Quantitative estimates of western ringtail possum (*Pseudocheirus occidentalis*) density and abundance at Karakamia Wildlife Sanctuary

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This thesis is presented as partial fulfilment of the requirements for the Degree of Bachelor of Science (Conservation Biology). I certify this thesis is my own work and that all material drawn from other sources has been fully acknowledged.

Signed Date

This work was conducted under animal ethics approval from The University of Western Australia

Cover Photograph: Western ringtail possum foraging in *Acacia saligna* at Karakamia Wildlife Sanctuary.

Photo taken by Wayne Lawler, Australian Wildlife Conservancy

Abstract

Distance sampling is an information-theoretic based approach, whereby model selection and use of the Akaike Information Criterion (AIC) is adopted to determine which model (or competing hypothesis) best describes the data. In this study, nocturnal spotlighting count data and distance sampling methods were used to estimate the abundance and density of a rare arboreal marsupial, the western ringtail possum (*Pseudocheirus occidentalis*), in two habitat types within Karakamia Wildlife Sanctuary, near Chidlow, Western Australia. Karakamia is a privately owned fenced sanctuary, which was the site of several *P. occidentalis* translocations over the period of 1995 to 2002. The vegetation over most of the sanctuary is open jarrah (*Eucalyptus marginata*) forest. However there is a limited area of riparian habitat, which was thought to be suitable for translocation of *P. occidentalis*. In the event that *P. occidentalis* had persisted at Karakamia, I predicted dispersal into the surrounding non-riparian habitat would have occurred. Also, on the basis of habitat structure and what is known of the biology of *P. occidentalis*, I predicted the current density would be greater in the riparian habitat than in the non-riparian forest habitat.

The model which best described the data resulted in population density estimates of 2.87 ha^{-1} and 2.02 ha^{-1} for the riparian and non-riparian habitat, respectively. Distance sampling also estimated a population size of 11 and 17 adult *P. occidentalis* for the riparian and non-riparian habitats, respectively. These findings indicated that the riparian habitat supported a higher density and was preferred by *P. occidentalis* at Karakamia. The findings also indicated that the *P. occidentalis* population has dispersed into the surrounding non-riparian habitats. This study provided the first quantitative estimates of *P. occidentalis* population size since the initial translocation to the sanctuary. Management decisions can now be based on reliable estimates with known confidence limits, however future monitoring of the Karakamia *P. occidentalis* population is required to detect population trends.

Table of Contents

List of Figures

List of tables

Acknowledgements

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Also, a big thank-you to Judy Clarke for helping me in the design of my survey, assisting in spotlighting possums and always providing me with helpful advice whenever I asked! Thanks must also be extended to Sean Garretson from the D.E.C. for helping me out with the pilot study, setting up my transects and collecting all of the equipment. I am also very grateful to all my spotlighters, Tim Walker, Terri Jones, Harriet Mills and of course Judy Clarke.

I would also like to express my gratitude to the Australian Wildlife Conservancy for providing me with the project, and a place to do it! Thank you to Manda Page and Jo Kuiper for all of your help and advice, particularly with putting the planks out to cross the creek!

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1. Introduction

Population estimates are crucial to develop conservation strategies for threatened wildlife species (Otis *et al.* 1978; Williams *et al.* 2002). However, information on population size is often lacking for such species. One example of a threatened species which requires further population estimates is the western ringtail possum (*Pseudocheirus occidentalis*, Thomas 1888). *Pseudocheirus occidentalis* is a small to medium size (700-1300 g) folivorous arboreal marsupial, endemic to south-west Western Australia (de Tores 2008). Internationally, *P. occidentalis* is listed on the IUCN Red List of Threatened Species as "Vulnerable" (IUCN 2009). *Pseudocheirus occidentalis* is also classified as vulnerable under the national *Environment Protection and Biodiversity Conservation Act 1999* and is listed as 'fauna that is rare or likely to become extinct' under the Western Australian *Wildlife Conservation Act 1950* (DEWHA 2009). Since European settlement, the species has seen a contraction in its range (Jones & Hilcox 1995; de Tores 2008). Once widely distributed throughout south-western Australia, populations of *P. occidentalis* are now patchy and restricted (Figure 1). *P. occidentalis* is threatened by introduced predators such as the fox *(Vulpes vulpes)* and cat *(Felis catus)*, as well as by habitat loss, high intensity fire and disease (de Tores *et al.* 2004; Wayne *et al.* 2005c).

Pseudocheirus occidentalis is most commonly recorded in coastal or near coastal habitat with peppermint trees (*Agonis flexuosa*) as a major component of the vegetation (Jones *et al.* 1994a; de Tores *et al.* 2004). They can give birth up to twice a year to one to three young, and have been recorded with a life span in excess of six years (Jones *et al.* 1994b; Wayne *et al.* 2005d; de Tores 2008). Population monitoring and improved understanding of the ecology and conservation status of *P. occidentalis* have been constrained by the relative difficulty of surveying the species in the wild (Wayne *et al.* 2005b; de Tores 2009). Conventional trapping techniques (ground and tree trapping) are ineffective for capture of *P. occidentalis* and usually result in no captures, even at sites of high density (de Tores *et al.* 2004). Therefore, estimating the density and abundance of the species requires a technique that does not involve conventional trapping (see Appendix 1).

The need for such specific *P. occidentalis* survey design requirements has seen *ad hoc* survey methods, such as nocturnal counts of animals and scat counts, become standard and accepted practice when estimating population size of *P. occidentalis* at sites proposed for clearing and development (de Tores & Elscot 2010). Such *ad hoc* survey methods do not provide quantitative estimates of *P. occidentalis* population size, and further do not provide

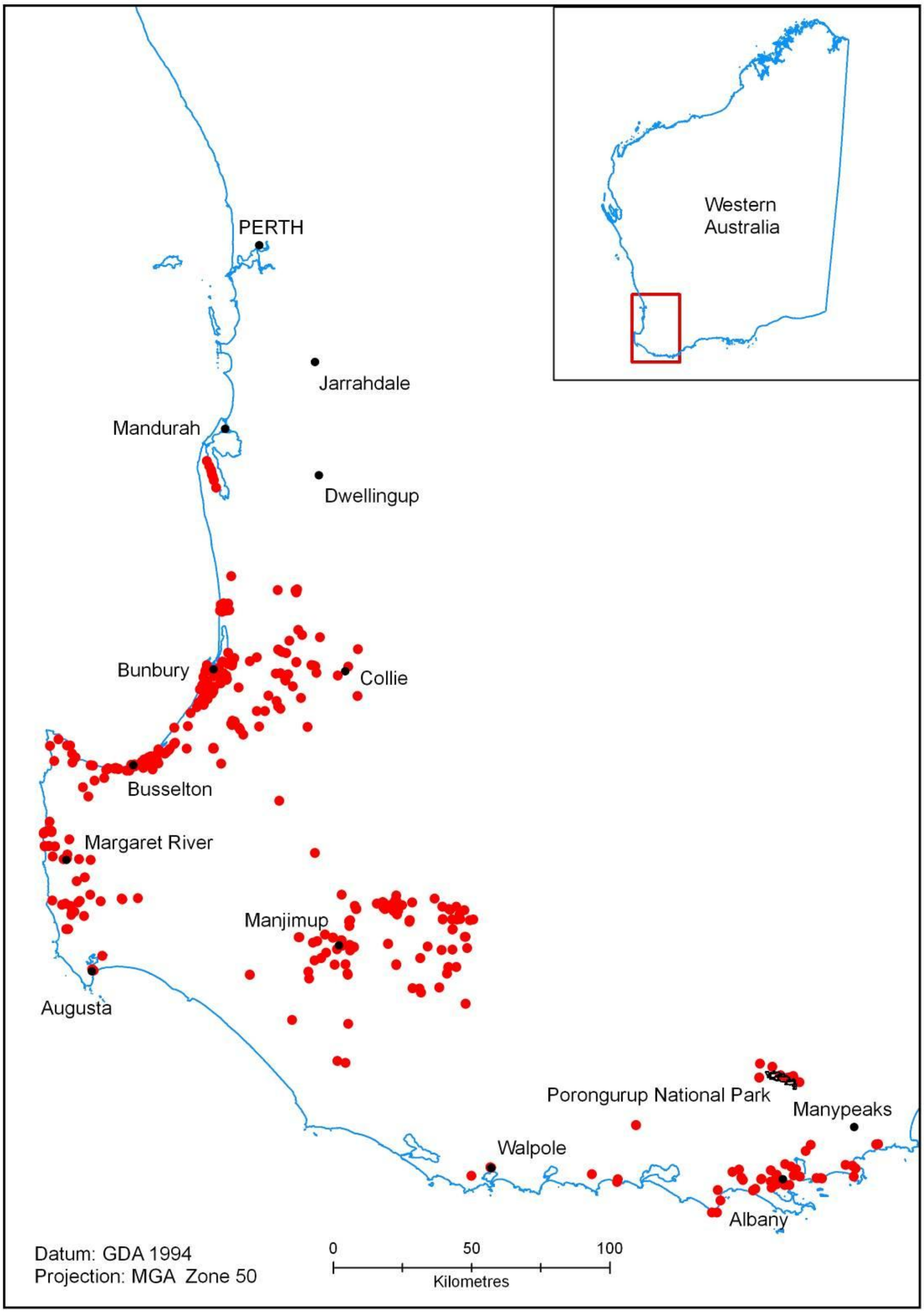

Figure 1: Known location records for the western ringtail possum (*Pseudocheirus occidentalis*) in the south-west of Western Australia.
Source: de Tores (2009)

managers with an indication of the level of uncertainty and/or variability associated with the estimates (de Tores 2009). Conversely, distance sampling has been shown to be an effective technique for estimating *P. occidentalis* population size, and has been particularly important in monitoring the success of several translocations of the species to date (de Tores *et al.* 2004; de Tores & Elscot 2010). Distance sampling is an approach used to estimate both abundance and density of wildlife populations (Buckland *et al.* 2001). Distance sampling is an information-theoretic based approach, whereby model selection and use of the Akaike Information Criterion (AIC) is adopted to determine which model (or competing hypothesis) best describes the data (Buckland *et al.* 2001; Thomas *et al.* 2010).

Since 1991, *P. occidentalis* have been translocated to several sites in south-west Western Australia (DEWHA 2010). The success of these reintroduction programs has relied heavily on the effective control of introduced predators. For example, translocation of *P. occidentalis* to Leschenault Peninsula Conservation Park, near Bunbury, Western Australia, was carried out in the presence of 1080 baiting for fox control (de Tores *et al.* 2004). The first translocation was in 1991, and by 1998 the population was thought to have become self-sustaining (de Tores *et al.* 2005a; 2005b; de Tores 2003). However a large decline in the population occurred between 1998 and 2002, with a drop in the number of sightings from 72 in 1996, 75 in 1997 and 102 in 1998, to two in 2002, for the same sampling effort. The cause of this decline appeared to be the result of a change in the fox baiting regimes at Leschenault Peninsula, where several months of baiting were missed between 1998 and 2002 (de Tores *et al.* 2004). In further attempts to re-establish viable populations of *P. occidentalis*, displaced possums have been translocated from development sites in the Busselton region to national parks further north (McCutcheon *et al.* 2007). However, the successes of such translocations are yet to be confirmed for many populations, including the population translocated to Karakamia Wildlife Sanctuary (de Tores *et al.* 2004, de Tores pers. comm.[1]), where the current study took place.

[1] Paul de Tores is a research scientist from the Department of Environment and Conservation, and is involved in *P. occidentalis* recovery and conservation.

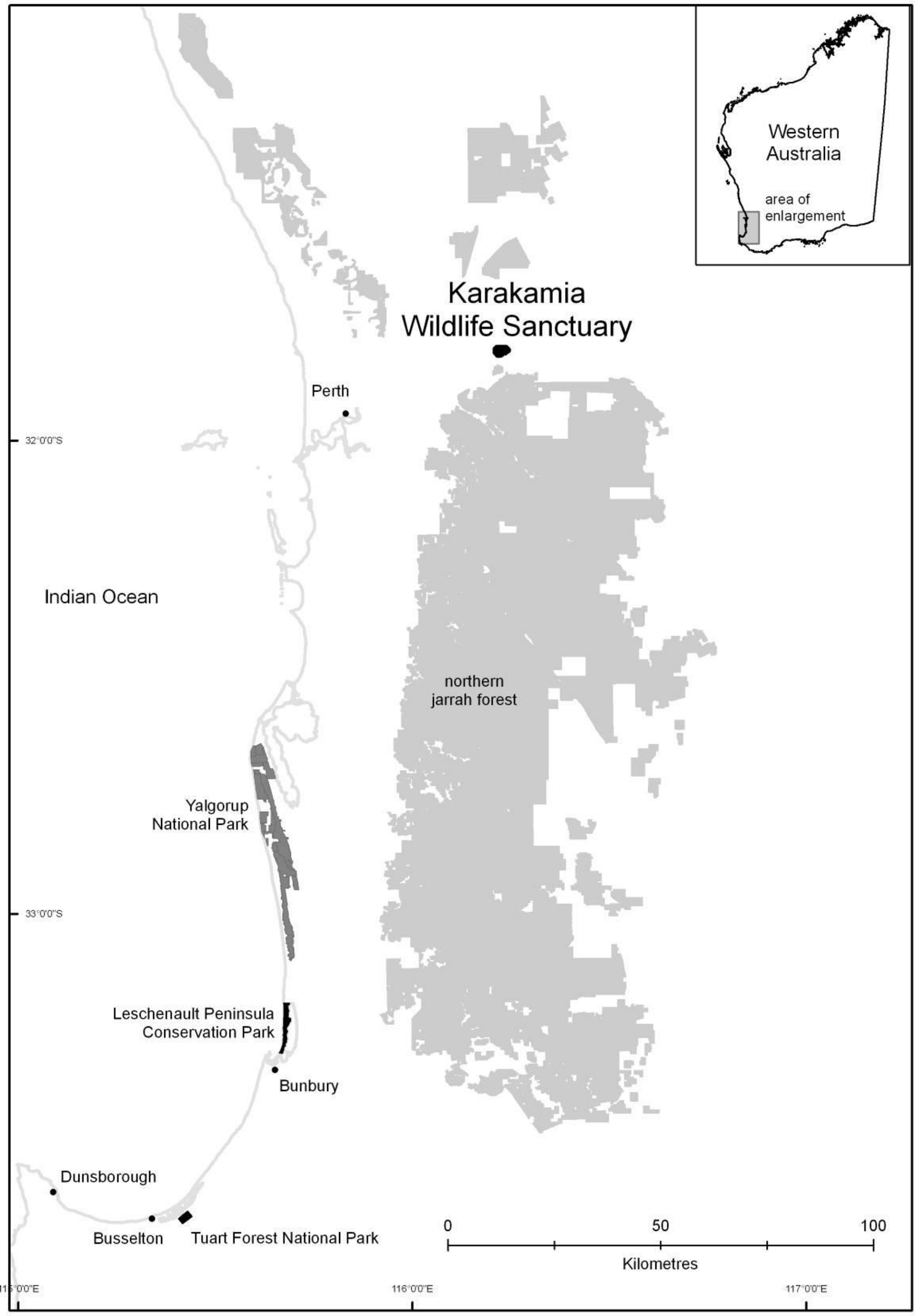

Figure 2: Location of Karakamia Wildlife Sanctuary and translocation release sites for the western ringtail possum, *Pseudocheirus occidentalis*, in south-west Western Australia.

Karakamia is a fenced, feral predator-free reserve at the northern end of the previous range of *P. occidentalis* (Figure 2). The site is unusual for the species in its present distribution, because it contains jarrah forest habitat without peppermint trees. However, *P. occidentalis* is known to occur at other sites in the absence of peppermint, such as the Perup population near Manjimup, and Karakamia is also within the species' former inferred geographic range (Jones *et al.* 1994a; Wayne *et al.* 2005a; see Appendix 1). Anecdotal evidence indicates that the sanctuary currently has an extant population of *P. occidentalis* (M. Page pers. comm.[2]). However, it is important to note that *P. occidentalis* has not been confirmed north of the Collie River and within the northern jarrah forest in surveys from the 1980s such as Dell and How (1988) or from more recent surveys. This lack of presence data of *P. occidentalis* populations within the northern jarrah forest, raises questions about the suitability of Karakamia for the species. This study was developed to confirm whether a population is still extant at Karakamia and, if so, derive a reliable estimate of population size and a reliable method for future monitoring.

Forty-two *P. occidentalis* were translocated to Karakamia between 1995 and 2002 (AWC 2005; 2006).The release site at Karakamia was within the limited riparian vegetation (Appendix 1) (Figure 2). This site was chosen because the riparian habitat is similar to the wetter areas associated with water courses and swamps preferred by *P. occidentalis* in other areas of the species distribution (Jones *et al.* 1994a; de Tores *et al.* 2004). The riparian habitat also has greater canopy connectivity, another habitat preference of *P. occidentalis* (Jones *et al.* 1994a). In the years since the translocations, anecdotal evidence has suggested that the *P. occidentalis* population has remained concentrated around the limited riparian habitat at Karakamia (AWC 2005).

Therefore, I predicted that there would be a greater density of *P. occidentalis* in the riparian habitat than in the jarrah forest and wandoo/marri woodland habitat at Karakamia Wildlife Sanctuary. I tested this prediction by using distance sampling to derive robust, quantitative estimates of population size (abundance and density) in the three available habitats at Karakamia—riparian, jarrah forest and wandoo/marri woodland.

[2] Manda Page is the South West Regional Ecologist for Australian Wildlife Conservancy, based at Karakamia Wildlife Sanctuary

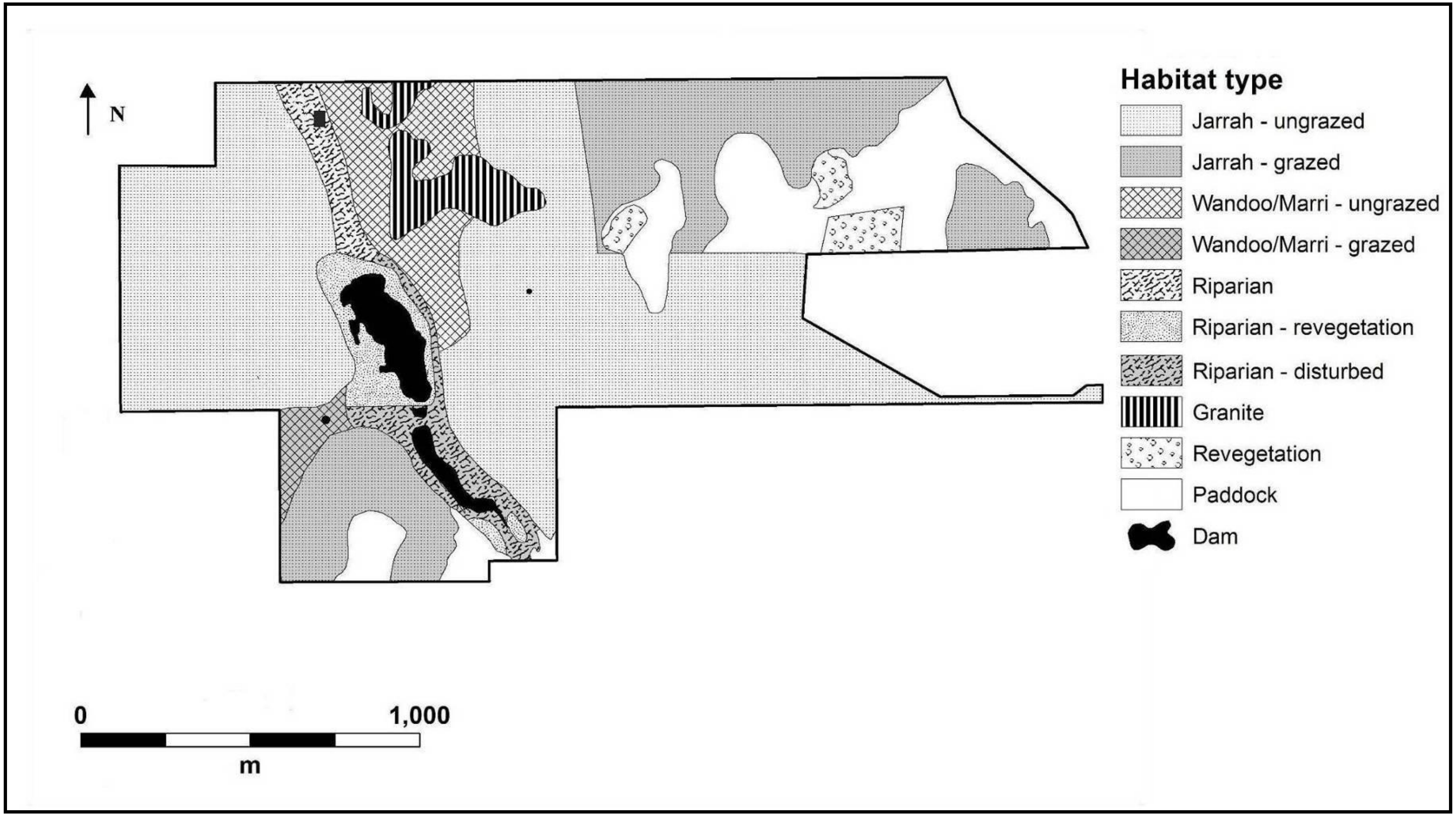

Figure 3: Map of Karakamia Wildlife Sanctuary showing the broad habitat types. The *P. occidentalis* release site in the riparian habitat is indicated by ▪

(Source: Australian Wildlife Conservancy).

2. Materials and Methods

2.1 Choice of technique for sampling population size and density

Acquiring a population estimate of a threatened wildlife species usually involves two parameters, abundance (absolute number of individuals) and density (number of individuals per unit area) (Buckland *et al.* 2000). The best density estimates are those that are both precise and unbiased (Thompson *et al.* 1998). An estimate which lacks precision, exhibited as high sampling variance, standard error and coefficients of variation, will have less power (Anderson 2003). Furthermore, *ad hoc* unquantified methodologies, such as raw counts, fail to account for differences in detection probability and can therefore result in biased estimates and misleading results (Buckland *et al.* 2001). Consequently, choosing an appropriate method for estimating density is imperative to reliably assess the population size of threatened species.

Distance sampling provides a viable technique to determine measures of variance of wildlife abundance or density estimates, and adopts a model selection and inference approach (Burnham & Anderson 1998; Buckland *et al.* 2001; Koenen *et al.* 2002). Distance sampling and the use of the software Distance (Thomas *et al.* 2010) is now widely accepted as a robust and repeatable method to estimate population density (Kacoliris *et al.* 2009; Thomas *et al.* 2010; Young *et al.* 2010).

Distance sampling comprises of a set of techniques based on sampling distances from a line or point, from which animal abundance is estimated (Buckland *et al.* 2000; Buckland *et al.* 2009). In line-transect distance sampling, observers traverse a series of transects and record the perpendicular distance to detected animals (Buckland *et al.* 2001; Young *et al.* 2010). This procedure enables the investigator to account for individuals present but not detected in the survey, provided that they are potentially observable during the survey (Ekblom 2010). The probability of detecting an animal is then modelled as a function of the observed perpendicular distances (Buckland *et al.* 2001; Young *et al.* 2010). This detection probability is then combined with the estimated encounter rate (the number of observations per unit effort), to calculate the abundance of individuals in the study area (Buckland *et al.* 2001). Density is also calculated if the total area of the study region is known. Thus, distance sampling incorporates the probability of detection to enable the density and abundance of an animal population to be estimated.

When there are too few observations at each site (strata) to fit a separate detection function, distance sampling also allows pooling of the observation data and density estimation for those strata (Thomas *et al.* 2010). Such stratification can also be used to account for differences associated with varying habitat types, changes between years, differences between species etc. (Buckland *et al.* 2001). Detection heterogeneity, such as from the observer or climatic differences, can also be accounted for through the use of covariates (Buckland *et al.* 2001). Moreover, distance sampling provides estimates of variance and confidence intervals for the derived estimates of abundance and density (Buckland *et al.* 2001).

There are a number of assumptions associated with distance sampling that must be met by the survey design, to produce a reliable and unbiased density estimate (Buckland *et al.* 2001). The three key assumptions of distance sampling are: (1) all individuals on the transect line are detected with certainty; (2) animals are detected at their original location; and (3) measurements are exact (Thomas *et al.* 2010). In the case of line-transect sampling, there is also the assumption that transects are independent and placed randomly with respect to animal distribution. This assumption of independence is ensured when there are adequate transects with randomized locations (Thomas *et al.* 2010). When one or more of the assumptions of distance sampling are violated, there is the possibility abundance and density estimates can be under- or over-estimated.

Data analysis in Distance is carried out in an information-theoretic framework, whereby *a priori* candidate models are compared (Thomas *et al.* 2010). Information-theoretic methods offer a more useful, general approach in the analysis of empirical data, rather than the testing of a null hypothesis (Anderson *et al.* 2000). As a result, the information-theoretic paradigm avoids statistical null hypothesis testing concepts, such as an arbitrary α-level, and focuses on the relationships of variables (via model testing) and on the estimation of effect size and measure of its precision (Anderson *et al.* 2000).

Information-theoretic methods allow selection of a “best” model and a ranking and weighting of the remaining models from the pre-defined set (Burnham & Anderson 2002). Formal inference can then be based on more than one of these models (multi-model inference). In this sense, there is no acceptance of a simple “true model” for the data of biological sciences. Rather, modelling is viewed as an exercise used to approximate the explainable information in the empirical data, following the context of the data being from some well-defined population or process. As a result, the science of the problem has to be brought into the modelling before one begins to search through the data (Burnham & Anderson 2002).

2.2 Survey techniques for Pseudocheirus occidentalis

Ad hoc survey methods have become standard and accepted practice when estimating population size of *P. occidentalis* (de Tores & Elscot 2010; Finlayson *et al.* 2010). Examples of such *ad hoc* methods include nocturnal and/or diurnal counts of animals, counts of dreys (nests built by *P. occidentalis*) or scat counts (de Tores 2009). However, as *ad hoc* methods do not account for variation in detection probability, and are not validated against more accepted robust techniques, they do not provide the best estimator of density and/or abundance, or provide data from which inference can be made (Burnham & Anderson 2002). In contrast, quantitative and robust estimates of *P. occidentalis* population size have been derived through use of distance sampling methods (Buckland *et al.*, 2001; 2004) at Leschenault Peninsula Conservation Park and Yalgorup National Park (Clarke, 2010; de Tores *et al.*, 2004), Locke Nature Reserve, Busselton (de Tores and Elscot, 2010) and private property within Busselton (de Tores and Elscot, 2010; Finlayson *et al.*, 2010). Therefore, it has been recommended that distance sampling protocols should be routinely applied to estimate *P. occidentalis* density to ensure decision makers and mangers are provided with reliable information on which to base management decisions (de Tores & Elscot 2010).

2.3 Study area

The study site was established at Karakamia Wildlife Sanctuary, near Chidlow in Western Australia (31^0 49’ 30” S and 116^0 14’ 30” E) (Figure 2). The sanctuary is owned by the Australian Wildlife Conservancy (AWC) and consists of 260 ha of mostly ungrazed Jarrah (*Eucalyptus marginata*) forest habitat surrounded by a predator-proof fence. In the past, sections of the sanctuary had been cleared for farming activities, such as cattle grazing. Initially 75ha in size, the sanctuary was expanded to 260ha as several adjacent private blocks of land were acquired (Richards *et al.* 2009). All non-native mammals, such a foxes and cats have been removed (Richards *et al.* 2009). Within the sanctuary there is a variety of habitat types, including jarrah forest, wandoo woodland, riparian vegetation, as well as open paddocks, granite outcrops, a dam and a creek.

Since 1995 several native mammal species have been released into the sanctuary as part of a multi-species translocation (AWC 2006; Richards *et al.* 2009). These species included woylies (*Bettongia penicillata*), tammar wallabies (*Macropus eugenii*), numbats

(*Myrmecobius fasciatus*), quendas (*Isoodon obesulus*), quokkas (*Setonix brachyurus*) as well as western ringtail possums (*Pseudocheirus occidentalis).* Forty-two *P. occidentalis,* in total, were released into Karakamia between 1995 and 2002 (AWC 2005; 2006).

Karakamia contains all the key vegetation associations found within the northern jarrah forest complex (Figure 3). The sanctuary is dominated by open jarrah (*Eucalyptus marginata*) forest, which occurs predominantly on the lateritic soil, marri (*Corymbia calophylla*) woodland on the slopes, and wandoo (*Eucalyptus wandoo*) woodland on clays weathered from exposed granite. Vegetation on the granite outcrops ranges from lichen and moss to herbfields such as *Borya sphaerocephala* and shrublands dominated by *Hakea undulata*, *Hakea elliptica*, *Calytrix depressa* and *Grevillea bipinnatifida*. Lower down the slopes stands of flooded gums (*Eucalyptus rudis*) make up the riparian community along the stream zone. This riparian area is also dominated by *Trymalium floribundum, Astartea fascicularis and Baumea spp*.

Prior to carrying out the study, individual *P. occidentalis* were infrequently reported within Karakamia, usually as a result of opportunistic sightings or sightings from ecotourism spotlighting events (AWC 2005; 2006). These animals were presumed to be individuals persisting from the original translocations and/or recruits to the translocated population (M. Page pers. comm.*).* A pilot survey was conducted to positively confirm presence and to identify areas for follow-up survey, if presence was confirmed. Presence of *P. occidentalis* was confirmed and most sightings were within a small area of riparian habitat and abutting areas of wandoo and jarrah forest. This riparian habitat was also the release site for all *P.occidentalis* translocations (M. Page pers. comm.*)*. The common brushtail possum (*Trichosurus vulpecula*) was also confirmed present in the same habitats as *P. occidentalis*. I therefore anticipated encountering *T. vulpecula* in my survey. The riparian habitat and abutting jarrah forest to the west and wandoo woodland to the east were targeted for the follow-up survey. The study site within Karakamia encompassed 12.3ha of the sanctuary, 3.81ha of riparian habitat and 8.49ha of non-riparian habitat (4.97ha jarrah forest and 3.52ha wandoo woodland) (Figure 4).

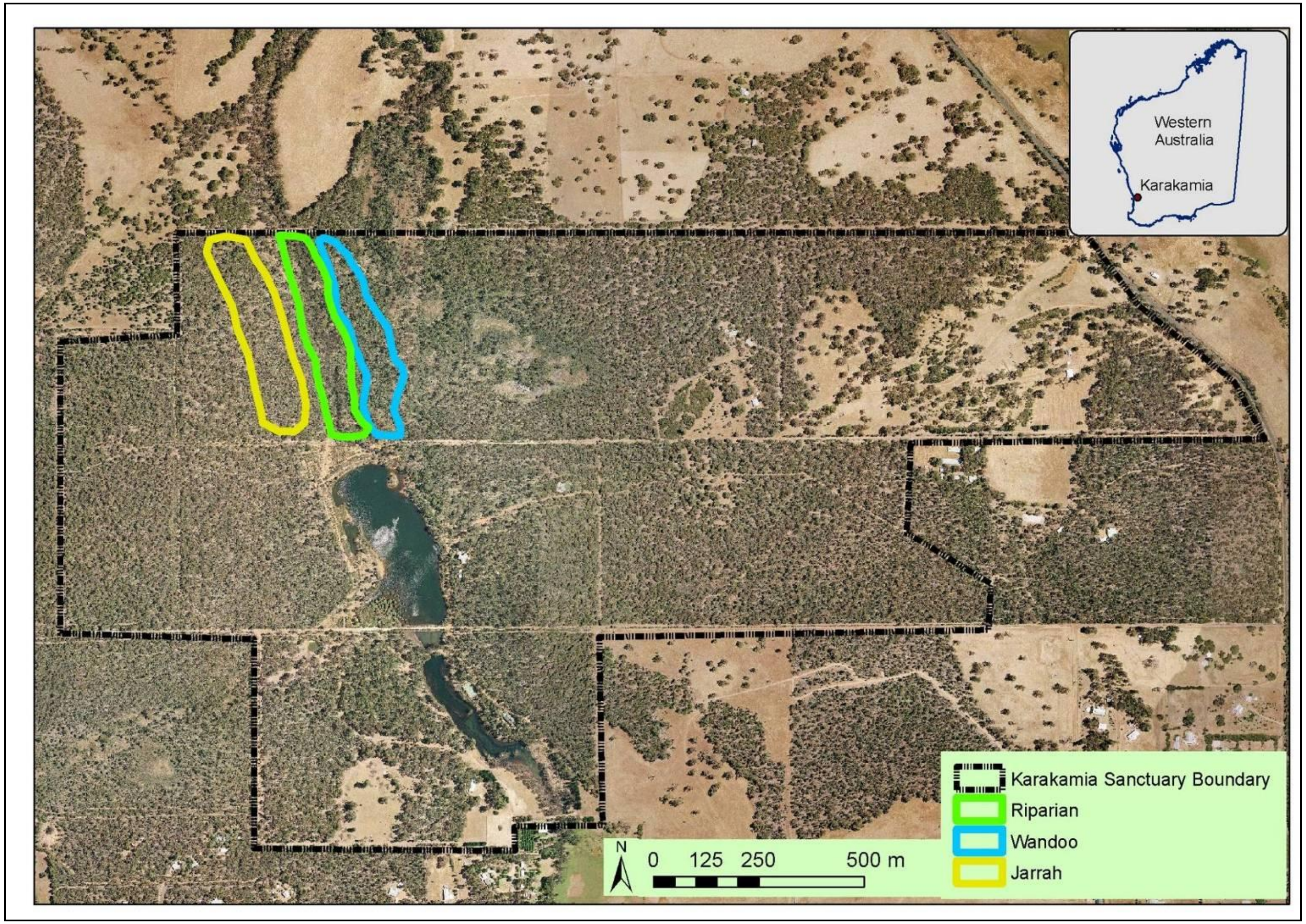

Figure 4: Orthophotograph of the 12.3ha area within Karakamia Wildlife Sanctuary surveyed to estimate the density and population size of *Pseudocheirus occidentalis.*

Each habitat zone is outlined (green: riparian, blue: wandoo, yellow: jarrah). Orthophotograph sourced from Australian Wildlife Sanctuary.

2.4 Survey Design

On the basis of the results of the pilot study, I established a series of transects within each of the riparian, wandoo and jarrah habitat zones. A total of 15 transects were established with an average length of 72.9m (Figure 5). Five transects were placed systematically parallel in each of the three habitat types (riparian, wandoo, jarrah), to provide complete coverage of the study area. The total study area was generally rectangular in shape, but not perfectly symmetrical as transect lines were of unequal length. I treated the jarrah and wandoo habitat types as one unit or zone, termed non-riparian. I compared the derived population estimates for *P. occidentalis* and *T. vulpecula* from this non-riparian zone with the estimates derived from the riparian zone (Figure 5).

Transects were spaced 90m apart. This spacing allowed a 45m distance for detection and therefore minimised the chances of double counting sightings, which would occur if animals were displaced by observers from one transect to the next. This resulted in a very good coverage of the survey area. Each transect line was ‘surveyed in’ by compass such that lines were parallel. Furthermore, the transects were set out in an east-west direction cutting across a potential density gradient in the *P. occidentalis* population.

Jarrah stakes (1.8m) were placed along each transect, with stakes spaced at approximately 10-15m. A band of reflective tape was placed around the top of each stake to enable easy nocturnal navigation between stakes (Figure 6). This was also to ensure the line was identifiable and effectively surveyed to meet the assumption of perfect detectability on the transect line. The location of the start and endpoint of each transect was accurately determined though the use of a differential (post processed) Global Positioning System receiver (GPS) (Thales MoblieMapper™). The location of each end point was only accepted as sufficiently accurate if the post-processed horizontal error was less than 1.0m (de Tores & Elscot 2010).

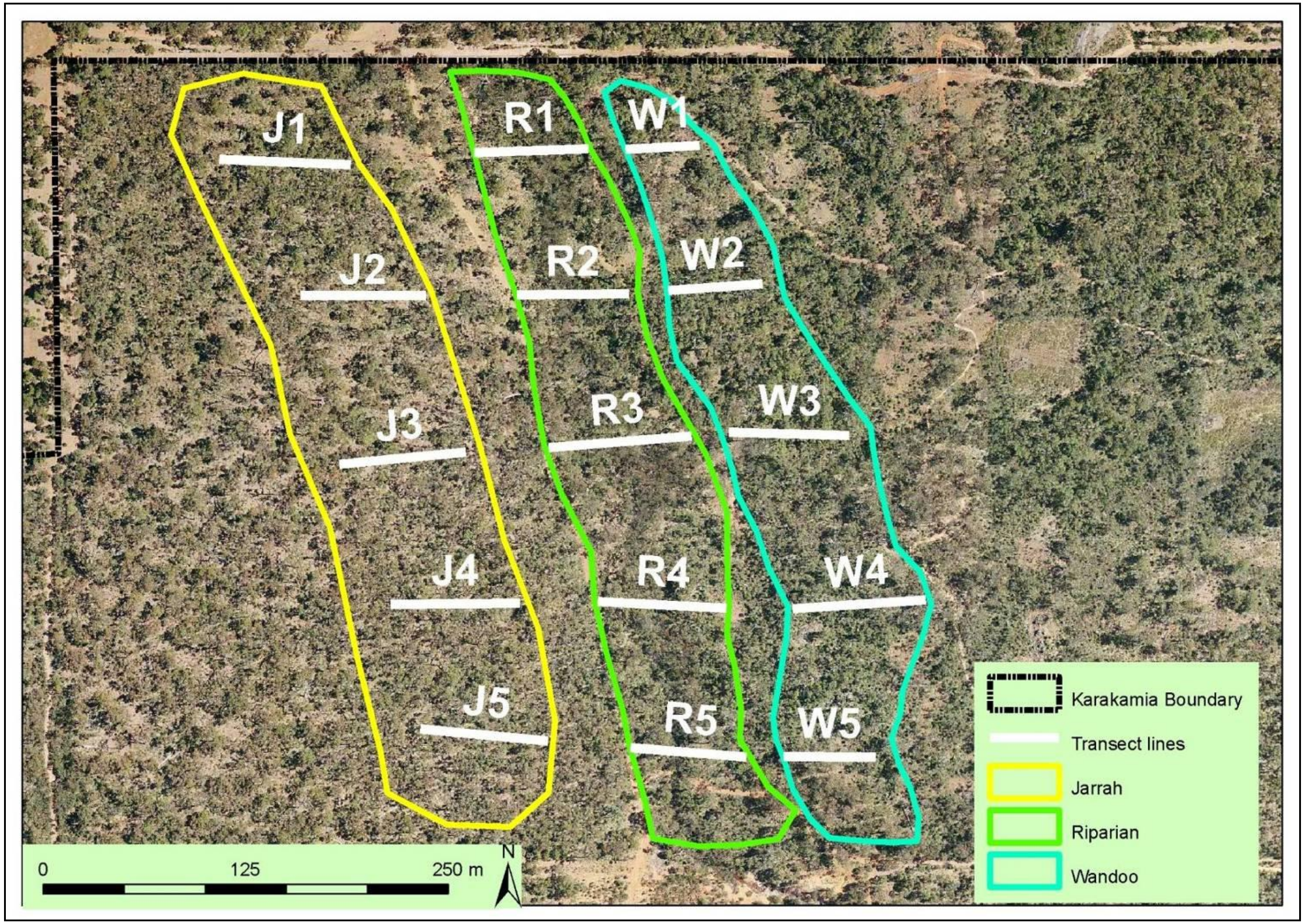

Figure 5: Orthophotograph of the study area within Karakamia Wildlife Sanctuary showing the location of the transect lines with labels used for identification.

Each habitat zone is outlined (green: riparian, blue: wandoo, yellow: jarrah). Orthophotograph sourced from Australian Wildlife Sanctuary.

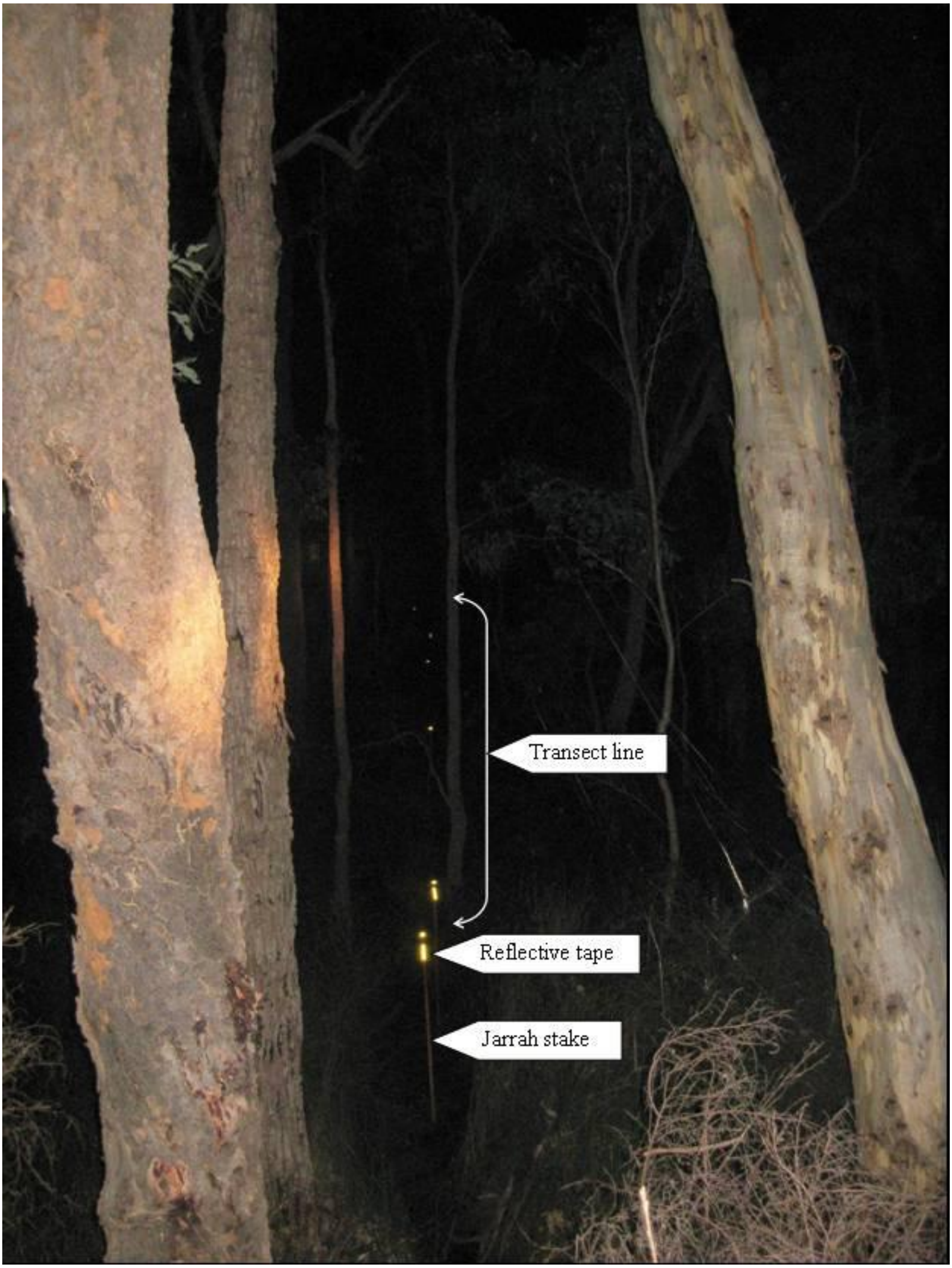

Figure 6: Example of nocturnal transect survey showing the visibility of the jarrah stakes along the transect line.

2.5 Site survey

The site survey involved using line transect methodology with nocturnal spotlighting. Distance sampling was conducted over 18 nights between the 20th of May and 2nd of August 2010. During each night of surveying, only one habitat section was completed (5 transects). The 15 line transects measured 1.09 km in total length. Buckland *et al.* (2001) suggested that a sample size of 60-80 sightings would be adequate to reliably estimate the average density within a study area. Therefore, the 15 transects in this study were repeatedly sampled until the minimum of 70 observations was achieved. As a result, each transect was surveyed 6 times, but only once on any given night. The order in which the transects were walked in was alternated (1 to 5 or 5 to 1) on each repeated night the habitat was surveyed to avoid bias (Figure 5). Transects in the non-riparian habitats were always walked in the same direction (wandoo: west to east, jarrah: east to west), while in the riparian the order alternated. Spotlighting surveys were conducted on all but one rain-free nights.

Spotlighting was carried out by a team of two spotlighters, each equipped with a Black Diamond ™ head torch, with a halogen globe, and a purpose built spotlighting system comprised of a rechargeable 12V battery pack and hand held focusable Lightforce™ spotlight, with a 30W halogen globe. I was present for every spotlighting survey, accompanied by one of four other spotlighters (J. Clarke, T. Walker, H. Mills, and T. Jones). *P. occidentalis* and the sympatric *T. vulpecula* were detected primarily by eyeshine (*i.e.* reflected light from the head torch or hand held spotlight). Individual possums were also located through detection of movement, sound or silhouette. If a possum was moving, the distance was measured to the point at which it was first seen. Once sighted, each animal was then approached and the location of sighting accurately recorded using a Thales Mobile Mapper™ GPS receiver. GPS data were logged for 5 minutes at each sighting point.

In addition to logging the GPS coordinates, each location was flagged using flagging tape and labelled. Locations were differentially corrected through post processing and only considered sufficiently accurate if the post processed horizontal error was less than 2m (de Tores & Elscot 2010). Where the horizontal error was 2m or more, the flagged position of the sighting was re-surveyed on another occasion. Ancillary data were also recorded (time, possum species, height of possum, observer, height of tree, side of transect, tree species, canopy connectivity, presence of ear tags). The phase of the moon and weather conditions were also noted for each transect. Rainfall data were obtained from the Bureau of Meteorology website (Bureau of Meteorology 2010) for the nearby weather station at Chidlow. Temperature data were recorded by two Hobo data loggers (using software

HobowareLite® Version 3.0 2002-2010) at the study site. One logger was placed in the riparian and one in the jarrah forest within the study area. The daily maximum and overnight minimum temperatures were then averaged from these two loggers to obtain a temperature representative of both habitat types.

Each transect was mapped in a Geographic Information System (GIS), ArcMap© 9.3 (ESRI 2004-2008) with the start and end points connected through use of the extension HawthsTools (Beyer, 2002-2006). Each sighting location was also mapped and the perpendicular distance from the transect to each sighting location determined in the GIS, using the extension ET GeoWizards (ET Spatial Techniques). These perpendicular distances were tabulated in MS Excel© 2007, along with ancillary information and then transferred into MS Access© 2007.

2.6 Hypotheses, models and model selection

I hypothesised detection and abundance of each possum species had the potential to be influenced by a range of variables. Abundance of both *P. occidentalis* and *T. vulpecula* are potentially a function of the habitat type. Detection was also potentially a function of one or more of a combination of the variables for the spotlighters, the overnight minimum temperature, the daily maximum temperature, the height of each sighting, the phase of the moon or the amount of daily rainfall. Furthermore, I hypothesised that there would be an effect of sighting height on detection. I expected there to be a higher probability of detection for animals at eye height when on the transect. The explanatory variables (candidate models) examined and the justification for each is given in Table 1. Models (or combinations of variables) were only included in the candidate model set if they were considered biologically plausible.

Candidate models included three different model sets. Model sets differed only in the choice of the key function, which was either half normal or hazard rate and in use of either a global detection function or a species specific detection function. The exploratory analysis (see below and results) supported the use of the half normal key, a global detection function and truncation at 50m. Therefore the candidate model set shown lists the model set using the half normal key and a global detection function (Table 1).

The model selection was based on Akaike's Information Criterion (AIC). A model with a lower AIC value indicates that it is better able to describe the data (Burnham & Anderson 2002). In my analysis, the second order of AIC, AICc, was used due to my small sample size. Therefore, AICc was used to choose between competing models. Models with a ΔAICc (difference in AICc) of ≤2 of the 'best' model are considered to have substantial support and should receive consideration in making inferences. Models with a ΔAICc within 4-7 of the best model have considerably less support, and models with a ΔAICc of 10 or more have essentially no support (Burnham & Anderson 2002).

Use of AICc was subject to each model within the candidate model set having an acceptable fit. The fit of the model was determined by examining quantile-quantile (q-q) plots, χ^2, Kolmogorov-Smirnov and Cramér-von Mises goodness of fit (GOF) tests. The q-q plot graphs the empirical distribution function (edf) for the fitted model against the cumulative distribution function (cdf) and allows for the plotting of ungrouped data. This provides an alternative to visually examine the fit of the data (Burnham & Anderson 2002). In a q-q plot, a good fit is revealed where the plotted cdf points fall on, or close to, the 1-1 line of the graph. As a result, q-q plots can assess how good a fit is for points close to the line, where the model fit is most important.

In contrast, the χ^2 GOF test places no additional importance for the model fit for points close to the line. Instead, the χ^2 GOF test subjectively determines the choice of intervals used for ungrouped data. Another GOF test, the Kolmogorov-Smirnov test, assesses the model fit based on the largest difference between the cdf and edf, and can only be used for ungrouped data. The last of the GOF tests, the Cramér-von Mises test, is more powerful than the Kolmogorov-Smirnov test and is based on the difference between the cdf and edf functions over their entire range (Buckland *et al.* 2004). The Cramér-von Mises test can further have uniform weighting, whereby greater weight is given to the tail of the detection function. In Distance, the Cramér-von Mises test can also provide a test statistic from weighting close to the line (Buckland *et al.* 2004).

Table 1: The *a priori* candidate models compared to determine which model(s) best described the data used to estimate population abundance and density for two species of arboreal marsupial, the western ringtail possum (*Pseudocheirus occidentalis*) and the common brushtail possum (*Trichosurus vulpecula*) within different habitat at Karakamia Wildlife Sanctuary, south-west Western Australia.

Model number	**Syntax**	**Description**	**Justification or rationale for inclusion**
Model 1	HN global strat	The Half Normal (HN) key function without adjustment terms, with a global detection function, stratified by habitat and possum species	I hypothesised detection and abundance would be a function of habitat only and further hypothesised P. occidentalis density would be higher in the riparian habitat and T. vulpecula density higher in the non-riparian habitat.
Model 2	HN global strat dailymax	The HN key function without adjustment terms, with a global detection function, and stratified by habitat and possum species. Also with an effect from the covariate for maximum daily temperature included.	As with Model 1, with an additional effect on detectability as a response to higher maximum daily temperature. The daily maximum temperature could affect P. occidentalis detectability as the species has been known to show signs of distress at high ambient temperatures (de Tores 2009), wildlife carers report a higher incidence of P. occidentalis being brought into care when ambient temperatures are high (de Tores 2009) and evaporative water loss is high at temperatures above 32.5^{0}C (Yin 2006).
Model 3	HN global strat overnightmin	The HN key function without adjustment terms, with a global detection function, and stratified by habitat and possum species. Also with an effect from the covariate for overnight minimum temperature included.	As with Model 1, with an additional effect on detectability as a response to lower overnight minimum temperature. The overnight minimum temperature could affect P. occidentalis detectability as a reduced number of sightings have been recorded on colder nights (Wayne *et al.* 2005a).

Table 1 (… cont.): The *a priori* candidate models compared to determine which model(s) best described the data used to estimate population abundance and density for two species of arboreal marsupial, the western ringtail possum (*Pseudocheirus occidentalis*) and the common brushtail possum (*Trichosurus vulpecula*) within different habitat at Karakamia Wildlife Sanctuary, south-west Western Australia.

Model 4	HN global strat spotlighter	The HN key function without adjustment terms, with a global detection function, and stratified by habitat and possum species. Also with an effect from the covariate for spotlighter included.	As with Model 1, with an additional effect on detectability as a result of each different spotlighter. I hypothesised that the spotlighter had the ability to influence detectability, as there may be skill differences between them (Wayne et al. 2005b). Furthermore, observer differences have been found when spotlighting P. occidentalis (Wayne *et al.* 2005b), and squirrel gliders (Goldingay & Sharpe 2004).
Model 5	HN global strat height	The HN key function without adjustment terms, with a global detection function, and stratified by habitat and possum species. Also with an effect from the covariate for height of the possum sighting included.	As with Model 1, with an additional effect on detectability as a result of the height of each sighting. I expected there to be a higher probability of detection for possums at eye height when on the transect. This is because at this height, possums would be easier to detect.
Model 6	HN global strat moon	The HN key function without adjustment terms, with a global detection function, and stratified by habitat and possum species. Also with an effect from the covariate for moon phase included.	As with Model 1, with an additional effect on detectability as a response to the phase of the moon. I hypothesised that fewer P. occidentalis would be detected on clear moonlit nights, near a full moon. When the phase of the moon is full, or near full, the extra light emitted could result in greater P. occidentalis detections. This may be due to a change in behaviour to avoid predators (Laurance 1990), or the effect of the extra light on detectability (Wayne *et al*. 2005a).

Table 1 (… cont.): The *a priori* candidate models compared to determine which model(s) best described the data used to estimate population abundance and density for two species of arboreal marsupial, the western ringtail possum (*Pseudocheirus occidentalis*) and the common brushtail possum (*Trichosurus vulpecula*) within different habitat at Karakamia Wildlife Sanctuary, south-west Western Australia.

Model 7	HN global strat rainfall	The HN key function without adjustment terms, with a global detection function, and stratified by habitat and possum species. Also with an effect from the covariate for daily rainfall included.	As with Model 1, with an additional effect on detectability as a response to the amount of rainfall during the day of survey. The rainfall could affect P. occidentalis detectability as fewer sightings have been recorded on nights following rainy days (Wayne *et al.* 2005a)
Model 8	HN global strat dailymax × overnightmin	The HN key function without adjustment terms, with a global detection function, and stratified by habitat and possum species. Also with an effect from the covariates for maximum daily temperature and overnight minimum temperature included.	As with Model 1, with an additional effect on detectability as a response to the combined effect of higher maximum temperature and lower minimum overnight temperature. The combination of these two variables could influence P. occidentalis detectability as greater distress caused by higher temperatures during the day combined with colder overnight minimum temperatures could result in less activity during spotlighting nights.

In conjunction with AIC ranking, the likelihood of the model, given the data, can also be used to make inferences concerning the relative strength of evidence for each of the models in the set (Burnham & Anderson 2002). The likelihood of model g_i, given the data, for each model in the set is calculated using:

$$\mathcal{L}(g_i|x) \propto \exp\left(-\frac{1}{2}\Delta_i\right)$$

where $L(g_i \,|\, x)$ is the likelihood of the model given the model (g) and the data (x) (Burnham & Anderson 2002). The log likelihood (log L) of this function is used to derive AIC values and compare the relative strengths of each model. The log likelihood function can also be used to derive a model weight (W_i).

Each model will also calculate an effective strip width (ESW) which is used to estimate abundance and density. The ESW is calculated so that the number of animals detected outside the ESW exactly equals the number of animals missed inside the ESW (Buckland *et al.* 2001).

The default variance estimator in Distance does not perform well if there is a trend in the population density across or within the survey and/or when the survey design is systematic (Fewster *et al.* 2009). As my survey design was systematic and there was the potential for a population density trend, I used the S2 variance estimator described by Fewster *et al.* (2009). This variance estimator pools the stratum specific variance estimates and weights the estimate by the total line length per stratum.

2.7 Estimates of abundance and population density

2.7.1 Data Structure

Survey data were imported into Distance in a layered structure with four main levels: 1) global layer (study area at Karakamia), 2) stratum layer (habitat type), 3) transect layer (repeat nights on five transects in each habitat) and 4) observations layer (perpendicular distance of individual sightings from the transect). The stratum layer for habitat type was stratified into riparian and non-riparian to allow for comparison of *P. occidentalis* density and abundance in the riparian habitat and jarrah and wandoo combined. The data were further stratified by possum species to enable a density and abundance estimate of each possum species for the riparian and non-riparian habitats.

2.7.2 Exploratory Analysis

Estimates of abundance and density were derived in Distance using the Multiple Covariate Distance Sampling (MCDS) engine. Exploratory analysis was carried out to investigate characteristics of data structure. The species data were pooled (i.e. total number of *P. occidentalis* and *T. vulpecula* sightings) due to the small sample size. The exploratory analysis of the data was carried out in Distance in two stages.

In the first stage, histograms of the distribution of perpendicular distances from the transect line were visually assessed for any obvious violations of assumptions (e.g. movement away from observers can be detected as a spike in the histogram). Data were grouped over a range of different distance intervals. Histograms were also visually assessed in order to identify any obvious truncation points (larger distances discarded), where there are natural breaks in the tail of the histogram. Tails occur where there are few records at large distances from the transect. Breaks in the tail indicate distance intervals with no recorded sightings (Buckland *et al.* 2001).

Untruncated histograms were also examined for the presence of a tail, and goodness of fit tests were used to assess the value of truncating the right hand tail of the data. Right truncation of the dataset is often valuable to improve the model fit near the line, where the fit is the most important for density estimation (Buckland *et al.* 2001).

The first stage of the exploratory analysis also involved derivation of probability detection functions from untruncated, ungrouped data using models with the Half Normal and Hazard Rate keys with and without an adjustment terms. The potential of detection heterogeneity as a result of having several different spotlighters was also assessed by plotting the distance of each observation against the categorical variable of spotlighter in the software package Stata (StataCorp 2006).

The second stage of the exploratory analysis involved assessment of options for use of a detection function for each of the four strata (*P. occidentalis* within riparian and non-riparian habitat and *T. vulpecula* within riparian and non-riparian habitat) or the use of a global detection function.

2.7.3 Final Model Selection

The exploratory analysis supported use of truncation and the Half Normal global detection function (see results). Each model from the candidate set (Table 1) was then compared and models were ranked on the basis of ΔAICc to select the model which best described the data.

3. Results

3.1 Sightings

A total of 27 *P. occidentalis* and 43 *T. vulpecula* sightings were recorded in the study area. In the riparian habitat 20 *P. occidentalis* and 18 *T. vulpecula* were sighted, while in the non-riparian habitat seven *P. occidentalis* and 25 *T. vulpecula* were sighted (Table 2). No evidence of clumping was found (Figure 7). However, a concentration of possum sightings was found in the riparian habitat (Figure 8).

Table 2: The number of *P. occidentalis* and *T. vulpecula* sightings in each of the three habitat (jarrah, wandoo, riparian) types at the study site

Possum species	Number of possum sightings Habitat type jarrah	 wandoo	 riparian	Total
P. occidentalis	2	5	20	27
T. vulpecula	9	16	18	43
Total	**11**	**21**	**38**	**70**
Total non-riparian	32			

3.2 Exploratory analysis

Visual assessment of histograms revealed no evidence of spiking or movement away from the observer (Figure 7a). The dataset was sparse and there was an obvious truncation point at 35m. The GOF tests revealed no lack of model fit, however further analysis showed truncating at 35m would result in a large loss of data (Figure 7b). Therefore, I decided to fit the preliminary model to the data, and compute the detection function $g(x)$ to find the value of x such that $g(x) = 0.15$. This method is recommended by Buckland *et al.* (2001). The result of this procedure resulted in truncation point of 50m (Figure 7c).

There was evidence of detection heterogeneity (differences in the mean distance to sightings) between spotlighters (Figure 9), as the mean distance to a sighted possum varied for each observer.

The results from comparing models with the half normal and hazard rate key supported use of the half normal key. Addition of adjustment terms (one cosine adjustment for the half normal key and one hermite polynomial adjustment for the hazard rate key) did not improve the model (Table 3). Therefore, the half normal key without adjustment terms was used for further analysis

Comparing use of a separate detection function for each stratum with a global detection function indicated a good fit for both options, as determined by the X^2, Kolmogorov-Smirnov and Cramér-von Mises GOF tests. However, the q-q plots suggested the model fit close to the line was better for the global detection function (Figure 10). The values for the coefficient of variation were also considerably lower (better) for the global detection function. Therefore, the model selection only included models derived from global detection function, with density estimates derived for each stratum.

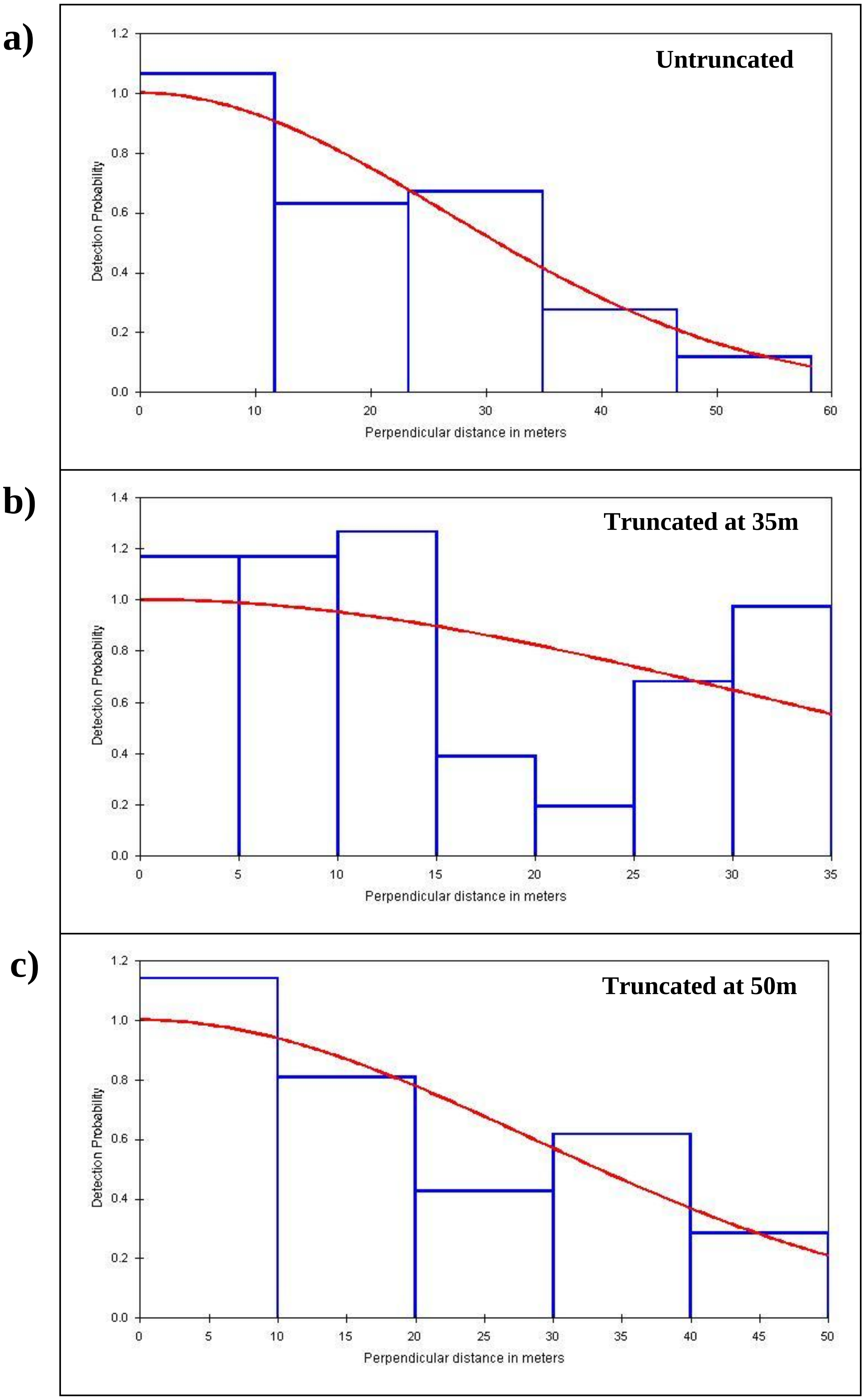

Figure 7: Frequency distributions of perpendicular distances of observations from the transect line for all habitats combined; a) Untruncated, b) truncated at 35m, c) truncated at 50m

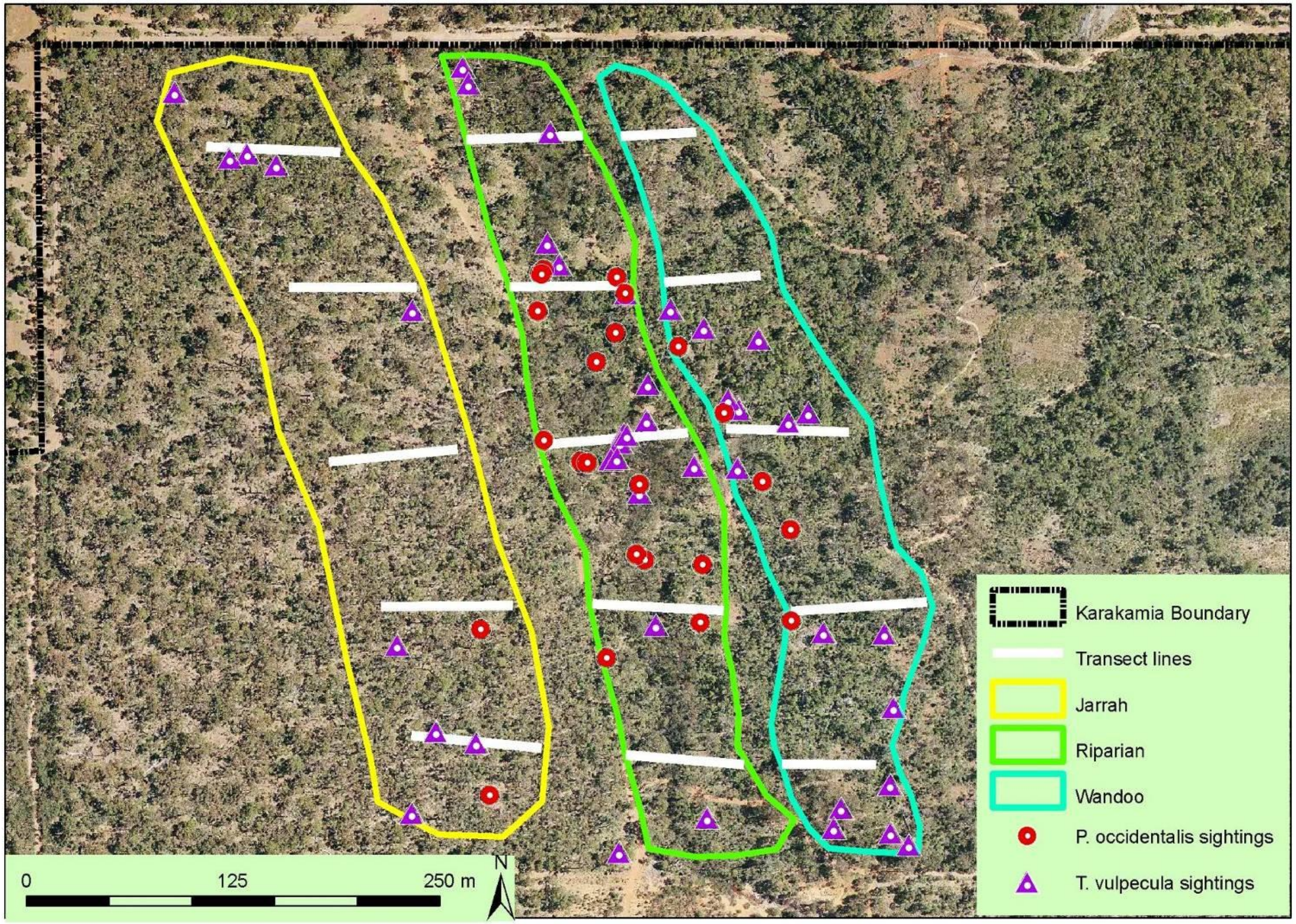

Figure 8: Orthophotograph of the study area within Karakamia Wildlife Sanctuary surveyed to estimate the density and population size of *Pseudocheirus occidentalis* showing the locations of all the sightings. Orthophotograph sourced from Australian Wildlife Conservancy

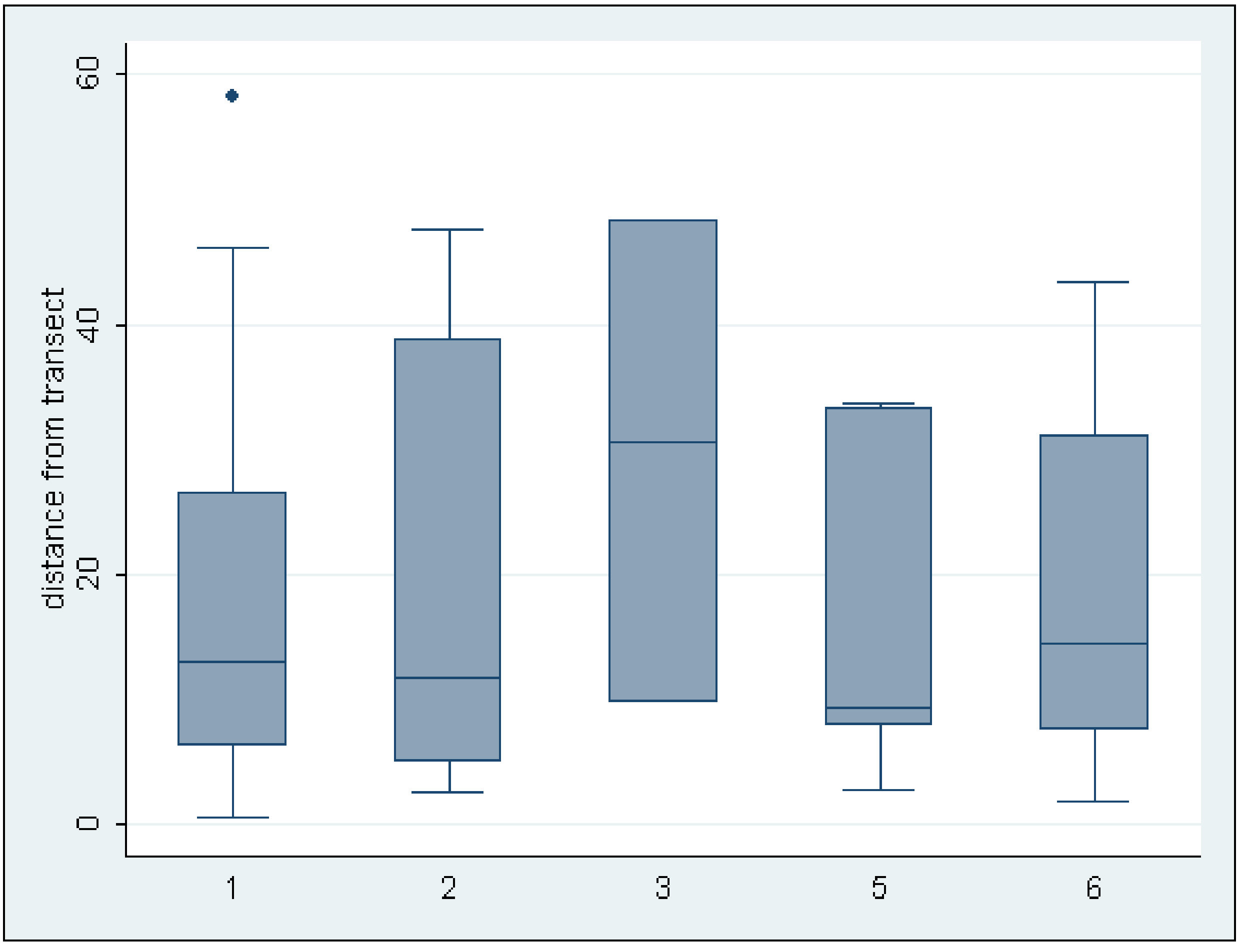

Figure 9: Box plot of the perpendicular distance of the sightings against the five spotlighters used to spotlight *Pseudocheirus occidentalis and Trichosurus vulpecula*.

Boxes define the 75th (upper) and 25th (lower) percentile.

Whiskers identify the upper and lower values, and the dot represents an outlier value.

Table 3: Models fitted to untruncated data for *Pseudocheirus occidentalis* and *Trichosurus vulpecula* with global (DF Global) and stratified (DF by Strata) detection functions, half normal (HN) and hazard rate (HR) keys, with and without adjustment terms. Models are ranked by ΔAICc values.

ESW = effective strip width, Log Ł = log likelihood

Model	**No of parameters**	**AICc**	**Δ AICc**	**ESW**	**LogŁ**
DF Global HN no adjustments	1	546.81	0	32.14	-272.38
DF Global HN 1 cosine adjustment term	2	548.67	1.86	29.96	-272.24
DF Global HR no adjustments	2	549.34	2.55	28.98	-272.59
DF by Strata HN no adjustments	4	551.59	4.77	N/A - listed by stratum	-271.07
DF Global HR 1 hermite polynomial adjustment term	3	550.58	3.77	29.17	-272.11
DF by Strata HR no adjustments	8	561.70	14.88	N/A - listed by stratum	-270.32

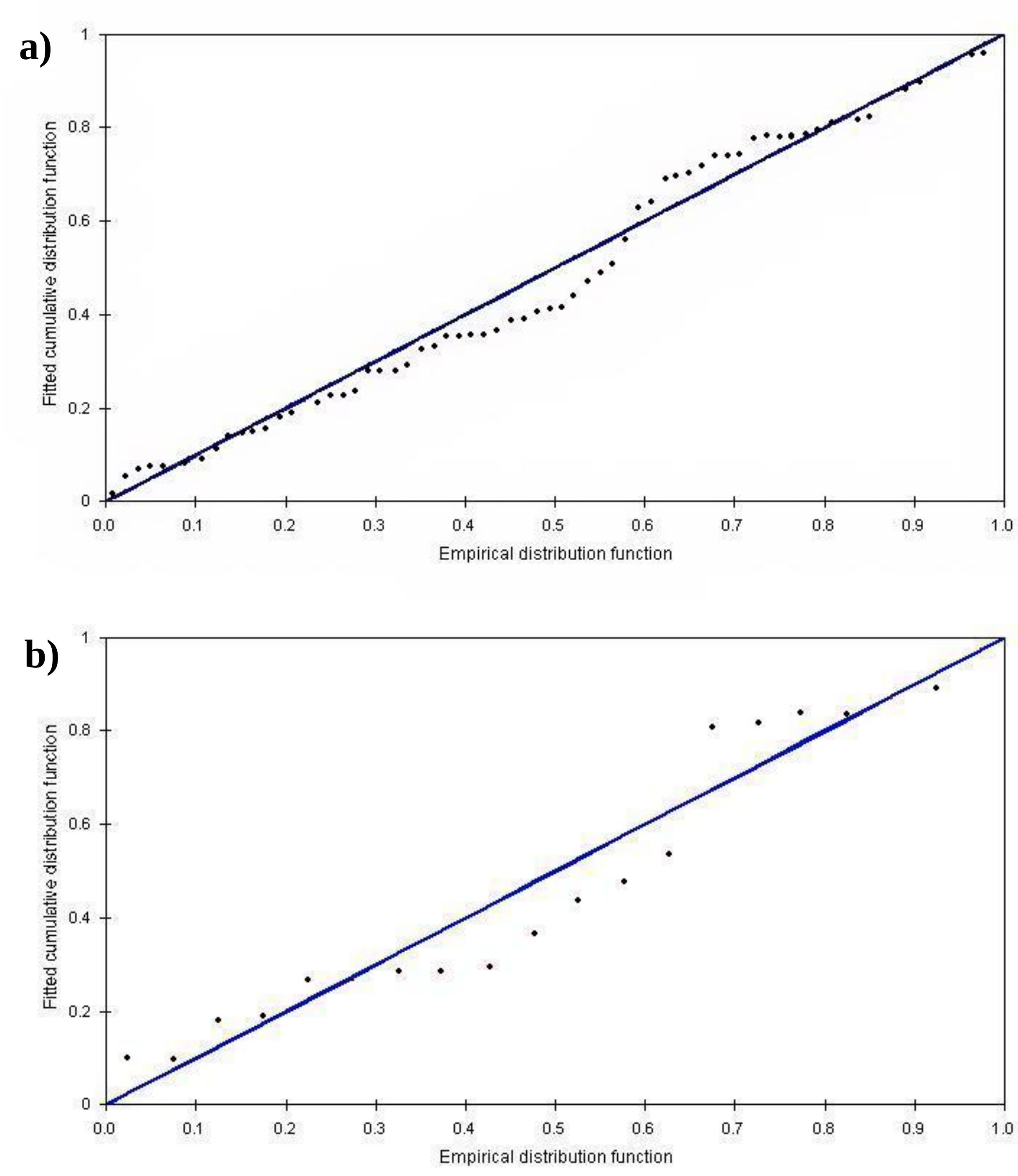

Figure 10: Q-q plots of the empirical distribution function vs. the cumulative distribution function for the untruncated models with a half normal key fitted to the *Pseudocheirus occidentalis* spotlighting data

a) using a global detection function

b) using a separate detection function for each stratum.

3.3 Density and abundance estimates

The output from Distance identified Model 2 as the preferred model (lowest AICc value) (Table 4). This model had a half normal global detection function and includes the covariate for the maximum daily temperature. There was equal support (ΔAICc < 2) for at least one other model (Model 1, Table 4). These two models differed only by inclusion of the covariate for daily maximum temperature. Lower maximum diurnal temperatures broadened the shoulder of the detection function (Figure 11), indicating increased detection probability at lower daily maximum temperatures.

A third model (Model 7, Table 4) also had a ΔAICc of <2 from the preferred model. This ΔAICc (of 1.96) would initially suggest substantial support for the model. However, Model 7 has a likelihood value very close to Model 1 and differs from Model 1 only by inclusion of one additional parameter (the parameter for the covariate for rainfall). Where models are within 0-2 AIC units and differ only by the addition of one parameter, have similar log-likelihood values, and where the larger model (the model with the additional parameter) does not improve (i.e. does not have a lower AICc value), it is not considered supported or competitive (Burnham & Anderson, 2002). Such models appear to be competitive simply because they are very similar to the model without the additional parameter. On this basis, the remaining models have no real support (Table 4).

The derived population estimates for *P. occidentalis* from the preferred model (Model 2) were 11 in the riparian habitat and 17 in the non-riparian habitat. The density estimate was 2.87 ha^{-1} for the riparian and 2.02 ha^{-1} for the non-riparian habitat (Table 5).

Furthermore, the model selection resulted in populations estimates of 10 *T. vulpecula* in the riparian habitat and 22 for the non-riparian habitat. The density estimates were relatively similar for the riparian habitat, 2.64 ha^{-1}, and the non-riparian habitat, 2.60 ha^{-1} (Table 5).

The effective strip width for the preferred model was 31.72 m (Table 4). The maximum sighting distance for *P. occidentalis* was 48.4m and for *T. vulpecula* 58.2m.

Table 4: Output from Distance, ranking of the candidate models to determine which best describes the data for density of the western ringtail possum, *Pseudocheirus occidentalis*, and the common brushtail possum, *Trichosurus vulpecula*, at Karakamia Wildlife Sanctuary, south-west Western Australia.

All models have a Half Normal global detection function with the data truncated at 50m. ESW = effective strip width, Log Ŀ = log likelihood, p = probability of detection. See text for additional abbreviations and explanation of models. The preferred model is shown in bold font and shaded.

Model number	**Model description** **Global HN detection function**	**# parameters**	**AICc**	**ΔAICc**	**ESW**	**Log Ŀ**	**p**
2	**Daily max temp**	2	529.27	0	31.72	-262.55	0.63
1	No covariates	1	530.35	1.07	32.75	-264.14	0.66
7	Rainfall	2	531.23	1.96	32.40	-263.52	0.65
3	Overnight min temp	2	531.35	2.07	32.37	-263.58	0.65
8	Dailymax temp + Overnightmin temp	3	531.50	2.28	31.66	-262.57	0.63
5	Height	2	532.36	3.08	32.73	-264.09	0.65
6	Moon	4	533.48	4.21	31.85	-262.43	0.64
4	Spotlighter	5	536.58	7.31	31.80	-262.82	0.64

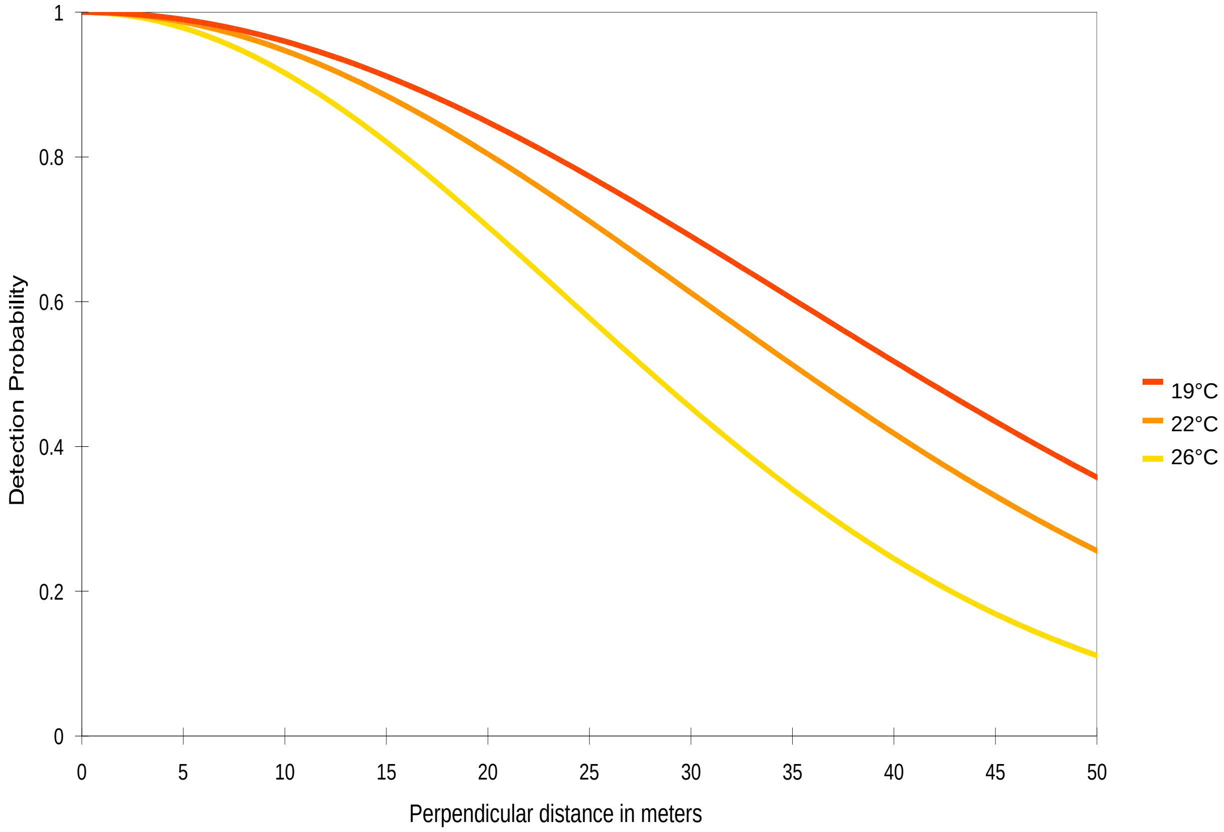

Figure 11: Plot showing the effect of daily maximum temperature on the detection probability of *Pseudocheirus occidentalis* and *Trichosurus vulpecula* derived from spotlighting data from Karakamia Wildlife Sanctuary, south-west Western Australia.

Table 5: Derived estimates of density and abundance for *Pseudocheirus occidentalis* and *Trichosurus vulpecula* from two habitat types (riparian and non-riparian) within Karakamia Wildlife Sanctuary, south-west Western Australia.

	Estimate	**%CV**	**df**	**lower 95% CI**	**upper 95% CI**
Stratum : *P. occidentalis*, riparian					
Density (no ha.$^{-1}$)	2.87	10.45	30.35	2.32	3.55
N (population estimate)	11	10.45	30.35	9	14
Stratum : *P. occidentalis*, non riparian					
Density (no ha.$^{-1}$)	2.02	7.92	70.44	1.72	2.36
N (population estimate)	17	7.92	70.44	15	20
Stratum : *T. vulpecula*, riparian					
Density (no ha.$^{-1}$)	2.64	15.65	12.21	1.88	3.70
N (population estimate)	10	15.65	12.21	7	14
Stratum : *T. vulpecula*, non riparian					
Density (no ha.$^{-1}$)	2.60	9.01	70.47	2.17	3.11
N (population estimate)	22	9.01	70.47	18	26

4. Discussion

4.1 Population estimates: Pseudocheirus occidentalis

The hypothesis of higher *P. occidentalis* density in the riparian habitat at Karakamia Wildlife Sanctuary is supported by the results from the preferred model in Distance. The density of *P. occidentalis* for the riparian habitat was estimated to be 2.87 ha^{-1} (95% CI 2.32-3.55) and 2.02 ha^{-1} (95% CI 1.72-2.36) for the non-riparian habitat. The slightly greater density found in the riparian habitat appears to indicate that it is the preferred habitat for *P. occidentalis* at Karakamia and that this area can support a population of the species. The lower density occurring in the non-riparian habitat further suggests that *P. occidentalis* is able to inhabit areas outside the riparian zone.

Additionally, the population estimate of 11 adult *P. occidentalis* in the riparian habitat was fewer than the population estimate of 17 for the combined non-riparian habitats (wandoo and jarrah). As the riparian habitat covered a smaller area of 3.81ha compared with the combined non-riparian of area 8.49ha, this would explain why a greater number of individuals were estimated to inhabit the non-riparian habitat. Therefore, the greater population size estimated for the non-riparian habitat can be attributed its larger area. These results provide evidence that a *P. occidentalis* population has established within Karakamia Wildlife Sanctuary.

Furthermore, the findings indicate that dispersal has occurred from the original riparian release site and that the surrounding wandoo and jarrah habitats can support populations of *P. occidentalis* at a lower density. It is possible that as breeding has occurred from the translocated population, individuals have dispersed from the riparian habitat. It is also possible that the dispersal has been very slow over time and/or there has been very slow population growth. Therefore the *P. occidentalis* population may have dispersed and in fact be more widespread throughout Karakamia, however my initial broadscale survey did not indicate this (see appendices).

There is also the possibility that, although the riparian habitat is structurally similar to the known preferred peppermint habitat of *P. occidentalis*, there is a reduced dependence on this habitat structure within Karakamia. This would be the case in the continuity of canopy provided protection from predation and in the absence of foxes and cats at Karakamia, *P. occidentalis* population may be less dependent on this protection from predation. These possibilities highlight the need for continued monitoring and evaluation of the translocation, to increase the understanding of how *P. occidentalis* interact with their environment at Karakamia.

A population size estimate of 28 individuals was calculated for the total area (12.3ha) covered by this study. The total number of individuals released into Karakamia over the seven year period up until 2002 was 42. The population estimate for this study is smaller than the total number of individuals released, however, the 42 translocated animals were released over a seven year period (1995 to 2002) and the population size possibly never exceeded 28 individuals. As the translocation was staggered over several years, most of the founder *P. occidentalis* individuals would have now died, leaving the current population to be made up of mainly recruits. This is further supported by finding that all the sightings in this study but one had no ear tags, indicating that they were most likely new recruits. Alternatively, they may be founding translocated individuals which have lost their ear tags. *Pseudocheirus occidentalis* has a recorded life span which may exceed six years (de Tores 2008), however this is thought to be the upper limit (P. de Tores, pers. comm.). Therefore, it is unlikely the observed *P. occidentalis* in this study were founder individuals and more plausible that breeding has been successful at Karakamia and a small population has established.

The *P. occidentalis* density estimates in this study were greater than those for other translocation sites in coastal regions of south-west Western Australia. In studies conducted by de Tores *et al.* (2004) and Clarke (2010) at translocation sites at Leschenault Peninsula, Preston Beach Road, White Hills Road and Martin’s Tank, *P. occidentalis* density estimates ranged from 0.12 ha^{-1} to 1.04 ha^{-1}. However, comparison with these translocated populations is limited, as they are outside fenced reserves. Therefore, it is only possible to speculate that the populations within the habitats surveyed at Karakamia are of a greater density than other translocated populations outside fenced reserves.

4.2 Population estimates: Trichosurus vulpecula

The densities of *T. vulpecula* within the riparian and non-riparian habitats were found to be similar, 2.64 ha^{-1} and 2.60 ha^{-1}, respectively. Accordingly, the population estimate of *T. vulpecula* was greater in the combined non-riparian habitats than in the riparian habitat. The similar density estimates suggest that *T. vulpecula* is less habitat specific than *P. occidentalis*, supporting previous literature (Wayne *et al.* 2005d). However, as the study area only covered a small portion of Karakamia, the density estimates may not be representative of the rest of the sanctuary. Further, the lower population size estimated for the riparian habitat was reflective of its smaller area compared with the non-riparian habitat. The population size estimates are only reflective of the study area, and further surveying would be needed to estimate the total *T. vulpecula* population size at Karakamia. As the study was primarily designed to estimate the population size of *P. occidentalis*, the population estimates provided should only be used as an indication that all three habitat types in the study area at Karakamia equally support populations of *T. vulpecula*.

However, spotlighting may not be the most effective method of estimating *T. vulpecula* population size. Wayne *et al.* (2005b) estimated that spotlighting detected only 5-13% of *T. vulpecula* captured in traps at the Perup Nature Reserve in south-west Western Australia. Such results would suggest that the population size and density estimates from this spotlighting study may be underestimates. However, distance sampling was not used to analyse the data and a detection probability was not included in the study by Wayne *et al.* (2005b). Therefore, using distance sampling to analyse spotlight data may provide a better representation of *T. vulpecula* population size than just spotlight counts.

Furthermore, it is important to examine whether the type of survey effort used is sufficient to determine abundance and density. Estimates of *T. vulpecula* density may require trap-based methods, particularly if the habitat they occupy causes *T. vulpecula* to be less detectable (Efford 2004). This result was found in a study conducted at Tuart Forest National Park, near Busselton, comparing spotlight distance sampling with trapping (using program 'Density') (H. Grimm, unpublished data 2010). Trapping was found to be a better technique than spotlighting at this site, due to *T. vulpecula* individuals tending to occupy areas high in the Tuart (*Eucalyptus gomphocephala*) forest canopy, where they were less visible. Therefore, trapping could provide a better technique to spotlighting to estimate *T. vulpecula* abundance at Karakamia if the species is preferring to occupy less detectable areas in the canopy.

In contrast, the same study found spotlighting to be a better technique than trapping for estimating *T. vulpecula* abundance at Gelorup, a local government managed reserve, near Busselton (H. Grimm[3], unpublished data). This result could reflect a low *T. vulpecula* underlying abundance at Gelorup, or that all of the individuals were not detected due to habitat type, predator or behavioural factors, resulting in lower precision in the estimates derived from trapping. Therefore, spotlighting can be utilised as a technique for estimating *T. vulpecula* abundance, but relative success most likely depends on the, habitat type, underlying population density etc. Consequently, as *T. vulpecula* tended to occupy habitat that didn't

[3] Helen Grimm is a PhD student at Murdoch University, in association with the Department of Environment and Conservation, completing her thesis on using sufficient effort to determine the abundance/density of *P. occidentalis* and *T. vulpecula* at Tuart Forest National Park and Gelorup reserve, near Busselton.

inhibit detection at Karakamia, there is the possibility that the spotlighting survey effort applied in this study was sufficient to determine reliable abundance and density estimates.

4.3 Model Selection

The preferred model included an influence of the daily maximum temperature on detectability, reflecting a difference in *P. occidentalis* detectability at different daily maximum temperatures. The results indicate *P. occidentalis* was more readily detected when the daily maximum temperature was lower. Lower daily temperatures provide more favourable conditions for *P. occidentalis* as the species is less capable at coping with high temperatures (Yin 2006). At these high temperatures, *P. occidentalis* experiences high evaporative water loss and it has been suggested that the species do not have an efficient mechanism for heat dissipation (Yin 2006). Therefore, the greater detectability of *P. occidentalis* at lower daily maximum temperatures in this study possibly indicates that due to their susceptibility to high temperatures during the day, *P. occidentalis* may be less detectable during following high diurnal temperatures . As *P. occidentalis* is nocturnal and most active at night (Jones *et al.* 1994b), higher temperatures experienced during the day would appear to be discouraging individuals from adventuring from tree-hollows or drey shelters and being detected during earlier hours of the night. Thus, survey efforts delayed to later night hours, when the temperature has dropped lower, may be more likely to detect *P. occidentalis*.

The daily maximum temperatures recorded in this winter study were not as extreme as would be experienced during summer months. The highest daily maximum for any of the survey days was 30°C. This temperature is likely to be much higher during summer, and therefore likely to have a greater influence on *P. occidentalis* detectability. In the study conducted by Yin (2006), *P. occidentalis* was observed to be easily overheated at moderately high temperatures (35°C). Therefore during summer, when the daily maximum temperature is likely to reach and exceed 35°C, it is predicted that fewer *P. occidentalis* will be detected in any surveys carried out.

In using distance sampling to estimate the *P. occidentalis* population size at sites near Busselton, de Tores and Elscot (2010) found that the covariate for 'observer effect' improved their preferred model. However, while my exploratory analysis suggested that the observer (spotlighter) covariate was important, addition of this covariate didn't improve the preferred model. This absence from the preferred model is probably a result of my sparse data set (70 sightings) and the large number of five parameters, as opposed to two, by inclusion of the categorical variable for the observer effect. Similarly, in using distance sampling to analyse spotlight monitoring data and estimate population size for *P. occidentalis* at Leschenault Peninsula Conservation Park and Yalgorup National Park, de Tores *et al.* (2004) found that adding the covariate for 'rainfall' improved the preferred model. However, in light of the fact that all but one spotlighting session in this study was carried out on rain-free nights, I was unable to assess the effect of rainfall in the current study.

After examining the histograms of the detection probability, the truncation point of the data was chosen at 50m. However, the searching width either side of each transect was designed to be 45m. Therefore sightings outside the 45m strip had the possibility of being double counted on the neighbouring transect. This was not a problem for the estimates derived in this study, as the effective strip width applied by the preferred model was 31.72m. Consequently, the preferred model didn't include sightings outside 31.72m to derive the abundance and density estimates, and the possibility of overestimating was thus avoided.

4.4 Validity of assumptions

The main concern with any distance sampling line transect study is assessing the validity of the three assumptions. Firstly, the perpendicular distance measurements in this study were accurate due to the use of differential (post-processing) GPS and an established base station. Hence, the assumption of accurate measurement of distance was satisfied. Secondly, the histogram plots did not suggest that evasive movement was present for either *P. occidentalis* or *T. vulpecula* in response to the observer. Therefore, the assumption that animals are detected at their original location appears to have been met. There is no statistical problem in observing the same possums more than once as long as their movement is not influenced by the observer (Focardi *et al.* 2002). However, the third assumption of perfect detectability on the line was difficult to achieve and verify. While every effort was to made to ensure every possum was detected on the transect line, the assumption was unlikely to be met. As *P. occidentalis* is difficult to capture, this study was not able to incorporate the use of radio collars, which would allow for the assumption of perfect detectability to be tested. Accordingly, further research into the use of the mark-recapture techniques and development of practical field techniques to enable incorporation of appropriate multipliers in the analysis of *P. occidentalis* spotlight data is recommended.

4.5 Survey design

Obtaining reliable results from a distance sampling survey depends critically on good survey design (Thomas *et al.* 2010). This relies on there being sufficient transect lines to ensure that variation in encounter rate (number of objects detected per unit survey effort) can be adequately estimated. The five concepts that need to be addressed in selecting line-transect layout are: 1) replication 2) randomization 3) sampling coverage 4) stratification and 5) sampling geometry. Firstly, a minimum of 10-20 replicate lines should be surveyed to provide a basis for an adequate variance of the encounter rate and reasonable number of degrees of freedom for construction confidence intervals (Buckland *et al.* 2001). The survey design in this study involved 15 replicate transect lines, and therefore provided adequate replication. Secondly, transects should be placed within the study region following some form of randomization to avoid sampling bias. Thirdly, transects should also be placed to ensure that all portions of the study area have an equal probability of being included in the sample (equal coverage probability) (Thomas *et al.* 2010). Therefore the parallel equally spaced transects with a random start used in this study, provided uniform coverage of the survey area and some randomization.

Fourthly, spatial stratification can be included in the survey deign to improve precision, and also to divide the study area into smaller areas. This division allows managers to obtain separate abundance estimates for each area (Buckland *et al.* 2001). In the case of this study, habitat stratification was used to provide precise estimates of two habitat types for the managers of Karakamia. Lastly, choice of sampling geometry depends on the survey region, logistics and efficiency, knowledge of density gradients or patterns and their interplay with other sampling concepts. Therefore, transects lines in this survey were placed parallel to a potential density gradient across the riparian habitat.

The density estimation process of distance sampling allows repeat sampling of replicate transects and repeat counting of the same individuals. In line-transect sampling, if an animal moves ahead of the observer and is counted repeatedly, abundance will be over-estimated. This is undetected movement in response to the observer and is an un-measureable source of bias (Buckland *et al.* 2001). Due to the sedentary nature of *P. occidentalis* and the species naivety to human presence (de Tores & Elscot 2010), the species is unlikely to move ahead in response to the observer without first being detected and unlikely to be counted more than once on the same unit of transect sampling. Further, it is of no concern if an animal is detected more than once on different occasions of sampling the same transect (Buckland *et al.* 2001), as re-sampling of the same transect is treated as an individual unit of sampling effort.

In contrast, double counting, by itself, is not a cause of bias if such counts correspond to different units of counting effort. Furthermore, this bias is likely to be small unless repeated counting is common during a survey (Buckland *et al.* 2001). During this study, transects were placed at a set distance of 90m apart to allow for a 45m strip for detection either side of the transect and to avoid double counting on neighbouring transects. Therefore, it is important to note that the procedures of repeated sampling of the same transect, and counting of the same individuals on separate occasions, are used to extend the total transect length traversed and therefore minimize the variance of the derived estimates.

4.6 Transect length requirements

Spotlighting surveys are the most useful method of estimating *P. occidentalis* density (Wayne *et al.* 2005b). Therefore in order to implement future spotlight surveys at Karakamia, the level of efficiency of the current survey needs to be assessed. To do this, one can look at the coefficient of variation for the estimates of *P. occidentalis* density and abundance in this study. For practical purposes, the suitable target coefficients of variation for density and abundance estimates are often in the range of 10%, with values greater than 20% likely to preclude reliable inference (Buckland *et al.* 2001).The coefficient of variation for the riparian habitat estimates for *P. occidentalis* was 10.45 and for the non-riparian, 7.92. The corresponding values for *T. vulpecula* were 15.62 and 9.01 (Table 5). Thus, the coefficients of variation for the density estimates in this study were less than 20%, and can be used for reliable inference.

Variance estimates can further be used to calculate the necessary survey effort (in transect length) to obtain density and population size estimates with specified levels of precision. The transect lengths used in the current survey therefore provided sufficient survey effort to produce close to 10% precision for the *P. occidentalis* density estimates. Buckland *et al.* (2001) suggested that a sample size of 60-80 sightings would be adequate to reliably estimate the average density within a study area. In this study, the level of survey effort used produced a coefficient of variation of 7.92-15.62%, with a sample size of 70 possum sightings. Therefore, precise density estimates can be obtained at Karakamia applying similar survey effort to acquire approximately 70 sightings.

The precision of population estimates is important for evaluating the effectiveness of survey measures, as wide confidence intervals complicate detection of any trends (Fisher *et al.* 2000). The confidence intervals were relatively small for both the *P. occidentalis* density and abundance estimates in this study. Therefore, the current estimates can be used for comparison and detection of population trends in the future.

As mentioned above, the precision of a distance sampling survey can be estimated from the field data. However, the accuracy of a survey can only be evaluated by comparison with the true population size. As is consistent with many surveys of mammal populations, the accuracy of this *P. occidentalis* survey cannot be measured as the true population size is not known. Nevertheless, one example of a study which critically evaluated the accuracy of distance sampling was conducted by Hounsome *et al.* (2005). The authors studied a high-density badger (*Meles meles*) population at Woodchester Park in Gloucestershire, U.K., which had been routinely and intensively live-trapped for the past 30 years. The estimates of abundance derived from the mark-recapture analyses of the live-trapping data were compared against estimates derived from distance sampling. Distance sampling produced an estimate of abundance that was close to that derived from the mark-recapture analysis, but involved considerably less effort. However, mark-recapture analysis still only provides an estimate of the population size, and not the true population size. Therefore, distance sampling methods can provide reasonably accurate estimates of population size, and should be utilized in *P. occidentalis* surveys.

To obtain the estimates of density and abundance in this study, the data for both possum species were pooled to calculate a detection function. A pooled detection function provides a less comprehensive analysis of the data compared to calculating a separate detection function for each species (Buckland *et al.* 2009). More sightings of *P. occidentalis* are required to enable calculation of separate detection functions for each species. Therefore, greater repetition of each transect would be required. However, due to the small size of the *P. occidentalis* population at Karakamia and the fact that the preferred habitat may be near carrying capacity, the encounter rate of *P. occidentalis* sightings is likely to be low. Therefore, further survey effort could become time-consuming and may not be practical for the managers of Karakamia.

4.7 Future population monitoring and management

While the purpose of this study was not to assess the success of the *P. occidentalis* translocation into Karakamia Wildlife Sanctuary, the population size estimates suggest that it has in fact been successful to some extent. A translocation project is considered a success if it results in a self-sustaining population (Griffith *et al.* 1989; Rout *et al.* 2007). Therefore, the success of the *P. occidentalis* translocation can be measured by the persistence of the population and evidence that breeding has occurred. Therefore, while this study may not provide evidence that the population is self-sustaining, the translocation can be considered successful as a small populations appears to have established.

This study is the first conducted to estimate *P. occidentalis* population size at Karakamia, 15 years after the original release. The estimates derived in this study therefore represent the population over a long time period. Obviously there have been fluctuations in the population since the original release, resulting in the current population size. Thus, the estimate of 28 individuals may be close to the carrying capacity for the habitat area surveyed in this study during 2010.

It has been recommended that *P. occidentalis* populations be monitored every 1-2 years (Clarke 2010). Such regular monitoring could be used to detect whether the population fails to remain self-sustaining in the longer term. An example of this was seen at the population at Leschenault Peninsula, where the reason for the rapid population decline was unknown due to lack of regular monitoring (de Tores *et al.* 2004). However, the most likely cause of decline was predation. Karakamia Wildlife Sanctuary is managed with the objective of remaining free of introduced mammalian predators (foxes and cats) and provides an opportunity to study a *P. occidentalis* population in the absence of predation by foxes and cats. Given the decline of *P. occidentalis* at Leschenault Peninsula Conservation Park occurred over a four year period (de Tores *et al.* 2004), monitoring at Karakamia is recommended to be carried out at intervals no longer than two years. These intervals should enable managers to detect a decline in the *P. occidentalis* population, should it occur, in sufficient time to identify the cause and take the appropriate management actions and ensure long term success of the resident population.

As there have been no prior population estimates of *P. occidentalis* abundance at Karakamia, the population estimates produced in this study cannot be used to give an indication of whether the population is growing or declining. However, the estimates can be used for comparison in future studies of the same habitats covered by this study. It is recommended that line transect distance sampling techniques be used to monitor the population at Karakamia in the future, following the same, or similar transects lines.

Furthermore, similar survey effort to that used in this study should be applied in any future monitoring surveys. This effort will enable reliable comparison of abundance and density estimates. However, for comparison with the current estimates, further surveys would need to be conducted around the same time of year (in winter) and in the same habitat areas. The limited time frame of this study only provided snap-shot view of the population in the study area, during the winter months. Therefore, comparisons with surveys conducted in different areas of Karakamia or during different seasons would not be comparable.

Habitat quality is an important to consider in the analysis of survey data because of its potential to influence detectability (Wintle *et al.* 2005). High quality habitat would be expected to support higher numbers of individual animals due to the higher quality of resources available. In the case of this study, the riparian vegetation represents the higher quality habitat for *P. occidentalis*, compared to the surrounding jarrah forest and wandoo woodland (Jones *et al.* 1994a). Accordingly, a greater number of *P. occidentalis* individuals were detected in the riparian habitat. Other studies have found differences in the detectability of *P. occidentalis* for different habitat structures (logged and unlogged) (Wayne *et al.* 2005a) and in the detectability of the greater glider and yellow-bellied glider for different habitat qualities (Wintle *et al.* 2005). Therefore, the influence of habitat quality at Karakamia has implications for the interpretation of survey data and for future survey effort. The probability of detecting *P. occidentalis* in marginal habitat, such in other parts of Karakamia, may be reduced because the density of animals is likely to be lower. Ideally, a comprehensive survey of other habitats at Karakamia should be carried out to try and confirm *P. occidentalis* absence from these areas.

Accordingly, in future monitoring, further surveys of other areas in Karakamia would be needed to confirm whether the current study area is in fact where the population is concentrated. If other habitats were found to support *P. occidentalis* individuals, then these would need to be incorporated into monitoring efforts. Density estimates are more efficient when more search effort is put into regions of greater expected *P. occidentalis* presence, using a stratified survey design such as seen in this study (Buckland *et al.* 2001). However due to the size of the riparian habitat, this study was not able to apply greater search effort to the survey design. Future surveys of *P. occidentalis* at Karakamia may be able to incorporate a stratified design if further transects were placed in other parts of the sanctuary where *P.*

occidentalis may be found in lower densities. However, combined with the expected survey effort required, such a survey design may not be feasible for the managers of Karakamia.

4.8 Conclusions

This study has provided the first quantitative estimates of *P. occidentalis* population size since the initial translocation to the sanctuary. The low coefficients for variation for the derived estimates demonstrate the efficacy of distance sampling, and provide evidence that the method should be used to obtain reliable estimates in the future. Management decisions can now be based on these reliable estimates, however future monitoring of the Karakamia *P. occidentalis* population is required to detect population trends. Applying distance sampling methods and the information-theoretic based approach will enable managers to effectively detect these trends.

With regular, efficient monitoring, the managers of Karakamia should be able to effectively detect *P. occidentalis* population trends in the future. This study provides insight into the abundance, density and habitat preference of translocated *P. occidentalis* in the northern jarrah forest. The population at Karakamia Wildlife Sanctuary now represents the northern extent of *P. occidentalis* geographic range (DEWHA 2010). Although habitat and resource requirements are likely to vary in different parts of the former distribution of this species, the establishment of *P. occidentalis* at Karakamia suggests that the species is capable of re-inhabiting areas within its former range. This may depend on the availability of essential resources and preferable habitat, as well as the removal of introduced predators. However, the findings of this study provide hope that *P. occidentalis* may be restored to parts of its former range in the future.

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Appendices

Appendix 1:

Research Proposal: Density and abundance of western ringtail possums (*Pseudocheirus occidentalis*) translocated in the jarrah forest of Karakamia Wildlife Sanctuary

Table of Contents

Summary

The density of western ringtail possums (*Pseudocheirus occidentalis*) at Karakamia Wildlife Sanctuary will be estimated using the line transect distance sampling technique. Western ringtail possums have been translocated into Karakamia since 1995 for approximately 10 years, however it is yet to be determined whether a substantial population persists today.

My project aims to calculate a reliable estimate of density of western ringtail possums, to assess the persistence of the translocated population. The significance of my project is that the persistence of every remaining population of western ringtail possums it vital to the species long-term survival. The translocated population within Karakamia is also one of few enclosed by a predator-proof fence, and may contain suitable habitat for the western ringtail possum. Therefore, my project will obtain further information for Australian Wildlife Conservancy (AWC), the managers of Karakamia, to manage and conserve the western ringtail possums within the sanctuary in the future.

2. Background

2.1 Introduction to the Western Ringtail Possum

An arboreal specialist folivore, the western ringtail possum (*Pseudocheirus occidentalis)* has experienced a substantial decline in its distribution since European settlement (Jones & Hilcox 1995; Wayne *et al.* 2006). Endemic to the south-west of Western Australia, the western ringtail possum further has been threatened by introduced predators, such as the fox (*Vulpes vulpes*) and the cat (*Felis catus*)(de Tores *et al.* 2004). Consequently, the western ringtail possum is listed as vulnerable under the Australian Government Environment Protection and Biodiversity Act 1999 (World Conservation Union 1994). In Western Australia, the western ringtail possum is also listed as ‘fauna that is rare or likely to become extinct’ under the Western Australian Wildlife Conservation Act (1950). These status listings demonstrate the need for future conservation efforts to ensure the long-term survival of the western ringtail possum (World Wildlife Fund 2004).

A significant contraction in the western ringtail possum’s range led to the species endangered classification by the Western Australian Government in 1980 (Jones & Hilcox 1995). Further contraction in its range resulted in increasing study of the species ecology, in order to try and understand the reasons behind it. However, an understanding of its ecology and conservation status has been constrained by the relative difficulty in surveying the species in the wild (Wayne *et al.* 2005b). This western ringtail possum is not amenable to traditional trapping methods and is naive, displaying minimal predator avoidance (de Tores 2008).

The western ringtail possum is a medium-sized arboreal browsing marsupial weighing up to 1.3 kilograms and approximately 40cm in body length (Jones & Hilcox 1995, DEWHA 2010). In appearance, the western ringtail possum has dark brown fur above and cream to grey fur underneath and a tail that grows to its body length, terminating in a white tip (Jones & Hilcox 1995). Similar in appearance to the common brushtail possum (*Trichosurus vulpecula*), the western ringtail possum can be distinguished by its smaller rounded ears and thin prehensile tail (DEWHA 2010). This difference in appearance allows reasonably easy identification from the common brushtail possum in the wild.

2.2 Distribution: past and present

Once widely distributed across south-western Australia, the distribution of the western ringtail possum is currently patchy and restricted to south-western Australia (Figure 1). The considerable reduction in distribution has occurred with extensive local declines in the northern and inland parts of the species original range (de Tores 2000; DEWHA 2010). Much of the former habitat of the species has been cleared or fragmented during the agricultural development of south-western Western Australia (Jones *et al.* 1994a, de Toes *et al*. 2004, Jones 2004). The threatening processes contributing to the species decline include habitat loss, degradation, introduced species, disease and competition with the common brushtail possum (de Tores *et al.* 2004).

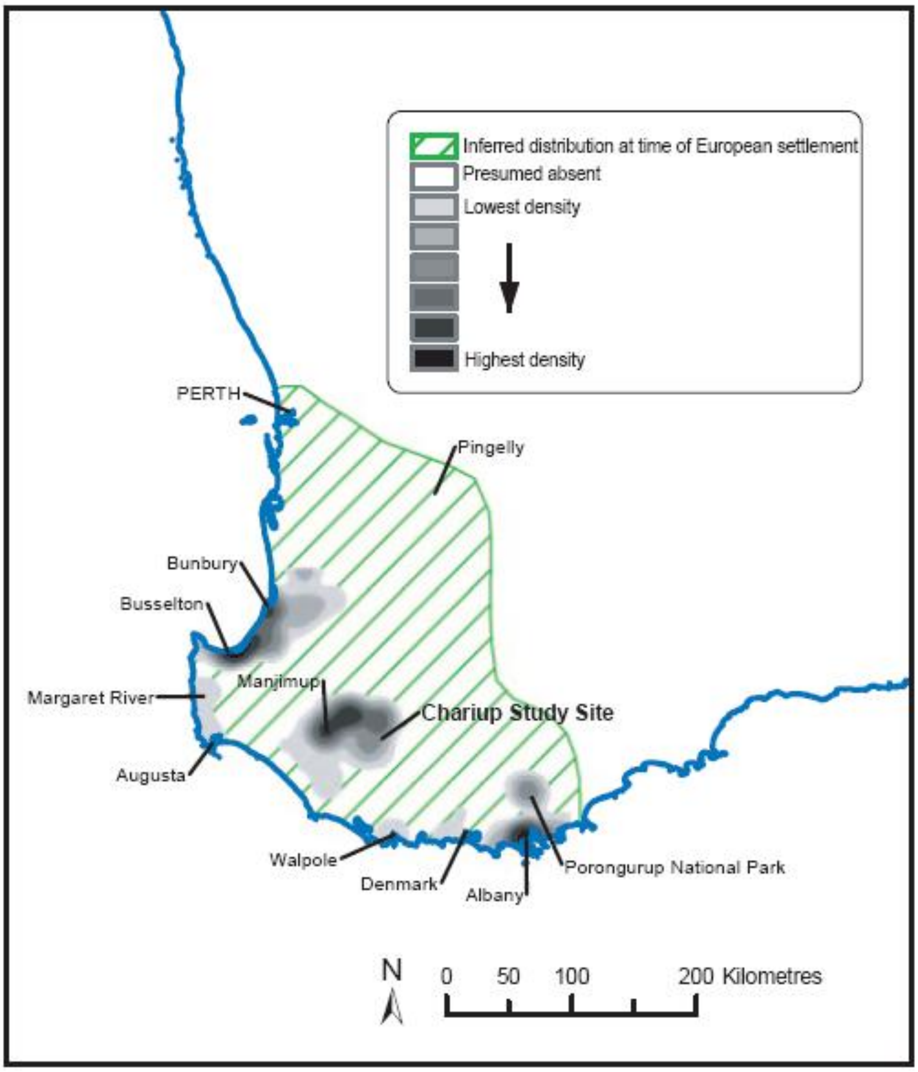

Figure 1: Present day distribution of the western ringtail possum (*Pseudocheirus occidentalis*) in the south-west of Western Australia. Green shading indicates the inferred distribution at European settlement. Darker shades of grey indicate higher density populations, and lighter shades of grey indicate lower density populations. Source: Wayne *et al.* 2005c

The species is most commonly recorded in coastal or near coastal habitat where peppermint trees (*Agonis flexuosa*) are a major component of the vegetation (Jones *et al.* 1994a; de Tores *et al.* 2004). This peppermint tree association may be linked to the level of nitrogen in the species diet (Jones & Hilcox 1995). Further, the abundance of hollow-bearing trees is considered to be a determinant of western ringtail possum's habitat (Jones & Hilcox 1995). The decline in western ringtail possum abundance has been most extensive in inland areas where canopies tend to be more dispersed than in coastal and near coastal habitat (Jones and Hilcox 1995). This preference for denser canopies may be explained by the possible need for possums living in open canopies use ground travel to access feeding and rest sites. On the other hand, those living in more dense canopies may access more of their resources within the home range canopy (Jones and Hilcox 1995). Therefore the western ringtail possum's contraction in distribution can be attributed, in part, to the species preference for dense, peppermint tree habitat.

However, peppermint trees are not always the dominant upper storey species in western ringtail possum habitat (DEWHA 2010). In 1993, a population at Ludlow, near Busselton, was considered to the most important population help by the Department of Environment and Conservation (DEC). At Ludlow the dominant upper storey species are Tuart trees (*Eucalyptus gomphcephala*). However, peppermint trees make up a prominent mid-storey at Ludlow, and therefore this may contribute to the habitat being suitable for the western ringtail possum (Jones & Hilcox 1995). One location, which is unique in terms of vegetation, is located at Perup Nature Reserve, near Manjimup. Perup is the most inland population and it the only population known to occupy forest that does not contain peppermint trees (DEWHA 2010).

The distribution of the species has become more fragmented and constricted with time. In 2007, records estimated the extent of occurrence to be 7155km^2, and the area of occupancy to be 3700km^2 (DEWHA 2010). Seven locations presently support populations and five support translocated populations (DEWHA 2010). However population information on the western ringtail possum is not consistently available from across its range, with some local populations having been subject to long-term study, whilst other areas have not received basic surveys to determine presence (de Tores *et al.* 2004). This presents a problem in terms of the conservation of all remaining populations.

Overall, detailed population numbers are unknown (no published estimates). Estimates are difficult to obtain due to difficulties in surveying the species, and the scattered nature of populations (de Tores *et al.* 2004; DEWHA 2010). However, the overall population trend is thought declining (DEWHA 2010). The main contributor to this decline has been the selective clearance of high value habitat for agriculture (Wayne *et al.* 2006). Furthermore, as western ringtail possums feed and are more predominantly in the canopy, they may be limited in their ability to disperse and recolonise areas across fragmented landscapes (DEWHA 2010). Therefore, overall, greater abundance of the western ringtail possum occurs in areas with limited anthropogenic disturbance (de Tores *et al.* 2004).

2.3 Habitat

Western ringtail possums are arboreal, spending most of their time in the canopy (Jones *et al.* 1994b; Wayne *et al.* 2005c). The main determinant of their habitat is the presence of the peppermint tree, either as the dominant tree species, or as an understory component (DEWHA 2010). Populations therefore occur in various habitats including coastal or near-coastal peppermint, Tuart forest with a peppermint understorey, jarrah forest, jarrah-marri forest associated with peppermint trees (near Collie de Tores *et al.* 2004) and riverine stands of peppermint (near Harvey River) (DEWHA 2010). These preferred peppermint habitats are typically located close to water courses, swamps, or on flood plains (Jones *et al.* 1994a; de Tores *et al.* 2004).

Western ringtail possums have a small home range, relative to other possum species (Gibbons & Lindenmayer 2002; Clinchy *et al.* 2004). An individual western ringtail possum's home range is usually less than five hectares, and in high density populations on the southern Swan Coastal Plain home ranges can be below one hectare (DEWHA 2010). The home range of the western ringtail possum is dependent on interacting factors such as population density, habitat productivity and climatic conditions, which can influence shelter selection (Jones *et al.* 1994a, 1994b, Jones & Hillcox 1995). Therefore, while the species home range is relatively small in comparison to other species, it does vary depending on location-specific factors.

During daylight hours, the western ringtail possum preferentially rests singly (or with young) in tree hollows or dreys (nests constructed from vegetation) (Jones *et al.* 1994b; Jones & Hilcox 1995). Although the western ringtail possum is predominantly arboreal, Jones *et al.* (1994b) suggested that on-ground rest sites (e.g. hollow logs) may be preferentially used during high temperatures. In the southern Swan Coastal Plain, areas with an understorey containing sword sedge and *Lepidosperma* species are also important habitat areas for the western ringtail possum (de Tores *et al.* 2008). Therefore, the requirement for a rest site influences which habitat within a location individuals may prefer.

Western ringtail possums also have a preference for habitat with little evidence of burning and a continuous upper or mid-storey strata canopy (Jones *et al.* 1994b, DEWHA 2010). Canopy continuity is therefore another important predictor of western ringtail possum abundance, with the highest density populations occurring in areas with higher canopy continuity (Jones *et al*. 1994a). Correspondingly, the highest density populations are generally found in mature peppermint remnants. Overall, Jones and Hilcox (1995) identified four habitat parameters for defining western ringtail possum habitat quality including; forest florisitcs, the abundance of foliage, the nutritional quality of the foliage and the abundance of tree hollows. Of these parameters, the nutritional value of foliage is identified as the most important habitat determinant for the species (Jones & Hilcox 1995).

2.4 Life History

The western ringtail possum has a low breeding capacity and breeding which is seasonal (DEWHA 2010). Females usually give birth once a year to one to three offspring and although breeding can occur at any time of year, it is most common during autumn (April-June) (Jones *et al.* 1994b). In the southern jarrah forest, however, western ringtail possum breeding has been found to occur between September and October (Wayne *et al.* 2005c). This variation in breeding season may be due to a different timing in availability of food. Autumn breeding ensures that lactation and weaning, which have the highest nutritional demands on females, occur in spring and summer when new shoots are abundant (Jones *et al.* 1994a, 1994b; Wayne *et al.* 2005c). The importance of nutrition in the timing of breeding has been shown by Ellis and Jones (1992). The authors monitored a captive colony of western ringtail possums which were fed on a diet of vitamin and nutritional and mineral supplements. Seasonality of breeding was linked to nutritional constraints (Ellis & Jones 1992). Therefore, breeding in western ringtail possums is highly seasonal and linked to nutritional availability.

The western ringtail possum also has a relatively short life span compared to other possum species (Lindenmayer *et al.* 2000, Wayne *et al.* 2005d).On average, the life span of a western ringtail possum is two to three years (DEWHA 2010), however they are thought to live three to five years in the wild (de Tores *et al.* 2004; de Tores *et al.* 2008). The oldest observed age of a western ringtail possum is the jarrah forest is four years (Wayne *et al.* 2005c). Therefore, from what has been observed of captive and wild populations, the western ringtail possum has a relatively short life span.

Western ringtail possum reproduction has also been studied in both captive and wild populations. After two to four weeks gestation, western ringtail possum young remain in the pouch for three to four months (Jones *et al.* 1994b). Young are weaned at six to eight months and disperse at eight to twelve months (Wayne *et al.* 2005c). The young gain independence at six to seven months, although young (especially females) may stay with their mother for up to four or more months. Both captive and wild populations in the Jarrah have been documented as breeding at 12 months (Ellis & Jones 1992; Wayne *et al.* 2005c). Given this breeding capacity, relatively short life-span and susceptibility to predation, all known populations are considered essential for species recovery and long-term survival (DEWHA 2010).

The sex ratio of populations also plays a role in the viability of each western ringtail possum population. At Abba River (part of the southern Swan Coastal Plain population), the sex ratio is documented as being equal in populations but by contrast, a significant female bias has been found at the Locke and State populations (Wayne *et al.* 2005c). Wayne and colleagues (2005c) speculate that an equal sex ratio is indicative of a stable population. They also hypothesised that female bias indicates an expanding population in high quality habitat and male bias may indicate marginal to declining resource conditions. Therefore, a combination of a male biased sex ratio may threaten each population's long-term survival.

2.5 Diet

Western ringtail possums have a diet that primarily consists of leaves of peppermint trees (around 90%, and to a lesser extent those of other myrtaceous plant species) (Sheppard *et al.* 1997, de Tores *et al.* 2008). Coastal populations, have been found to have the typical peppermint leaves dominant in their faecal samples, and subsequently their diet (Sheppard *et al.* 1997). However, at Perup Nature Reserve, where peppermint trees are not present, faecal samples were found to only contain jarrah forest canopy species (Sheppard *et al.* 1997). Of these species, jarrah leaves were the predominant species eaten, but western ringtail possums also have been known to supplement their diet with seeds, flowers and insect larvae (Sheppard *et al.* 1997). Western ringtail possums also have a low field metabolic rate, much lower than expected for their body size (Sheppard *et al.* 1997). With such low energy requirements, the species shows decreased mobility compared with more active brushtail possums (Sheppard *et al.* 1997). Consequently, a western ringtail possum's diet varies according to their habitat.

2.6 The Western Ringtail Possum as a rare species

Threatening processes

Many processes have contributed to the decline in western ringtail possum abundance. The main threatening process acting on the species has been clearing and habitat fragmentation (Burbridge & de Tores 1998; Wayne *et al.* 2006). Clearing of coastal peppermint woodland, particularly in the Busselton and Albany areas, has contributed to fragmentation habitat loss (DEWHA 2010). Loss of habitat combined with the species specialised habitat and diet requirements, has resulted in a significant decline in population size.

Another threatening process central to the species decline has been the introduction of the European fox. The species is classified as being in the critical weight range (35g-5.5kg) and has therefore been significantly affected by fox predation. The importance of fox predation has been demonstrated by successful re-introductions to Leschenault Peninsula Conservation Park and Yalgorup National Park, where fox control is in place (de Tores *et al.* 2004). As a result, a combination of loss of habitat and an introduced predator has seen the western ringtail possum put under severe threat.

Logging and prescribed burning also significantly threaten populations. On a local scale, western ringtail possum abundance is negatively associated with greater fire intensity (Wayne *et al.* 2006).The species is present in areas that have been logged and which have been subjected to several prescribed burns, e.g. near Perup and near Collie (Wayne *et al.* 2006). Inions *et al.* (1989) found that a single high intensity fire at Perup destroyed 38% of trees previously inhabited by western ringtail and common brushtail possums. However, the total number of trees used by possums increased after the fire, with new hollows being created (Inions *et al.* 1989). Therefore, logging and fire regimes threaten the western ringtail possum, however, fire has the potential to create new rest sites.

Another threatening process is urbanisation, especially around the Busselton area. The development of urban areas has results in fragmentation of populations, and individuals re-invading newly developed areas (DEWHA 2010). Anthropogenic disturbance has also been negatively associated with western ringtail possum abundance (Wayne *et al.* 2006). As a result of increased urbanisation and dissection of the species habitat, road kills now also present a threat. In addition, disease also has the potential to threaten the species survival (DEWHA 2010). A number of animals that died at both the Busselton populations and at translocation sites have shown moderate-high levels of endo and/or ecto parasitism (McCutcheon *et al.* 2007). Inter-specific competition with the common brushtail possum also poses a threat to populations, competing for resources. Therefore, a number of threatening processes exist for the western ringtail possum with the most important and current threat being habitat loss.

Translocations

Due to continued habitat loss a translocation programme has been in operation since the mid-1990s (McCutcheon *et al.* 2007). Displaced western ringtail possums have been translocated from building development sites in the Busselton region to national parks further north in an attempt to re-establish viable populations within their previous range (McCutcheon *et al.* 2007; DEWHA 2010). To date, most western ringtail possum site-specific research has been associated with translocation release programs of rehabilitated orphaned and/or injured possums and possums displaced as a result of vegetation clearing and habitat loss at development sites (de Tores *et al.* 2004).

Translocation may also not always be the preferred option for management of the western ringtail possum (de Toes *et al.* 2008). Translocation success is yet to be confirmed for several populations including; Leschenault Peninsula Conservation Park, Yalgorup National Park (Whitehill and Preston beach), Lane Poole Reserve and Keats Forest Block (within northern Jarrah forest south-east of Dwellingup), Karakamia Wildlife Sanctuary and Locke Nature Reserve (de Tores *et al.* 2004). The translocation to Leschenault Peninsula was initially successful, but a population crash occurred at that site between 1998 and 2002 for indeterminable reasons (de Tores *et al.* 2004). However, tentative results suggest viable populations have been established at Yalgorup National Park and Lane Poole Nature Reserve (de Tores *et al.* 2008). Therefore, translocation as conservation tool is yet to be determined for a number of populations.

Karakamia Wildlife Sanctuary is a small enclosed sanctuary 50k east of Perth, Western Australia. The translocation site presents a case where western ringtail possums were released at the northern end of their previous range. While no published data exists to demonstrate the success of the translocation (de Tores *et al.* 2004), Karakamia provides a situation in which to study translocation success on a small scale. Data are available that indicate the sanctuary has an extant population (P. Gardner & J Richards pers. comm.*).* After the initial translocation phase and follow up, limited study has been conducted on the released individuals and into whether recruitment is occurring.

2.7 Review of the methods used to estimate Western ringtail possum abundance

Traditional

Traditional sampling of rare arboreal species include direct (trapping, spotlighting) and indirect (scat counts, hair tubes etc.) counts (Davey 1990; Wayne *et al.* 2005b; Wintle *et al.* 2005). Spotlighting is the most widely used field technique for studying Australian arboreal marsupials (Lindenmayer *et al.* 1999; Soderquist & MacNally 2000; Kanoski *et al.* 2001; Lindenmayer *et al.* 2001; Van der Ree & Loyn 2002; Wormington *et al.* 2002), and this is the case for the western ringtail possum (de Tores *et al* 2004; Wayne *et al.* 2005b). However, there are some limitations associated with the spotlighting technique, such as the risk of poor detection efficiency (data return for effort) (Wayne *et al.* 2005a).

The low population size of most western ringtail possum populations means that traditional survey mark-recapture-release methods are not applicable (Burbridge & de Tores 1998; Wayne *et al.* 2005b). In addition, the western ringtail possum is not amenable to conventional trapping techniques and therefore these have therefore not been applied in survey efforts (de Tores *et al.* 2004). Therefore, in order to efficiently estimate western ringtail possum density, the best approach is to use a spotlighting technique.

Spotlighting for western ringtail possums also should take place between October and April, due to greater detection efficiency in these months (Wayne *et al.* 2005a). Furthermore, spotlight surveying should be avoided on rainy days or during particularly cold weather (i.e. winter in the jarrah forest) (Wayne *et al.* 2005a). As a result, certain conditions will result in a greater western ringtail possum spotlighting detection efficiency.

Distance Sampling

The sampling of rare species presents a number of challenges. First is that of how to successfully locate the species while still maximising effort (Rhodes *et al.* 2006). The second is that the results of rare species surveys often lack a statistical basis and therefore provide no further reliable understanding of a population of a species (Berkunsky & Williams 2009).

Therefore, in order to obtain a reliable density estimate of a rare species, a systematically based mathematical approach is required.

Distance sampling is a technique used for estimating the size or density of biological populations. The distance sampling technique comprises of a set of methods in which distances from a line or point to detections are recorded, from which density and/or abundance of animals is estimated (Thomas *et al.* 2010). Most distance sampling surveys are analysed using the software Distance. Distance sampling follows a specific survey design and applies a mathematical model to the data (Thomas *et al.* 2010). Therefore, distance sampling follows an information theoretic approach, statistically fitting distance data to an appropriate model (Thomas *et al.* 2010).

In the distance sampling technique, the observer would travel along each line, recording any animals detected within a distance *w* from the line (Buckland et al. 2001). In the standard method, we assume all animals on the line are detected, but detection probability decreased with distance from the line (Thomas *et al.* 2010). Therefore, not all animals in the strip half-width *w* need to be detected. Also, the distance of each detected animal is recorded. The distribution of these distances is then used to estimate the proportion of animals in the strip that are detected, which allows an estimation of animal density and abundance (Thomas *et al.* 2002; Thomas *et al.* 2010).

The lack of quantified abundance data for western ringtail possums reflects the difficulty of deriving reliable estimates for this species. Consequently this calls for a survey method with a good detection efficiency, or greater return for effort. Distance sampling provides a method to reliably estimate density and abundance. In the case of applying this to a population of western ringtail possums, distance sampling may be pertinent to the results, as they have implications for conservation.

Further understanding of the western ringtail possum has been constrained by the relative difficulty in surveying the species in the wild. In order to improve this understanding, survey methods depend on being able to detect reasonable numbers of individuals. Utilizing the distance sampling technique for a spotlighting survey will therefore increase the chances of obtaining a reliable density estimate, as the analysis calls for a minimum number of sightings, not individuals (Buckland *et al.* 2001).

Paul de Tores and colleagues (2004) estimated the density of western ringtail possums at Leschenault Peninsula Conservation Park (1996-1998) using distance sampling, and found an increasing population trend. However, for a further sampling year in 2002, only two possums were sighted. de Tores *et al.* (2004) stated that this demonstrated the translocation to Leschenault was yet to be shown to be an efficient management tool. The potential in this case for translocation to succeed appeared to have been compromised by changes to the fox baiting regime. The study by de Tores *et al.* (2004) highlights the need for long-term surveying of density of western ringtail possums. The use of distance sampling methods for calculating the density of at Leschenault Peninsula provides justification for utilizing the methodology in future studies of western ringtail possums.

Distance sampling has utilized to provide density estimates of a variety of rare species, such as the numbat (*Myrmecobius fasciatus*). Vieria *et al*. (2007) used distance sampling to estimate the numbat density at Scotia Sanctuary in New South Wales to determine if the translocation had been successful. Very few numbats were sighted, resulting in a density estimate of 1.24 individuals per 100ha (SE=0.56). The study therefore found that the species was capable of re-inhabiting areas within its former range.This, in turn, had implications for future management decisions relating to how and where to reintroduce the species in other parts of its former range.

2.8 Karakamia Wildlife Sanctuary

Karakamia Wildlife Sanctuary is owned by Australian Wildlife Conservancy and is located in the northern jarrah forest 50 km east of Perth, Western Australia. Initially 75 ha in size, the sanctuary was expanded to 260 ha as several adjacent private blocks of land were acquired. Landforms within Karakamia consist of an undulating lateritic plateau dissected by winter flowing watercourses feeding Cookes Brook, a permanently flowing stream that runs through the southern third of the sanctuary (P. Gardner & J Richards pers. comm.*)*.

Karakamia contains all the key vegetation associations found within the northern jarrah forest complex (Figure 2). The sanctuary is dominated by open jarrah forest, which occurs predominantly on the lateritic, marri *Corymbia calophylla* woodland on the slopes, and wandoo *E. wandoo* woodland on clays weathered from exposed granite. Vegetation on the granite outcrops ranges from lichen and moss to herbfields such as *Borya sphaerocephala* and shrublands dominated by *Hakea undulata*, *H. elliptica*, *Calytrix depressa* and *Grevillea bipinnatifida*. Lower down the slopes, flooded gum (*E. Rudis*) dominates and a riparian community occurs along the stream zones dominated by *Trymalium floribundum, Astartea fascicularis and Baumea spp*.

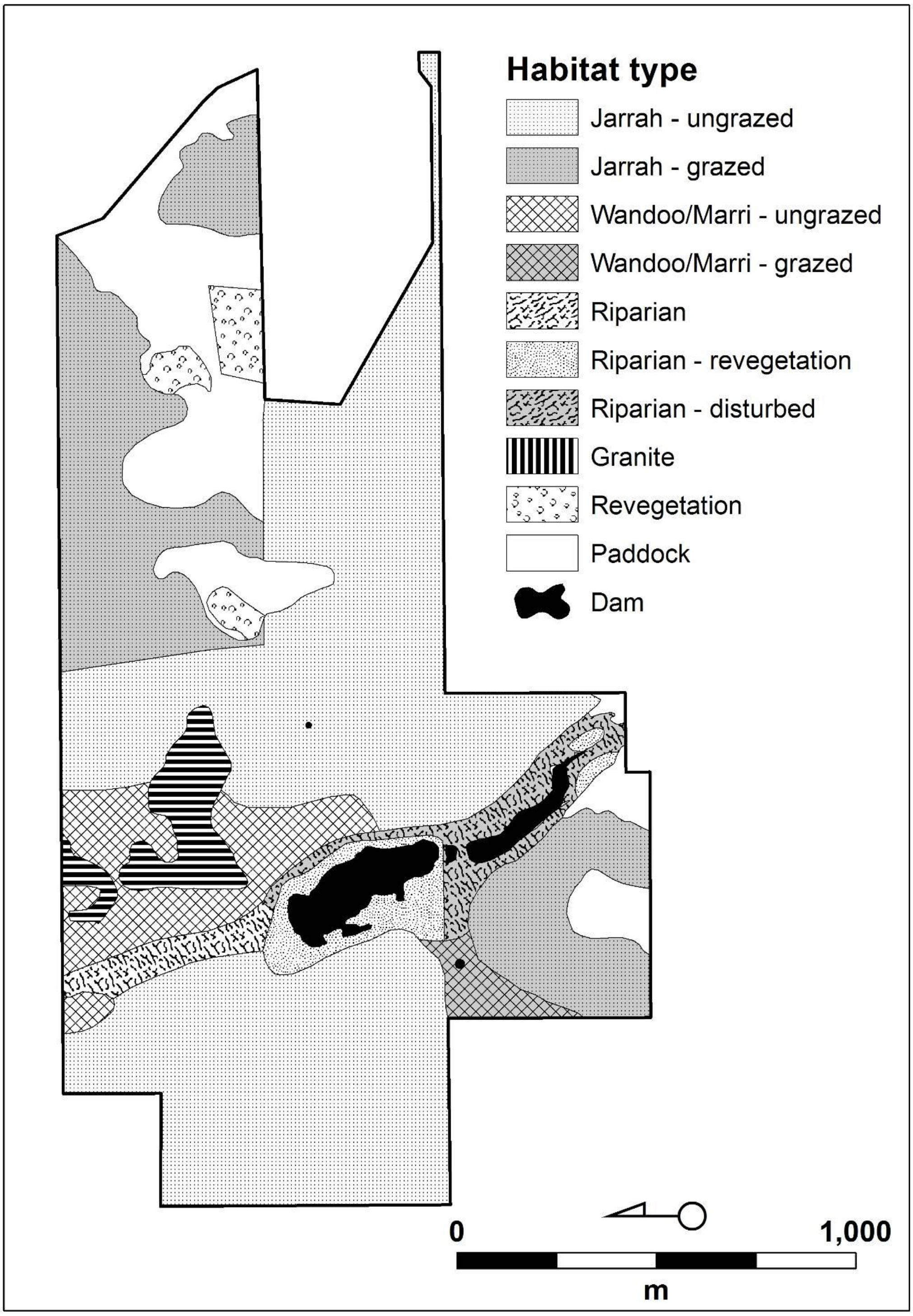

Figure 2: Broad habitat types within Karakamia Wildlife Sanctuary.

Source: Australian Wildlife Conservancy

Fire History

Karakamia was severely burnt in extensive regional bushfires in 1961, which followed years of fire exclusion. Other significant wildfires occurred in: 1971, which burnt the eastern slope of mixed jarrah, marri and wandoo forest and granite outcrops; 1988, which burnt out a small area on the eastern edge; and 1996, which burnt the south-eastern portion. Small prescribed burns have been conducted by AWC since 1991 on an annual rotational basis (P. Gardner & J Richards pers. comm.).

Control of introduced species

Karakamia was surrounded by a predator proof fence constructed in 1994. Prior to the construction of the fence, the eradication of European rabbits (*Oryctolagus cuniculus),* foxes (*Vulpes vulpes*) and feral cats (Felis catus) took place. This occurred between April and August 1994. Rabbits were baited with 1080 (sodium monoflouroacetate) laid at a bait station in areas of high rabbit activity. Karakamia was then baited for foxes using 1080 dried meat baits while feral cats were eradicated with a combination of trapping with large cage traps, baiting and shooting (P. Gardner & J Richards pers. comm.).

Translocations of western ringtail possums

Five ringtail possums were transported to Karakamia in nest boxes and placed in aviary style enclosures in riparian habitat (Figure 3). The animals were penned for up to four months while a dietary study was conducted in order to determine if suitable palatable species were available in the sanctuary. They were fed initially with *Agonis flexuosa* leaves, which comprised a significant part of their diet of the source population in the Busselton region (Jones *et al*. 1994a). A wide range of other species were supplied due to the paucity of *A. flexuosa* within Karakamia, including *Eucalyptus rudis*, *E. wandoo, E. patens, E. marginata, Corymbia calophylla*, *Taxandria linearifolia* and *Acacia saligna*. Once the ringtail possums were readily consuming a variety of plant species, the pens were opened and animals soft released in the riparian habitat near Cookes Brook (Figure 3). Supplementary food was provided in the pens for four weeks after release (P. Gardner & J Richards pers. comm.*)*. An additional 37 individuals were added to the population in subsequent years, including 14 from a development site and the remainder from wildlife carers. All animals were of Busselton provenance (DEWHA 2010).

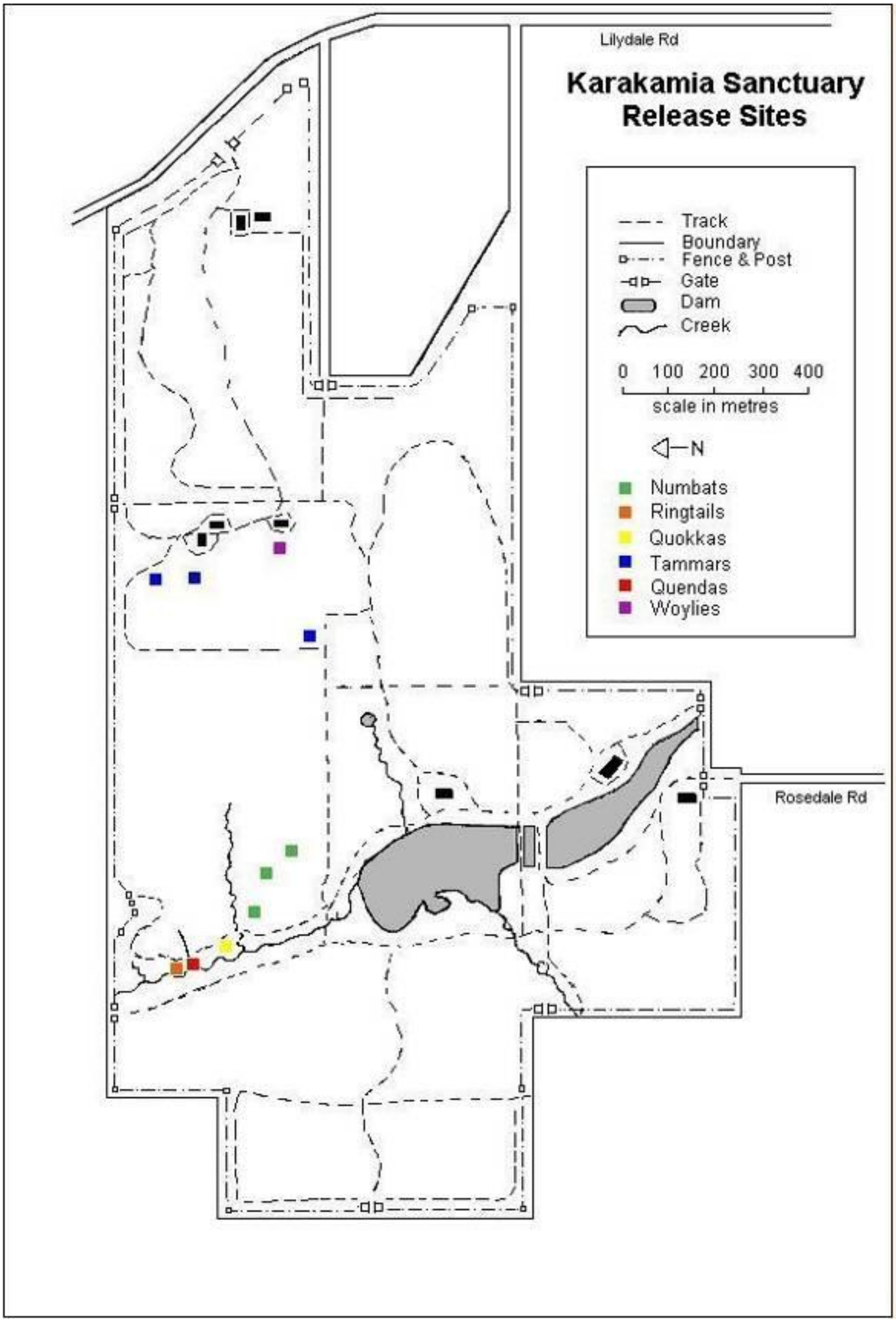

Figure 3: Map of Karakamia Wildlife Sanctuary showing the initial release sites of translocated mammals. Translocated western ringtail possums were released in the riparian habitat along Cookes Brook (shown by orange square).

Source: Australian Wildlife Conservancy

Translocated western ringtail possums were monitored with a combination of radio tracking and spotlighting as they were difficult to trap due to high inter-specific competition for traps from brushtail possums and woylies. Capture of ringtail possums was predominantly by hand from their nests during the day.

Translocation results: 1995-2000

All five ringtail possums survived translocation and at one month post release all had survived. At 12 months post release two of the founders were known to be alive and one had successfully produced a pouch young. An additional five founders were added to the population within the first 12 months. At three years' post release, several unmarked adults had been observed. Breeding of the second generation was confirmed with the capture of an untagged lactating female. At five years' post release, the population was extant (P. Gardner & J Richards pers. comm.*)*.

Thirty six individual ringtail possums were captured between 1995 and 2000. During this period radio tracking revealed high mortality rates: five from unknown causes, two by carpet python predation, two by goshawk predation, one from unknown predation, three from disease and one from wildfire. Heat stress was thought to be the most likely factor as 12 of the 14 individuals were found dead during the warmest months (December to March). Observations of ringtail possums on hot days > 35^{0}C at Karakamia found individuals obviously heat stressed and in exposed conditions (P. Gardner & J. Richards pers. comm.). As ringtail possums were difficult to trap, monitoring was conducted predominantly using radio tracking and spotlighting. Problems with collars were encountered, especially premature collar failure.

As predicted animals predominantly inhabited the riparian zone (Jones *et al.* 1994a) but were found also in other habitats within the sanctuary (M. Page pers comm.). Other habitats included two individuals located near the visitors centre and two individuals in the thick *Casuarina obesa* understorey. The riparian habitat favoured by the western ringtail possum is also the coolest location within the sanctuary. This is due to Cookes Brook flowing all year round. The habitat is very limited within Karakamia, and does not contain peppermint trees (P. Gardner & J. Richards pers. comm.). In Busselton where the population was sourced, this species is found to make up 79-100% of their diet. However, *Eucalyptus calophylla* and *Eucalyptus marginata* has been found to be a suitable substitute in other locations without *Agonis flexuosa* (Jones *et al.* 1994b). There is no evidence of whether this changed diet may reduce the species ability to cope with other changed conditions associated with the translocation

In the future, it is possible that the population will not persist due to low recruitment, high mortality rates, competition with brushtail possums, limited riparian habitat and dispersal out of the sanctuary (P. Gardner & J. Richards pers. comm.). It is also important to note that western ringtail possums were not documented from the northern jarrah forest in recent historic data by earlier studies such as Dell (1988),however earlier records do exist. This lack of recent data regarding the presence of the western ringtail possum within the Darling Scarp raises questions about the suitability of Karakamia as habitat for the species. Therefore a present-day study of abundance is needed to determine whether a population is still extant within Karakamia.

3. Aims and significance

3.1 Aims

The aims of this project are:

- To obtain a density estimate of western ringtail possums in Karakamia Wildlife Sanctuary
- To assess the persistence of the translocated population of western ringtail possums at Karakamia Wildlife Sanctuary

The hypothesis of this project is:

- There will be a greater density of western ringtail possums in the riparian habitat than in the Jarrah forest habitat at Karakamia Wildlife Sanctuary

3.2 Significance

Conservation of every remaining population of western ringtail possums is vital to the recovery and long-term survival of the species. As Karakamia is an enclosed sanctuary without the presence of predators and therefore may be an area where a small population may be able to persist. However, the translocation of the western ringtail possum is yet to be proven successful for population persistence in Karakamia Wildlife Sanctuary. Consequently, estimating the density of western ringtail possum population within Karakamia will provide an indication of translocation success.

The sanctuary is also situated in the northern most limit of the species previously inferred range, near Childow Western Australia. The site is unusual for the species in its present distribution, in that it contains jarrah forest habitat without the presence of peppermint trees. However, due to previous inhabitation, the western ringtail possums must have lived in the northern jarrah forest without peppermint trees in the past. Therefore, study of this population and a comparison of western ringtail possum density to other locations with peppermint trees will offer greater understanding of whether the habitat within Karakamia is suitable.

The results will provide the first density estimate of a translocated population of western ringtail possums within a enclosed sanctuary. As the predominant threats of habitat loss and fox predation have been removed at Karakamia Wildlife Sanctuary, it provides an opportunity to study whether a population can survive in the future.

Another aim of this study is to assess the persistence of the translocated population of western ringtail possums released between 1995 and 2005. In order to make an assessment of persistence, the reproductive biology of western ringtail possums needs to be taken into account. In the literature, the western ringtail possum has been noted as having a relatively short life-span and low breeding capacity. In order to fully assess the persistence, the growth rate and recruitment would need to be measured for the Karakamia population. As western ringtail possums are not amenable to trapping methods, this information cannot be determined. Therefore, for the translocation for the populations to be considered persistent at least 100% of the original number of translocated individuals would need to be present within the sanctuary.

The criteria set by AWC for a successful translocation included having population persistence or increase five years post translocation, with no requirement for additional translocations. This is supported in the literature as a way of determining success of a translocation (Richard & Short 2003; Mawson 2004). Therefore, assessing the persistence of the western ringtail possum population at Karakamia with provide AWC with an assessment of the success of their translocations.

My study will aim to estimate density and determine whether, in fact, the population size is the same or greater than the number of translocated individuals. Therefore, I will also compare my estimate with other populations in south-western Australia, to establish whether the population within Karakamia is persistent after translocation. The persistence of the population will also have consequences for its management in the future by AWC. Therefore, by determining the density of western ringtail possums, this estimate can be used as a comparison for future density surveys.

4. Research Plan, Methods, and Techniques

4.1 Research plan

The research project will be divided into two phases. Initially, a pilot study will be carried out at Karakamia Wildlife Sanctuary to determine the presence and habitat preference of the western ringtail possum. The second phase will involve a spotlighting transect survey to determine the density of the western ringtail possum in Karakamia.

A brief outline of the experimental procedures to be used during the course of this project is set out in the following sections.

4.2 Methods and techniques

Study area

I will conduct this study at a site within Karakamia Wildlife Sanctuary. Since 1995 woylies (*Bettongia penicillata*), tammar wallabies (*Macropus eugenii*), numbats (*Myrmecobius fasciatus*), quendas (*Isoodon obesulus*) and quokkas (*Setonix brachyurus*) have been released into the sanctuary as part of a multi-species translocation (P. Gardner & J. Richards pers. comm.). A total of 42 western ringtail possums have been released at Karakamia between 1995 and 2005 (P. Gardner & J. Richards pers. comm.)

Pilot Study

I will conduct a pilot study at Karakamia Wildlife Sanctuary in April 2010. The aim of this study will be to determine the presence of western ringtail possums in the sanctuary. The pilot study will also determine the habitat preference of western ringtail possums and confirm whether the highest concentration of individuals occurs in the riparian habitat surrounding Cookes Brook.

I will carry out a presence survey of Karakamia through conventional survey techniques,following the roads within the sanctuary travelling by foot and by vehicle. I will spotlight for western ringtail possums using a 30-W spotlight provided by DEC. Any individuals sighted will be recorded with a GPS (Global Positioning System) receiver. The tree species, sex and age of each individual will also be recorded if possible.

The GPS locations of each individual sighted will then be plotted in the Geographic Information System (GIS) program ArcMap™ over a vegetation map of Karakamia. This will enable me to determine where the western ringtail possums are mostly occurring within the sanctuary. The pilot study will therefore provide the layout of the survey design for my project, based on the need to sample more intensely in better quality habitat.

Spotlight sampling techniques

I will conduct most fieldwork between May and June 2010. To obtain density estimates I will conduct surveys by walking transects within selected areas of Karakamia Wildlife Sanctuary. Following de Tores *et al.*(2004), I will walk transects after sunset. One observer will be consistent for every survey (L. Zimmermann) and a minimum of one other observer will be present per survey. For each individual sighting I will record the coordinates of the animal's location using a GPS.

Adult western ringtail possums (*Pseudocheirus occidentalis*) will be spotlighted along fifteen transects of varying length. For each possum observed, the perpendicular distance to the transect line will be recorded. Possums will not be recorded while away from the transect line, but rather once I have returned to the transect. Perpendicular distance will be determined using a handheld GPS connected to a baseline GPS set up at the office at Karakamia. Accurate readings from the baseline GPS will allow the exact perpendicular distance to the transect line to be calculated, through post-processing. Line transects will be established through the site, which will then be walked and the distances from the transect line to observed possums recorded. Each transect will be walked once per survey night. Each survey night will be as consecutive as possible to provide a density estimate for the particular season that is similar in terms of environmental variables such as temperature, wind and rainfall. This will minimise bias of sampling for the season over a longer time period.

At this stage in the project, the exact positioning of the survey sites has not been determined as it the presence of western ringtail possums within Karakamia to be confirmed and the preferred habitat to be assessed with the conclusion of the pilot study. Habitat will be defined on the basis of floristic and structural vegetation and mapped using GIS software ArcMap™.

Once the pilot study has been conducted, the exact location of the transects will be plotted. This will ensure that sampling effort is concentrated in habitat preferred by the western ringtail possums at Karakamia. To accurately estimate the density of possums the distance sampling technique requires 60-80 sightings. As only 42 western ringtail possums were released in the sanctuary, a larger sampling effort is needed to assess the possibly small population at present. Therefore, I will continue to survey the population until I reach the minimum number of 60 sightings.

In order to ovoid overestimation of the population, the survey will take place in autumn and winter (May-June), when few young are present in the population. Even though Wayne *et al.* (2005a) found the greatest survey efficiency for western ringtail possums to be between the months of October and April, due to the timing of my project, I will only be able to carry out my study between May to June.

As many factors can affect the proportion of the population observed during a survey**,** environmental conditions will be recorded on each survey night. For example, time of day or can affect the results and therefore will be standardized for every survey night. Also as cited in the literature, western ringtail possums are difficult to detect in rain and particularly cold temperatures, and so nights with these conditions will be avoided.

Data analysis

I will estimate western ringtail possum density using Distance Sampling protocols and the Line Transect option of the software DISTANCE 6.0 Release (Thomas *et al.* 2009).

Model selection in the program DISTANCE is based on the Information-Theoretic approach and Akaike Information Criteria (AIC). A series of *a priori* candidate models will be compared and the best estimator based on minimum Akaike Information Criterion (AIC) will be fit to the data.

5. Timetable

The project will be conducted from February until October 2010. All practical components will be completed by the end of July.

	February				*March*				*April*				*May*				*June*				*July*				*August*				*September*				*October*			
Activity	*1*																																			
Research and preparation of proposal																																				
Conduct literature review																																				
Risk Assessment																																				
Pilot study presence survey																																				
Submission and presentation of proposal																																				
Implement distance sampling survey																																				
Analyse results																																				
Compile results for draft thesis																																				
Thesis writing																																				
Hand in draft thesis to supervisor																																				
Submission and presentation of final thesis																																				

6. Budget

Item	Units needed	Price	Total Cost
Jarrah stakes for marking transects	100	$100	$100
Reflective tape	1 roll	$118.80	$118.80
GIS ArcView 1 year license	1	$280	$280
PAWES course	1	$50	$50
Transport cost (UWA to and from Karakamia): $11.25 per trip			
- Preliminary Project (2-3 days)	2-3 trips	$11.25	$33.75
- Distance surveys (~16days)	16	$11.25	$180
Printing cost for copies of research proposal and thesis		$100	$100
Total			**$862.55**

6.2 Justification

The expense expected to be incurred in the proposed research fall into three main areas:

- Field travel expenses- to Karakamia Wildlife Sanctuary
- Fieldwork materials
- Expenses associated obtaining licence for GIS program ArcMap™

Travel to the field site is required to carry out surveys, however at this stage the exact number of surveys is not known. As I require a minimum of 60 sightings for my analysis, the number of trips will vary depending on how many western ringtail possums I detect per night.

The cost of fieldwork materials will be subsidised by the Department of Environment and Conservation. Fieldwork facilities and fieldwork materials will be also provided by AWC and therefore are not included in the budget.

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