UNIVERSITY OF TECHNOLOGY, SYDNEY

Impact of roads on swamp wallaby populations on Sydney's Northern Beaches

COMMERCIAL IN CONFIDENCE

This Project Report has been prepared by the University of Technology, Sydney (“UTS”) as represented by the Centre for Compassionate Conservation.

UTS CRICOS Provider Code 00099F

This report should be cited as:

Ramp, D., Dougherty, E. & Bino, G. (2014). Impact of roads on swamp wallaby populations on Sydney’s Northern Beaches. Final report to the Roads and Maritime Services. Centre for Compassionate Conservation, University of Technology, Sydney.

FOR THE ATTENTION OF:

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ABSTRACT

The persistence of wildlife in altered environments is increasingly problematic as urbanised and production landscapes become ever more developed. The tipping point for many wildlife populations is the expansion of road networks and subsequent fragmenting effects. Patches of remnant vegetation become restricted in size and become infiltrated by roads with impacts that extend beyond the road edge. The primary mitigation method for addressing this complex problem is to either improve roads to allow safe passage of wildlife (through installation of under- or over-passes) and/or to prevent access to roads by wildlife to reduce rates of road-kill (through fencing). Crossing structures are expensive and governments require confidence that costs directly secure population persistence. Although not as expensive, fencing carries with it ongoing upkeep costs and can restrict wildlife movement, thereby increasing isolation. The question of how to distribute resources to both crossing structures and anti-crossing structures is often site and situation specific, yet decision frameworks for examining the trade-off between the two are in their infancy.

Here we present the case of a medium-sized mammal, the swamp wallaby (*Wallabia bicolor*), that appears to be eminently suited to surviving in urban remnants. However, roads are a major contributor to annual mortality rates. We surveyed four adjacent sub-populations living in reserves surrounded by suburbia, with each sub-population segregated by roads. We estimated population densities and annual fatality rates and used this information to examine the sensitivity of each sub-population to different levels of connectivity (movement across roads). We found that Ku-Ring-Gai Chase National Park was acting as a source population for Garigal National Park and surrounding regions. At current rates of fatalities and without connectivity, the likelihood of localised extinction in sink populations (Garigal National Park and surrounds) was high. With a large enough source population, continual migration to threatened sub-populations is likely to maintain sub-populations, but increasing isolation from the source will result in greater sensitivity to on-going fatalities.

Our approach enabled us to identify targets for different amounts of connectivity and fatality reduction that may be used in similar situations by environmental planners. Our findings suggest that complacency over the impacts of increased road usage on wildlife populations may lead to localised extinction due to the complexity of how populations function across landscapes.

INTRODUCTION

Land-use is the greatest threat to the conservation of biodiversity (Pimm and Raven 2000). Landscapes of all types are being altered to support a burgeoning human population that is resource hungry (Imhoff *et al.* 2004; Foley *et al.* 2005), while the acceleration of extinction rates of species globally highlights the pressing need to address threats to biodiversity and ecosystem processes and services (Sekercioglu *et al.* 2004). The challenge for conservationists is to identify threats to biodiversity (Possingham and Wilson 2005; Wilson *et al.* 2005), to quantify their impact on ecosystems and establish mechanisms to mitigate those threats.

Over the next few decades, the world's rapid urbanisation will be one of the greatest challenges to ensuring human welfare and a sustainable global environment. Three billion people live in cities world-wide and this number is expected to double over the next 50 years (Crane and Kinzig 2005). The number of urban areas with over one million people is expected to grow by over 40% between 2000 and 2015 (Crane and Kinzig 2005). There is now a growing concern for the conservation of biodiversity and the condition of global ecosystems stemming from this urban expansion (Daily 1997). In the USA, urbanisation has been identified as a primary cause, singly or in association with other factors such as roads, for declines in more than half the species listed as threatened or endangered under the U.S. Endangered Species Act (Czech *et al.* 2000; Miller and Hobbs 2002). Refugia in these urban landscapes are of vital importance for biodiversity conservation (Adams 2005). Sustainable land-use practises and pro-active protection of remnant fragments are essential to the preservation of biodiversity in these expanding regions (Margules and Pressey 2000). But these urbanised regions now have severely reduced wildlife populations, with many remaining species considered endangered (Flannery 2004). Roads present a major threat to the viability of populations restricted to these refugia (Ramp and Ben-Ami 2006; Ramp *et al.* 2006).

For the most part, remnant habitat exists as isolated refugia on the periphery of urban areas (termed peri-urban), impacted upon by a wide variety of disturbances such as landscape fragmentation, edge effects, truncated food webs, pollution, invasion from feral and domestic animals, invasive weeds, changes to natural disturbance regimes such as fire, high visitation rates and vehicular traffic (McDonnell 1997; Adams 2005). Urban remnants often provide the last pockets of refuge for species that were once widespread and are important for both local and regional biodiversity (Margules and Pressey 2000; Cornelis and Hermy 2004), often harbouring endangered species. Urban expansion poses a significant threat to these refugia, not only caused by habitat loss but also by the increase in transportation necessary to facilitate this expansion. Roads will likely increase in significance as a form of disturbance over the coming decades. It is therefore essential that knowledge of this process is obtained immediately so that conservation priorities can be set in place prior to major biodiversity loss in peri-urban landscapes.

The density, configuration and placement of roads in the landscape are important determinants of their overall impact on biodiversity (Forman and Alexander 1998; Jaeger 2001). However, their impact on the persistence of species has been largely neglected, although there has been a recent increase in landscape level studies on populations (Beaudry *et al.* 2008; Eigenbrod *et al.* 2009; Roger and Ramp 2009). The carrying capacity of a habitat and its quality will also affect species persistence. The size of habitat that a species needs to survive is central to where they can persist in the landscape and the ability of populations to cope with fragmenting of suitable habitat (Holling and Meffe 1996). The smaller the habitat patch, the lower the carrying capacity. This leads to smaller populations vulnerable to disease and seasonal fluctuations in conditions. Hence, the location of the roads, as well as their density, greatly impacts on the size of patches of land that are considered suitable habitat.

The linking of meta-population theory with landscape and community ecology is an essential step in conserving landscapes and preventing biodiversity loss (Margules and Pressey 2000; Burgman *et al.* 2005). It enables a focus on indicator species and incorporates spatial and temporal variation (Wagner and Fortin 2005), as well as uncertainty (Burgman *et al.* 2005), in decision making. The key issues for wildlife populations are how individual-based movement decisions around roads alter population-level use of space, how those decisions change with distance from the road, and the implications of these decisions, coupled with fatalities, on the viability of roadside populations.

Conceptual framework

Habitat-selection theory is uniquely placed to solve many conservation and wildlife management issues (Morris 2003). Accounting for dispersal and movement, favoured by natural selection (Morris 1991; Kokko and Lundberg 2001), enables a landscape-level approach to habitat selection (Morris and Brown 1992). Landscapes are inherently spatially heterogeneous, with habitat patches varying in suitability for wildlife (Ramp and Coulson 2002; Ramp and Coulson 2004; Maguire *et al.* 2005). Critical to evolutionary stable strategies ((ESS) Maynard Smith 1982) is the ability to disperse, without which there are serious implications for population and community dynamics (Morris *et al.* 2004). The dominant theoretical frameworks describing dispersal and movement are the source-sink model (Holt 1984; Pulliam 1988) and the balanced dispersal model (McPeek and Holt 1992; Doncaster *et al.* 1997). In the source-sink model, an ESS occurs when source populations produce emigrants for sink habitats, while in the balanced dispersal model, an ESS occurs when moving individuals are equivalent among patches. A new theory of reciprocating dispersal (Morris and

Diffendorfer 2004; Morris *et al.* 2004) has also been described that predicts that dispersal should be reciprocated among habitats. During times of population increase, the flow of individuals should be from one habitat to another, while during times of population decrease the flow should be reversed. These theories are perfectly suited to addressing the issues of conservation of roadside environments but surprisingly have not yet been used.

To understand this it is important to contextualise how species utilise landscapes of varying quality habitat. Patch dynamics suggests that animal populations can assess the distribution of resources in landscapes and respond accordingly (Pulliam and Danielson 1991). This has been called Optimal Foraging Theory (Fretwell and Lucas 1970). Not all patches within the landscape are the same: some are suitable for foraging and breeding (sources), while others are suitable only for foraging (sinks). Sinks are thought to be utilised until resources decline to the point where the costs of staying within the sink become too great. Sinks with negative consequences for fecundity and mortality have been termed ecological traps (Kokko and Sutherland 2001) or attractive sinks (Falcucci *et al.* 2009). Macropods have been shown to successfully utilise a form of optimal forage theory, called Ideal Free Distribution theory (Ramp and Coulson 2002; Ramp and Coulson 2004; Maguire *et al.* 2005), while source-sink dynamics in response to road fatalities have been shown to influence population persistence in common wombats (Roger *et al.* 2011). Habitat acting as an ecological trap or attractive sink (through high levels of road fatalities) is unlikely to support populations unless fed by source habitat.

Key to the ongoing functioning of meta-populations, comprised of both source and sink habitats, is the ability of individuals to distribute among sub-populations, fostering genetic variation through exchange of individuals. This exchange allows migration of individuals to areas which have suffered declines in population numbers, thereby making individual sub-populations more stable (Hansson 1991). However, road upgrades and increasing road density may reduce connectivity among sub-populations (Stamps *et al.* 1987) and decrease the availability of available habitat. Smaller habitats support fewer individuals; with smaller sub-populations more susceptible to extinction from threatening processes, like disease or changes to the environment (Lande and Shannon 1996; Hels and Nachman 2002). Reduction in movement due to roads among sub-populations may increase the level of inbreeding, resulting in offspring with higher rates of mortality and reduced fecundity (Lacy 1997). Declines in fecundity can result in fewer breeding individuals, further decreasing the stability of the populations (Reijnen and Foppen 1995).

Yet there has been little research that has attempted to quantify the trade-off between allowing for connectivity among source and sink habitats, which without ensuring safe passage increases the risk population decline, and reducing the frequency of road fatalities on overall population persistence.

This is a major impediment to decision making on road planning and modification in sensitive landscapes. The only way to scientifically address this question is to quantitatively assess how species are distributed throughout the landscape and to robustly test various scenarios of connectivity and road mortality on meta- and sub-population functioning.

Population density and abundance estimation

Knowledge of population density and abundance is often required as a first step in many conservation efforts, including those that seek to determine the viability of a certain population, the effects of various management techniques, or the rate of change in the face of increasing selective pressures (O'Connell *et al.* 2011). Different methods have emerged to estimate these parameters when a direct census count cannot be performed easily. Some of these techniques, such as transect counts and distance sampling, utilise direct counts through observation to extrapolate to a broader area (Buckland 2001). Other techniques, such as mark-recapture studies, actively trap individuals on multiple occasions to determine the total population size (Seber 1986). More recently, camera traps have been integrated into these approaches to minimise the effect of humans on the study population.

Algorithms have recently been developed to enable the estimation of animal density using camera traps that overcome the inability to discriminate amongst individuals (Rowcliffe *et al.* 2008; Rowcliffe *et al.* 2011). Rowcliffe *et al.* (2008) formulated a model to quantify the density and abundance of a species that does not exhibit individual markings. To do this, the investigators set cameras in an animal park with known abundances of four species including the red-necked wallaby (*Macropus rufogriseus*), a similar sized macropod to the swamp wallaby, and compared the density estimates obtained using camera trap data to those acquired by a census count of the study site, and concluded that their model was successful as long as camera placement was unbiased.

Here we present a new Derived Probability of Detection model, which incorporates a number of animal and camera-specific variables to obtain estimates of density and abundance for wildlife species moving within well-defined home ranges. This alternative probabilistic model accounts for the particular ecology of our study subject. Since non-random movement patterns are common in many wildlife species, our aim was to obtain more accurate density and abundance estimates of solitary animals with selective habitat usage in the form of well-defined home ranges with core areas of activity.

Project aims

This research sought to quantify the abundance and movement of swamp wallabies in reserves and national parks in Sydney's northern beaches area. The motivation was to assess how the wallaby population functions at the landscape scale, contrasting the importance of connectivity in maintaining stable meta-populations with the loss of individuals through collisions with vehicles on roads.

METHODS

Study location

Garigal National Park (GNP) (33°42′21″S, 151°14′11″E) is located 20 km north of Sydney in New South Wales, Australia (Figure 1). It is surrounded by high-density urban housing but is connected to the adjacent Ku-Ring-Gai Chase National Park (KCNP) via two narrow corridors of land. Mona Vale Road, a highly trafficked four-lane road, separates the two parks. GNP is a relatively small park at 22.54 square kilometres and contains extensive hiking and mountain biking trails that are open to the public throughout the year. GNP is further divided into two halves (east and west) by Forest Way, which itself is surrounded by residential and commercial developments. Wakehurst Parkway further segregates Dee Why West Recreation Reserve from East GNP. Several distinct habitat types occur, ranging from open forest in gullies to open woodland, scrub and heathland in flatter areas.

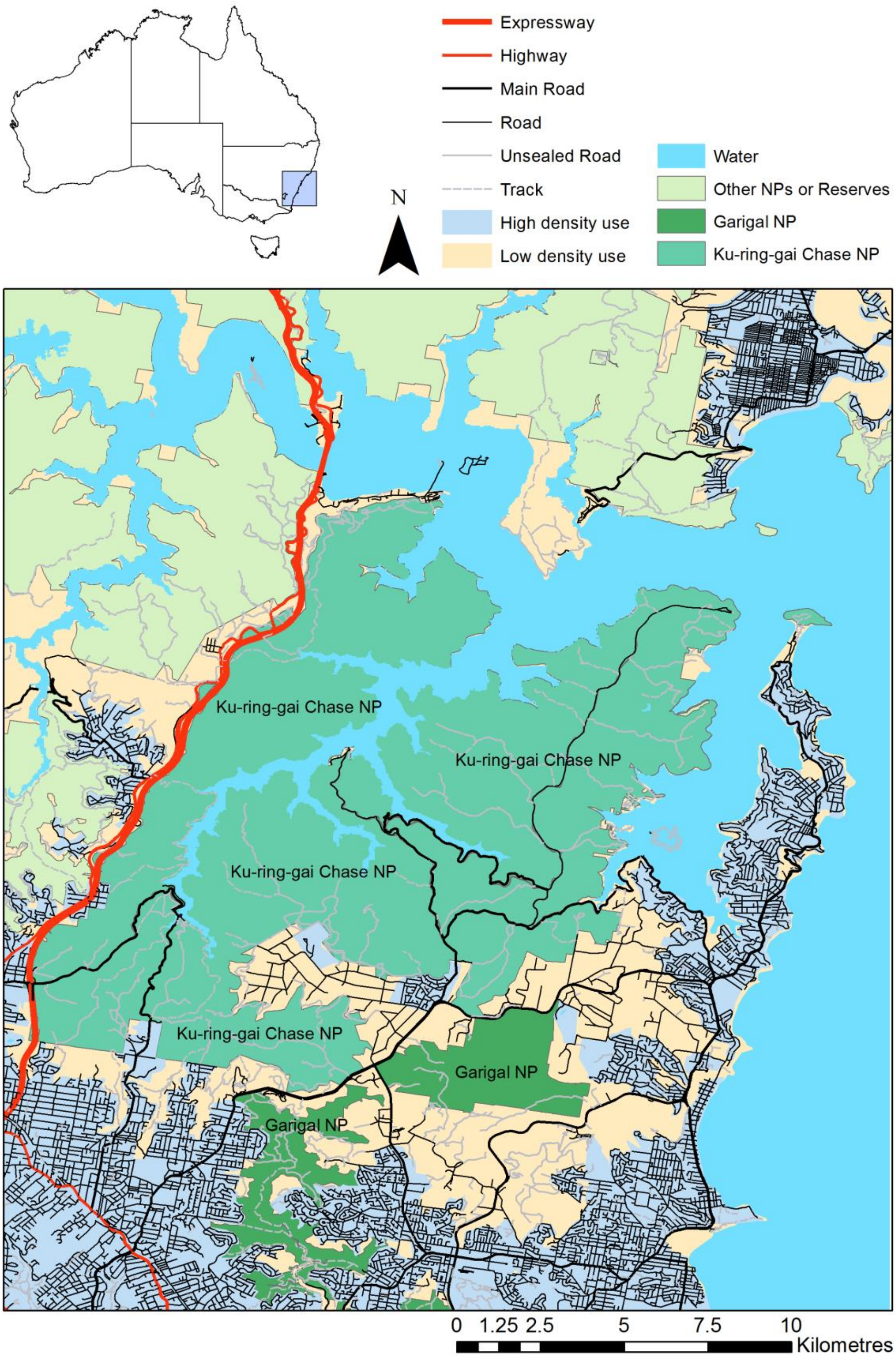


Figure 1 Northern Beaches study area 20 kilometres north of the city of Sydney, New South Wales. Urban areas dominated by housing and peri-urban areas with fewer houses are represented by grey/blue and cream respectively. The M1 Pacific Motorway forms an impenetrable barrier to movement from surrounding parks and bushland into Ku-ring-gai Chase National Park. Garigal National Park is further separated from Ku-ring-gai by Mona Vale Road but swamp wallabies currently disperse across this road.

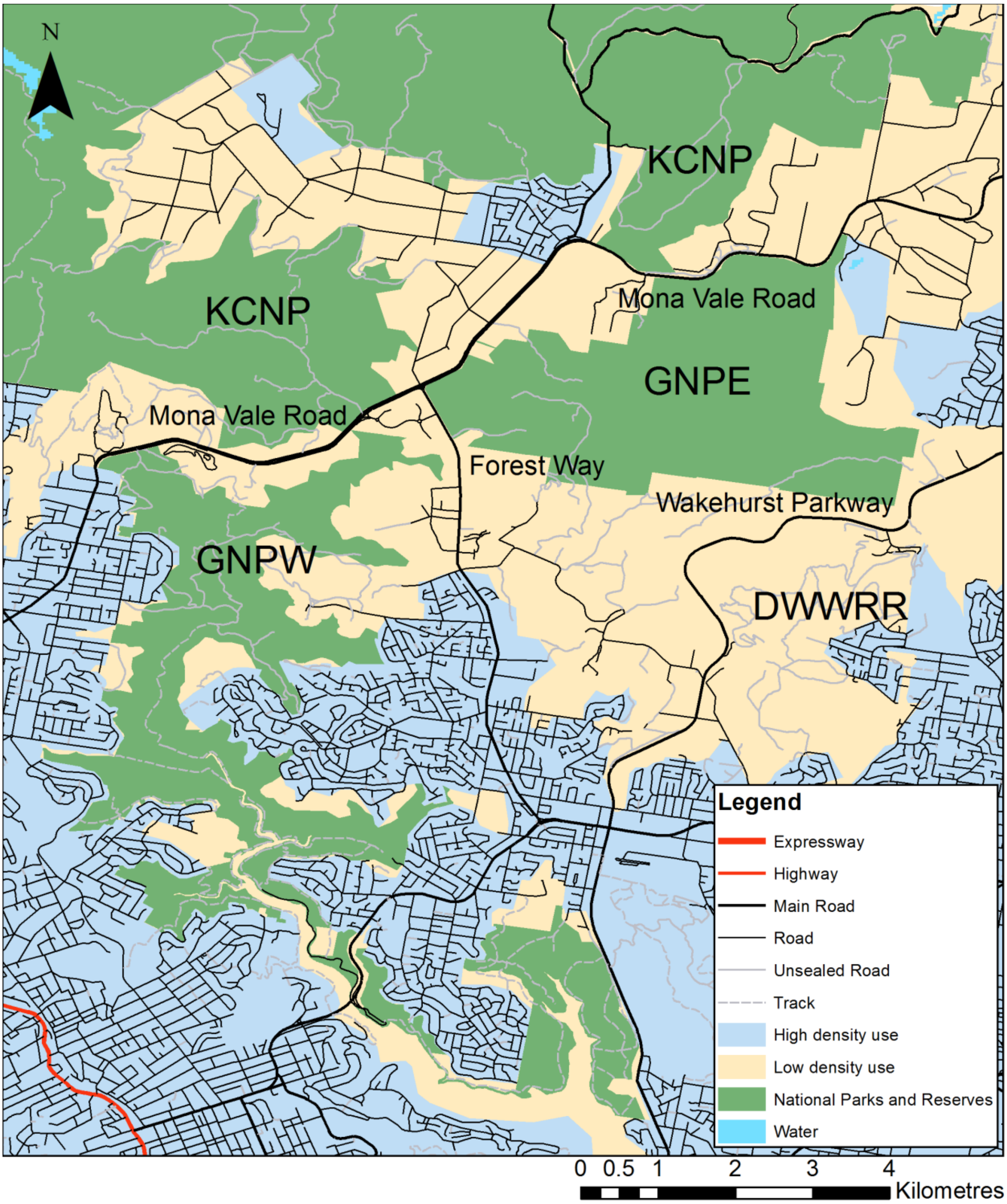


Figure 2 The study area was divided into four sub-populations: Ku-ring-gai Chase National Park (KCNP) to the north, Garigal National Park, divided into east (GNPE) and west (GNPW) sections, and the Dee Why West Recreational Reserve (DWWRR) to the south.

Study species

The swamp wallaby (*Wallabia bicolor*) is the only member of the monophyletic clade *Wallabia* and has vastly different dentition, genetic, reproductive, and behavioural characteristics from other wallabies (Merchant 2002). While the range of swamp wallabies is thought to be extensive, from Cape York to southern Victoria, their abundance within this range is currently unknown. Although often regarded as being resilient to disturbance (Ben-Ami 2005), their preferred habitat typically coincides with land occupied by humans and they are often abundant near roads, resulting in fatalities through collisions with vehicles (Osawa 1989; Coulson 1997; Taylor and Goldingay 2004; Ramp *et al.* 2006), placing increasing pressure on the sustainability of local populations (Ramp and Ben-Ami 2006). Swamp wallabies are the only macropod species currently found in GNP, though the red-necked wallaby (*Macropus rufogriseus*) and the common wallaroo (*M. robustus*) were previously found there.

Swamp wallabies are a sexually dimorphic species, with males on average between 12 and 20 kg and females between 10 and 15 kg (Merchant 2002). They are typically solitary, except for loose feeding aggregations and mother-young associations (Kirkpatrick 1970; Kaufmann 1974; Edwards and Ealey 1975). Often, each gully is home to a set group of individuals, with only minimal social interaction between groups in different gullies (Ben-Ami 2005). While many individuals may have overlapping home ranges within gullies (Troy and Coulson 1993; Ben-Ami 2005), they often avoid one another as much as possible, occupying overlapping areas at different times. Documented sizes of home ranges have varied, ranging from 8.7 to 39.0 ha in different environments (Troy and Coulson 1993; Ben-Ami 2005; Di Stefano *et al.* 2011). They browse on a diverse range of native vegetation (Harrington 1976; Hollis *et al.* 1986; Osawa 1989; Di Stefano and Newell 2008) and appear to target exotic plants on the periphery of human development (Osawa 1989; Watson and Dawson 1993). They prefer habitats with dense forest cover and thick understorey (Lunney and O'Connell 1988; Troy and Coulson 1993; Ben-Ami 2005; Di Stefano *et al.* 2009) and exhibit cryptic and mostly solitary behaviour, avoiding open spaces and congregating in groups. Births have been noted to peak in spring (Harrington 1976; Robertshaw and Harden 1986; Paplinksa 2005), but under predation pressure are also known to breed continuously (Kirkpatrick 1970; Robertshaw and Harden 1986). Pouch young are weaned at eight to nine months (Kirkpatrick 1970; Merchant 2002).

The populations of swamp wallabies across the study area are fragmented by major roads and their habitat abuts housing on all sides. Furthermore, they are predated upon by domestic dogs and red foxes (*Vulpes vulpes*). Predation represents a threat to the swamp wallaby, though attacks are rare. A baiting program of deploying toxic baits of 1080 poison for foxes is ongoing.

Abundance data

The distribution and abundance of swamp wallabies within GNP, adjacent reserves and crown land, and the nearby regions of KCNP, was determined using infrared wildlife monitoring cameras. Twenty motion detection cameras (Moultrie GameSpy i60 digital cameras) were placed as randomly as possible within the target area of the sampling effort. Satellite imagery was used to divide the study area into sections. GIS software (ArcGIS) was then used to locate random points over those sections which had vehicle access. We initially selected more points on the map than the number of cameras we had to set, providing some flexibility in actually locations chosen. Once in the area defined by the random point, we utilized signs of presence, such as scat and tracks through the brush, to determine the direction that we would face the camera. We therefore attempted to maximise the likelihood of capturing an image of a wallaby at each point.

Cameras were set 0.50 m above the ground as swamp wallabies have an average height of 0.75 m. The cameras were set to capture three images whenever the sensor was triggered, each one separated by 13 seconds. A one-minute delay was used to prevent the separate image recording. Cameras were left at each location for seven days then moved to a new location. The frequency of individuals observed over the week was recorded from a total of 1,151 trap nights over 63 days, distributed across the study area. Photographs from each location were viewed and the presence of animals, wallaby or others, was recorded, noting the time of day and climatic conditions.

Density estimation

Random encounter model (REM)

The Rowcliffe *et al.* (2008) model of density estimation uses the following equation:

$$D = \frac{y}{t}\frac{\pi}{vr(2+\theta)}$$

Equation 1

where D is density of individuals (km^{-2}), y/t is the number of photographs per unit time, v is the animal speed of motion, r is the detection distance of the camera, and θ is the angle of detection in radians. This value can be multiplied by the area of the park being sampled to determine the abundance of the target species throughout.

The speed of movement (v) of the swamp wallaby was attempted to be recorded using a remote camera on video mode to estimate the distance travelled over 30 seconds. Unfortunately, no suitable video footage was able to be obtained during the study period. Thus, our estimate of the speed of movement for the swamp wallaby was based on previous studies that have attempted to determine the value of this parameter. An upper limit was set at 1.06 km in 12 hours using data from Edwards and Ealey (1975) in which they actively chased swamp wallabies to determine their speed. Under these stressful conditions, it can be assumed that the individuals were moving at an irregularly fast past. A lower limit was set at 0.21 km in 12 hours using data from Garvey *et al.* (2010) in which radio-collared swamp wallabies were tracked during prescribed fires. Though some fled, others stayed within the safe confines of their gullies, unable to venture far because of the nearby burn. The average of these two values was 0.635 km per 12 hours, a value that was corroborated by a number of other values from these papers when conditions were less extreme. For the purposes of this study, this value was converted to a 17-hour day, leading to a value of 0.900 km day^{-1}.

Detection distance was given by the manufacturer of the camera as 12 m ± 1.5 m, though field testing led us to reduce this value to 8 m. The angle of the detection (field of view) for the cameras was 52 degrees (0.908 radians). Due to the activity levels of swamp wallabies, camera days were considered 15 hours rather than 12 hours or 24 hours. Only images captured between the hours of 4pm and 9am were taken into account when estimating density and abundance.

Derived probability of detection model (DPDM)

The Rowcliffe *et al.* (2008) model assumes that movement of individuals is random. To overcome this assumption, we developed a new model for calculating density from camera traps when individuals are not recognisable. To represent the unique habitat usage pattern of swamp wallabies, a half-normal distribution was used, with the x-axis representing the distance from the centre of the home range in kilometres. The half-normal distribution has long been employed to represent the habitat utilisation pattern of animals within their home ranges (e.g. Van Winkle 1975). This distribution illustrates the idea that an individual is more likely to be located closer to the centre of its home range than on the fringe. In this case we used a folded half-normal curve because it dismisses the sign of the observation and treats values as absolute. Thus, if an individual wallaby moved 10 metres away from the centre due east or 10 metres from the centre of the home range due west, the two values would be treated the same way. The half-normal distribution was given a mean of zero (representing the exact centre of a home range) and a standard deviation value of one-third of the radius of the average home range (three standard deviations account for 99% of an animal's kernel distribution), and takes the form:

$$P(x) = \frac{1}{\sigma\sqrt{2\pi}}\, e^{-(x-\mu)^2/(2\sigma^2)}$$

Equation 2

Due to the fact that we cannot be certain where, within this home range, our camera is set, we must consider the likelihood that it will be placed in the centre versus on the fringe. To represent this, we choose a linear function with the following equation, where *r* is the distance of detection in kilometres, θ is the angle of detection in degrees, and *H* is the area of the home range in km^2:

$$y = 2\pi x + \frac{\left(\frac{\pi r^2}{360/\theta}\right)}{H}$$

Equation 3

To obtain the slope of the line, we assumed that the home range area was circular. This idea is supported by work by Ben-Ami (Ben-Ami 2005), which showed that circular (or slightly oblong) home ranges were the most common shape in swamp wallabies in a nearby park to the study area. For the purposes of this study, 16.0 hectares (0.16 km^2) was chosen as the size of the swamp wallaby home range based on a synthesis of a number of previous studies. Troy and Coulson (1993) estimated the home ranges of radio-collared swamp wallabies using three different methods of analysis: Minimum Convex Polygons, Fourier Transform Estimation, and Harmonic Mean Isopleths. The result of the first of these methods was 16.0 hectares, and this has been the home range size cited in many subsequent papers. In addition, the average of the results from studies conducted by Johnson and Jarman (1987), Troy and Coulson (1993), Ben-Ami (2005), and Ben-Ami *et al.* (2006) was a value of 16.17 hectares, so the rounded value of 16.0 was deemed appropriate. The radius of this home range is 0.226 km, with one-third of this distance (one standard deviation) 0.075 km.

To estimate the probability of detection two parameters require estimating: the probability of sighting and the probability of capture. The probability of sighting (*s*) is essentially the chance that an individual and the camera will be in the same place at the same time. Because the y-axis of the half-normal distribution is arbitrary in units, the area of overlap between the curves is not sufficient to calculate *s*. Therefore, this value must be scaled using the total area under both curves up to the edge of the home range, effectively providing a proportion of the area where the individual and the camera can co-occur (Equation 4). To do this, the point of intersection between the two equations (2 and 3) must be found by solving for x. The area under the curves can be found by integrating to find values for A, B, C and D in Figure 3.

$$s = (B + C)/(A + B + C + D)$$

Equation 4

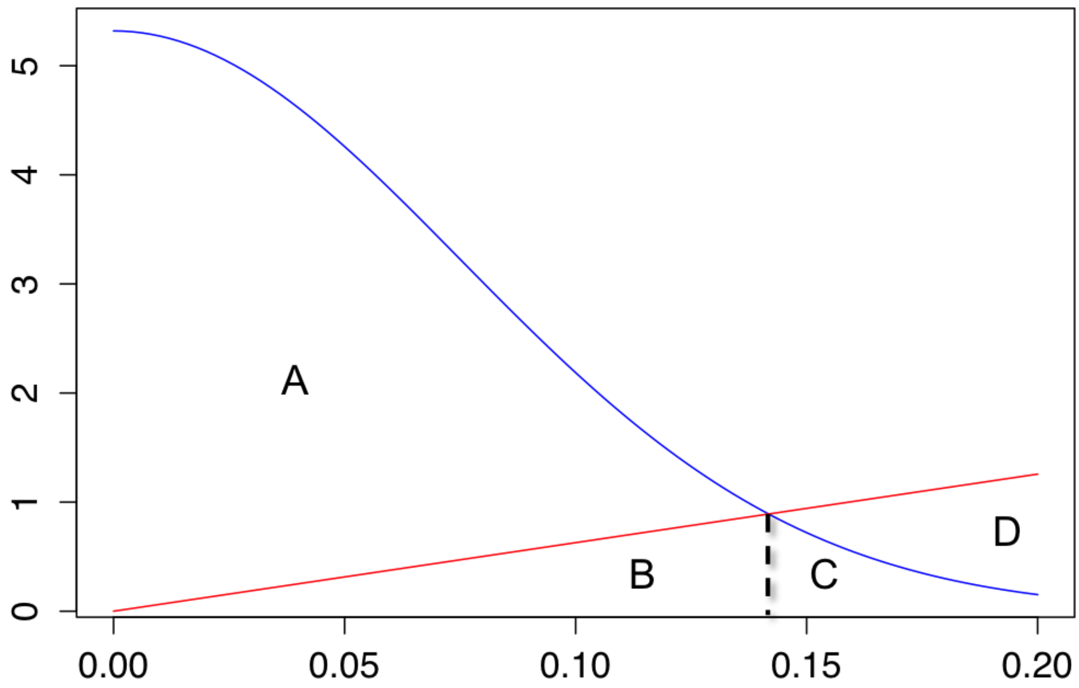


Figure 3 Folded half-normal curve (blue line, Equation 2) and the line representing the location of the camera within the home-range (red line, Equation 3). The letters A, B, C and D represent areas under the curves used to calculate the probability of sighting (s) using Equation 4. The point of intersection between the two functions is found when the two functions equal one another.

The probability of capture (*c*), a camera-specific parameter, was estimated by examining all of the images gathered during sampling to determine whether photographs that had no subject were the cause of motion by the vegetation or by a wallaby that moved too quickly to be caught by the camera. To do this, all cameras that took more than 50 images were excluded from examination, as it was assumed that light or some other interference was causing false positives (i.e. images when there was no movement in the frame). Next, each set of three images in which there was no wallaby was examined to determine whether any of the vegetation had shifted dramatically between the photographs, perhaps due to wind. If this were the case, the set would not be considered in the determination of *c*. Finally, those sets in which there were no visible movements in the background were considered "misses", where a wallaby had been present, but for some reason, was not captured by the camera. The number of sets of misses was added to the number of sets with a wallaby in the frame, and a proportion of successful captures was determined to act as the value *c*. In this case, a value of 0.69 was obtained using the images collected.

Multiplying the probability of sighting (*s*) by the probability of capture (*c*), which describes the chance that the camera will be able to take a picture of a wallaby that it encounters, one can determine the overall probability of detection (*P(d)*):

$$P(d) = sc$$

Equation 5

The probability of detection can then be used to obtain an estimate of density of individuals (km^{-2}), where *D* is density, *C* is the modified trapping rate (number of individuals photographed per unique camera placement), and *P(d)* is the probability of detection:

$$D = \frac{C}{P(d)}$$

Equation 6

Density can be multiplied by the area of the total park (in square kilometres) to determine the absolute abundance of individuals in the target region.

The modified trapping rate (*C*) is determined from the number of camera locations rather than the number of camera days (i.e. per some unit of time) as used in the Rowcliffe *et al.* (2008) model. The model requires that the length of time for which each camera is set at a given location reaches a threshold that represents the point at which enough time has passed for the half-normal distribution to be representative of the movement patterns of the study subject. In the case of swamp wallabies it was expected that one week was long enough for the kernel probability density (a graphical representation of the location of the animal at any point in time) to closely match the half-normal distribution.

Road-kill data

Collection of spatially-referenced road-kill information was collected by community groups and concerned individuals in the region sporadically over a 3 year period. These data are sporadic in nature as they vary in their spatial and temporal coverage. However, they provide an estimate of the minimum number of individuals killed in the region and their locations for the period. All carcases once recorded were removed from the roadside to avoid double counting. Two primary sources of road-kill information were obtained: from the Wildlife Information and Rescue service (WIRES) and from the Northern Beaches Roadkill Committee (NBRC). By far the most consistent in survey effort was conducted by the NBRC, and most of the main roads (Mova Vale Road, Forest Way, Wakehurst Parkway) were surveyed daily during the study period.

Population modelling

Information on abundance, movement, and road fatalities was used to explore a range of mitigation scenarios on swamp wallaby population persistence, focusing on the long-term viability of the wallaby population existing within GNP and DWWRR. Analyses were conducted to examine the trade-off between reducing road-kill and facilitating connecting (i.e. enabling safe dispersal) between Ku-ring-gai Chase National Park and Garigal National Park.

Demographic parameterisation

To facilitate population modelling it was necessary to describe the demographic structure of the populations, based on published information on demographics in the scientific literature and our own data.

Table 1 Values used to define the demographic parameters of the baseline scenario in the population model.

Category	Parameter	Value
Scenario settings	No. of iterations	50
	No. of years	50
	Extinction definition	No animals of one or both sexes
	No. of populations	4
Dispersal settings	Min age at dispersal	2
	Max age at dispersal	5
	Sex biased dispersal	Both
	% survival of dispersers	100
	Dispseral function	D*(1+(S='M'))
Reproductive system	Type of mating system	Polygynous
	Age of first offspring for Females	2
	Age of first offspring for Males	2
	Max age of reproduction	10
	Max no. of progeny/year	1
	% males at birth	50
Reproductive rates	% Females in the breeding pool	70 (10)
	% Males in the breeding pool	70
Mortality rates	Mortality from 0-1 yr	15 (3)
	Mortality from 1-2 yr	10 (3)
	Mortality after 2 yr	10 (3)
Catastrophes	Type	Wildfire
	Frequency	4%
	Reproduction	70
	Survival	90
Initial population sizes*	KCNP	2800
(* determined from camera trap survey)	GNP East	200
	GNP West	200
	DWWRR	70

Scenarios

Simulations of different scenarios were run explore the importance of maintaining the connectivity of KCNP with Garigal via Mona Vale Road versus the importance of reducing road fatalities. Population modelling was conducted using Vortex version 10 (Lacy *et al.* 2005).

A total of 3,000 iterations were performed with each variable sampled across its distribution, with increments defined using a Latin-Hypercube sampling methodology (Table 2). Values were derived by examining data on road-kill and population sizes obtained in this study as well as comparing to published literature. Road-kill values reflected the numbers of individuals killed each year, for each sex and for two age classes. Hence, a value of 5, for example, reflects a total of 20 individuals killed per annum. For dispersal, values are percentages of the total population size. Hence, a value of 2 percent from the source is equivalent to 56 animals (taken from initial population size of 2800). Likewise, a value of 10 percent within the sink is equivalent to 20 animals dispersing (taken from an initial population size of 200).

Table 2 Sensitivity parameters used to compare the effects of road-kill and dispersal on population persistence.

	Base value	Minimum	Maximum	Increments
Road-Kill GNPE	0	0	30	0.01
Road-kill GNPW	0	0	20	0.007
Road-kill DWWRR	0	0	20	0.007
Dispersal from source	0	0	4	0.001
Dispersal within sink	0	0	20	0.007

RESULTS

Camera data

In total, 7,490 images were recorded over a two month period from 99 different locations, representing 1,151 trap nights. Many of the images did not contain wildlife and were instead triggered by weather conditions, primarily by rain, changes in light and wind-blown vegetation. Of those, 1,820 contained images of wildlife (24.2% of images). Multiple images of individuals were taken when they were detected (Table 3), hence events were classified as different if images were more than 10 minutes apart.

Table 3 The numbers of image frames individuals were detected by.

Species	1	2	3	4	5	6	7
Unknown	4						
Bandicoot	6	1	2				
Brush-tailed possum	3	1	2				
Cat			1				
Rabbit			1				
Swamp wallaby	35	34	121	1		3	1
Bush rat	12	2	3				
Brush turkey	2						
Lyrebird	2				1		
Red fox	1						
Grand Total	**65**	**38**	**130**	**1**	**1**	**3**	**1**

A total of 239 different wildlife events were therefore recorded of nine different species, 195 of which were from swamp wallabies (Table 4). Most of the swamp wallaby sightings in KCNP occurred on the western side of the sampling area, while most in GNP occurred in east.

Table 4 Number of individuals captured by cameras during the study across the study area, encompassing Ku-ring-gai Chase National Park (east and west), Garigal National Park (east and west) and the Dee Why West Recreation Reserve.

Species	KCNP E	KCNP W	GNP E	GNP W	DWWRR	Total
Unknown			3		1	**4**
Bandicoot		3	1	4	1	**9**
Brush-tailed possum		5		1		**6**
Cat					1	**1**
Rabbit					1	**1**
Swamp wallaby	24	80	36	18	37	**195**
Bush rat		14			3	**17**
Brush turkey			2			**2**
Lyrebird			2		1	**3**
Red fox			1			**1**
Grand Total	**24**	**102**	**45**	**23**	**45**	**239**

Figure 4 Examples of images captured during the study. All are images shown are of swamp wallabies except for one of a lyrebird.

The sighting of swamp wallabies was not constant during the day, as would be expected for animals considered to be crepuscular (Figure 5). Activity during the middle of the day, between the hours of 9am and 4pm was generally lower than at other times, and was therefore used to reduce the movement calculations of individuals to periods of activity (i.e. excluding 9am to 4pm).

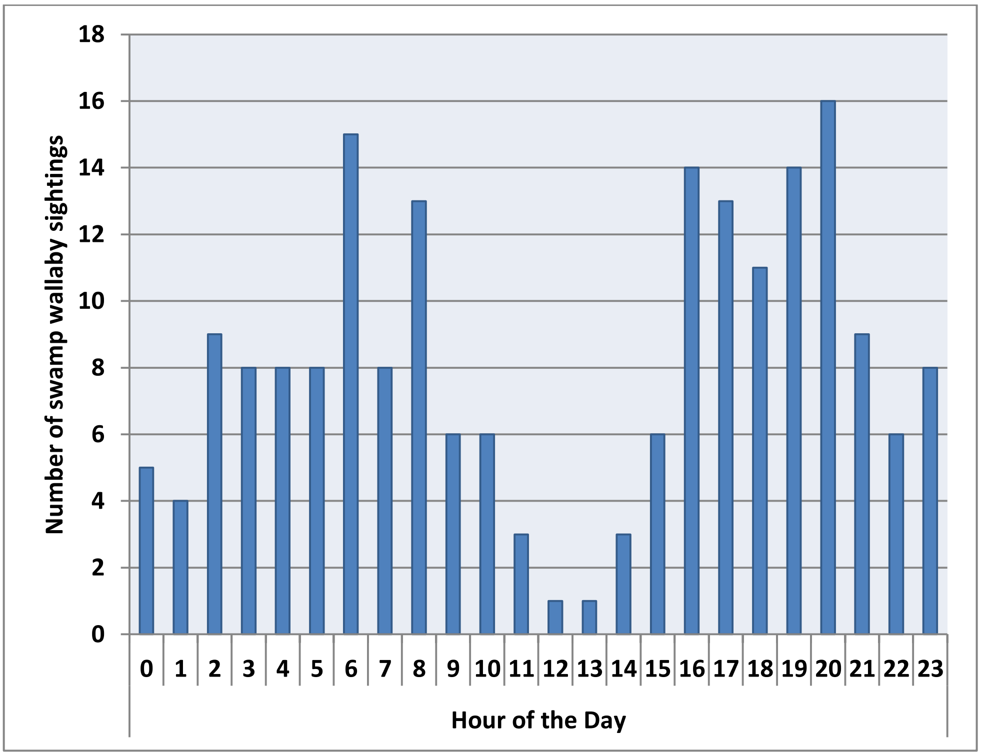


Figure 5 Number of wallaby sightings per hour sampled across all locations in the study area.

The trapping rate of swamp wallabies was spatially varying and ranged between 0 and 1.14 per camera per day (Figure 6).

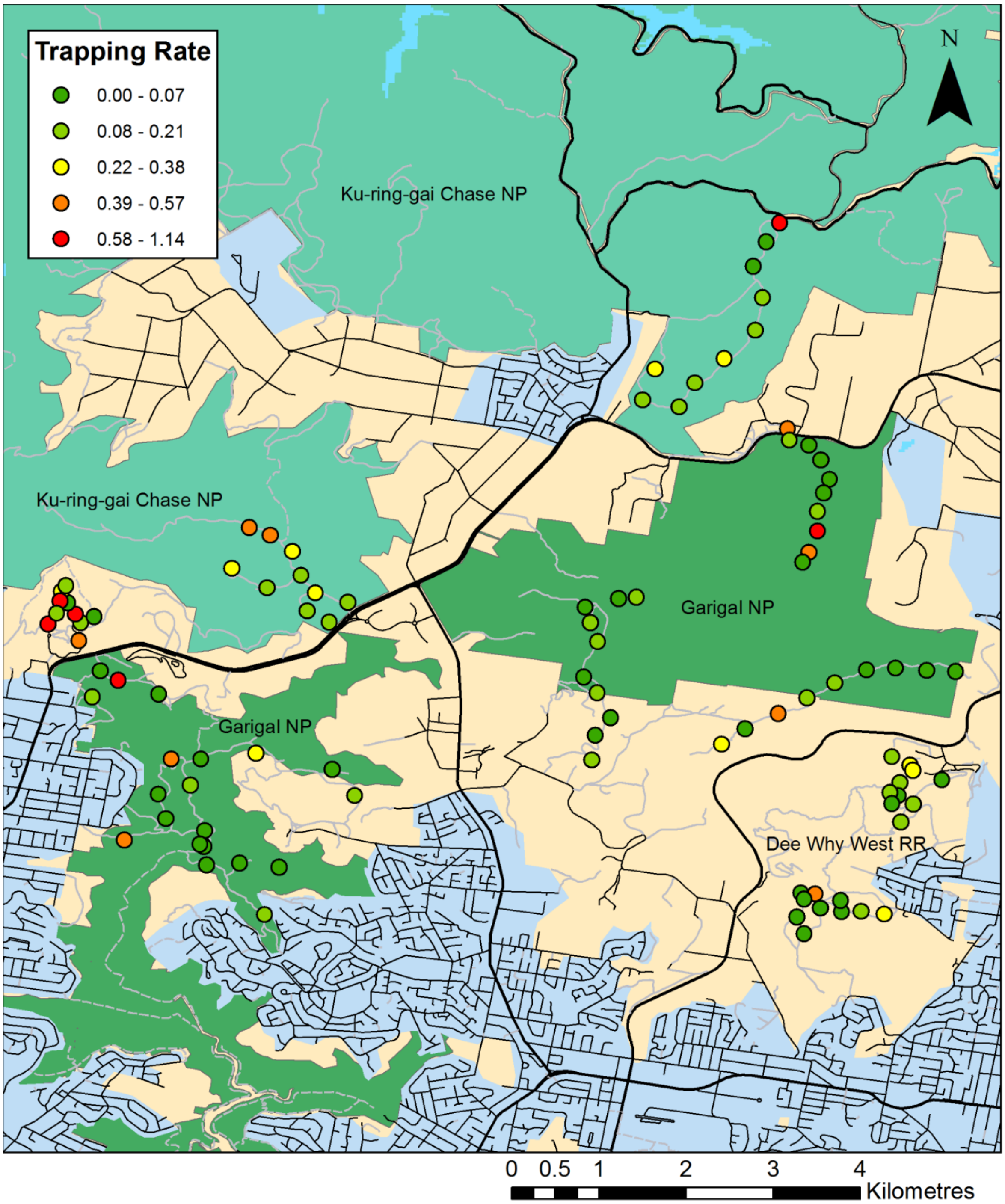


Figure 6 Location and trapping rate of swamp wallabies per day from 99 locations across Garigal National Park, Dee Why West Recreation Reserve and areas of the adjacent Ku-ring-gai Chase National Park. Locations are coloured in 5 categories identified using Jenk's Natural Breaks rule.

Trapping success was significantly higher adjacent to the Ku-ring-gai Wildflower Garden in west KCNP (Figure 7), and significantly lower in the gully region of Cascades Track of west GNP just north of Davidson and Belrose. Significance was determined by calculating the Getis-Ord statistic (Gi*) in ArcGIS to identify clusters of significantly different trapping rates.

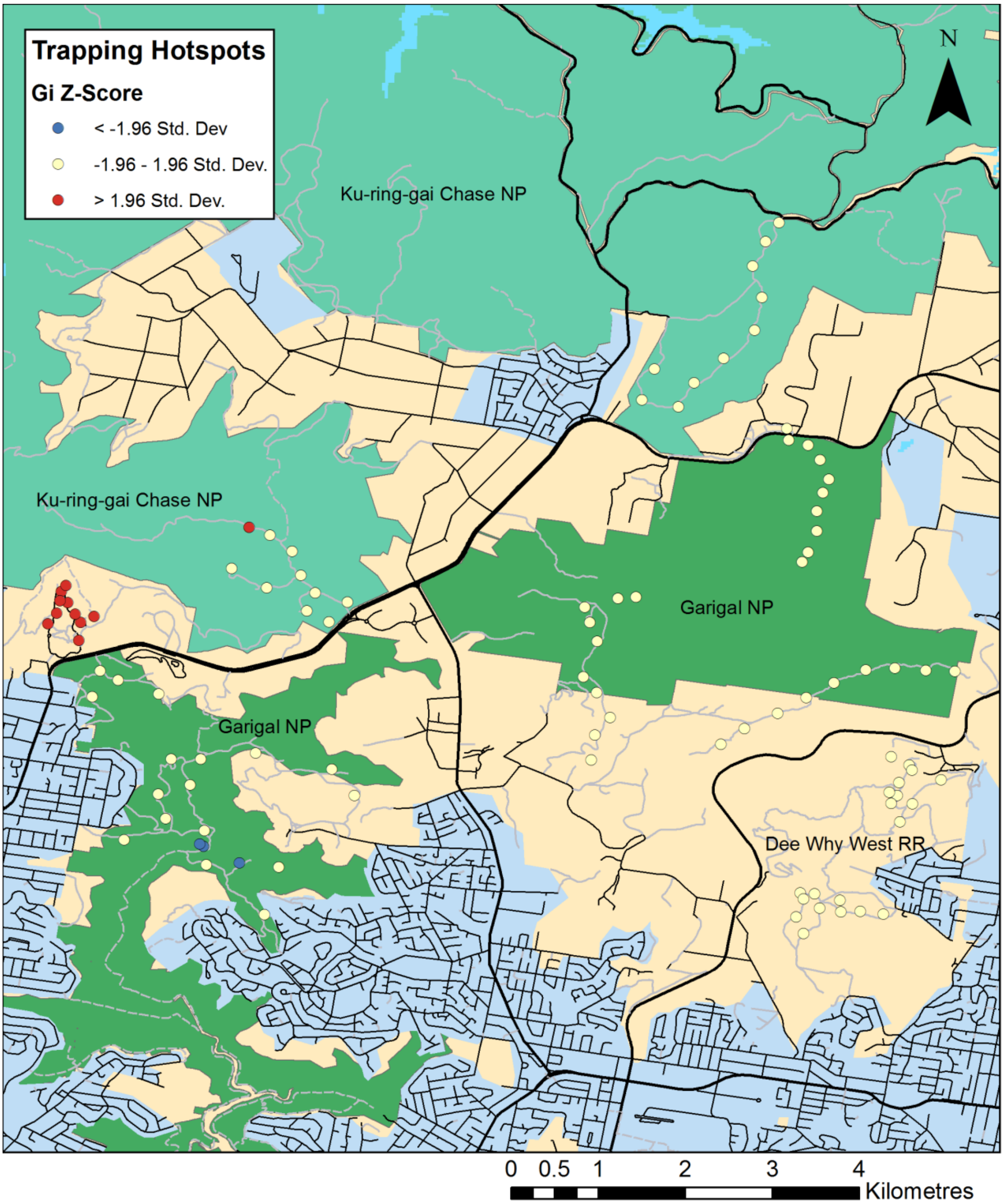


Figure 7 Hotspots for trapping of swamp wallabies in the study area. Regions with significantly higher than average trapping rates are represented by red circles, while regions with significantly lower than average trapping rates are represented by blue circles. Yellow circles imply trapping rates were not significantly different from the mean rate.

Density estimation

Random movement model (RMM)

Using the random movement model of Rowcliffe *et al.* (2008), the predicted densities of swamp wallabies within the sub-populations were derived from the number of wallabies sighted, the trapping rate, and the amount of habitat available (Table 5). The trapping rate was lowest in the DWWRR population and highest in the west side of KCNP.

Table 5 Number of wallabies sighted, the trapping rate, and the predicted density of individuals per km^2 determined using the random movement model.

Location	Wallaby sightings	Sightings within 17h	Trap nights	Probability of trapping	Density km^{-2}
KCNP E	24	22	140	0.16	23.58
KCNP W	80	68	280	0.24	36.44
Garigal E	36	30	275	0.11	16.37
Garigal W	18	17	140	0.12	18.22
DWWRR	37	32	316	0.10	15.19

Derived probability detection model (DPDM)

Densities of swamp wallabies predicted by the Derived probability of detection model were lower than those predicted by the RMM. Densities were significantly lower in habitats within DWWRR, while densities determined from locations on the west side of KCNP were significantly higher (Figure 9).

Population estimates

The total area available to swamp wallabies in GNPE, GNPW and DWWRR were determined to be 18, 15 and 5.5 km^2 respectively. Assuming even habitat availability and preference (a compromise in this case as detailed microhabitat preferences were not available), population estimates for these areas were determined for both of the two density estimation methods (Table 6).

Table 6 Population estimates determined using the two density estimation methods.

Location	Area (km^2)	RMM (#)	DPDM (# ± 95% c.l.)
GNPE	18	295	174 ± 69
GNPW	15	273	165 ± 96
DWWRR	5.5	84	48 ± 22

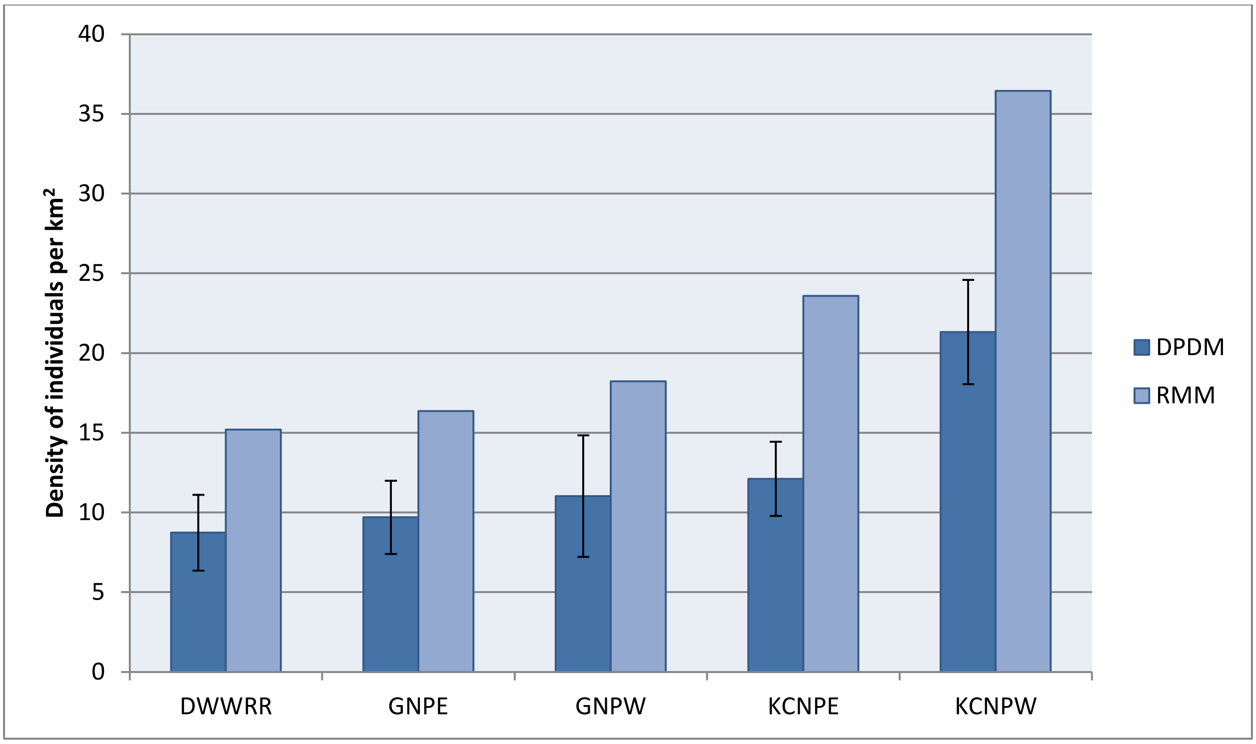


Figure 8 Densities of individuals per km^2 with standard errors for the surveyed locations estimated using the Derived probability of detection model (DPDM) and estimates derived using the random movement model (RMM).

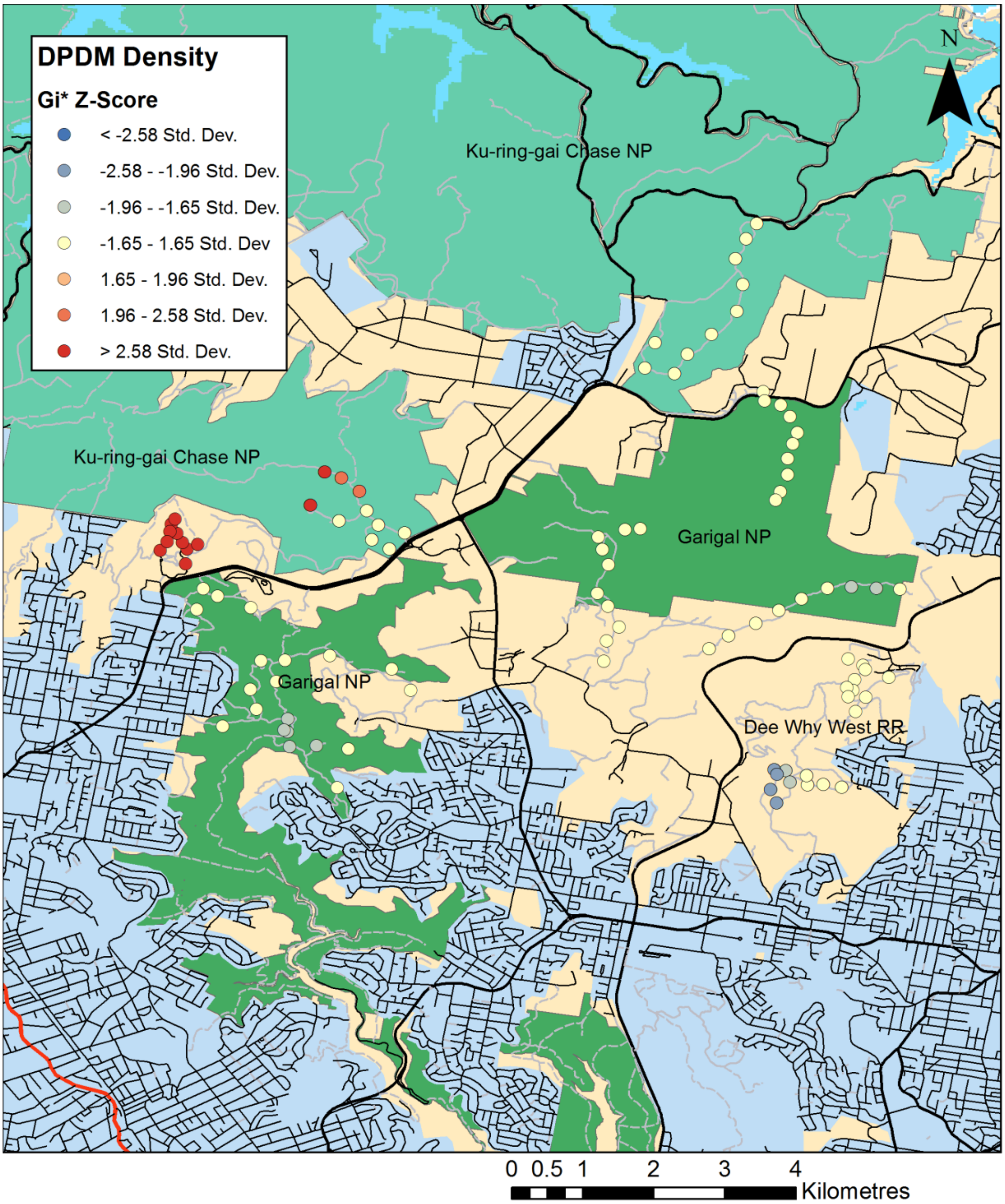


Figure 9 Getis-Ord z-scores for derived probability of detection estimates of swamp wallaby density. Blue circles are significantly lower than average densities, while red circles are significantly higher than average densities.

Road-kill

Although road-kill data was available from January 2009, it was not until April 2011 that a concerted effort was made by the NBRC to traverse most of the major roads each day (Figure 10). Sampling over a period of just over 12 months (5 April 2011 to 30 April 2012) by the NBRC identified 25 different species, including individuals of vulnerable species like the powerful owl (*Ninox strenua*), the grey-headed flying fox (*Pteropus poliocephalus*) and the yellow-bellied sheathtail-bat (*Saccolaimus flaviventris*). Additional records identified road-kill of an Eastern pygmy possum (*Cercartetus nanus*) (recorded after 2012). Bandicoots were not recorded in a way to distinguish between the two species that occur in the area, but it is possible that some of those killed were of the endangered southern brown bandicoot (*Isoodon obesulus obesulus*). Threatened status in NSW is listed as either vulnerable (V) or endangered (E) where appropriate. In total 979 individuals were recorded over the 12 months (Table 7). The most commonly killed species were common brushtail possums, common ringtail possums and swamp wallabies, between them accounting for 78% of all fatalities.

No major seasonal trends in fatalities were observed (Figure 11), although the numbers of possums (common brushtail and common ringtail possums combined) killed in the early parts of 2012 put the daily kill rate at over 2 per day.

Locations of road-killed fauna were assessed to detect hotspots. Point densities of the locations over 12 months for both possum species and swamp wallabies highlighted a number of key areas. Swamp wallabies were killed at six major hotspots representing just over 39% of all wallabies killed, although it should be recognized that these were not the only locations (Figure 12). Each of the hotspots recorded between 10 and 17 killed over the 12 months, with each being roughly 1km in length. Statistical analyses on these data were not performed.

Most of the road-killed common brushtail possums were killed in a hotspot adjacent to the Dee Why West Recreation Reserve and the Sydney Academy of Sport and Recreation and one towards the southern end of the Wakehurst Parkway (Figure 13), while common ringtail possum fatalities occurred on the southern Wakehurst Parkway hotspot (Figure 14).

Table 7 Number and type of road-kill individuals identified by the Northern Beaches Road-kill Committee between April 2011 and April 2012 on roads intersecting Ku-ring-gai Chase National Park, Garigal National Park and Dee Why West Recreation Reserve.

Species	Scientific name	Status	Class	Total
Pacific black duck	*Anas superciliosa*		Aves	3
Superb fairy wren	*Malurus cyaneus*		Aves	1
Brown cuckoo dove	*Macropygia amboinensis*		Aves	1
Brown quail	*Coturnix ypsilophora*		Aves	1
Australian brush-turkey	*Alectura lathami*		Aves	5
Laughing kookaburra	*Dacelo novaeguineae*		Aves	5
Lewins honeyeater	*Meliphaga lewinii*		Aves	1
Rainbow lorikeet	*Trichoglossus haemotodus*		Aves	1
Powerful owl	*Ninox strenua*	V	Aves	1
Sulphur-crested cockatoo	*Cacatua galerita*		Aves	9
Tawny frogmouth	*Podargus strigoides*		Aves	12
Australian wood duck	*Chenonetta jubata*		Aves	2
Bandicoot sp.	*Perameles nasuta* or *Isoodon obesulus obesulus*	E	Mammalia	98
Short-beaked echidna	*Tachyglossus aculeatus*		Mammalia	9
Grey-headed flying fox	*Pteropus poliocephalus*	V	Mammalia	1
Possum sp.	*Pseudocheirus peregrinus* or *Trichosurus vulpecula*		Mammalia	24
Common ringtail possum	*Pseudocheirus peregrinus*		Mammalia	258
Common brushtail possum	*Trichosurus vulpecula*		Mammalia	287
Swamp wallaby	*Wallabia bicolor*		Mammalia	195
Yellow-bellied sheathtail-bat	*Saccolaimus flaviventris*	V	Mammalia	1
Eastern blue-tongued lizard	*Tiliqua scincoides scincoides*		Reptilia	3
Eastern long-necked turtle	*Chelodina longicollis*		Reptilia	2
Diamond python	*Morelia spilota spilota*		Reptilia	3
Gecko sp.			Reptilia	1
Lace monitor	*Varanus varius*		Reptilia	3
Unknown				26
Grand Total				**979**

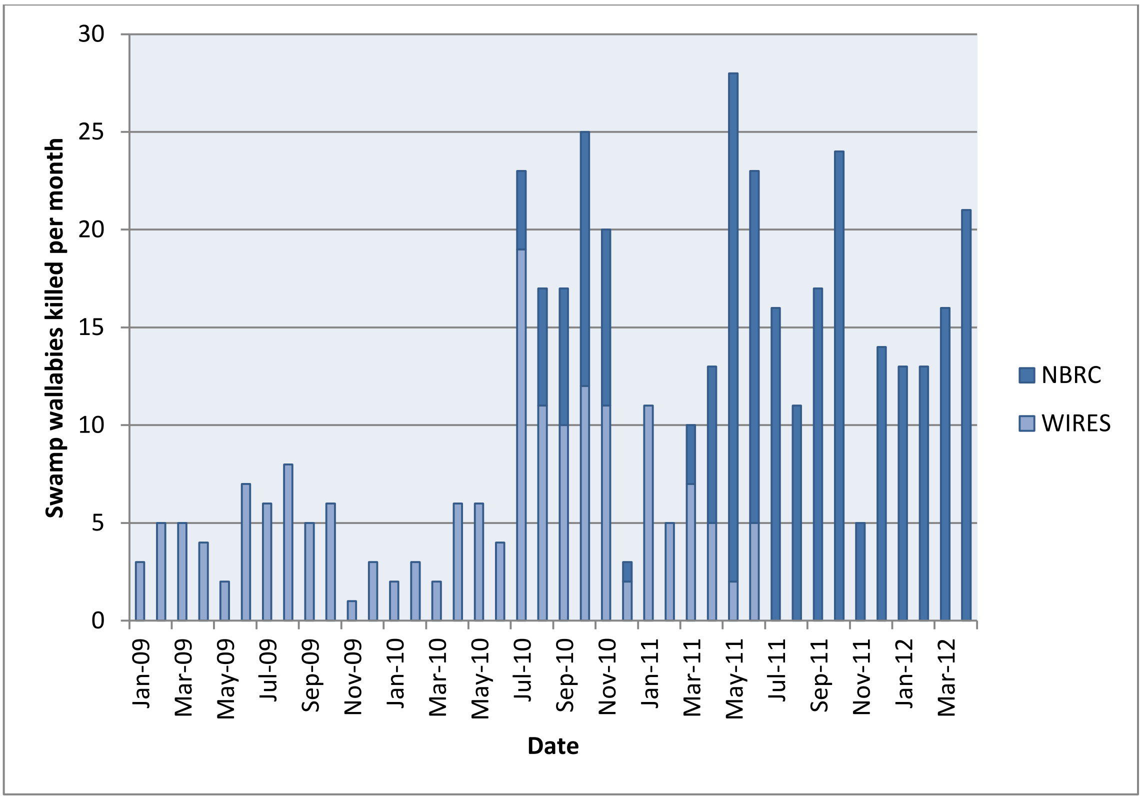


Figure 10 Number of road-killed swamp wallabies as recorded by WIRES and the Northern Beaches Road-kill Committee over 3 years. The figure provides

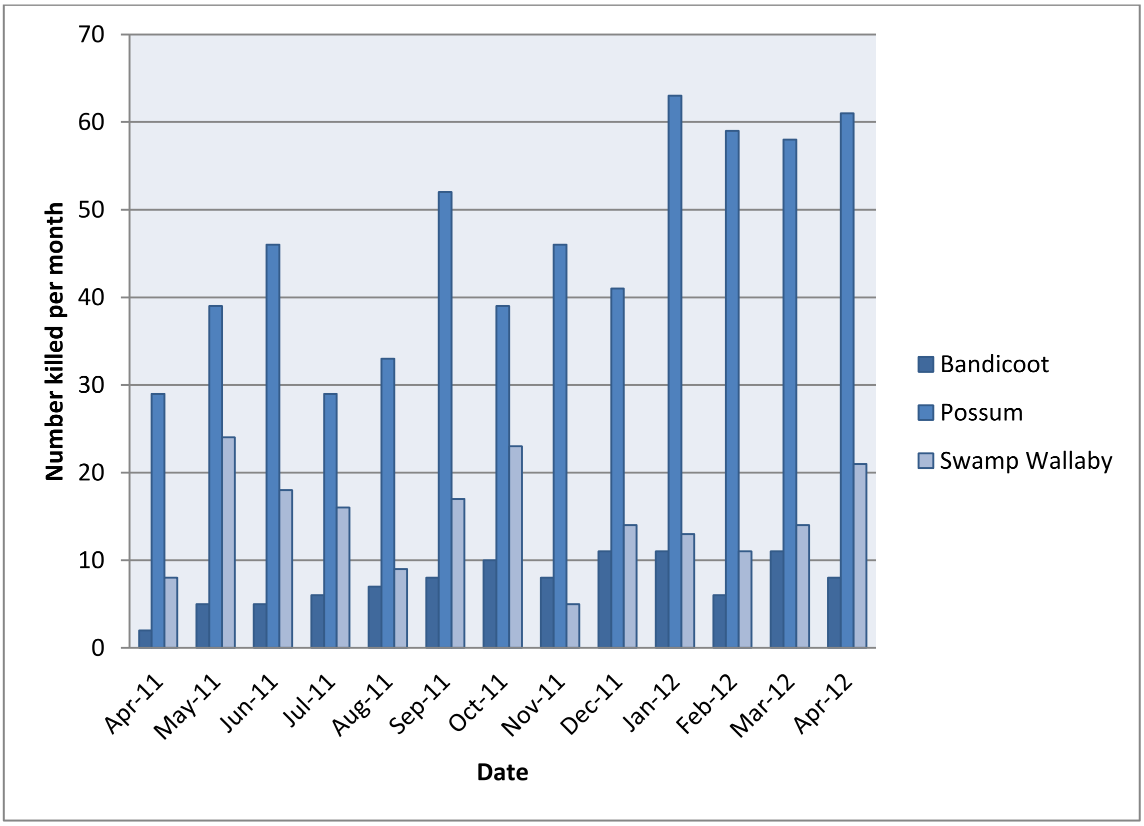


Figure 11 Monthly numbers of road-killed bandicoot sp., possum sp. and swamp wallabies over 12 months as recorded by the Northern Beaches Road-kill Committee.

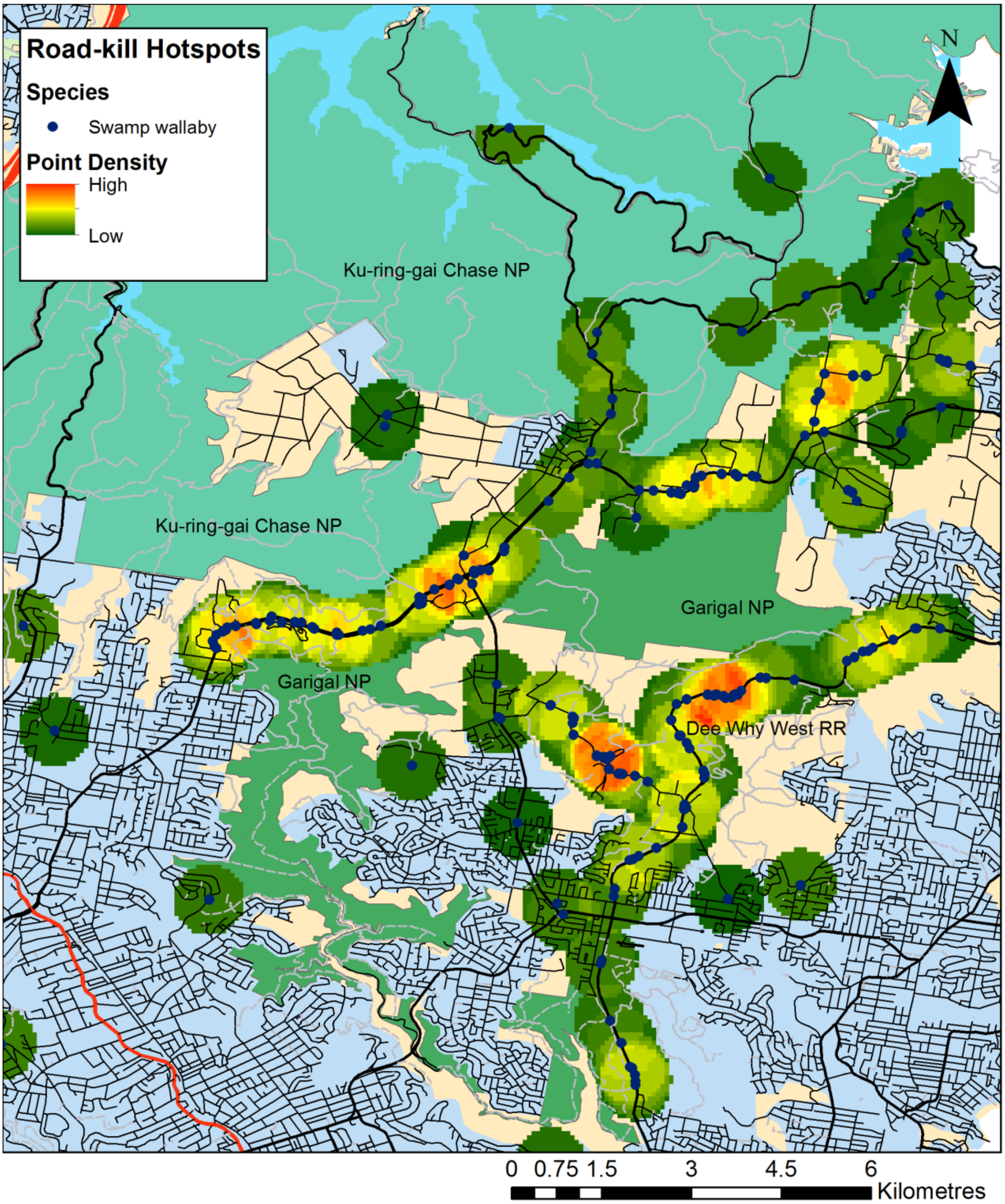


Figure 12 Location of road-killed swamp wallabies over 12 months as recorded by the Northern Beaches Road-kill committee. Point density calculations show hotspots (red).

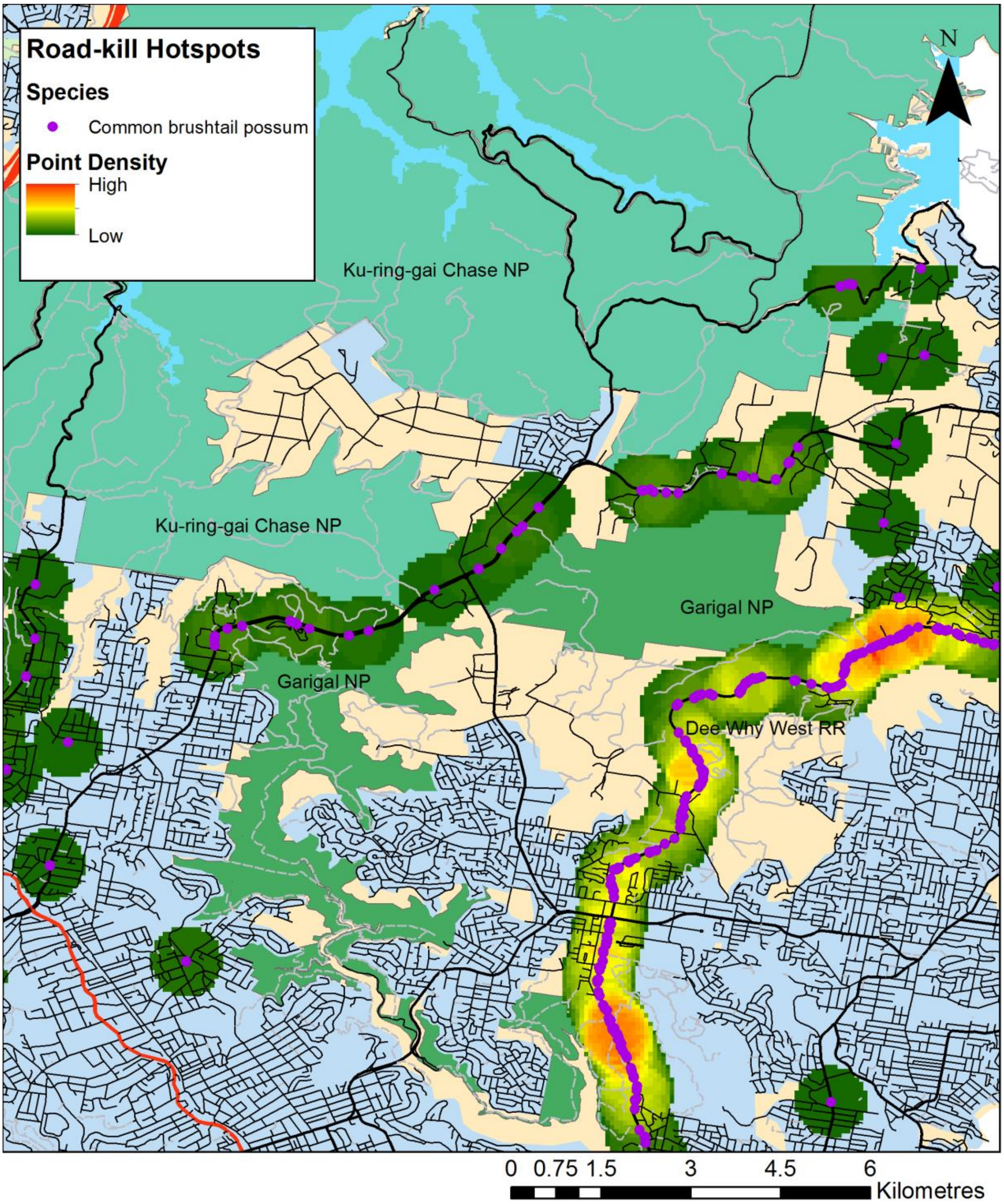


Figure 13 Location of road-killed common brushtail possums over 12 months as recorded by the Northern Beaches Road-kill committee. Point density calculations show hotspots (red).

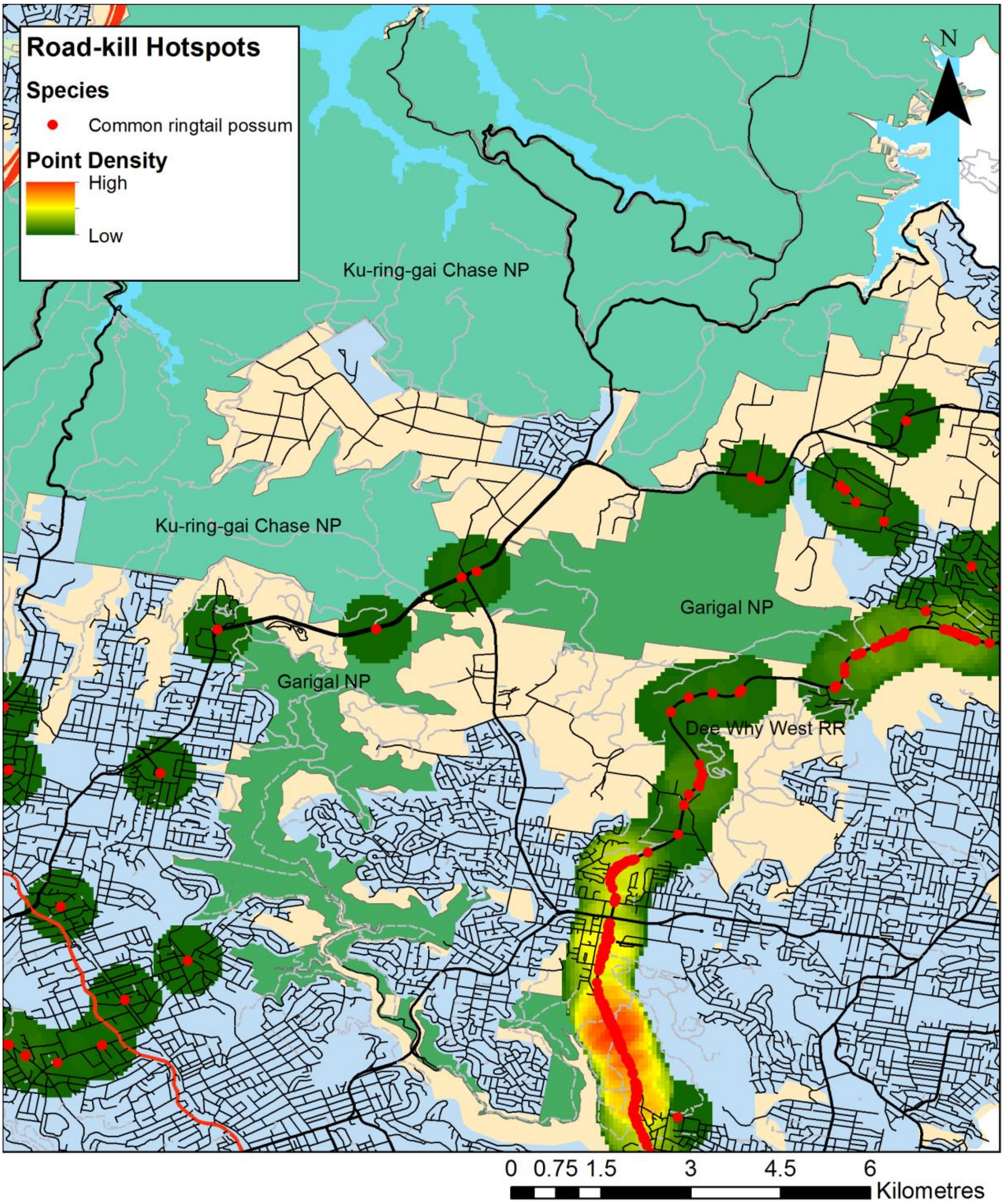


Figure 14 Location of road-killed common ringtail possums over 12 months as recorded by the Northern Beaches Road-kill committee. Point density calculations show hotspots (red).

Population modelling

Modelling of the populations without road-kill or allowing for dispersal between sub-populations showed that the populations were relatively robust over the 50 years of simulations (Figure 15). These were achieved by using conservative estimates of population sizes, slightly higher than the mean densities described by the derived probability of detection model (which is felt to be more accurate).

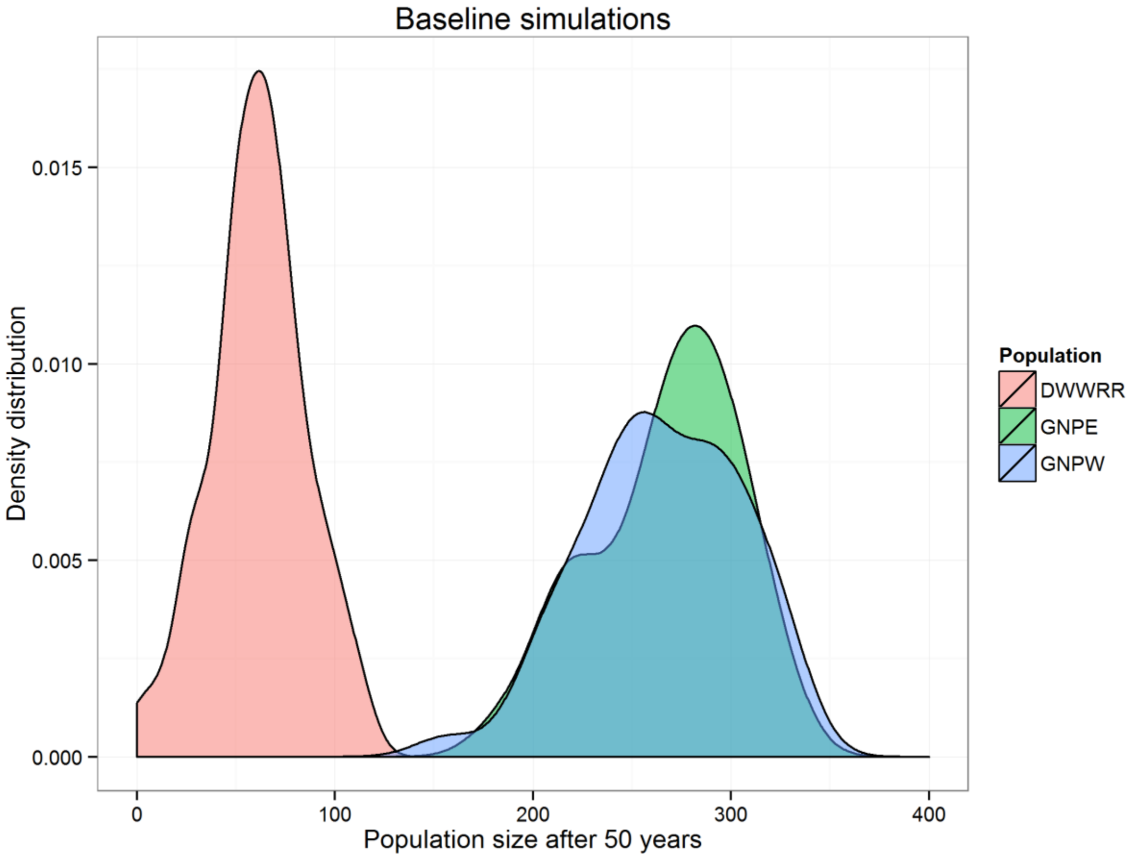


Figure 15 Baseline simulations of the populations of Garigal National Park and the adjacent Dee Why West Recreation Reserve. No dispersal or road-kill formed part of the simulations.

In contrast, once road-kill and dispersal were included in modelling scenarios, each of the populations had fewer animals within them over 50 years than in the baseline scenario (Figure 16). The DWWRR population was observed to go extinct more often than not, while both of the two Garigal populations also crashed often.

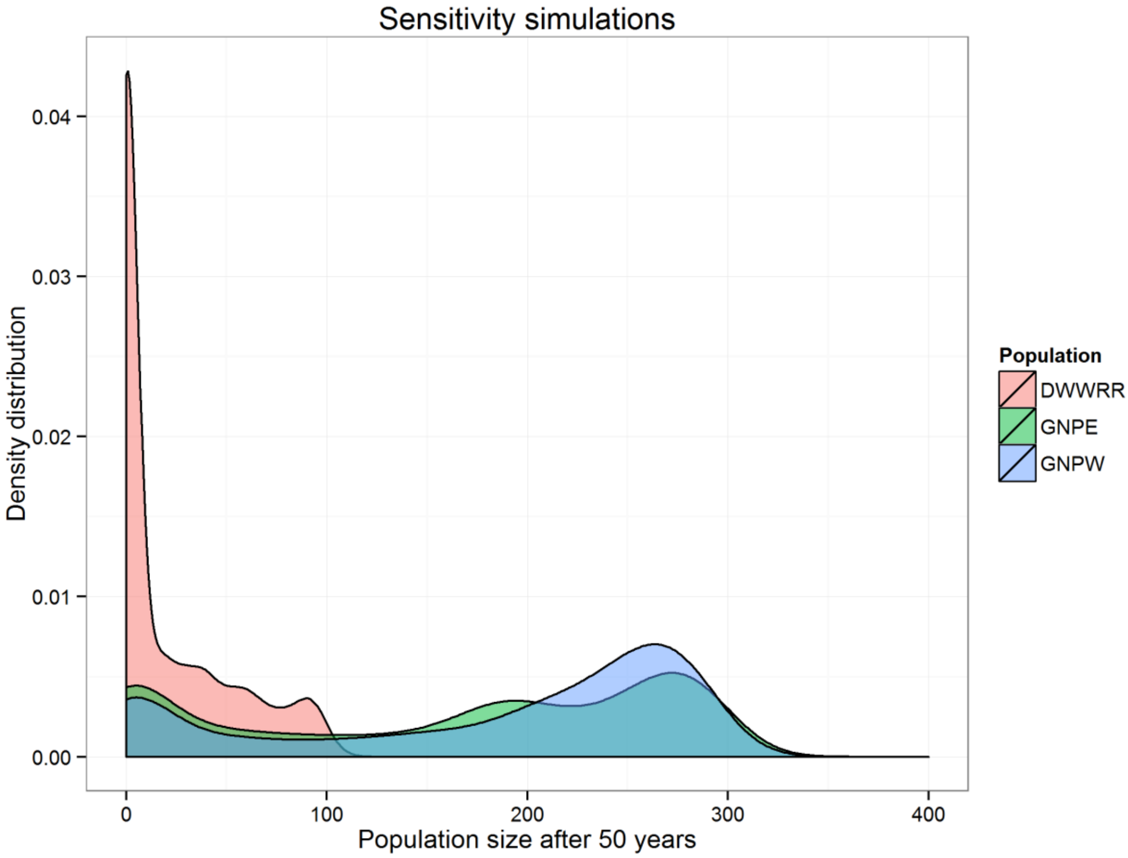


Figure 16 Simulations of the populations of Garigal National Park and the adjacent Dee Why West Recreation Reserve across a range of road-kill and dispersal scenarios.

Modelling of the probability of extinction in GNPE, GNPW and DWWRR identified that dispersal between the source population (KCNP) and the sink populations (Garigal and Dee Why West) was vital to preventing road-kill from crashing the populations (Table 8, Figure 17). This strongly suggests that the populations are not viable if dispersal is prevented and road-kill persists.

Table 8 Model statistics for the Generalised Additive Model of the probability of extinction for five variables simulated in the sensitivity analysis.

Variable	Estimated DF	Chi Squared	P-value
Road-kill in GNPE (SV1)	2.396	594.21	< 0.001
Road-kill in GNPW (SV2)	1	316.86	< 0.001
Road-kill in DWWRR (SV3)	6.838	374.32	< 0.001
Dispersal from source (SV4)	3.501	1621.48	< 0.001
Dispersal within sink (SV5)	1.554	40.42	< 0.001

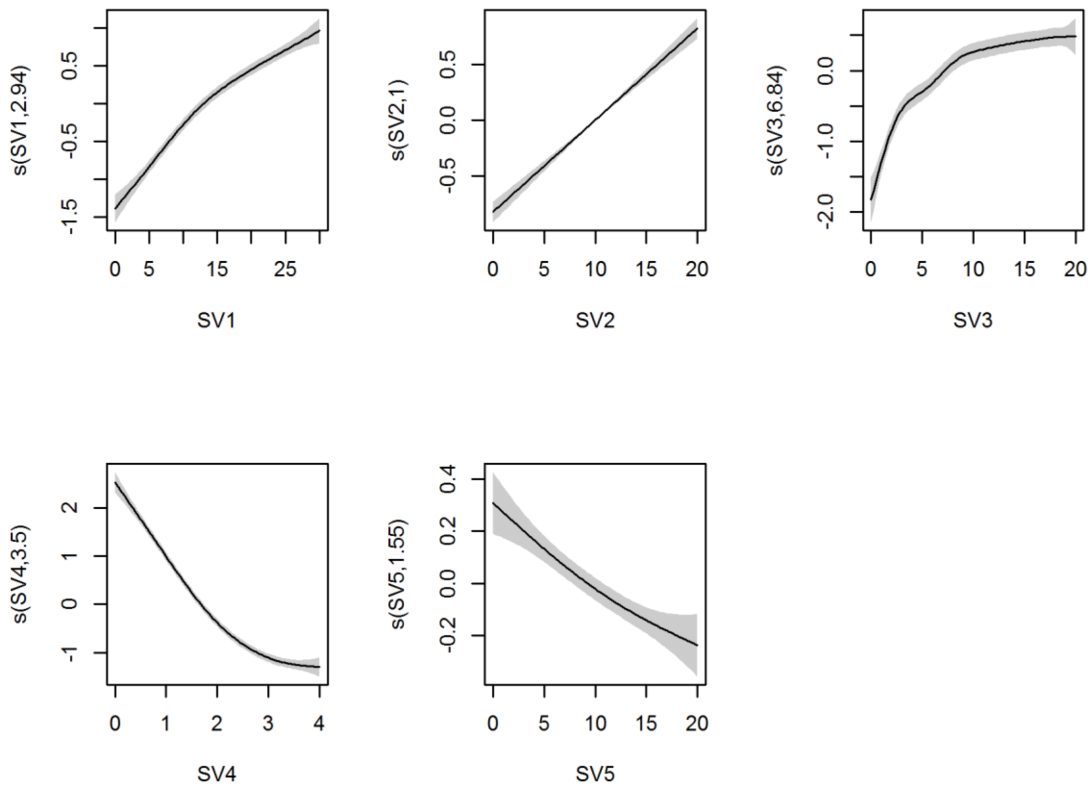
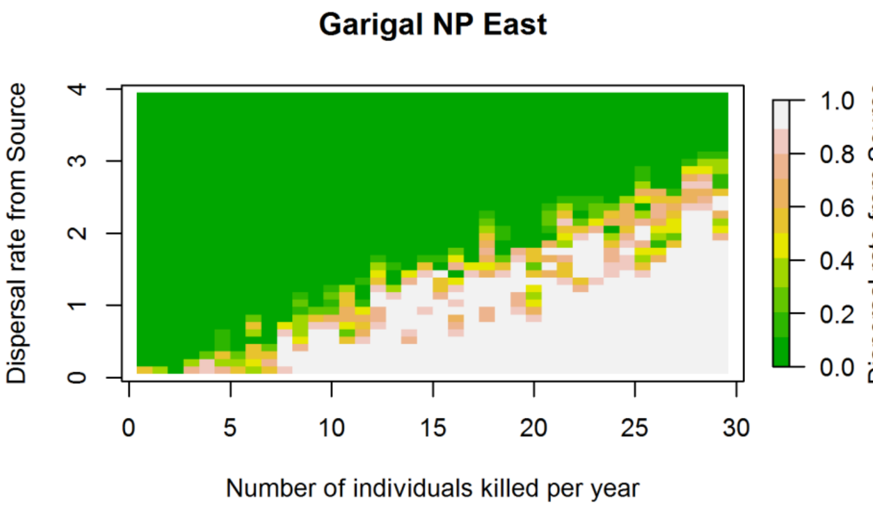
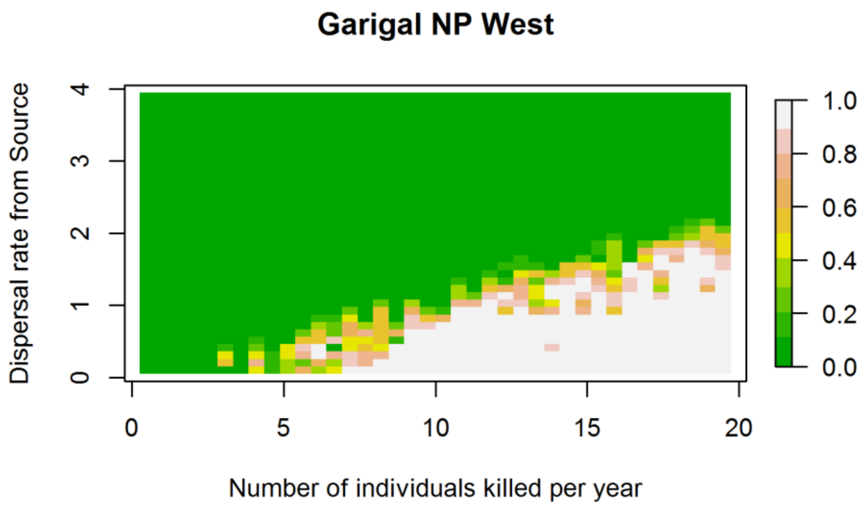


Figure 17 Partial plots of the Generalised Additive Model of the probability of extinction for five variables simulated in the sensitivity analysis. SV1 = Road-kill in Garigal NP East; SV2 = Road-kill in Garigal NP West; SV3 = Road-kill in Dee Why West Recreation Reserve; SV4 = Dispersal from Ku-ring-gai Chase NP into Garigal NP; SV5 = Dispersal between Garigal and Dee Why West populations. Shaded areas represent 95% confidence intervals.

Comparison of simulations of scenarios across a range of each of the variables highlighted trade-offs between road-kill and dispersal needed to ensure that populations persisted. Survival in the two Garigal populations is clearly being facilitated by dispersal from the source population in KCNP (Figure 18). Road-kill was simulated by harvesting of animals from both sexes and two age classes each year. Hence, a value of 5, for example, reflects an actual loss of animals to road-kill of 20 for that population each year. Dispersal rates from KCNP into the Garigal populations above 2 percent was necessary reduce the probability of those populations going extinct within 50 years to an acceptable level. A dispersal rate of 2 percent is roughly equivalent to 56 animals dispersing into Garigal each year. The DWWRR population was less sensitive to fluctuations in dispersal from KCNP and was more dependent on the two Garigal populations persisting.

Within sink dispersal (i.e. among Garigal and the DWWRR populations) was the least important predictor of the probability of extinction (Table 8), although the relationship was represented by a significant negative correlation. Given the small population size estimated for DWWRR, even small amounts of road-kill resulted in the population predicted to go extinct within 50 years (Figure 19).





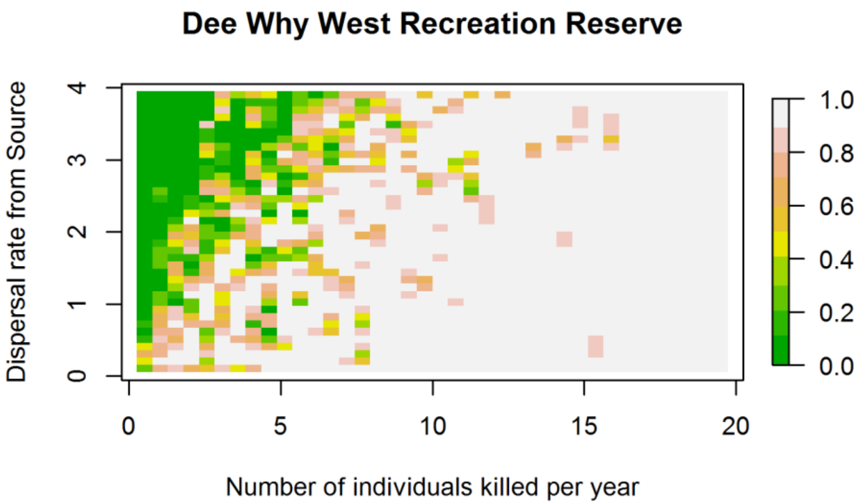


Figure 18 Plots of the probability of extinction for each of the three sink sub-populations plotted against dispersal from the source population (Ku-ring-gai Chase National Park). The three sub-populations show values for the probability of extinction (grey – green) for combinations of road-kill each year (x-axis, values are numbers of each sex and two age classes killed – e.g. 5 = 20 individuals) and dispersal (% of population).

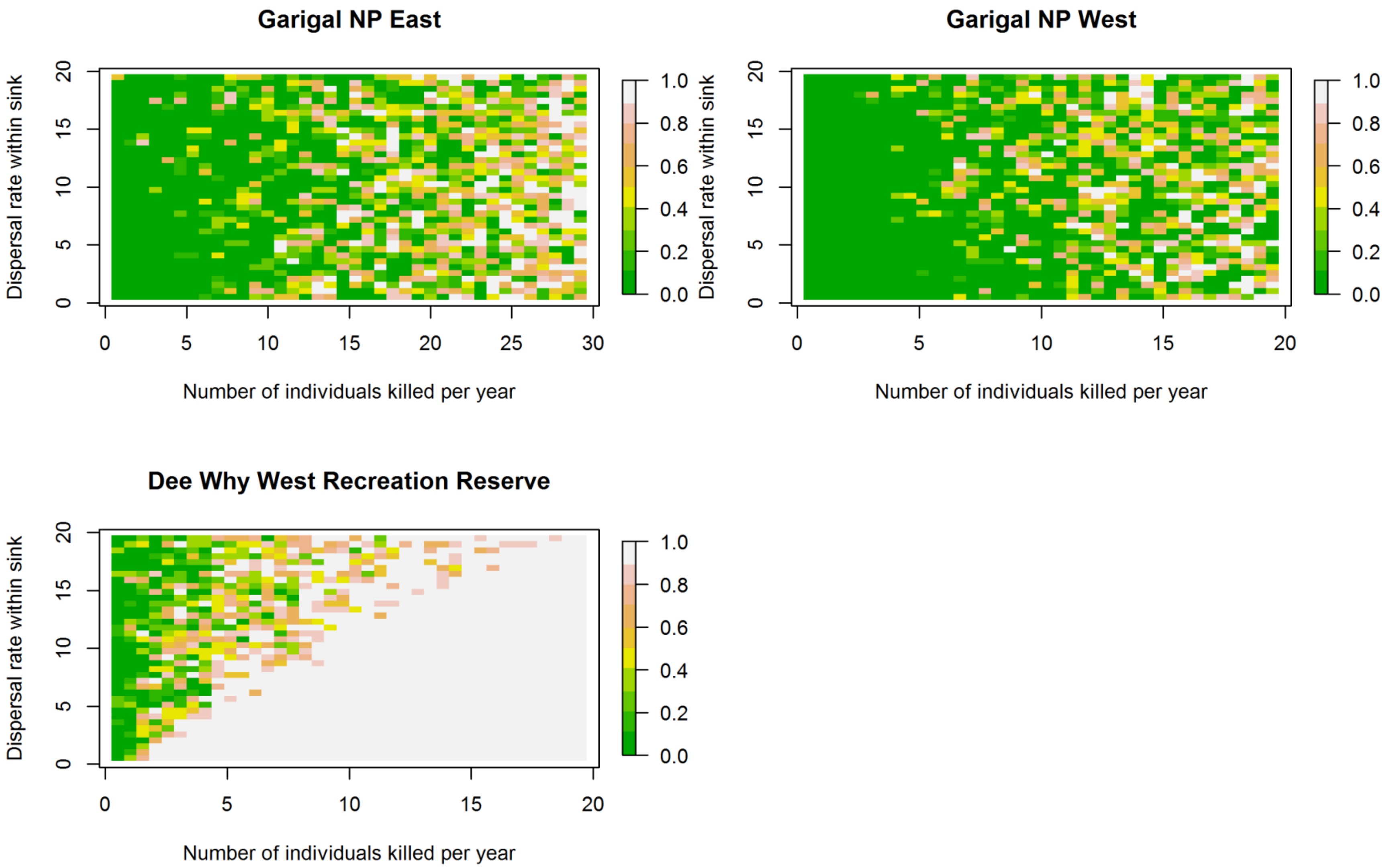


Figure 19 Plots of the probability of extinction for each of the three sink sub-populations plotted against dispersal within sink populations. The three sub-populations show values for the probability of extinction (grey – green) for combinations of road-kill each year (x-axis, values are numbers of each sex and two age classes killed – e.g. 5 = 20 individuals) and dispersal (% of population).

DISCUSSION

Swamp wallabies are tenacious macropods, able to persist in peri-urban landscapes where other fauna have long since disappeared (Ben-Ami 2005). This tenacity, however, cannot be taken for granted. Without careful consideration and protection of the ecological dynamics necessary to maintain populations, even swamp wallabies may no longer be a part of the landscapes they now characterise. Where impacts from a single source are extreme, the need to protect and conserve can be obvious and relatively straightforward (Chapple *et al.* 2011). Problematically, threatening processes are frequently not that simple, and small but incremental impacts can have deleterious consequences for species populations (Roger *et al.* 2011).

It is essential to recognise that not all areas of habitat provide the same contribution to meta-population persistence. Some areas are more important than others. Source populations are vital for ongoing persistence, while sink populations are dependent upon source populations for survival. When sink populations attract individuals from source populations they may be considered as attractive sinks (Delibes *et al.* 2001). Where this attractive sink also has a hidden risk (i.e. road-kill), it may be considered as an ecological trap. How populations persist under these conditions is case specific, but it is clear that persistence must be considered at a landscape scale where population dynamics are able to be evaluated robustly.

The aim of this study was to survey the populations of swamp wallabies residing in reserves on the northern beaches of Sydney and to examine the functioning of the meta-population. Although anecdotally the populations of swamp wallabies were assumed to be stable, indications from the toll of fatalities on roads in the region suggested that this loss of life must be of concern to their long-term persistence. Importantly, with increasing housing development and upgrades of roads, a timely examination of their population dynamics was deemed necessary.

Through the use of camera-traps distributed throughout Ku-ring-gai Chase National Park, Garigal National Park and the area described as Dee Why West Recreation Reserve, density estimates for swamp wallabies were obtained using two algorithms, one of which was specifically developed as part of this study. Estimates of up to 200 individuals were obtained for the east and west sides of Garigal National Park, while another 70 or so may be utilising Dee Why West Recreation Reserve south of Wakehurst Parkway. These values correspond well with published literature on typical densities of swamp wallabies, although further work is necessary to refine the approach. In particular, both density estimation algorithms assume that habitat quality is spatially equivalent. It is highly likely, however, that this assumption does not hold. Evidence for this was obtained via the hotspot analysis of density estimates which showed depressed densities in areas surrounding Wakehurst Parkway. This assumption could be rectified by applying a spatially-varying density estimate across a habitat suitability model for the region.

Although the derived probability density model is sound theoretically, parameterisation of the different components would be improved through targeted analysis from additional camera-trap surveys. Further work is needed to test the model under a range of conditions and species, and the NSW WildCount surveys would be very suitable for this purpose.

Over the course of 12 months, the local community (via the Northern Beaches Road-kill Committee) was encouraged to increase surveys in a repeatable and scientific manner. This resulted in the first reliable estimate of fauna fatalities on the major roads surrounding the reserves of interest. Although every effort was made to encourage volunteers to survey every road on a daily basis, this effort was always unlikely to be possible. Hence, estimates represent the minimum number of animals killed during the survey period and it is likely that more animals were killed during this time.

Of the 195 swamp wallabies recorded as being killed over the 12 months of sampling, 65 of those occurred on Mona Vale Road, while 48 occurred on the Wakehurst Parkway. These two major sources of fatalities together represent 58% of the total killed. Clearly, efforts to reduce fatalities on these two major roads will go a long way to ensuring the persistence of swamp wallabies in the region.

The population in Garigal NP west is primarily being threatened by fatalities on Mona Vale Road. If fatalities remain at current levels, survival of this sub-population could be offset by a dispersal rate from Ku-ring-gai Chase NP by 1% (or around 28 individuals per year) (Figure 18). This target does not, however, consider the need for dispersing individuals necessary to prevent extinction of the Garigal NP east and Dee Why West Recreation Reserve populations. If the only migration into Garigal NP east, and therefore also Dee Why West Recreation Reserve, is via Garigal NP west, then dispersal from Ku-ring-gai Chase NP would need to be higher.

From an estimated population of around 200 individuals, the Garigal NP east population had 61 individuals recorded as being killed on roads over 12 months. This value was derived by assuming that of animals killed on roads adjoining other populations (e.g. Ku-ring-gai Chase NP, Garigal NP west, and Dee Why West Recreation Reserve), only 50% were attributed to Garigal NP east. Without a fully marked population and known home ranges, this assumption was necessary, and may in fact be an underestimate of fatalities if the source-sink theory holds true. A dispersal rate from Ku-ring-gai Chase NP of over 60 animals per year was necessary to reduce the probability of extinction to an acceptable level (i.e. below 0.4) (Figure 18). Keeping Forest Way open for dispersing individuals between east and west Garigal NP is also essential, while simultaneously ensuring that fatalities on the road do not increase in number.

Clearly, the eastern population of Garigal NP and the population south of Wakehurst Parkway are the most susceptible to localised extinction risk. This study identified that enabling safe dispersal from Ku-ring-gai Chase NP while reducing fatality rates to more than half current levels are needed to ensure the populations persist.

This is a simulation study, based on estimates of demographics from the literature, densities from short-term camera-trap surveys, and a 12 month community driven road-kill survey. The findings should be placed within this context.

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