between seasons were related to this sudden decrease in activity following breeding season and leading into the warmer, harsher summer days.

Because spring trapping was undertaken over four weeks commencing at the end of the South Australian breeding season, it is possible that bobtail lizards were still breeding at some trapping webs but that breeding had ceased at others. It is recommended that in future studies of bobtail lizard abundance spring trapping be carried out earlier (preferably October/November) to ensure that trapping is carried out during the likely breeding season at all trapping webs. This would allow for greater consistency between webs and for more reliable comparisons to be made.

There appears to be a trend of higher captures at trapping webs with an open vegetation structure. The Belvidere, Preston Beach Road and Martin's Tank trapping webs provided the highest capture success rates for both seasons with the exception of Preston Beach Road which actually produced the lowest capture rate in summer. It may be expected that bobtail lizard activity is highest at these trapping webs due to the openness of the vegetation allowing a higher proportion of sunlight to penetrate through to ground level. Further analyses using a software package such as DISTANCE (Thomas *et al.*, 2005), which permits for data that is more sparse than with MARK analyses, would be required to determine if the differences between dense and openly vegetated trapping webs reflect differences in population size or relate only to differential capture probability.

4.2 Common brushtail possums at Belvidere

A comparison of spring and summer trapping sessions at Belvidere revealed a striking difference in the number of common brushtail possums captured (Figure 13). While only one individual possum ('Ducky') was trapped on two occasions in spring, twenty one individuals were trapped on 50 occasions in summer. This included Ducky who was caught on all five trapping nights. No other trapping web showed such a marked difference in brushtail possum capture rates between seasons. Given the high capture rate only three months later, it is highly unlikely Ducky was the only possum in the area during the spring trapping. Barring a mass immigration event in between seasons it is reasonable to assume the difference in capture rates arose from a difference in trappability of possums, not a difference in abundance.

One difference between seasons at Belvidere of note was that while the capture rate of possums increased markedly from spring to summer, the number of bobtail lizards trapped dropped. This brings forth the hypothesis that a higher bobtail capture rate resulted in fewer cage traps remaining open and available for possums to enter.

A comparison of cage traps occupied by bobtail lizards, brushtail possums and other species with the total number of traps occupied during the spring (Figure 14) and summer (Figure 15) trapping sessions revealed a definite dominance by bobtails in spring and by brushtail possums in summer. It may be assumed that bobtails, being diurnal, entered the traps earlier in the day than possums, which are nocturnal. However, it is possible that some bobtails may have entered traps in the early morning immediately prior to traps being checked. Assuming the number of these occurrences was relatively low, it is clear that, given the high capture success rate of bobtail lizards in spring (2.88 total captures per 100 trap nights), fewer traps would have remained available for possums to enter than in summer. When bobtails were trapped at a lower frequency (0.82 total captures per 100 trap nights in summer), brushtail possum capture rate increased drastically (from 0.41 to 10.31 total captures per 100 trap nights).

While evidence for this theory is in no way conclusive, it is preferred to avoid such bias in population estimates by adjusting the trapping protocol so traps are cleared of animals at dusk as well as in the mornings. By releasing all animals trapped during the day the maximum amount of traps would be available for nocturnal animals. This would also have ethical benefits as diurnal animals would spend less time in traps.

Another potential explanation is a difference between seasons in food availability. A particular food source (or sources) may have been plentiful in spring but reduced in summer. In spring, this would have reduced the area traversed by brushtail possums while foraging as well as deterred them from seeking out baits, reducing their capture rate. If this was the case at Belvidere it certainly didn’t appear to be so at Leschenault, only a short distance away. This difference in food availability may have been a result of differences in vegetation structure and composition (Table 1). However, no studies on the diet of the brushtail possums or detailed vegetation analyses were undertaken concurrently with the trapping as this was outside the scope of the project.

Testing of this hypothesis would require such data to be collected simultaneously with future trapping of prey species.

The suggestion that brushtail possums may have had a lower capture success rate in spring due to the large number of bobtail lizards occupying traps raises a new question; why were there so many bobtails captured in spring and so few in summer? Reasons for this were hypothesised previously in section 4.1.

4.3 Common brushtail possums and program MARK

More common brushtail possums were captured than any other species. However, the only data sufficiently robust to enable analysis through Capture-Mark-Recapture (CMR) techniques were for the brushtail possum captures for the summer trapping session. This indicates the data constitute "sparse" data. In this case the models fail to estimate one or more parameter(s). This is often used as the rationale for simplistic presentation of results through capture success and/or minimum number known to be alive (MNA) estimates. Such simplistic presentation of results restricts interpretation and any inference from the data. Given the high frequency of studies which present data in the form of MNA and/or capture success as described previously (Chapter 1), there is a clear need for evaluation of methods which provide a large amount of information obtained from sparse data.

For the data from this study which was sufficiently robust for CMR analysis (i.e. the brushtail possum summer trapping session), analysis in program MARK (White, 2001) revealed a clear preference for models containing vegetation and baiting effects and/or a relationship between the two. It is clear that these two factors strongly affect the behaviour of brushtail possums at LPCP and YNP and therefore affect encounter probabilities.

There was strong support for models in which encounter probabilities were different for baited trapping webs with dense vegetation, baited webs with open vegetation and for unbaited webs. This suggests brushtail possums adopt predator avoidance behaviour strategies. Where the perceived risk of predation is high (in unbaited areas), brushtail possums are likely to restrict their movements and prefer areas of dense vegetation.

There was strong evidence for capture heterogeneity as revealed by the high ranking of models containing a constant probability of recapture (Table 8) as well as by examination of real function parameter estimates (Table 10). Probability of recapture was higher than that of initial capture, suggesting trap-happiness. This finding is consistent with previous studies which have observed such behaviour (Efford, 2004; Wayne *et al.*, 2005a).

Examination of β values also revealed a dichotomy of probability of capture between sexes. While males were more likely to be captured initially, females were decidedly more likely to be recaptured. This is likely related to the differences in home range size and mobility of the two sexes. Females generally have smaller home ranges and show greater site fidelity within their home range than males (Clout & Efford, 1984; Cowan & Clout, 2000; How & Hillcox, 2000). Males on the other hand can travel relatively large distances while dispersing and while moving about their home-ranges and seeking out mates (Clout & Efford, 1984; Cowan & Clout, 2000). It is intuitive that the larger the area traversed, the higher the likelihood of encountering a trap and subsequently being captured. Males can therefore be expected to have a higher initial capture rate than females. However, it follows that being more mobile males are also more likely to move away from the trap location or leave the trapping area. This would decrease the probability of being recaptured. Females, on the other hand, have a higher probability of recapture as they are more likely to remain in the vicinity of the trap.

The primary hypothesis tested within this project was that brushtail possum populations are highly affected by predator-prey relationships. The data derived from estimates of population size using a multi-model approach support this. The highest population estimates were derived at baited trapping webs with open vegetation with fewer brushtails occurring in the baited webs with dense vegetation. Unbaited webs produced the lowest population estimates for brushtail possums. Of these, the trapping web with dense vegetation produced the higher population estimate. These results are in accordance with the hypothesis which stipulated that where the risk of predation was low (in baited areas) brushtail possums would prefer vegetation which provides optimum habitat and that where predation risk is higher (in unbaited areas), the possums would select vegetation which provides a predator refuge.

Brushtail possums are known to spend a large amount of time on the ground as ground vegetation may constitute a large proportion of their diet (Kerle, 1984; Pickett *et al.*, 2005). The importance of den trees (preferably old or dead trees containing hollows) is also well known (Inions *et al.*, 1989; Jones

& Hillcox, 1995). It is intuitive therefore, that brushtail possums would prefer habitat with a high availability of tree hollows and an open vegetation structure, such as at the “open” trapping webs at which tuarts are a dominant canopy species. However, when active on the ground, brushtail possums are susceptible to increased predation risk. Several studies have observed a decrease in activity levels and increased preference for dense and closed vegetation in areas where the risk of predation is high (Dickman, 1992; Korpimäki *et al.*, 1996; Longland & Price, 1991).

The results of this study confirm these trends in brushtail possum behaviour. Population estimates obtained through program MARK indicate a preference for areas with an open vegetation structure which also exhibit a high incidence of old or dead tuarts which may be suitable for provision of den sites. Where there is an additional risk of predation from foxes, the availability of shelter from predators becomes more important in habitat selection. Where predation risk is high, common brushtail possums were found to prefer areas of dense vegetation.

These results may be indicative that the presence of introduced predators such as foxes may be affecting brushtail possum populations by restricting their home ranges to areas of sub-optimal habitat. The importance of vegetation to brushtail populations also highlights a clear requirement for habitat management. The importance of tree hollows should be acknowledged as well as the relative importance of dense groundcover in non-1080 baited areas.

4.4 Comparison of methods for quantifying trapping data

Based on the comparison of the minimum number known to be alive (MNA) and population estimates derived using Huggins Closed Captures models for the data analysed, there appears to be some merit to using the MNA approach when quantifying data. In this case, population estimates using program MARK (White, 2001) and using the MNA method were certainly comparable and followed roughly the same trends. One major difference, however, is that the MNA for Belvidere and Spyridium trapping webs were the same (21 possums) with Leschenault being slightly higher (22 possums). The program MARK (White, 2001) derived population estimates would suggest that population size was possibly higher at Belvidere than at the remaining two trapping webs. Although this can not be said for certain due to the high standard error at Belvidere, this is potentially representative of lost information if relying solely on MNA.

It is generally accepted that capture success rates are not to be used as a measure of population size but rather as an abundance index. Indices may be used to compare populations between study sites and between trapping sessions based on the assumption that fluctuations in capture rates reflect fluctuations in the population. Although, in this case, MNA values were comparable to population estimates using Huggins Closed Capture models, generally MNA values tend to underestimate the actual size of the population as detection probability is not considered and the estimate cannot be higher than the total number of individuals detected. There is considerable value in adopting a model-selection and multi-model inference approach. This allows valuable insights to be made on a wide variety of issues to obtain the maximum amount of information from the data at hand. The Information-Theoretic approach allows variables to be modelled and inferences drawn as to which factors affect population size and probability of detection the most. This particularly applies to animals which may display trap heterogeneity, such as trap happy brushtail possums (Efford, 2004; Wayne *et al.*, 2005a).

Program MARK (White, 2001) allows a vast amount of information to be inferred from even a small dataset. Minimum number known to be alive, as the name suggests, provides only the minimum number of animals known to be alive within the sample area. No further inferences may be made from the data and it is not known how many individuals may be present in the study area, but not detected. Based on the results from the summer trapping of common brushtail possums at LPCP and YNP it is highly advised to adopt a robust data approach (such as in program MARK) to analysing CMR data where logistically possible. In the situation where data are too sparse to enable CMR analyses, alternative I-T analyses such as DISTANCE sampling techniques (Buckland *et al.*, 2005; Thomas *et al.*, 2005) may be more appropriate and will enable inference to be made from the data.

4.5 Evaluation of trapping web design

One of the major limitations of many studies is that density estimates are difficult to obtain. This is because density estimation requires knowledge of the effective size of the area sampled. Trapping is often undertaken in a small area representative of a larger area occupied by the species of interest. In these cases it is highly likely that transient animals from outside the trapping area will also be captured, resulting in overestimation of the density (Buckland *et al.*, 1993). The species is therefore effectively sampled from within the trapping area as well as an undefined area surrounding the trapping grid/web (Sutherland, 2006).

Various methods have been used to account for this ambiguity. Burnham and Overton (1978) developed an non-linear least squares regression approach which assumes the density and mobility of the animals is constant over the whole grid and the surrounding area (Wilson *et al.*, 2007). Effective trapping area may therefore be estimated based on a nested grid approach – a method applied in program CAPTURE. This method requires the trapping grid to be square with equal spacings between trap points (Sutherland, 2006).

Trapping webs allow estimation of density on a similar principle, but allow estimation of density directly using distance sampling methods rather than relying on determining the sampling area and population size separately (Anderson *et al.*, 1983; Buckland *et al.*, 1993). Trapping webs consists of a set of concentric circles which, ideally, lie a constant distance apart with the innermost circle having a radius equal to half that distance (Sutherland, 2006). Programs such as DISTANCE (Thomas *et al.*, 2005) allow density estimates to be derived from Capture-Mark-Recapture data collected from trapping webs.

As illustrated in Figure 5, the design of the trapping webs utilised in this project incorporated a single trap point in the centre of the web with equal spacing between all trap points (i.e. 15 m between the centre point and points 01, as well as between points 01 and 02). The "centre" of the trapping web for the purposes of this project was therefore defined as area contained by all "01" trap points which includes the centre point. For any future studies utilising trapping webs it is recommended these are designed so that there is no centre trap point and the distance from the centre of the web to the first circle of traps is equal to half the distance between all other trap points (Lukacs *et al.*, 2005).

Estimation of density using trapping webs is based on the assumption that animals which are near the centre of the web are more likely to be captured than those on the outer edges (Sutherland, 2006). This relates to the simple fact that there is a higher density of traps in the centre of the web. However, the central traps are so close together that they interfere with each other to an unknown degree (Sutherland, 2006). As a result of this unquantified interference factor distance sampling methods are required to obtain density estimates.

Use of trapping web design in obtaining density estimates relies on the general assumptions pertaining to point transect sampling theory (Buckland *et al.*, 1993; Sutherland, 2006):

- All animals at the centre are captured at least once. I.e. probability of capture = 1 at the centre of the web and trapping continues until no new animals are captured in the centre of the web;

- Migration through the web does not occur as animals move over distances that are small relative to the size of the web. Trap spacing is therefore species dependent and considers the size of the home-ranges.

- The trapping of one animal is independent of the trapping of its neighbours;

- All animals are equally likely to be captured;

- Density is constant across the web.

The first of these assumptions requires trapping sessions be long enough for no new animals to be captured in the centre (Sutherland, 2006). New captures are to be expected on the outer edges of the trapping web since these traps, being spaced further apart, attract animals from a much larger area (Sutherland, 2006). This assumption is easily tested by examining the number of new individuals trapped near the centre of the web over the trapping occasions (Buckland *et al.*, 1993; Buckland *et al.*, 2005).

Based on all new captures for common brushtail possums in the trapping web centre, with data pooled over trapping webs and sessions, this assumption was supported (Figure 21). Most of the new animals in the centre were captured on the first day of trapping (six out of seven captures), and all new captures had been made by day four. This would suggest that it is feasible to limit trapping sessions to four days in length if logistics require this.

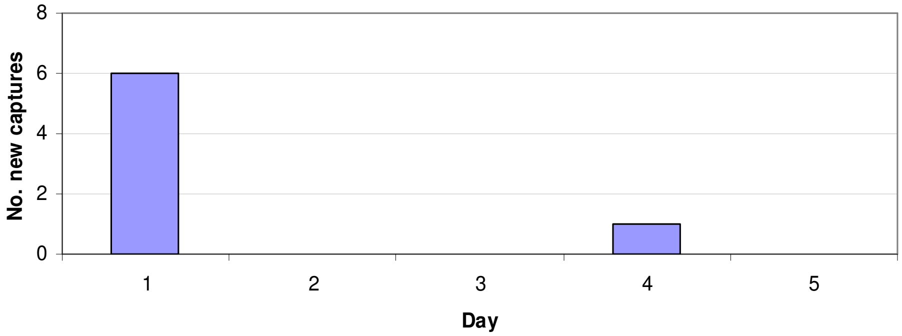


Figure 21: Number of new captures of common brushtail possums at the centre of the trapping web. Data is pooled across spring and summer seasons and across six trapping webs at Leschenault Peninsula Conservation Park and Yalgorup National Park.

The second requirement of trapping webs for density estimation is that no migration through the web occurs. Trapping webs perform best if the home range of the animal of interest is small compared to the size of the web (Buckland *et al.*, 1993). If trap spacing is too small overestimation of density can be expected due to the high concentration of traps at centre (Buckland *et al.*, 1993).

At the Preston Beach Road site, trapping webs did not prove to be independent of each other for brushtail possums and quendas. In the summer trapping session three quendas and one brushtail possum were captured at the Preston Beach Road web after having been previously captured at Spyridium. All quendas were male and the brushtail possum was a sub-adult female with a high ectoparasitic load.

A study of a Victorian population of southern brown bandicoots (*I. obesulus*) estimated home ranges to be 0.82 – 3.15 ha (Lobert, 1990). This was comparable to other studies, although Heinsohn (1966) recorded home ranges of 4.9 ±1.7 ha. In a study on common brushtail possums, How and Hillcox (2000) recorded 4.9 ha as the largest home range for a male and 4.3 ha for a female. Based on these data it home range size appears to be comparable for quendas and brushtail possums.

Given the violation of the assumption of independence between the Preston Beach Road and Spyridium trapping webs there is potential for overestimation of the population size at these trapping webs. It is recommended that, if DEC is to continue monitoring of prey availability at the Preston Beach Road site, one of the trapping webs be moved to increase the separation of trapping webs. Given the large home ranges for brushtail possums and quendas (up to ~5 ha) relative to that of the trapping webs (10.17 ha) it is also advisable that the distance between trap points be increased in future designs of trapping webs.

4.6 Conclusions

The benefits of using multi-model inference and I-T approach in quantifying trapping data are clear. The reliability of population estimates may be increased by adopting a multi-model approach which accounts for model uncertainty. This method is clearly more informative to that of using simple counts to provide an index of abundance. Data collected from trapping is generally difficult to obtain and requires a large amount of time, effort and resources. It is intuitive that one should aim to obtain the maximum amount of information from this data.

Program MARK (White, 2001) allows detailed and comprehensive information to be collected on an array of factors by incorporating knowledge of the biology of the species of interest. Where data are sufficiently robust, the use of software packages such as program MARK (White, 2001) in analysis of CMR data is highly recommended in future trapping studies aimed at estimating species abundance.

In this case, population estimates were obtained for common brushtail possums at western ringtail translocation sites. By modelling variables such as vegetation type, baiting regime, sex and by accounting for capture heterogeneity, it was possible to derive a large amount of information from a relatively sparse dataset (one trapping session only). Based on model-averaged population estimates obtained using the Huggins Closed Captures model type, the data indicates 1080 baiting for foxes is benefiting brushtail possum populations and highlights the value of open vegetation with a high availability of tree hollows.

Although data was collected from trapping webs which met the conditions of Lukacs, Anderson and Burnham (2005), i.e. that at least 90 traps are run for five consecutive nights and that the probability of capture is 1 at the centre, only one trapping session for one species (spring data for common brushtail possums) had data robust enough to allow analysis using program MARK (White, 2001). Program DISTANCE (Thomas *et al.*, 2005) allows similar analyses to be carried out using a smaller dataset by utilising the trapping web design. Program DISTANCE (Thomas *et al.*, 2005) has the added benefit that it enables estimation of population densities as the design of the trapping web overcomes the limitations of an unknown effective trapping area.

One of the major recommendations for future studies is to conduct further analysis of the trapping data using program DISTANCE (Thomas *et al.*, 2005). This would allow analysis of a greater range of data. Population densities can be obtained for five of the seventeen species trapped in this study. These include common brushtail possums (for spring as well as summer data), quendas, house mice, bobtail lizards and south-western crevice skinks. Analysis of a greater array of species would allow evaluation of the availability of alternative prey species to predators of the western ringtail possum. A continuation of the trapping conducted as part of this pilot study could be compared to results of future dietary analyses for foxes, feral cats and south-west carpet pythons. Dietary analyses may then be compared to the population estimates of available prey to determine if predators actively select for western ringtail possums as a preferred prey species.

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Appendix

Appendix 1

Table 11: List of all mammal, reptile and amphibian species potentially occurring in or near the study sites at Leschenault Peninsula Conservation Park and Yalgorup National Park.

Mammals

Antechinus flavipes leucogaster	Yellow-footed Antechinus
Phascogale tapoatafa	Brush-tailed Phascogale
Sminthopsis crassicaudata crassicaudata	Fat-tailed Dunnart
Sminthopsis griseoventer	Grey-bellied Dunnart
Cercartetus concinnus	Western Pygmy-Possum
Tarsipes rostratus	Honey Possum
Mus musculus	House Mouse
Rattus fuscipes fuscipes	Bush Rat
Rattus norvegicus	Brown Rat
Rattus rattus	Black Rat
Parantechinus apicalis	Southern Dibbler
Dasyurus geoffroii	Chuditch
Myrmecobius fasciatus	Numbat
Isoodon obesulus obesulus	Southern Brown Bandicoot (Quenda)
Pseudocheirus occidentalis	Western Ringtail Possum
Trichosurus vulpecula	Common Brushtail Possum
Macropus eugenii	Tammar Wallaby

Macropus fuliginosus	Western Grey Kangaroo
Macropus irma	Western Brush Wallaby
Setonix brachyurus	Quokka

Skinks

Egernia kingii	King Skink
Egernia luctuosa	Western Morning Skink
Egernia napoleonis	South-western Crevice Skink
Tiliqua rugosa rugosa	Bobtail
Acritoscincus trilineatum	South-western Cool Skink
Cryptoblepharus plagiocephalus	Fence Skink
Morethia lineoocellata	Western Pale-flecked Morethia
Morethia obscura	Obscure Skink
Menetia greyii	Common Dwarf Skink
Ctenotus catenifer	Heath Ctenotus
Ctenotus labilladieri	Labillardière's Skink
Ctenotus australis	Australian Striped Ctenotus
Ctenotus fallens	Coastal Ctenotus
Ctenotus impar	Eleven-striped Ctenotus
Hemiergis peronii	Lowlands Earless Skink
Hemiergis quadrilineata	
Lerista distinguenda	
Lerista elegans	
Lerista lineata	Lined Skink

Geckos

Diplodactylus spinigerus spinigerus	Western Spiny-tailed Gecko
Phyllodactylus marmoratus marmoratus	Marbled Gecko

Underwoodisaurus milii	Barking Gecko

Pygopods

Aprasia repens	
Delma fraseri	Fraser's Gecko
Delma grayii	Gray's Gecko
Lialis burtonis	Burton's Legless Lizard
Pygopus lepidopodus	Common Scaly Foot

Dragons

Pogona minor minor	Bearded Dragon

Monitors

Varanus goudlii	Gould's Monitor
Varanus rosenbergi	Southern Heath Monitor
Varanus tristis	Black-headed Monitor

Snakes

Ramphotyphlops australis	Southern Blind Snake
Ramphotyphlops pinguis	Rotund Blind Snake
Morelia spilota imbricata	Carpet Python
Morelia stimsoni stimsoni	Stimson's Python
Demansia psammophis reticulata	Yellow-faced Whipsnake
Notechis coronatus	Crowned snake
Notechis curtus	Bardick
Notechis scutatus occidentalis	Tiger snake
Pseudonaja affinis affinis	Dugite
Rhinoplocephalus goudlii	Gould's snake
Rhinoplocephalus nigriceps	Black-backed snake
Vermicella bertholdi	Jan's Banded snake

Vermicella bimaculata	Black-naped snake
Vermicella calonotos	Black-striped snake
Vermicella semifasciata	Southern Shovel-nosed snake

Frogs

Crinia georgiana	Quacking Frog
Geocrinia leai	Lea's Frog
Heleioporus eyeri	Moaning Frog
Heleioporus inornatus	Plain Frog
Limnodynastes dorsalis	Bullfrog or Banjo Frog
Litoria adelaidensis	Slender Tree Frog
Litoria moorei	Motorbike Frog
Myobatrachus gouldii	Turtle Frog
Neobatrachus pelobatoides	Humming Frog
Pseudophryne guentheri	Günther's Toadlet
Ranidella glauerti	Glauert's Froglet
Ranidella insignifera	Squelching Froglet

Appendix 2

Summary of program MARK (White, 2001) output of model averaging of derived population estimates for summer trapping of common brushtail possums at Leschenault Peninsula Conservation Park and Yalgorup National Park.

Table 12: Model averaged derived population estimates using Huggins Closed Population Estimation and an adjusted $\hat{c}$ of 1.5 for summer trapping of common brushtail possums at Leschenault Peninsula Conservation Park and Yalgorup National Park.

Derived Parameter	Weighted Average Estimate	Weighted Average Standard Error	Unconditional Standard Error	95% CI for Weighted Average Estimate	Percent of Variation Attributable to Model Variation
1	22.48	0.91	1.03	20.46 - 24.50	22.12%
2	27.24	9.53	22.48	-16.82 - 71.31	82.03%
3	21.49	0.95	1.05	19.43 - 23.55	19.14%
4	34.32	5.19	5.87	22.82 - 45.82	21.87%
5	16.53	1.10	1.23	14.11 - 18.95	21.22%
6	14.69	1.07	1.49	11.77 - 17.60	48.18%

Appendix 3

Full program MARK (White, 2001) output of model averaging of derived population estimates for summer trapping of common brushtail possums at Leschenault Peninsula Conservation Park and Yalgorup National Park. The data type used was Huggins Closed Population Estimation with an adjusted $\hat{c}$ of 1.5.

Derived Parameter 1

Model	Weight	Estimate	Standard Error
{p(g)c(.) g1=g3, g2=g4, g5=g6}	0.13733	22.3188831	0.7633285
{p(v)c(v)}	0.12538	22.4605143	0.9167071
{p(g)c(g) g1=g3, g2=g4, g5=g6}	0.10507	22.3188831	0.7633288
{p(g+s)c(.) g1=g3, g2=g4, g5=g6}	0.09227	22.3617540	0.8262158
{p(v*b)c(.)}	0.08045	22.3188831	0.7633268
{p(v*b+s)c(.)}	0.06858	22.3918252	0.8699234
{p(v)c(.)}	0.05800	22.4605129	0.9167964
{p(v+s)c(v+s)}	0.05006	22.5311509	1.0074018
{p(g)c(.)}	0.04823	22.1466475	0.5208729
{p(g+s)c(g+s) g1=g3,g2=g4,g5=g6}	0.04135	22.3617548	0.8263021
{p(v+s)c(.)}	0.03462	22.5311517	1.0074026
{p(v*b)c(v*b)}	0.02628	22.3188831	0.7633267
{p(g+s)c(.)}	0.02571	22.1797471	0.5902319
{p(b)c(.)}	0.02366	23.6921912	1.9146271
{p(s)c(.)}	0.02105	23.4056206	1.6921276
{p(s)c(s)}	0.01650	23.4056188	1.6921269

{p(b+s)c(.)}	0.01380	23.8528778	2.0659646
{p(v*b+s)c(v*b+s)}	0.01345	22.3918260	0.8698964
{p(b)c(b)}	0.00923	23.6921897	1.9146255
{p(b+s)c(b+s)}	0.00401	23.8528785	2.0658857
{p(g)c(g)}	0.00375	22.1466475	0.5208734
{p(g+s)c(g+s)}	0.00121	22.1797484	0.5900750

Weighted Average		22.4751450	0.9079634
Unconditional SE			1.0288280

95% CI for Weighted Average Estimate is 20.4586421 to 24.4916478

Percent of Variation Attributable to Model Variation is 22.12%

Derived Parameter 2

Model	Weight	Estimate	Standard Error
{p(g)c(.) g1=g3, g2=g4, g5=g6}	0.13733	26.1859150	5.2523647
{p(v)c(v)}	0.12538	23.6287296	2.7713564
{p(g)c(g) g1=g3, g2=g4, g5=g6}	0.10507	26.1859053	5.2523591
{p(g+s)c(.) g1=g3, g2=g4, g5=g6}	0.09227	26.4687077	5.3864866
{p(v*b)c(.)}	0.08045	26.1859189	5.2523562
{p(v*b+s)c(.)}	0.06858	26.8025506	5.7184160
{p(v)c(.)}	0.05800	23.6287264	2.7711377
{p(v+s)c(v+s)}	0.05006	23.8157406	2.9432171
{p(g)c(.)}	0.04823	55.2596308	86.476529
{p(g+s)c(g+s) g1=g3,g2=g4,g5=g6}	0.04135	26.4687135	5.3865716
{p(v+s)c(.)}	0.03462	23.8157404	2.9432168
{p(v*b)c(v*b)}	0.02628	26.1859189	5.2523569
{p(g+s)c(.)}	0.02571	42.8858779	40.838373
{p(b)c(.)}	0.02366	22.6152735	1.8597634
{p(s)c(.)}	0.02105	22.3189994	1.6252815
{p(s)c(s)}	0.01650	22.3189977	1.6252808
{p(b+s)c(.)}	0.01380	22.7401912	1.9807827
{p(v*b+s)c(v*b+s)}	0.01345	26.8025643	5.7184326
{p(b)c(b)}	0.00923	22.6152720	1.8597619
{p(b+s)c(b+s)}	0.00401	22.7401918	1.9807112
{p(g)c(g)}	0.00375	55.2593188	86.475413
{p(g+s)c(g+s)}	0.00121	42.8858148	40.838086
Weighted Average		27.2413174	9.5316863
Unconditional SE			22.482208

95% CI for Weighted Average Estimate is -16.8238105 to 71.3064453

Percent of Variation Attributable to Model Variation is 82.03%

Derived Parameter 3

Model	Weight	Estimate	Standard Error
{p(g)c(.) g1=g3, g2=g4, g5=g6}	0.13733	21.3043884	0.7429393
{p(v)c(v)}	0.12538	21.4395818	0.8923516
{p(g)c(g) g1=g3, g2=g4, g5=g6}	0.10507	21.3043884	0.7429396
{p(g+s)c(.) g1=g3, g2=g4, g5=g6}	0.09227	21.3565668	0.8197085
{p(v*b)c(.)}	0.08045	21.3043884	0.7429377
{p(v*b+s)c(.)}	0.06858	21.3877518	0.8650879
{p(v)c(.)}	0.05800	21.4395805	0.8924351
{p(v+s)c(v+s)}	0.05006	21.5213758	0.9976445
{p(g)c(.)}	0.04823	21.5942467	1.2278812
{p(g+s)c(g+s) g1=g3,g2=g4,g5=g6}	0.04135	21.3565676	0.8197882
{p(v+s)c(.)}	0.03462	21.5213766	0.9976453
{p(v*b)c(v*b)}	0.02628	21.3043884	0.7429376
{p(g+s)c(.)}	0.02571	21.5983510	1.2276923
{p(b)c(.)}	0.02366	22.6152735	1.8597634
{p(s)c(.)}	0.02105	22.3745602	1.6746874
{p(s)c(s)}	0.01650	22.3745583	1.6746868
{p(b+s)c(.)}	0.01380	22.8097719	2.0412357
{p(v*b+s)c(v*b+s)}	0.01345	21.3877526	0.8650612
{p(b)c(b)}	0.00923	22.6152720	1.8597619
{p(b+s)c(b+s)}	0.00401	22.8097728	2.0411579
{p(g)c(g)}	0.00375	21.5942506	1.227884
{p(g+s)c(g+s)}	0.00121	21.5983493	1.2276917
Weighted Average		21.4944593	0.9452929
Unconditional SE			1.0512634

95% CI for Weighted Average Estimate is 19.4339830 to 23.5549356

Percent of Variation Attributable to Model Variation is 19.14%

Derived Parameter 4

Model	Weight	Estimate	Standard Error
{p(g)c(.) g1=g3, g2=g4, g5=g6}	0.13733	36.1615017	6.8925315
{p(v)c(v)}	0.12538	32.6301504	3.5089520
{p(g)c(g) g1=g3, g2=g4, g5=g6}	0.10507	36.1614883	6.8925242
{p(g+s)c(.) g1=g3, g2=g4, g5=g6}	0.09227	35.8053016	6.4318226
{p(v*b)c(.)}	0.08045	36.1615071	6.8925189
{p(v*b+s)c(.)}	0.06858	36.0807852	6.6921781
{p(v)c(.)}	0.05800	32.6301460	3.5086229
{p(v+s)c(v+s)}	0.05006	32.5052889	3.4249417
{p(g)c(.)}	0.04823	31.3608973	3.1028821
{p(g+s)c(g+s) g1=g3,g2=g4,g5=g6}	0.04135	35.8053075	6.4320048
{p(v+s)c(.)}	0.03462	32.5052878	3.4249410
{p(v*b)c(v*b)}	0.02628	36.1615071	6.8925199
{p(g+s)c(.)}	0.02571	31.6765112	3.4840043
{p(b)c(.)}	0.02366	31.2306157	2.2854324
{p(s)c(.)}	0.02105	30.6230439	1.8191921
{p(s)c(s)}	0.01650	30.6230421	1.8191915
{p(b+s)c(.)}	0.01380	31.1546185	2.2478726
{p(v*b+s)c(v*b+s)}	0.01345	36.0808002	6.6921958
{p(b)c(b)}	0.00923	31.2306137	2.2854303
{p(b+s)c(b+s)}	0.00401	31.1546184	2.2477867
{p(g)c(g)}	0.00375	31.3608953	3.1028829
{p(g+s)c(g+s)}	0.00121	31.6765072	3.4838078
Weighted Average		34.3204959	5.1868946
Unconditional SE			5.8682767

95% CI for Weighted Average Estimate is 22.8186736 to 45.8223183

Percent of Variation Attributable to Model Variation is 21.87%

Derived Parameter 5

Model	Weight	Estimate	Standard Error
{p(g)c(.) g1=g3, g2=g4, g5=g6}	0.13733	16.3149344	0.7796736
{p(v)c(v)}	0.12538	16.3349195	0.7644372
{p(g)c(g) g1=g3, g2=g4, g5=g6}	0.10507	16.3149334	0.7796730
{p(g+s)c(.) g1=g3, g2=g4, g5=g6}	0.09227	16.3756080	0.8759924
{p(v*b)c(.)}	0.08045	16.7832989	1.5664181
{p(v*b+s)c(.)}	0.06858	17.2241728	2.2795136
{p(v)c(.)}	0.05800	16.3349185	0.7644935
{p(v+s)c(v+s)}	0.05006	16.3752391	0.8217675
{p(g)c(.)}	0.04823	16.7832989	1.5664212
{p(g+s)c(g+s) g1=g3,g2=g4,g5=g6}	0.04135	16.3756091	0.8759914
{p(v+s)c(.)}	0.03462	16.3752397	0.8217682
{p(v*b)c(v*b)}	0.02628	16.7832989	1.5661790
{p(g+s)c(.)}	0.02571	17.0117650	1.9434453
{p(b)c(.)}	0.02366	16.3149344	0.7796738
{p(s)c(.)}	0.02105	16.9970146	1.3779582
{p(s)c(s)}	0.01650	16.9970134	1.3779576
{p(b+s)c(.)}	0.01380	16.3538179	0.8414940
{p(v*b+s)c(v*b+s)}	0.01345	17.2241706	2.2795155
{p(b)c(b)}	0.00923	16.3149324	0.7795166
{p(b+s)c(b+s)}	0.00401	16.3538190	0.8414965
{p(g)c(g)}	0.00375	16.7832978	1.5664213
{p(g+s)c(g+s)}	0.00121	17.0117642	1.9434467
Weighted Average		16.5257980	1.0961074
Unconditional SE			1.2349483

95% CI for Weighted Average Estimate is 14.1052994 to 18.9462966

Percent of Variation Attributable to Model Variation is 21.22%

Derived Parameter 6

Model	Weight	Estimate	Standard Error
{p(g)c(.) g1=g3, g2=g4, g5=g6}	0.13733	14.2755676	0.7197934
{p(v)c(v)}	0.12538	15.7524864	2.0974860
{p(g)c(g) g1=g3, g2=g4, g5=g6}	0.10507	14.2755667	0.7197927
{p(g+s)c(.) g1=g3, g2=g4, g5=g6}	0.09227	14.3863036	0.8948191
{p(v*b)c(.)}	0.08045	14.0676727	0.3468428
{p(v*b+s)c(.)}	0.06858	14.0804414	0.3826881
{p(v)c(.)}	0.05800	15.7524843	2.0973572
{p(v+s)c(v+s)}	0.05006	16.0917047	2.4209072
{p(g)c(.)}	0.04823	14.0676727	0.3468432
{p(g+s)c(g+s) g1=g3,g2=g4,g5=g6}	0.04135	14.3863048	0.8948181
{p(v+s)c(.)}	0.03462	16.0917050	2.4209073
{p(v*b)c(v*b)}	0.02628	14.0676727	0.3468429
{p(g+s)c(.)}	0.02571	14.0733589	0.3630096
{p(b)c(.)}	0.02366	14.2755676	0.7197935
{p(s)c(.)}	0.02105	14.9904545	1.3915239
{p(s)c(s)}	0.01650	14.9904531	1.3915233
{p(b+s)c(.)}	0.01380	14.3571284	0.8492340
{p(v*b+s)c(v*b+s)}	0.01345	14.0804414	0.3826885
{p(b)c(b)}	0.00923	14.2755658	0.7196629
{p(b+s)c(b+s)}	0.00401	14.3571297	0.8492367
{p(g)c(g)}	0.00375	14.0676736	0.3468451
{p(g+s)c(g+s)}	0.00121	14.0733595	0.3629731
Weighted Average		14.6888524	1.0709499
Unconditional SE			1.4877646

95% CI for Weighted Average Estimate is 11.7728337 to 17.6048711

Percent of Variation Attributable to Model Variation is 48.18%