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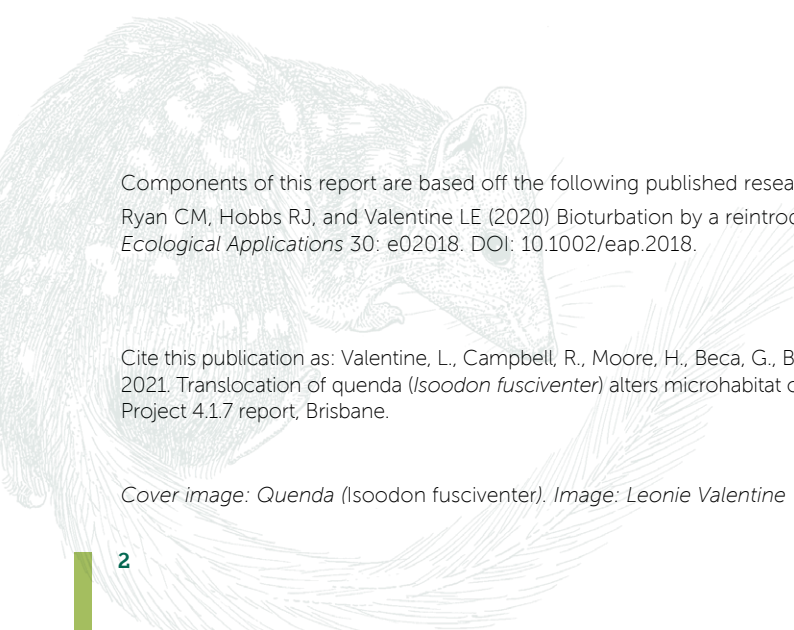
National Environmental Science Programme



Translocation of quenda (*Isoodon fusciventer*) alters microhabitat of urban bushland reserve

Leonie Valentine, Rebecca Campbell, Harry Moore, Gabrielle Beca, Daniel Bohórquez Fandiño, Bryony Palmer, Alice Reaveley, Catherine Ryan and Richard Hobbs

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Cover image: Quenda (*Isodon fusciventer*). Image: Leonie Valentine

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Executive summary

In 2013 a total of 46 quenda (*Isoodon fusciventer*) were translocated into Craigie Bushland, a ~42 ha predator proof enclosure located 19 km north of the Perth CBD. Between the initial release and December 2019, a total of 22 quenda trapping sessions occurred, where quenda were captured, measured, marked, and released. Across all trapping sessions (1980 total trap nights), a total of 1096 quenda captures (535 females, 548 males and 13 unknown captures that escaped prior to identification) were recorded. These captures comprised at least 185 unique individuals (91 females and 94 males), including 21 individuals (11 females, 10 males) from the original translocation. The most recent population estimates indicate there are likely to be a total of 69 (female = 39, male = 30) animals within the reserve, which is 39.1% larger than the founder population. Recent population estimates for both male and female quenda populations are less than the maximum recorded across the entire trapping period (female = 59, male = 53), which suggests populations may have peaked above the carrying capacity of the reserve and are now in the process of stabilising. Male populations peaked (February 2017) much earlier than the female population peak (December 2018), likely because the estimated home range of male quenda (~9.2 ha) inside Craigie Bushland is larger than the estimate for females (~2.1 ha), and thus resource competition for males is likely to be greater. The capacity of the monitoring program at Craigie Bushland to detect changes in quenda abundance is strongly influenced by the number of trapping sessions conducted per year. Trapping once a year will detect a population decline of 50% with 80% power.

The translocation of quenda into Craigie Bushland has substantially modified the microhabitat. Quenda foraging is prolific throughout the reserve. The digging activities of quenda increased two types of bare ground measurements: 1) the percent cover (%) of bare ground created by quenda, which was visually assessed; and, 2) the area (m²) of quenda digs, which was determined by measuring the bare ground created by quenda when they forage (e.g., measuring the spoil heap). The increase in quenda-created bare ground has subsequently reduced litter cover and litter depth; and the composition of litter is obviously patchy where quenda actively forage. The foraging activities of quenda are likely to increase litter burial into the soil, and thus aid litter decomposition, as buried litter is more likely to decompose due to the moister environment and increased exposure to microbial decomposers. Litter decomposition is one of the main ways nutrients that are stored in dead plant material are returned to the soil (and subsequently available once again for uptake by plants). In areas where quenda foraged, we detected higher amounts of phosphorus (a nutrient important for plant growth), but lower amounts of ammonium nitrogen. During a one-off survey, when quenda density had been high, we detected substantially lower estimated surface fuel loads where quenda could forage (likely due to the reduced litter cover and depth). Translocations of digging mammals into urban reserves are fairly rare; but represent an excellent opportunity to examine some of the accompanying changes that digging mammals can create on microhabitat. However, the population of quenda at Craigie Bushland are quite likely an isolated population and may be susceptible to a number of risks, including genetic constraints and stochastic events.

Introduction

Conservation translocations are an important tool for the conservation management of threatened species (Fischer and Lindenmayer, 2000; Seddon *et al.*, 2012; IUCN/SSC, 2013). Although conservation translocations have been essential for reducing the risk of extinction for several species (Seddon *et al.*, 2014; Hoffman *et al.*, 2015), translocations encounter many challenges and have a historically high failure rate (Fischer and Lindenmayer, 2000; Perez *et al.*, 2012; Berger-Tal *et al.*, 2020). Despite an increasing reliance on translocated fauna for multiple conservation purposes, post-translocation monitoring of individuals remains an uncommon practise (Fischer and Lindenmayer, 2000; Sheean *et al.*, 2012; Bubac *et al.*, 2019). Furthermore, monitoring the effects of translocations on broader habitat variables is even more limited (Armstrong and Seddon, 2008; Hale and Koprowski, 2018; Palmer *et al.*, 2020). Because the consequences of a translocation at the release site could range from beneficial to detrimental (Byers *et al.*, 2006), understanding how translocated species interact with their ecosystem is vital. Within an urban context, there is even less information on the success and impacts of fauna translocations. Here, fauna may be moved from one environment to another to reduce human-wildlife interactions, as part of conservation programs, and increasingly, as part of biodiversity offsets (Fischer and Lindenmayer, 2000; Perez *et al.*, 2012). Here, we monitor a population of quenda (*Isoodon fusciventer*) and the subsequent modifications they create in microhabitat following their translocation into an urban bushland reserve.

Australia's digging mammals

Due to their prolific digging activities, many of the mammal species translocated within Australia are considered ecosystem engineers (Palmer *et al.*, 2020). Animals that regularly dig for food or create burrows and whose soil turnover activities affect the availability of resources for other organisms are termed ecosystem engineers (Jones *et al.*, 1994). Through soil displacement, digging animals can modify biotic and abiotic habitat components, influencing many ecosystem functions (Davidson *et al.*, 2012; Fleming *et al.*, 2014; Mallen-Cooper *et al.*, 2019). During their excavation activities these animals break the soil surface layer, often reducing soil compaction while increasing soil moisture (Halstead *et al.*, 2020) while simultaneously capturing litter in excavated foraging pits or warrens (Garkaklis *et al.*, 2003; James *et al.*, 2009). This can subsequently alter soil nutrients and microbial properties as well as enhance seedling recruitment (James *et al.*, 2010; Eldridge *et al.*, 2016; Valentine *et al.*, 2018); all of which are likely to influence vegetation and landscape structure at broader scales (Whitford and Kay, 1999). Furthermore, animals that disrupt the surface litter layer through their behaviours have recently been implicated in altering fuel loads (Hayward *et al.*, 2016; Smith *et al.*, 2016; Foster *et al.*, 2020; Ryan *et al.*, 2020).

Digging for subterranean food or creating burrows alters the amount and distribution of bare ground and surface litter, which subsequently could alter fuel loads (Martin, 2003; Foster *et al.*, 2020). The surface fuel layer is an important determinant of fire behaviour (Sullivan *et al.*, 2012) and small scale variations in the combination of surface litter and bare ground can change fire behaviour (Atkins and Hobbs, 1995; Thaxton and Platt, 2006). Animals can create bare ground that may alter surface fuel loads and fire behaviour in many ways. Through wallowing, foraging, nesting and burrowing, animals can alter surface fuel layers. For example, American bison (*Bison bison*) were shown to create numerous deep wallows without vegetation that had an impact on reducing fire spread in North American grasslands (Knapp *et al.*, 1999). In Australia, seminar research in this field showed that litter mass, cover and predicted fire spread were lower inside fences where three threatened digging mammals had been reintroduced ten years previously (Hayward *et al.*, 2016).

Many of Australia's mammals have suffered range and population declines in the last 200 years (Woinarski *et al.*, 2015), including many digging mammals. Species within the critical weight range category (35–5500 g) are highly susceptible to predation by introduced red foxes (*Vulpes vulpes*) and feral cats (*Felis catus*), in addition to habitat loss and inappropriate fire regimes (Johnson and Isaac, 2009; Woinarski *et al.*, 2015). The loss, decline and range contraction of these digging mammals is likely to have drastically altered the surface heterogeneity of Australian landscapes (Fleming *et al.*, 2014). Translocations have played a critical role in the conservation management for these threatened species. Previous research indicates that the failure of reintroduced mammals to establish populations is linked to a number of issues, especially predation by introduced predators (Moseby *et al.*, 2011). However, reintroductions can be successful where predators are controlled (Richards and Short, 2003; Legge *et al.*, 2018) or where suitable refugia occur within the environment (Long *et al.*, 2005). In areas where Australian digging mammals have been reintroduced, their impact on the habitat is considered to assist in restoring components of ecosystem processes (James and Eldridge, 2007), including turning substantial soil (Munro *et al.*, 2019) that alters soil properties (Halstead *et al.*, 2020; Palmer *et al.*, 2021), changes fungal diversity (Clarke *et al.*, 2015; Dundas *et al.*, 2018), enhances seedling recruitment (James *et al.*, 2010) and alters fire behaviour (Hayward *et al.*, 2016; Ryan *et al.*, 2020). Given their role in ecological interactions and ecosystem processes, greater understanding of how translocations of digging mammals may alter their habitat is required.

Urban reserves and digging mammals

A developing field of research and conservation practice has focused on advocating the active return of nature into cities (Mata *et al.*, 2020). The concept of bringing back nature to cities is often embraced by urban sustainability and biodiversity conservation approaches. High quality urban reserves are a core feature of this concept, with urban reserves typically having the added goal of engaging people with nature (Standish *et al.*, 2013). However, very few translocations of digging mammals have occurred in urban areas. Mulligan's Flat in Canberra is an excellent example of where a threatened digging mammal, the eastern bettong (*Bettongia gaimardi*) has been successfully returned to a predator-proof enclosure on the city's outskirts (Munro *et al.*, 2019; Ross *et al.*, 2020). Translocation in urban areas may require fenced areas to reduce the likelihood of predation by both feral and domestic cats and dogs (*Canis familiaris*), as well as urbanised foxes. As most urban reserves are isolated and small, they may not support multiple species reintroductions, especially where the reintroduced species use similar resources. Fenced populations of translocated species in general may also suffer issues with over population (Moseby *et al.*, 2018) or inbreeding depression (Ottewell *et al.*, 2014) as well as stochastic events (Ramalho *et al.*, 2018), which may be pronounced in small, urban reserves.

Contemporary prescribed burning aims to manage fuel loads and fuel structure to mitigate effects of wildfire and has been used in Australia since the 1950s (Burrows and McCaw, 2013), although anthropogenic fire has been used in Australia for thousands of years, typically creating patchy mosaics of different aged vegetation (Hassell and Dodson, 2003). The cessation of traditional Aboriginal burning practices and altered fire management (Bowman, 1998; Abbott and Burrows, 2003), coupled with the loss of digging mammals, has likely altered the landscape of Australia (Martin, 2003). From a fire management perspective, urban and peri-urban reserves with remnant native vegetation, that are close to human habitations and infrastructure, offer challenging fire management conditions, often with competing objectives (Gill and Stephens, 2009; Driscoll *et al.*, 2010). From a human safety perspective, there is a desire to minimise fuel loads in reserves to reduce the risk of wildfire (Gibbons *et al.*, 2012). From a conservation management perspective, there may be a desire to maintain older post-fire aged habitat to cater to the resource requirements of some wildlife, such as endangered Carnaby's cockatoos (*Calyptrorhynchus latirostris*) (Valentine *et al.*, 2014). The ability of reintroduced native digging mammals to reduce fuel loads may have value to reduce fuel levels in between prescribed burning activities. However, it is unknown if a single reintroduced digging mammal species can influence microhabitat characteristics enough to influence fuel loads.

Quenda – a digging mammal around Perth

Although previously common, by the 1940s quenda, a medium-sized (adults: ~ 800 – 1200 g) marsupial bandicoot (Family Peramelidae) endemic to south-western Australia, had declined throughout much of their range due to a combination of threats, including habitat loss and predation by foxes and cats (Abbott, 2008). Home range estimates for quenda vary from 0.5 – 6.0 ha; and although these animals are typically solitary, they often have overlapping home ranges (Broughton and Dickman, 1991). Quenda are omnivorous, feeding upon invertebrates, including earthworms, beetles and their larvae, as well as underground fungi and plant material. They regularly forage upon subterranean fungal fruiting bodies and may play an important role in the dispersal of mycorrhizal fungi (Tay *et al.*, 2018). Like many of Australia's digging marsupials, they are predominantly nocturnal and search for their food via olfaction, creating ~45 distinctive conical foraging pits per night, with each individual turning over nearly 4 tonnes of soil annually (Valentine *et al.*, 2013). Foraging pits created by quenda are on average 7 – 8 cm deep and 10 cm wide, and have an associated spoil heap of evacuated soil, also known as the ejecta mound (Valentine *et al.*, 2013). Their foraging activities can alter soil and litter properties that subsequently influence seedling recruitment and growth (Valentine *et al.*, 2017; Valentine *et al.*, 2018). Although quenda are susceptible to predation by introduced foxes and cats, they persist in landscapes without predator control (Valentine *et al.*, 2013) and occur in peri-urban and urban reserves around Perth. The quenda is listed as a Priority 4 (Rare, Near Threatened or other species in need of monitoring) species under the WA Biodiversity Conservation Act 2016; and is the only commonly occurring digging marsupial within remnant habitat on the Swan Coastal Plain. Modelling also suggests that quenda in urban reserves in the greater Perth region are likely to be highly vulnerable to demographic decline, genetic deterioration and local extinction with increasing habitat loss (Ramalho *et al.*, 2018).

Much of the remnant vegetation on the Swan Coastal Plain includes banksia woodlands (Ritchie *et al.*, 2021). Banksia woodlands are comprised of a sclerophyllous layer of low trees (or tall shrubs) dominated by one or more *Banksia* spp., often interspersed with emergent *Eucalyptus* spp. and a species-rich understorey of sclerophyllous shrubs, sedges and herbs. Due to ongoing threats, including habitat clearing, invasive species and fire regime changes (Ritchie *et al.*, 2021), many of the banksia woodlands on the Swan Coastal Plain are part of a threatened ecological community listed as Endangered under federal legislation through the *Environment Protection and Biodiversity Conservation Act 1999* (Threatened Species Scientific Committee, 2016). These banksia woodlands are considered some of the most flammable woodlands in Australia (Burrows and Abbott, 2003), and prescribed burning is undertaken to reduce the risk of wildfire to human settlement.

In this study, we follow a population of quenda translocated into an urban bushland reserve in Perth, Western Australia and explore whether their foraging behaviour alters microhabitat characteristics. Specifically, we address the following questions:

1. How does the population of translocated quenda change over time? Associated with this question we also examine how often managers would have to conduct wildlife trapping in order to detect changes within the translocated population.
2. What is the influence of translocated quenda on microhabitat over time?
3. What is the effect of translocated quenda on estimated surface fuel loads?

Methods

Study site

The study took place at Craigie Bushland, a 53 ha urban reserve approximately 20 km north of Perth, Western Australia (Figure 1), on the traditional lands of the Whadjuk Noongar People. The reserve is considered to be an important remnant of bushland containing good condition habitat (Bush Forever site #303; Government of Western Australia, 2000) of the endangered Banksia Woodlands of the Swan Coastal Plain ecological community (Threatened Species Scientific Committee, 2016) and is managed by the City of Joondalup. The reserve is located within the Swan Coastal Plain bioregion (Thackway and Cresswell, 1995) mostly on Spearwood dune systems with an open banksia woodland dominated by *Banksia attenuata* and *B. menziesii*, interspersed with jarrah (*Eucalyptus marginata*), tuart (*E. gomphocephala*) and marri (*Corymbia calophylla*). Prominent understorey species include *Xanthorrhoea preissii* and the sedge *Mesomelaena pseudostygia*. The area has not experienced widespread fire for at least 20 years (K. Armstrong, City of Joondalup, pers. comm.). Few, if any, quenda were thought to occur in the bushland prior to the reintroduction.

The City of Joondalup constructed a predator proof fence (~ 42 ha) in 2010 to exclude introduced cats and foxes from the reserve, although the public can access the site and its walking trails. Although Craigie Bushland can currently be considered a secure reserve, up until late 2014 the pedestrian access gates and emergency access gates were often left open; and pedestrians could walk their dogs on lead through the reserve. Post 2014, dogs were excluded from the reserve and dual pedestrian-access gates were constructed to reduce the likelihood of incursion by cats, foxes or dogs. Despite the exclusion of introduced predators, potential native predators of quenda (especially juvenile quenda) exist in the reserve and include birds of prey, goannas (e.g., *Varanus gouldii*) and snakes. The reserve was rabbit-free for most of the population monitoring period, although in 2018, a few (approximately 2 – 3 individuals) domestic rabbits were observed within the reserve and are presumed to have been deliberately discarded individuals. Only three western grey kangaroos (*Macropus fuliginosus*) were known to occur in the reserve (observed occasionally) and their impact on our study plots was anticipated to be minimal.



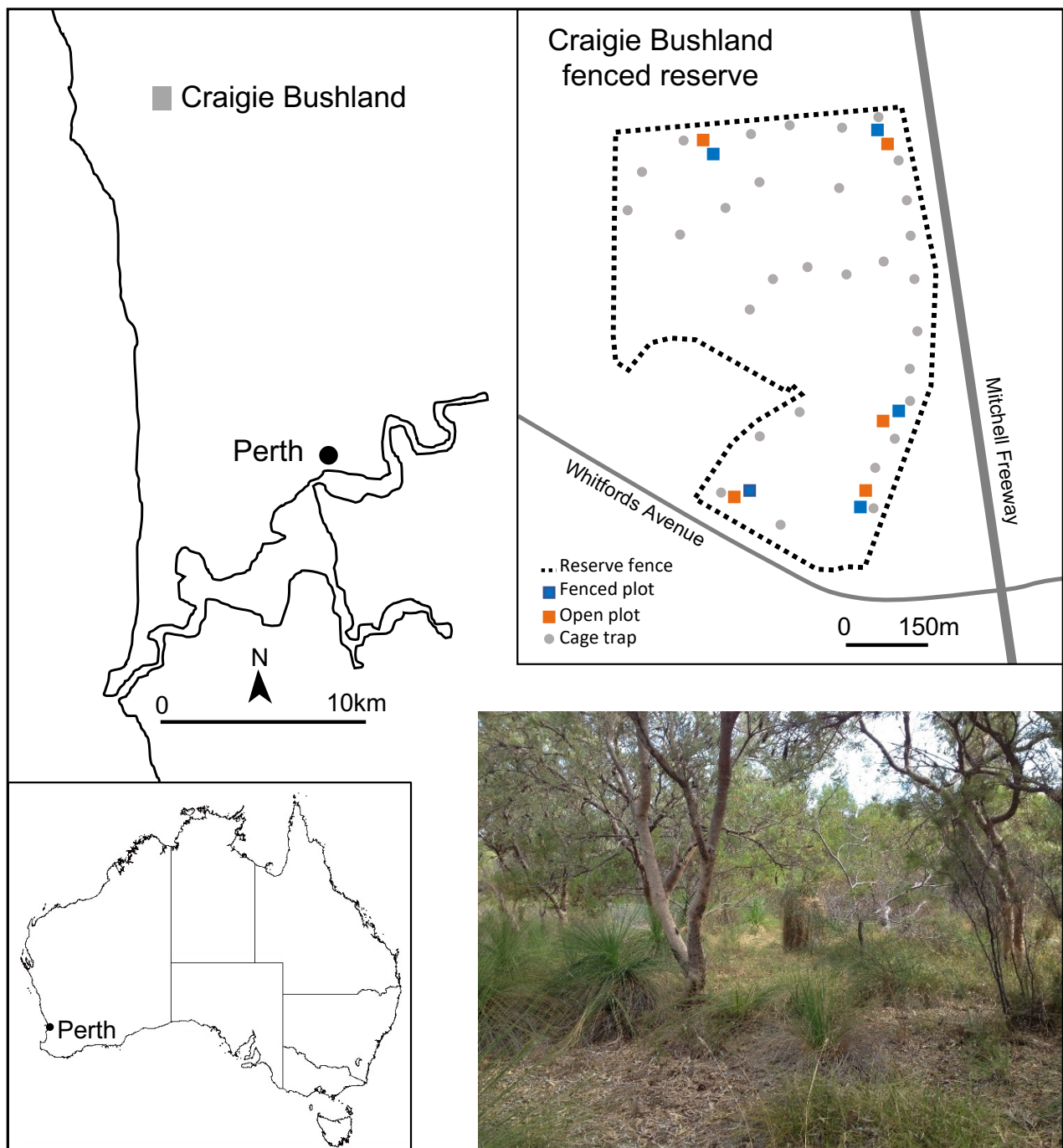


Figure 1. Location of Craigie Bushland, north of Perth, Western Australia. Inset shows predator-proof fence (indicated by dashed line) where quenda (*Isodon fusciventer*) were translocated in 2013. The inset also indicates the location of cage traps and the five paired plots located throughout the reserve. Paired plots consist of an open plot (orange square) where quenda can forage and a fenced plot (blue square) that excludes quenda. Image shows banksia woodland typical of the reserve.

Quenda translocation and population monitoring

The Western Australian Department of Biodiversity, Conservation and Attractions (DBCA) conducted a quenda translocation to Craigie Bushland during mid-2013. Initially, 42 quenda (24 females, 18 males) were translocated from Ellenbrook Nature Reserve into Craigie Bushland in May 2013, with a subsequent 4 quenda (1 female, 3 males) translocated from Twin Swamps in September 2013. The translocation occurred as part of the recovery actions for conservation management of the critically endangered Western Swamp Turtle (*Pseudemydura umbrina*), aimed to reduce the potential quenda predation upon turtle nests (Burbidge et al 2010).

Quenda population monitoring

Between August 2013 and November 2017, 3-monthly quenda monitoring trapping sessions occurred within Craigie Bushland. Between November 2017 and November 2019, 6-monthly quenda monitoring trapping sessions occurred. Trapping sessions occurred approximately every February, May, August and November, or every May and November, though due to logistical challenges, occasionally the trapping session occurred in a different month (see Table 1). Monitoring sessions employed 30 'Sheffield' wire cage traps (310 x 310 x 700 mm, Sheffield Wire Products, Welshpool WA) distributed throughout the reserve along fence lines and internal pathways (Figure 1). Traps were open for three consecutive nights and were baited with universal bait (peanut butter, rolled oats and sardine mix) and covered with hessian bags to provide shelter for captured animals. Traps were open and baited in late afternoon and all traps were inspected within 3 hours of sunrise the following morning. Captured animals were scanned for uniquely identifying microchips (LID-560 ISO Multireader, Microchips Australia, Keysborough VIC) inserted in previous trapping sessions. Upon their first capture, animals were individually marked with mini microchips (ISO FDXB Implants, Micro Products Australia, Southern River WA) introduced subcutaneously on the back of the neck, and a DNA sample (ear-hole punch, stored in 100% ethanol) was collected. Quenda that were originally translocated into the reserve were microchipped prior to their release. In addition, all animals were weighed (g) using pesolas and measurements were obtained for the head length and long pes length (mm, using calipers). Observations were also taken on the sex of individuals, and if female, the number of pouch young present. We also categorised whether the quenda was adult or subadult, based on a subjective assessment of the overall size of quenda and the condition of the pouch if female (if the pouch had been used we considered the quenda an adult) and relative size of testes if male. Sheffield cage traps are not effective at capturing juvenile quenda (< ~400 g). Where available, we collected scats from the base of cage traps by placing them in plastic vial containers (labelled with date and unique animal ID and stored frozen for future possible dietary analysis; samples hosted by Anna Hopkins, Edith Cowan University). During some trapping sessions, especially those when quenda carried pouch young, to reduce the amount of time quenda spent in cage traps we checked cages during the late evening. On these occasions, if there were no animals captured, cages were left open (and checked the following morning). If a quenda was captured, the animal was processed as usual and released; and the cage trap was kept closed (to maintain consistent trap effort).

Changes in microhabitat from quenda foraging

In April 2014, five paired fenced-open plots (each plot was 10 x 10 m), with a minimum of 180 m between pairs, were constructed (Figure 1) to examine how foraging activity of quenda may alter microhabitat variables. Locations of paired plots were randomly selected from quenda trapping points established by DBCA along fire break tracks, with each paired plot located a minimum of 30 m from the track edge and visually matched for broad habitat characteristics. The corner of each plot was marked using galvanised steel markers (One-Steel fence droppers, ~94 cm high). One plot in each pair was surrounded by a fence to exclude quenda (with a minimum 0.5 m buffer between the plot and fence) while the other plot remained open and was accessible by quenda. Fences were constructed from Waratah Longlife Blue Netting (1.8 m high x 4.0 cm diameter holes; 1.4 mm wire diameter mesh) and Waratah Galstar Extreme Steel posts (180 cm). The wire netting stood ~120 cm above the ground and had a 60 cm skirt that was pinned into the ground with 200 mm weed staples. Although we did not evaluate the effect of the fence itself, we do not believe the fences significantly influenced microhabitat in our study (e.g. by altering wind or run-off of litter) because i) banksia woodlands are relatively flat, with deep, sandy soils and little water run-off; and ii) we did not visually detect build-up of litter on the fence perimeters and we allowed a buffer zone between the fence and the internal plots to account for subtle effects of the fence on wind movement of litter. As the plots were constructed ~11 months after quenda were first translocated into the bushland reserve, quenda foraging pits were present in all plots initially. We periodically checked the integrity of the plots for any incursions into the fenced plots (3-monthly checks on fence integrity and visual scans for signs of fresh/new diggings). In 2018, we detected a minor incursion (evidence of recent quenda digging) in two of the fenced plots.

We surveyed microhabitat attributes in the fenced plots on four different occasions: April 2014 (within a few weeks of the fences being erected), October 2015, April 2017 and October 2018. During each survey we counted the number of detectable quenda digs within each plot by one person (LV) slowly circumnavigating the plot (ensuring minimal disturbance to ground) identifying foraging pits by their spoil heaps. To determine the amount of bare ground exposed by quenda while foraging, we measured the area of quenda digs. Firstly, the exposed soil of the spoil-heap for each dig was measured (length and width; cm) using a ruler. As spoil heaps were typically oval, the equation for the area of an ellipse was used to estimate the bare ground for each foraging pit. Secondly, the total amount of bare ground quenda created while foraging (area of quenda digs, m²) was calculated as the sum of each spoil-heap area converted m² plot⁻¹ (Equation 1):

$$\text{Equation 1: area of quenda digs} = \sum \frac{\left(\frac{(\text{spoil heap length cm})}{2} \right) \times \left(\frac{(\text{spoil heap width cm})}{2} \right) \times \pi}{10000}$$

The influence of quenda digging on microhabitat composition was also measured from five 0.5 m x 0.5 m quadrats in each plot (one quadrat in each plot quarter and one in the centre of each plot). In each quadrat, we visually estimated the percentage (%) surface cover of litter cover, quenda bare ground (bare ground created by quenda while they forage), bare ground (not identifiably created by quenda), coarse woody debris, and vegetation. Litter depth (nearest mm, with a ruler averaged over three locations within quadrat) was also recorded.

During the October 2018 microhabitat survey, in addition to the microhabitat response variables surveyed, we collected topsoil from each plot for soil-nutrient analyses. Using a clean hand trowel, we collected ~200 grams of topsoil (within the top 10 cm) per plot, bulked from each plot quarter and the centre of each plot. Standard soil nutrient analyses, undertaken by CSBP Soil and Plant Analysis Laboratory (Bibra Lake, Western Australia) examined the following soil nutrient quantities: nitrate nitrogen (mg Kg⁻¹), ammonium nitrogen (mg Kg⁻¹), phosphorous (mg Kg⁻¹; Colwell), potassium (mg Kg⁻¹; Colwell), sulphur (mg Kg⁻¹; KCl 40), organic carbon (%; Walkley-Black), electrical conductivity (dS m⁻¹, which provides an indication of the level of nutrient salts present) and pH level (CaCl₂).

Quenda foraging and surface fuel loads

Between April – June 2017, we collected additional information on the microhabitat changes quenda foraging created, with the specific focus of comparing estimated surface fuel loads between plots with and without quenda foraging. These data have been published (Ryan *et al.*, 2020), and here we present a subset of these data. At the time of sampling for these data (April-June 2017), the plots had been in place for three years and we had not detected any incursions into the fenced plots.

Fuels can be divided into four main types of layers on their vertical arrangement: canopy, elevated fuel, near surface fuel and surface fuel (Gould *et al.*, 2011). As digging mammals may alter ground level microhabitat characteristics, this study focuses on assessing surface fuel loads only. Surface fuels were comprised of litter, twigs, bark and other fine fuel lying on the ground (Gould and Cruz, 2012) and were assessed by adapting fuel sampling techniques described in Fontaine *et al.* (2012) and Ruthrof *et al.* (2016). To characterise fuel loads, the fuel is often sorted into different size categories that represent time-lag fuel classes that indicate the time it takes, under specified conditions, for a fuel particle to lose (or gain) ~ 63 % of the difference between its initial moisture and its equilibrium moisture (Gould and Cruz, 2012). Standardised time-lag fuel classes (Gould and Cruz, 2012) were adapted in this study and included: 1-hour time lag fuels (< 6 mm), 10-hour time-lag fuels (6 mm – 25 mm) and 100-hour time lag fuels (> 25 mm – 70 mm); with the combined classes representing total fuels.

To estimate biomass of surface fuel loads in plots in a non-destructive manner, we developed a linear regression between litter depth (cm) and litter mass (g) using destructive sampling techniques similar to Fontaine *et al.* (2012) from outside the plots. In ~2 ha area surrounding the paired plot locations, we assessed litter loads using 25 cm x 25 cm quadrats (n = 12-14 quadrats paired plot⁻¹, total of 65 quadrats) from locations with visibly varying litter cover (e.g., litter cover varying 10-100 %). Within each quadrat litter depth was measured with a ruler (mm, average of three places in the quadrat). The litter was then collected to mineral earth level, placed in paper bags, dried for 48 hours at 60 °C, weighed (providing total fuels, g), sorted into different time-lag fuel size categories (1-hour, 10-hour and 100-hour, described above) and weighed (g). No samples contained 100-hour fuels and this size category was excluded from subsequent analysis. The derived linear regression models between litter depth and fuel load consisted of the correct functional form with normally distributed errors and without heteroscedasticity and auto-correlated errors.

We collected information on surface fuels within plots using the point interception method (Elzinga *et al.*, 2001). At the point of intersection every 2 m along the edge and within the interior of each 10 x 10 m plot (n = 36 plot⁻¹) we measured microhabitat (% cover: vegetation, litter, coarse woody debris and bare ground; and litter depth, mm) in a 25 cm x 25 cm quadrat. To estimate fuel loads within each plot, we used the subset of quadrats (range: 21-27) where litter was recorded as a variable, and inserted the measured values for litter depth (mm) in the derived linear regression models (described above; Equations 3 and 4 in Results). The resultant value for surface fuel (total and time lag fuel size categories) in g quadrat⁻¹ were converted to tonnes ha⁻¹ and standardised in relation to the surface area available for digging within each plot.

Statistical analyses

Estimating population size and home range

Population abundance refers to the number of individuals of a given species within a given population. For the sake of this analysis, the population of interest can be defined as the quenda individuals enclosed within the predator proof fence at Craigie Bushland. In ecology, the most commonly used methods to measure abundance are capture-recapture (CR) models (Schwarz and Seber, 1999; Royle *et al.*, 2018). CR methods involve capturing a portion of a population, marking individuals and then releasing them (Royle and Converse, 2019). The proportion of marked individuals that are captured during a second trapping session is then used to estimate total population size, based on the assumption this proportion should be equivalent to the proportion of marked individuals within the population as a whole (Amstrup *et al.*, 2010).

Whilst traditional CR models can produce accurate population estimates, there are two fundamental limitations associated with this approach. The first limitation is that heterogeneity in capture probability between traps as a result of individual home range centre locations is not accounted for (Royle *et al.*, 2013). For example, a trap placed two metres from an animal's home range centre is probably more likely to capture that animal than a trap placed 200 metres from its home range centre, and this can influence abundance estimates. The second limitation of traditional CR models is that the effective sampling area from which abundance estimates are derived from is imprecise (Royle *et al.*, 2013) – in order to measure the abundance of animals within a pre-defined area, one must assume that the trapping array used is capable of capturing all animals within that area. In the case of the quenda population within Craigie Bushland, this is unlikely to be true (Appendix 1).

Spatial capture-recapture (SCR) models are increasingly used to overcome the limitations of traditional CR models. SCR models are an extension of traditional CR models that use animal spatial capture history data to make inferences about population densities that can later be used to derive abundance estimates (Borchers, 2012; Efford and Fewster, 2013). SCR models were used to estimate quenda density, and later abundance, within the fenced area of Craigie Bushland using the 'secr' package (Efford, 2015) in R version 3.5.3 (R Core Team, 2019). A secr model is comprised of two nested models. The first model is the detection model, which relates the probability of detecting an individual at a particular detector to the distance of the detector from a central point in each animal's home range (Efford, 2015). The second model is the state model, which describes the distribution of animal's home range centres (Efford, 2015).

SCR models were fit to the detection data using a hazard half-normal detection function, which is described by the detection model and controlled by the parameters λ (lambda) and δ (sigma). Lambda describes the per capita detection probability per unit effort. Sigma describes the spatial extent of an individual's use of the landscape, such that animals with large home ranges have large sigma values. The parameter D describes animal density. SCR models allow for the fact that individual covariates may influence model parameters lambda, sigma or density. We made apriori assumptions that lambda, sigma and density were all likely to vary by quenda sex (female, male) and quenda age (subadult, adult). It was also expected that quenda density was likely to vary across monitoring sessions. Models were run with all combinations of individual covariates, and then compared using Akaike Information Criterion (AIC).

All models were implemented such that the predicted location of quenda home range centres was constrained by a 320m buffer from traps, and also by the fence surrounding the enclosure. The buffer distance of 320m was selected based on initial estimates of sigma as recommended in Efford (2019), and should incorporate the home range centres of all animals captured in traps. Population size estimates were calculated by scaling density estimates to the total area enclosed within the fenced section of Craigie Bushland.

We were also interested in examining potential variables, such as year of survey, time of year of survey and rainfall, that may be useful for predicting quenda population (based on the secr population estimates above). Predictor variables included were: i) year: year in which trapping session occurred (2013 – 2019) which is also a surrogate for time since translocation; ii) quarter: the quarter in the calendar year in which the trapping session occurred; iii) quarter precipitation: precipitation in the quarter in which a trapping session occurred; and iv) previous quarter precipitation: precipitation in the quarter prior to the quarter in which a trapping session occurred. Precipitation records were collected from nearby Australian Bureau of Meteorology (BOM) weather station (Wanneroo: 009105). We used generalised linear models (GLM) fit with all combinations of predictor variables to estimate our response variables: quenda abundance, female quenda abundance and male quenda abundance, based on the secr abundance estimates (described above). Models were compared using Akaike Information Criteria adjusted for small sample size (AICc) and we report only the most parsimonious models with $\Delta AICc < 2$. All model selection was conducted using the dredge function in R package MuMIn (Barton, 2013).

We used predicted values of sigma from the most parsimonious secr models, fit with a hazard half-normal detection function, to estimate quenda home range sizes. For models fit with hazard half-normal detection functions, 95 % individual spatial activity is predicted to occur within a circle where the radius is equal to sigma multiplied by 2.44. Sigma values were derived from secr models where individual covariates 'Sex' and 'Age' were included as predictors for sigma, such that estimates could be calculated for the following categories: subadult female, adult female, subadult male and adult males.

Quenda metrics

To further explore attributes that may influence quenda population dynamics (e.g., recruitment), we generated response variables for the average body mass (g) of i) female quenda per session, ii) male quenda per session, iii) the proportion of breeding females (e.g., female quenda carrying pouch young) per session and the total number of pouch young (summed across all females) observed per session. As male quenda are typically heavier than females, and the mass of females may vary depending on whether they are carrying pouch young, we analysed male and female quenda separately. We used GLM fit with all combinations of the predictor variables described above (year, quarter, quarter precipitation and previous quarter precipitation) and included the overall quenda population estimates per session (derived from the best secr model above). Models were compared using AICc and we report only the most parsimonious models with $\Delta AICc < 2$. All model selection was conducted using the dredge function in R package MuMIn (Barton, 2013).

Detecting population changes with trapping effort

Often, a primary objective of ecological monitoring programs is to detect changes in a variable of interest through time (Lindenmayer and Likens, 2010). Being able to detect changes in population abundance can be of assistance for land managers – in terms of detecting potential declines. In the case of the quenda monitoring program at Craigie Bushland, changes in the quenda population size from year to year can also be expressed as changes in quenda density. The amount of confidence that we can invest in the Craigie Bushland monitoring program to detect changes in quenda density from year to year, or statistical power, is dependent on the precision with which yearly estimates of density are measured (Guillera-Arroita and Lahoz-Monfort, 2012). The amount of precision with which quenda density is measured within a year, should be dependent on trapping effort. For example, using three live trapping sessions per year (nights * traps = 270 trap nights) is likely to produce a more accurate yearly estimate of density than using one live trapping session (90 trap nights), and therefore as a method, should hold more statistical power.

To test the effect of increasing or decreasing yearly trapping effort on the statistical power of the Craigie Bushland quenda monitoring program, a power analysis was conducted. The first step in measuring statistical power was to measure the precision with which monitoring designs measure density, using different levels of trapping effort. To do this, we simulated live trapping data for four different monitoring designs –one session per year, two sessions per year, three sessions per year and four sessions per year. Simulations were conducted using the R package '*secdesign*' (Efford, 2018), with 500 simulations run for each monitoring design. All relevant data required to simulate data such as spatial array of traps, nightly replicates, and estimates of density, g and sigma, was sourced from null secr models described above. We then fit spatially explicit capture-recapture models for each of the simulated datasets using the hazard half-normal detection function. From the fitted models, we summarized the estimated population density across the replicates and assessed precision of each monitoring design. We used relative standard error (RSE) as the measure of precision. RSE was converted to statistical power using formulae outlined in Appendix 3 of Efford and Boulanger (2019). Power estimates of 0.80 or over are generally acceptable for most ecological monitoring programs (Di Stefano, 2003).

Changes in microhabitat from quenda foraging

To examine changes in microhabitat between treatments (fenced and open) and over time (during different surveys: April 2014, October 2015, April 2017 and October 2018) we used generalised linear mixed-effect models (GLMMs) and linear mixed-effect models (LMMs) from the '*lme4*' package (Bates *et al.*, 2015) in R Statistical Software version 4.0.4 (R Core Team, 2019). We included the paired plot as a random effect in all models. We used a GLMM, with a poisson distribution, to examine differences in the number of quenda digs between treatments and over time by modelling all combinations of the predictor factors, including the null model. Model selection was then conducted using the '*model.sel*' function in the R package MuMIn (Barton, 2013), with models compared using AICc and here we report only the most parsimonious model with $\Delta AICc < 2$. Differences in other microhabitat response variables (area of quenda digs, litter depth and the percent cover estimates: quenda bare ground, litter and vegetation) to treatment and over time were examined with LMMs with a Gaussian distribution. For all response variables, when either year of survey or the interaction term were significant ($P < 0.05$), pairwise comparisons (either between survey year, or between treatment within each survey year) were conducted using the '*emmeans*' package (Lenth, 2020). Area of quenda digs and litter depth were log transformed while percent cover estimates underwent a logit-transformation on the proportional data.

To examine differences in soil nutrient response variables between treatment (fenced and open), we used LMMs with a Gaussian distribution and included paired plot as a random effect in all models. All soil nutrient response variables (nitrate nitrogen, ammonium nitrogen, phosphorous, potassium, sulphur, electrical conductivity and Ph CaCl₂) were log-transformed, except for organic carbon (%) which underwent a logit transformation on the proportional data.

Quenda foraging and surface fuel loads

To compare surface fuel loads between fenced and open plots, we used LMMs in the 'lme4' package in R (Bates *et al.*, 2015) with paired-plot as a random effect. Wald chi-squared tests were used to estimate p-values from the 'car' package in R.

Results

Between the initial post-translocation survey (August 2013) and December 2019, a total of 22 trapping sessions occurred, where quenda were captured, measured, marked, and released (Table 1). Across all trapping sessions (1980 total trap nights), a total of 1096 quenda captures (535 females, 548 males and 13 unknown captures that escaped prior to identification) were recorded. These captures comprised at least 185 unique individuals (91 females and 94 males), including 21 individuals (11 females, 10 males) from the original translocation from Ellenbrook NR. None of the individuals translocated from Twin Swamps NR were ever recaptured. Not all quenda individuals were captured at every trapping session. Fifty-eight individuals (29 females) were only captured during one trapping session. However, several individuals were repeatedly trapped, with 81 individuals (41 females) captured in four or more trapping sessions. One female, first captured at Craigie Bushland a year after the translocations, was trapped on 15 different trapping sessions. Similarly, a male quenda (first captured at Craigie Bushland more than 18 months after the translocations) was captured during 13 different trapping sessions. Of the quenda that were originally translocated into Craigie Bushland in May 2013, the last time an individual (a female) was captured was February 2017, indicating she would have been at least 4 years old as she was an adult when translocated. Of the new individuals captured at Craigie Bushland (165 individuals), 35 were considered subadults at first capture, though all new adults captured were often smaller than adults captured during earlier trapping sessions.

Table 1. Summary information from the quenda (Isodon fusciventer) population monitoring conducted at Craigie Bushland. Quenda were initially translocated into the reserve in May 2013. From August 2013 to December 2018, trapping sessions were conducted approximately every 3 months, and then approximately every 6 months until November 2019. A trapping session consisted of 30 cage traps open for 3 consecutive nights. New quenda refer to quenda that have not been captured previously; recaptured quenda refer to quenda that have been captured in previous trapping sessions (or were translocated individuals) identified by their unique microchip.

Date	Trapping Session	New quenda	Recaptured quenda	Total Known Quenda
Aug-13	1	0	20	20
Nov-13	2	4	11	15
Feb-14	3	7	14	21
May-14	4	4	13	17
Aug-14	5	5	10	15
Nov-14	6	9	14	23
Feb-15	7	10	17	27
Jun-15	8	12	23	35
Aug-15	9	6	37	43
Nov-15	10	5	24	29
Feb-16	11	11	29	40
May-16	12	11	32	43
Aug-16	13	7	40	47
Nov-16	14	6	32	38
Feb-17	15	4	47	51
May-17	16	5	39	44
Aug-17	17	9	38	47
Nov-17	18	6	36	42
May-18	19	7	26	33
Dec-18	20	14	26	40
May-19	21	8	30	38
Nov-19	22	15	22	37

Estimating translocated population size and home range

The most parsimonious secr model for predicting quenda density in Craigie Bushland included 'Sex' and 'Age' as predictors for sigma, lambda and density (Table 2). Based on this model, the combined population estimate at the final trapping session (November 2019) was 69 individuals (Lower Confidence Interval, LCI = 49; Upper Confidence Interval, UCI = 96; Figure 2). This estimate was 25.7% lower than the maximum predicted population estimate ($n = 95$, LCI = 71, UCI = 126), which was recorded for the session occurring February 2017. The female population estimate for November 2019 was 39 individuals (LCI = 25, UCI = 60), which was 23% higher than the male population estimate for the same session ($n = 30$, LCI = 18, UCI = 49). The highest female population estimate was recorded in December 2018 ($n = 59$, LCI = 42, UCI = 85), and the highest male population estimate was recorded in February 2017 ($n = 56$, LCI = 39, UCI = 80) (Figure 2).

The best generalised linear model for predicting quenda population size across both sexes indicated year of survey had a significant effect (Table 3, Figure 3), with higher number of quenda being predicted in later survey years (ie. population of quenda increased with time since reintroduction). Predicted quenda population size was 53.6% higher in 2019 ($n = 82$, LCI = 69, UCI = 95) than it was in 2013 ($n = 38$, LCI = 26, UCI = 50). Year of survey was also a significant positive predictor for female abundance (Table 3, Figure 3). The best model for predicting male population size was the null model.

Table 2. Detection parameters from the best 'secr' model for predicting quenda (*Isodon fusciventer*) density in Craigie Bushland. The parameter lambda describes the per capita detection probability per unit effort. Sigma describes the spatial extent of an individual's use of the landscape, such that animals with large home ranges have large sigma values.

		Estimate	SE	LCL	UCL
Sub-adult female					
Lambda		0.11	0.07	0.03	0.34
sigma		47.52	16.90	24.15	93.47
Adult female					
Lambda		0.23	0.02	0.19	0.28
sigma		67.58	5.13	58.25	78.39
Sub-adult male					
Lambda		0.04	0.03	0.01	0.16
sigma		98.85	35.00	50.40	193.89
Adult male					
Lambda		0.10	0.01	0.08	0.13
sigma		140.59	11.44	119.90	164.85

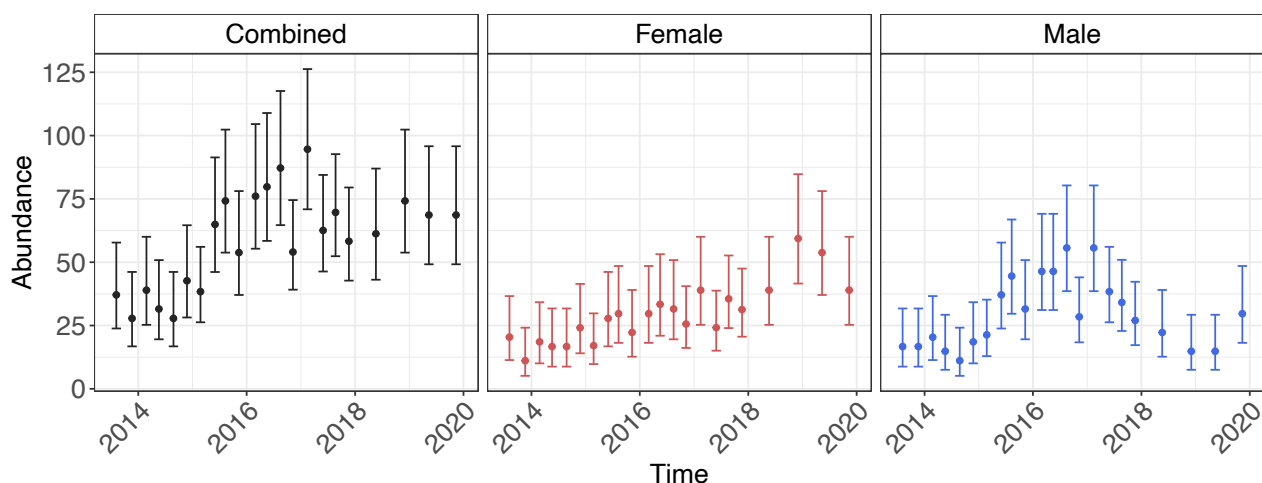


Figure 2. Population estimates for translocated quenda (*Isodon fusciventer*) at Craigie Bushland derived from spatially explicit capture-recapture (secr) models.

Table 3. Outputs from the best generalised linear models for predicting i) overall quenda (*Isoodon fusciventer*) abundance and ii) female quenda abundance in Craigie Bushland. Important effects are marked in bold. An important effect was recognised if $p < 0.05$.

		Estimate	SE	p
	<i>Quenda abundance</i>			
(Intercept)		-14763.76	3597.16	<0.05
Year		7.35	1.78	<0.05
	<i>Female quenda abundance</i>			
(Intercept)		-11320.00	1581.00	<0.05
Year		5.63	0.78	<0.05

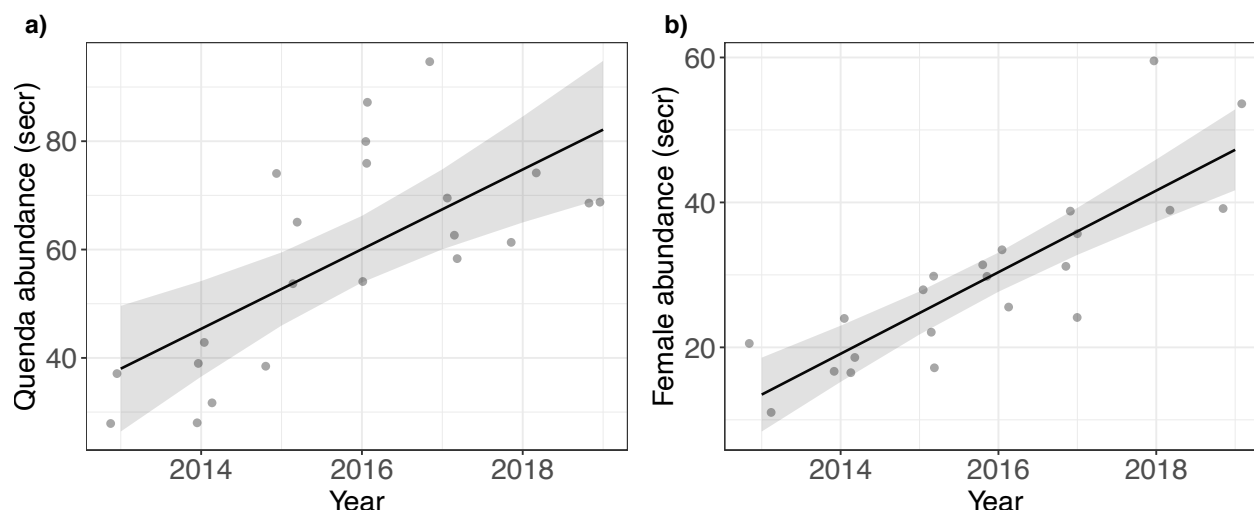


Figure 3. Model outputs from generalised linear models for predicting the abundance of a) all quenda (*Isoodon fusciventer*) and b) female quenda with year of survey. The line represents model predictions (shaded areas indicate 95% confidence intervals) and grey dots represent the raw data (which have been jittered).

The spatial activity (home range) estimates for quenda varied substantially depending on both sex and age of the quenda (Table 4). The largest home range estimates were for adult males (9.24 ha, CI 6.72 – 12.70 ha), followed by sub-adult males (4.57 ha, CI 1.19 – 17.57 ha), adult females (2.13 ha, 1.59 – 2.87 ha), and sub-adult females (1.05 ha, CI 0.27 – 4.08 ha).

Table 4. Home range (ha) estimates derived from the best secr models for predicting quenda (*Isoodon fusciventer*) density in Craigie Bushland. LCL – Lower confidence limits, UCL – Upper confidence limits.

Sex	Age	Home range (ha)	LCL	UCL
Female	Sub-adult	1.05	0.27	4.08
Female	Adult	2.13	1.59	2.87
Male	Sub-adult	4.57	1.19	17.57
Male	Adult	9.24	6.72	12.70

Quenda metrics

The best generalised linear model for predicting average mass of female quenda included precipitation in the previous quarter, but this variable was not a significant predictor (Table 5). The best model for predicting average mass of male quenda included year and precipitation in previous quarter, though this was not significant (Table 5). Year was a significant predictor of male quenda mass and predicted average quenda mass declined by 2.95% (35.2 g) every year since 2013 (the year of translocation; Figure 4). The best model for predicting the proportion of female quenda carrying pouch young included previous quarter precipitation as the only predictor. Model predictions indicated that for every 100ml of rain that fell in the previous yearly quarter, the proportion of females carrying pouch young in the current quarter increased by 11.08% (Table 5, Figure 5). The best model for predicting the total number of pouch young observed in a trapping session included yearly quarter as the only predictor. The average total predicted number of pouch young was much higher in the third and fourth quarters, and lower in the first two quarters (Table 5; Figure 5).

Table 5. Output from the most parsimonious generalised linear models for predicting the average mass (g) of i) female quenda (*Isoodon fusciventer*) and ii) male quenda per trapping session; as well as the iii) proportion of female quenda carrying pouch young and iv) the total number of quenda pouch young observed per trapping session in Craigie Bushland. Important effects are marked in bold. An important effect was recognised if $P < 0.05$. PQP = Precipitation in previous quarter; Quarter 1 = Jan – March, Quarter 2 = April – June, Quarter 2 = July – September, Quarter 4 = October – December.

		Estimate	SE	p
	<i>Female quenda mass</i>			
(Intercept)		862.97	28.43	<0.05
PQP		0.20	0.12	0.11
	<i>Male quenda mass</i>			
(Intercept)		72179.00	24047.85	0.01
PQP		0.31	0.15	0.05
Year		-35.29	11.93	0.01
	<i>Proportion of breeding females</i>			
(Intercept)		0.15	0.08	0.08
PQP		0.00	0.00	<0.05
	<i>Number of quenda pouch young</i>			
(Intercept)		3.00	5.88	0.62
Quarter2		5.67	7.59	0.47
Quarter3		27.60	7.89	<0.05
Quarter4		27.43	7.37	<0.05

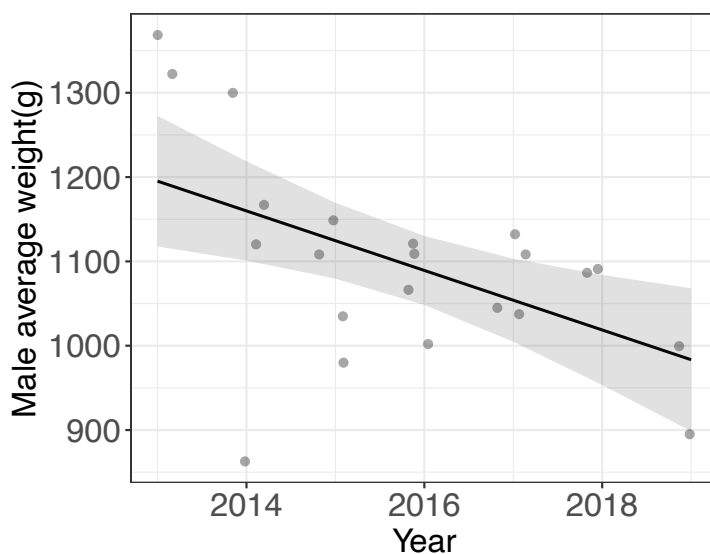
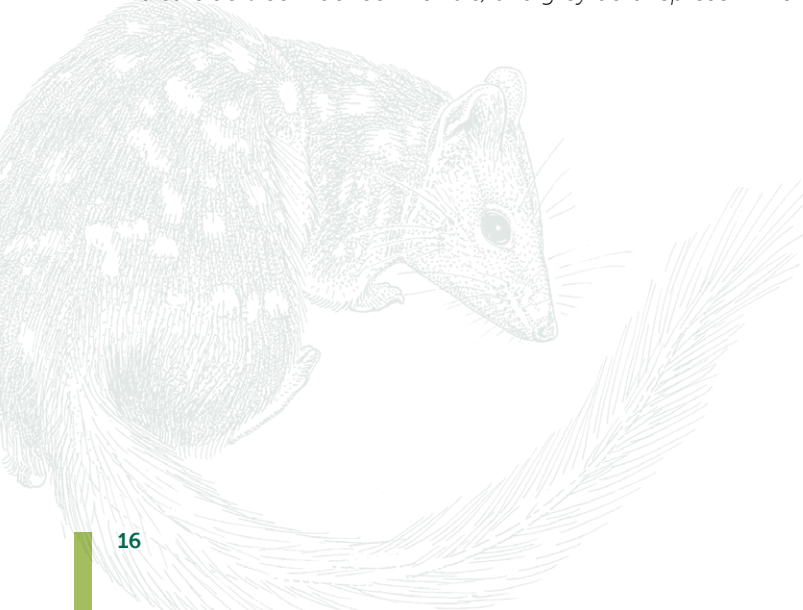


Figure 4. Model outputs from generalised linear model for predicting the response of average male quenda (*Isoodon fusciventer*) body mass (g) with year of survey. The line represents model predictions (shaded areas indicate 95% confidence intervals) and grey dots represent the raw data (which have been jittered).



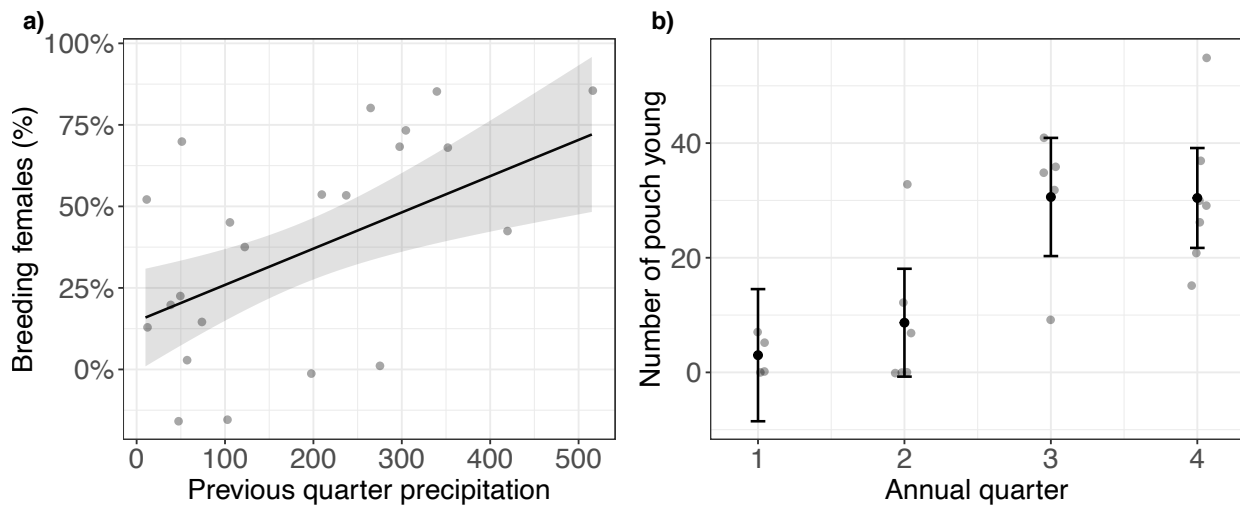


Figure 5. A) Model outputs from generalised linear models for predicting the response of proportion of breeding female quenda (*Isoodon fusciventer*) with precipitation in the previous quarter (mm). The line represents model predictions (shaded areas indicate 95% confidence intervals) and grey dots represent the raw data (which have been jittered). B) The average ($\pm$ 95% CI) total number of quenda pouch young observed per trapping session with annual quarter, (when survey was conducted during the calendar year: 1 = Jan – March; 2 = April – June; 3 = July – September; 4 = October – December).

Detecting population change with trapping effort

We simulated live trapping data using parameters from the secr null model (density = 0.70 quenda per ha, λ = 0.19, σ = 95m) to calculate the required number of yearly replicates to achieve 80% statistical power in detecting various magnitudes of quenda density decline. The precision of density estimates (RSE mean) was lower when trapping effort was lower (Table 6, Figure 6). To detect declines of 50% or more between years, a single trapping session per year (3 nights with 30 traps) is sufficient to achieve 80% power. To detect a 30% decline in quenda density between years, at least three quenda trapping sessions are required to achieve 80% power.

Table 6. Relative standard error (RSE) and power estimates for different scenarios of trapping intensity (sampling occasions per year) for detecting varying magnitudes of decline in quenda (*Isoodon fusciventer*) density in Craigie Bushland. Power estimates of 0.80 or over are generally acceptable for most ecological monitoring programs (Di Stefano, 2003)

Density decline	Sampling occasions per year	RSE (mean)	RSE (sd)	power
30%	1	0.23	0.01	0.57
30%	2	0.16	0.01	0.72
30%	3	0.13	0.01	0.82
30%	4	0.11	0.01	0.88
50%	1	0.23	0.01	0.84
50%	2	0.16	0.01	0.95
50%	3	0.13	0.01	0.99
50%	4	0.11	0.01	1.00
80%	1	0.23	0.01	0.99
80%	2	0.16	0.01	1.00
80%	3	0.13	0.01	1.00
80%	4	0.11	0.01	1.00

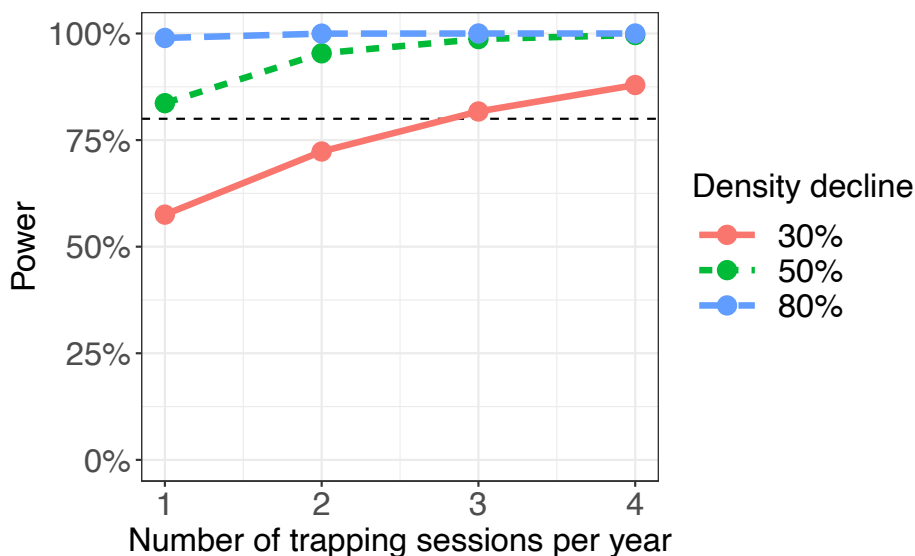


Figure 6. The statistical power of monitoring designs in detecting magnitudes of decline in quenda (*Isoodon fusciventer*) density in Craigie Bushland. The dotted line represents 80% statistical power.

Changes in microhabitat from quenda foraging

Within 18 months of the fenced plots being erected, there were visible differences between the fenced and open plots. Most notably, casual observation indicated more visibly disturbed bare ground (by quenda digs) in the open plots. When examining differences in the number of quenda digs created, the best model, carrying >99% of the model weight (conditional $R^2 = 0.98$), was the full interaction model (Table 7). The number of quenda digs varied significantly between fenced and open treatments at every survey, except for the first survey in April 2014 (Figure 7) when fences had recently been erected (Appendix 2). However, within 3 years of the fences being in place, quenda digs were 17 times greater in the open than the fenced plots (April 2017; Figure 7, Appendix 2). The area of quenda digs followed a similar pattern (Figure 7, Appendix 2), with greater amounts of bare ground created by quenda in every year except for the first survey. Within 3 years of the fences being erected, the area of quenda digging was 23 times greater in the open than the fenced plots (Figure 7). There was also a significant interaction term between treatment and survey for litter depth (Appendix 2). In the last two survey years, litter depth in the open plots was less than half of the litter depth in the fenced plots (Figure 8, Appendix 2). The percentage cover of litter was also lower in the open plots compared to the fenced plots in all but the first survey year (Figure 8, Appendix 2). In contrast, with the exception of the first survey, the percent cover of quenda created bare ground was 4 – 10 times higher in the open plots compared to the fenced plots (Figure 8, Appendix 2). The greatest difference between fenced and open plots was also during the April 2017 survey. Non-quenda created bare ground was infrequently observed, and not analysed. Typically, most of the bare ground we observed was created by quenda. The vegetation cover did not vary with treatment but did differ between year of survey being highest in the final year of survey (Appendix 2). The cover of CWD was infrequently detected and not analysed (Appendix 2).

Table 7. Model selection output from generalised linear mixed models (GLMMs) examining the number of quenda (*Isoodon fusciventer*) digs with treatment (fenced and open plots) and survey year (April 2014, October 2015, April 2017, October 2018). The best model ($\Delta AICc < 2$) is highlighted in bold.

Model	Intercept	Df	LogLik	AICc	$\Delta AICc$	weight
Interaction	3.898435	9	-239.09	502.1791	0	1
Additive	3.01791	6	-422.939	860.424	358.2449	1.61E-78
Treatment only	3.186917	3	-501.69	1010.047	507.868	5.22E-111
Survey only	3.912523	5	-866.047	1743.858	1241.679	2.36E-270
Null	4.081534	2	-944.798	1893.92	1391.741	6.13E-303

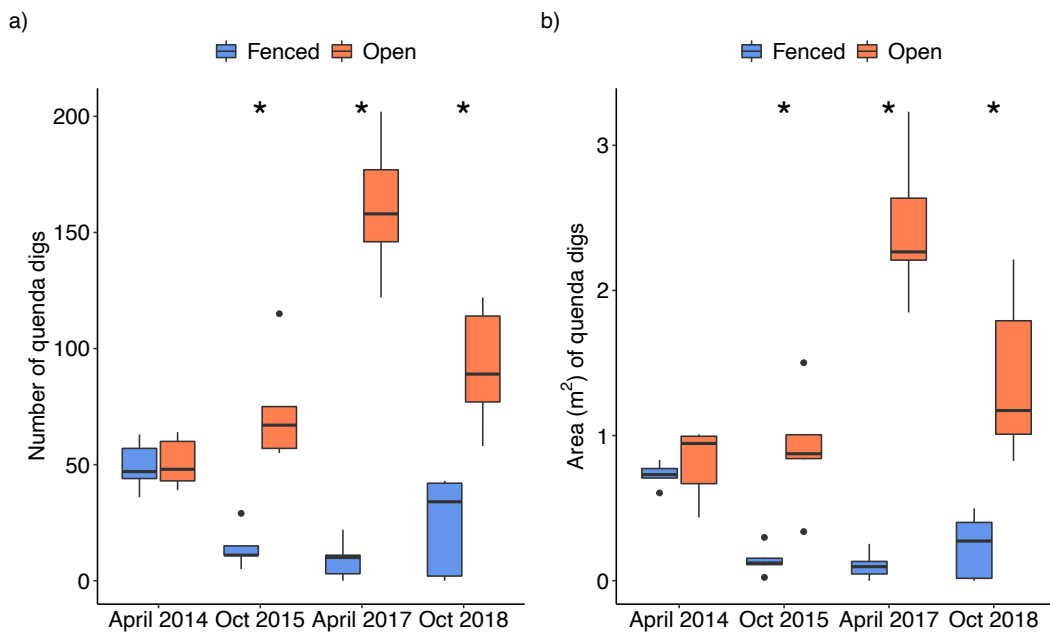


Figure 7. The a) number of quenda digs and b) area of quenda digs varied in fenced (quenda excluded) and open (quenda accessible) plots in different survey years at Craigie Bushland. Fences were erected in April 2014, approximately 11 months after quenda (*I. fusciventer*) were first translocated into the reserve. Asterisks indicate significant post hoc pairwise comparisons between treatments within each survey.

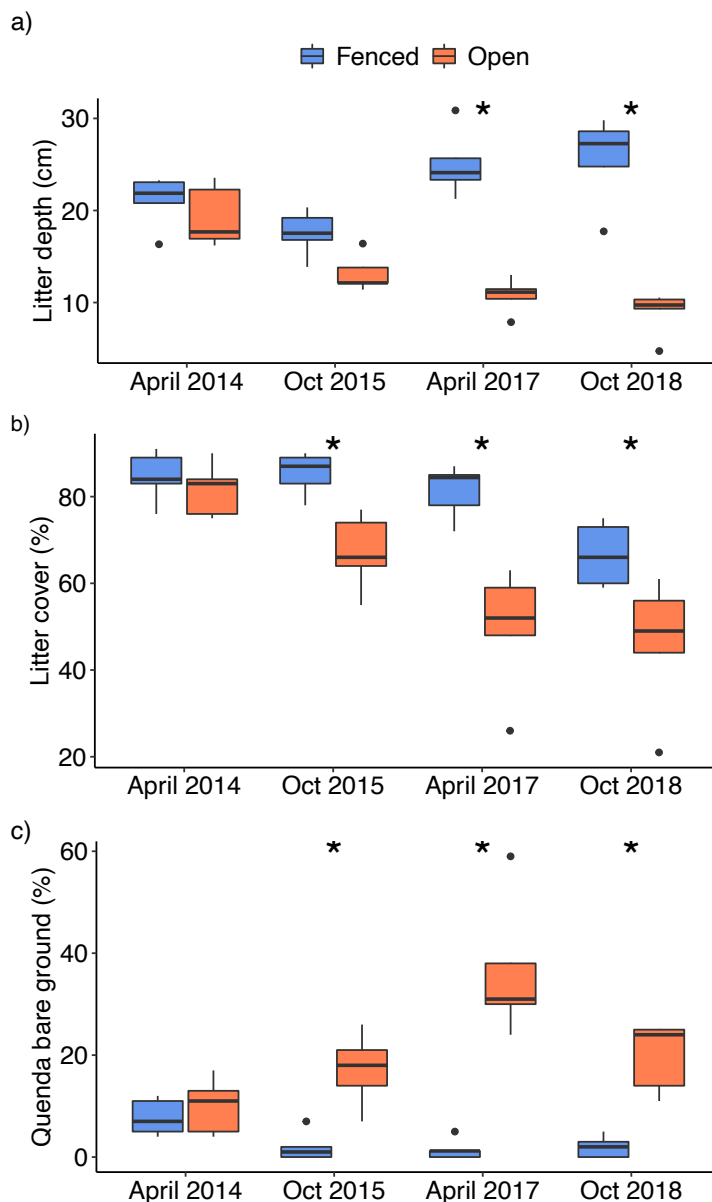


Figure 8. Differences in the a) litter depth, b) litter cover and c) quenda-created bare ground cover in fenced (quenda excluded) and open (quenda accessible) plots in different survey years at Craigie Bushland. Fences were erected in April 2014, approximately 11 months after quenda (*I. fusciventer*) were first translocated into the reserve. Asterisks above boxplots indicate significant post hoc pairwise comparisons ($P < 0.05$) between treatments within each survey.

Soil nutrient samples collected in October 2018 indicated that most variables, except two, were similar between open and fenced plots (Table 8). However, the concentration of ammonium nitrogen was significantly higher in the fenced plots compared to the open plots while the concentration of phosphorus was higher in the open plots (Figure 9).

Table 8. The back-transformed estimated marginal means ($\pm$ standard error) and statistics from linear mixed models on soil nutrient response variables collected in fenced (quenda-excluded) and open (quenda accessible) paired-plots in Craigie Bushland. Soil collection was conducted in October 2018, 4.5 years after the fences were erected and 5.5 years after the quenda (*Isodon fusciventer*) were translocated into the reserve. Statistically significant differences ($P < 0.05$) are shown in bold.

	Fenced ($\pm$ SE)	Open ($\pm$ SE)	df	F
Nitrate nitrogen (mg Kg ⁻¹)	0.44 (0.08)	0.52 (0.10)	1,5	0.463
Ammonium nitrogen (mg Kg⁻¹)	5.39 (0.62)	4.16 (0.48)	1,5	14.283*
Phosphorous (mg Kg⁻¹)	2.22 (0.16)	3.04 (0.21)	1,5	68.221***
Potassium (mg Kg ⁻¹)	28.0 (4.41)	35.5 (4.11)	1,5	0.224
Sulphur (mg Kg ⁻¹)	2.04 (0.28)	1.76 (0.24)	1,5	0.958
Organic carbon (%)	0.017 (0.002)	0.0180 (0.002)	1,10	0.071
Electrical conductivity (dS m ⁻¹)	0.029 (0.003)	0.026 (0.002)	1,5	2.541
pH level (CaCl ₂)	4.81 (0.16)	4.71 (0.16)	1,5	1.400

Asterisks indicate significant (* $P < 0.05$, *** $P < 0.001$) result.

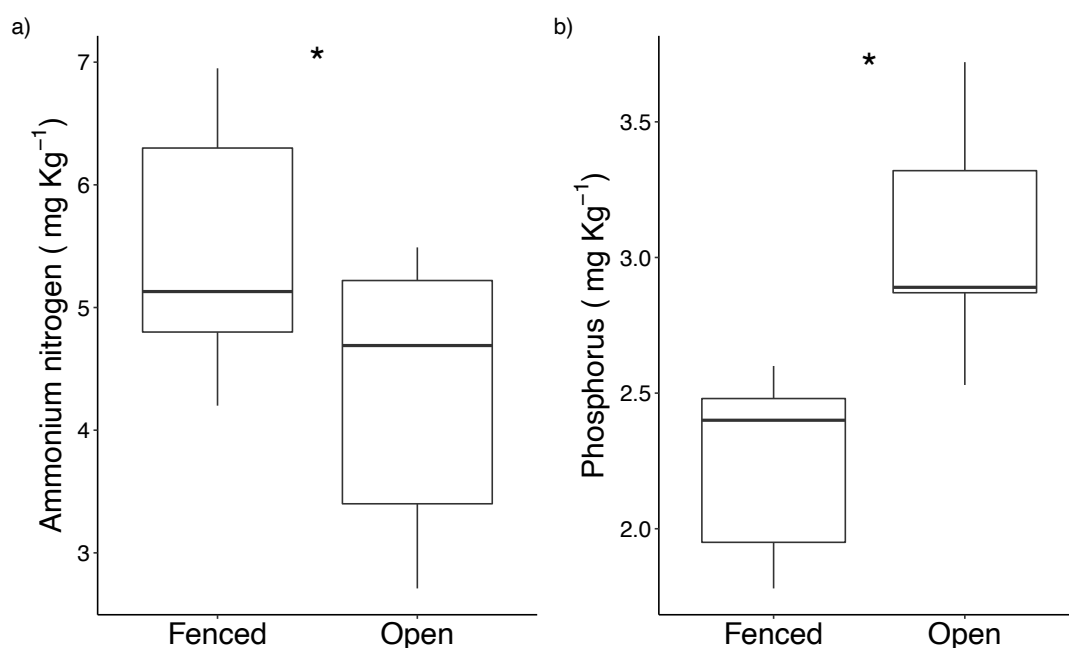


Figure 9. Differences in the concentration of a) ammonium nitrogen and b) phosphorus in soil samples collected in fenced (quenda-excluded) and open (quenda accessible) paired-plots in Craigie Bushland. Soil collection was conducted in October 2018, 4.5 years after the fences were erected and 5.5 years after the quenda (*I. fusciventer*) were translocated into the reserve. Asterisks above boxplots indicate significant ($P < 0.05$) differences between treatments.



Quenda foraging and surface fuel loads

A strong relationship was observed between litter depth and litter mass in bushland outside the paired plots ($r^2 = 0.82$, $P < 0.001$) resulting in correlative equations to predict fuel loads inside the plots (Figure 10) for 1-hour fuel (Equation 3) and total fuel (Equation 4). Heteroscedasticity and non-normal errors in the data for 10-hour fuel resulted in an unreliable model for predicting 10-hour fuel mass and was therefore excluded from further analysis. The correlative equations for fuel mass (g per 25 x 25 cm quadrat) were:

$$\text{Equation 3: 1 hour fuel (g)} = 2.52 (\text{depth (mm)}) - 19.2$$

$$\text{Equation 4: Total surface fuel (g)} = 2.70 (\text{depth (mm)}) - 19.4$$

To determine surface fuel loads within the paired-plots, we used the average litter depths we recorded during point-intercept sampling of the plots in the equations above (Appendix 3). There was significantly more estimated surface fuel in the fenced plots compared to the open plots for both 1-hour fuel and total fuel load estimates (Table 9, Figure 11); with open plots, where quenda foraged, containing nearly half of the fuel load of fenced plots.

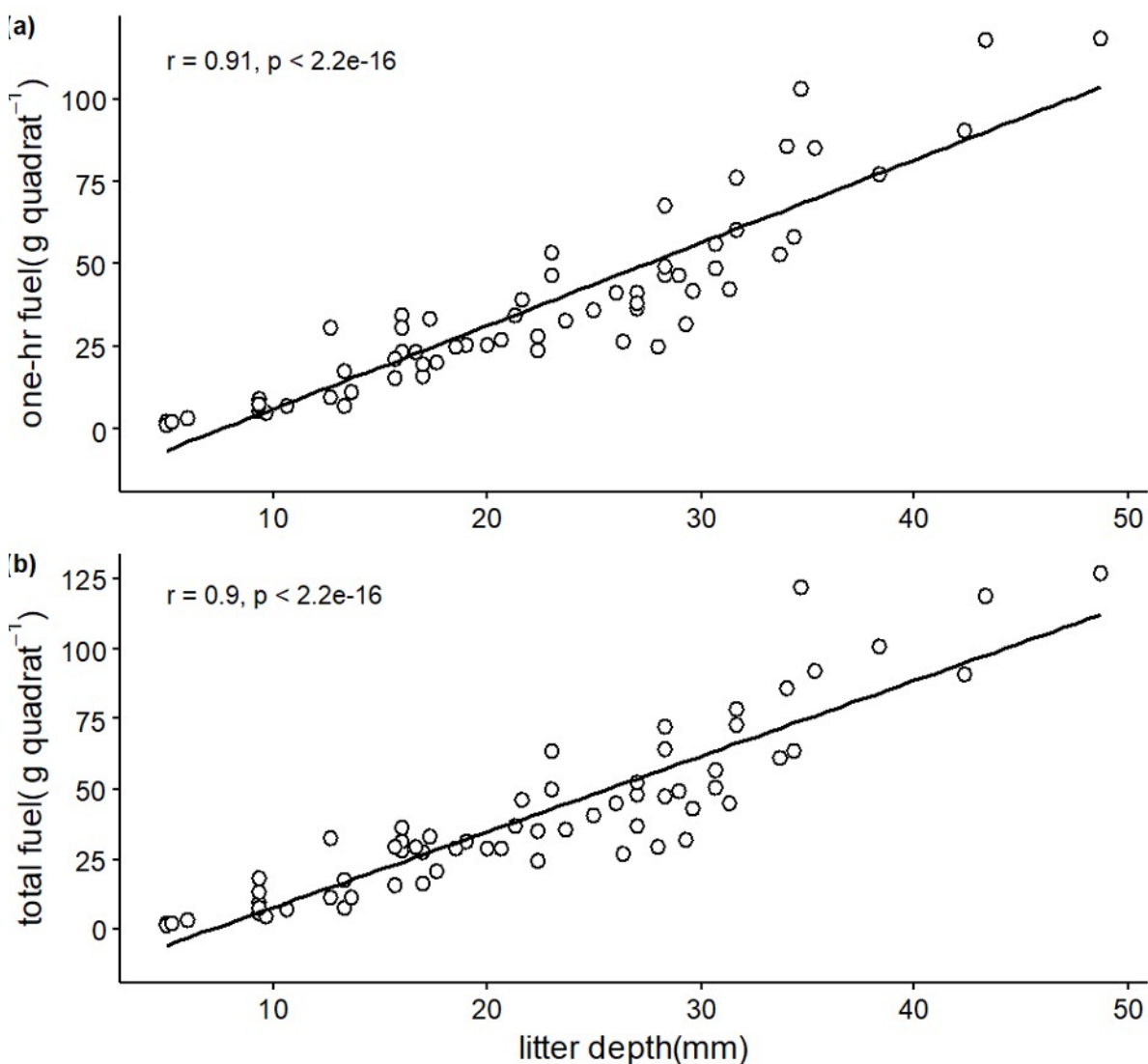


Figure 10. The relationship between litter mass (g quadrat⁻¹) and litter depth (mm) for a) one-hour fuel and b) total surface fuels derived from destructive sampling ($n = 64$) at Craigie Bushland.

Table 9. Results of linear mixed effects models on estimated fuel loads (tonnes ha⁻¹) for one-hour and total surface fuel load estimates between fenced (quenda excluded) and open (quenda accessible) treatments at Craigie Bushland. At the time of sampling for these data (April-June 2017), the plots had been in place for three years.

	Treatment	Coefficient	95 % CI	t-value	Chisq
1 -hour fuel (tonnes ha ⁻¹)	Fenced	5.8	5.2 - 6.4	20.4	60.2***
	Open	3.3	2.0 - 4.5	-7.8	
Total surface fuel (tonnes ha ⁻¹)	Fenced	6.4	5.7 - 7.1	19.2	59.5***
	Open	3.6	2.3 - 5.0	-7.7	

*** $P < 0.001$

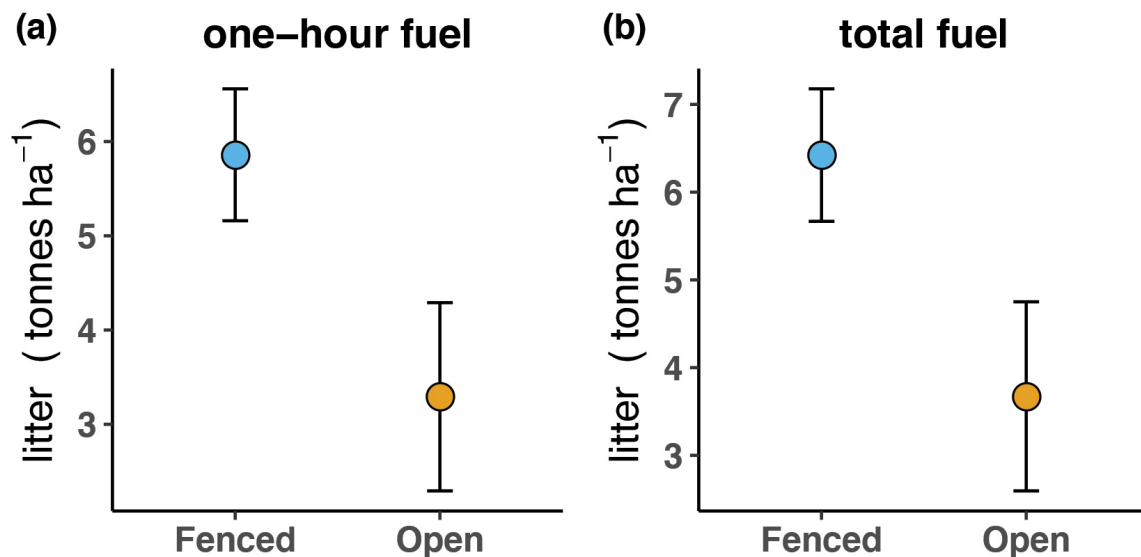


Figure 11. The amount of estimated litter (tonnes ha⁻¹ $\pm$ 95 % CI) in paired fenced (quenda excluded) and open (quenda accessible) treatments for a) one-hour fuel and b) total surface fuels at Craigie Bushland. At the time of sampling for these data (April-June 2017), the plots had been in place for three years.

Discussion

Changes in quenda population with time

The quenda population at Craigie Bushland reserve has increased markedly since the first animals were translocated in 2013, and the translocation can be considered successful within the timeframe of this study. We recaptured nearly 50% of all of the original translocated individuals. In addition, we regularly found evidence of recruitment (females carrying pouch young, new recruited individuals) and the population increased within the reserve. The most recent estimates indicate there are likely to be a total of 69 animals within the reserve, with a bias towards female individuals. Although quenda population size increased linearly with year since translocation, we also observed clear evidence of recent declines in female and male quenda populations within the reserve. For example, female abundance appeared to increase in a near linear fashion from the initial translocation up until December 2018, after which a steady 23% decline was observed over the following year. Similarly, the population size of male quenda increased steadily until February 2017, after which it declined by 52.9% in the space of 27 months. Population size estimates from the most recent trapping session indicate that the male quenda population may now be stabilising. The longest trapped individual quenda was at least four years of age, which is similar to observations made by (Ramalho *et al.*, 2018). A closely related species, the southern brown bandicoot (*I. obesulus*), also typically survive for less than four years in the wild (Menkhurst and Knight, 2004).

Initial increases in population estimates following the 2013 translocation indicate that habitat quality within the reserve is sufficient to support a population of quenda at least equal in size to that of the founder population (46 individuals), but likely more. Initial population increases also demonstrate that the predator proof fence surrounding the reserve is sufficient in lowering/eliminating predation to the extent that non-negative population growth is possible. Given this information, we suggest that the subsequent male and female declines are most likely a result of population size exceeding the maximum carrying capacity of the reserve — the maximum population size that can be reached in a given area when accounting for the limited availability of resources like food, shelter, water and mates. Similar trends have been observed following the translocation of other marsupial species in Australia. For example, Griffiths *et al.* (2017) found a northern quoll (*Dasyurus hallucatus*) population translocated to predator free Astell Island (NT) rapidly increased in size in the five years following their introduction, but then declined and stabilised at reduced size several years later. The authors suggest declines were likely resultant of quoll related food depletion, as evidenced by the fact that quoll body condition was inversely correlated with quoll abundance (Griffiths *et al.*, 2017). At Craigie Bushland, it is likely that intraspecific competition may also be an important factor influencing the population size of quenda.

Declines in the male population of quenda within Craigie Bushland were evident almost two years before declines in the female population, suggesting the carrying capacity of Craigie Bushland is sex specific, with higher numbers of female quenda being supported. Supporting this hypothesis are the home range sizes of adult male quenda (~9.2 ha), which are nearly five times larger than adult female quenda (~ 2.1 ha) in Craigie Bushland. With larger home ranges, male quenda are more likely to overlap with other individuals than female quenda, potentially resulting in male quenda experiencing elevated levels of intraspecific competition. To further investigate the influence of overlapping spatial use on intraspecific competition within the quenda population in Craigie Bushland, we suggest future studies track individual quenda movements using GPS or VHF tracking collars.

We found the average mass of quenda declined significantly with years since translocation, particularly for male quenda. Declines in body mass may have resulted from higher levels of intraspecific resource competition within the reserve, which likely increased over time with increasing population size. Similar trends in body mass changes have been observed in other translocated populations of bandicoot; Dunlop and Morris (2018) found golden bandicoots (*Isodon auratus barrowensis*) increased in mass when moved from a high density site to a low density site, presumably due to a reduction in intraspecific competition. In addition to years since translocation, we found female, male, and combined average body mass all increased to some extent with increasing previous quarter precipitation, although this relationship was not significant in any model. However, previous quarter precipitation did have a significant effect on the proportion of female quenda that were carrying pouch young. Female quenda were more likely to be carrying pouch young when rainfall in the previous quarter was higher, probably because resource availability subsequently became higher following rainfall. Similar results have been found in other bandicoot studies (Gordon, 1971; Barnes and Gemmell, 1984; Friend, 1990).

Detecting population change with trapping effort

Unsurprisingly, increasing trapping effort enhanced the ability to detect population declines. At least three trapping sessions are required per year to detect 30% declines in quenda abundance (between years) with 80% power. However, only one trapping session is required to detect 50% declines in quenda population with 80% power. The monitoring design of trapping twice a year (as happened in 2018 and 2019) would still be able to detect 30% declines in abundance with a reasonable level of statistical power (power = 72%). Essentially this means that if trapping were to continue twice a year at Craigie Bushland, there is a 28% chance a decline in quenda abundance of 30% would not be detected, but a very high chance a decline of 50% would be detected. Based on abundance estimates calculated as part of this report, it's clear that the quenda population at Craigie Bushland is subject to fluctuations in abundance that occasionally exceed 30%. Changes of this magnitude may not necessarily warrant management concerns. However, larger changes, such as declines of 50% between years, are likely to be of concern, and could trigger further investigations. Such declines should still be detected (with 80% power) with only one trapping session per year.

Changes in microhabitat from quenda foraging

Quenda visibly altered the microhabitat within Craigie Bushland over time. There was a substantial amount of digging and foraging by the population of quenda in Craigie Bushland, which is the main driver for the differences in microhabitat characteristics observed between open and fenced plots in later years. Initially, when the fences were first erected there were no differences in any of the microhabitat variables. However, within 18 months of the fences being erected, there was a greater number of foraging pits in the open plots compared to the fenced plots. The presence of old quenda digs in the fenced plots indicates that some quenda digs may persist for years after creation, at least three years in our study. Most quenda digs are probably more ephemeral and degraded quickly; and potentially the old digs we observed in the fenced plots were originally larger digs, and hence less likely to be entirely covered by litter fall in the intervening years.

Quenda foraging activities substantially increased the area of bare ground (area of quenda digs) and bare ground cover while also reducing litter cover and litter depth. These differences were particularly pronounced in 2017 when quenda population was predicted to be at its greatest within the reserve. However, the microhabitat variables were still significantly different when quenda population estimates were lower. A key finding of this study was that litter depth and litter cover were all higher in the fenced plots within 18 months of the fences being erected, while bare ground cover was higher in the open plots as a consequence of quenda digging. When quenda dig for food, they simultaneously discard soil, disrupting the microhabitat layer by exposing soil at the digging site, and burying organic matter and litter under the spoil heap (Valentine *et al.*, 2017). By mixing litter and soil together, the foraging by quenda probably contributes to litter decomposition; as the burial of litter is an important component in litter decomposition (Beare *et al.*, 1992; Austin *et al.*, 2009). Consequently, in places where animals have foraged extensively, or create mounds, there is often an increase in bare ground (Fleming *et al.*, 2014). The micro-scale disturbances created by digging mammals may be incredibly important for facilitating microscale heterogeneity (Mallen-Cooper *et al.*, 2019). One concern, that we did not investigate, is that the increased disturbance created by the foraging activities of quenda (and other digging mammal species) may enhance recruitment of non-native plants. Further research into the interactions between native digging mammals and non-native plants is warranted.

We also detected that quenda foraging influenced the concentrations of two soil properties, with higher amounts of phosphorus, but lower amounts of ammonium nitrogen detected in open plots where quenda could forage. The engineering activities of animals that enhance litter decomposition can subsequently facilitate a change in soil nutrients (see Platt *et al.*, 2016 for review). However, these are often inconsistent, varying among organisms, bioturbation type and intensity of digging (Yu *et al.*, 2017). For example, higher potassium concentrations were observed in the spoil heaps of quenda foraging pits compared to undug soil (Valentine *et al.*, 2018). Whereas foraging pits created by woylies (*Bettongia penicillata*) did not alter potassium concentrations (Garkaklis *et al.*, 2003). Previous work indicates that many burrowing animal species are associated with an increase in phosphorus at the burrow location, possibly due to the upward movement and mixing of deeper soil, with variable responses of ammonium nitrogen (Platt *et al.*, 2016). Fewer studies have examined how soils are altered by foraging pits. In contrast to our findings, short-beaked echidna (*Tachyglossus aculeatus*) foraging pits have been associated with a decrease in phosphorus (Eldridge and Mensinga, 2007). However, similar to our findings, simulated woylie foraging pits were associated with decreases in ammonium nitrogen (Garkaklis *et al.*, 2003). Our study clearly shows that foraging by translocated quenda can alter microhabitat characteristics and soil nutrients

Quenda foraging and surface fuel loads

Quenda foraging reduced litter cover and litter depth that subsequently had a substantial impact on the estimated surface fuel loads. Foraging by the superb lyrebird (*Menura novaehollandiae*), using its large legs and feet to rake through surface litter layers for invertebrates in eucalypt woodlands of southern Australia, resulted in similar changes in microhabitat, with litter loads reduced by 25% compared to areas where the birds couldn't forage (Nugent *et al.*, 2014). Hayward *et al.* (2016) also found reduced litter mass and litter cover inside large reserves where multiple digging mammal species had been reintroduced. By mixing the litter and sediments, quenda are likely burying litter, potentially aiding in its decomposition and subsequently reducing the fine fuel material. Our results indicate that within a short time frame (< 4 years since translocation, 3 years since fence construction) the successful translocation of a single digging mammal species can alter surface fuel load characteristics in an urban area.

From a fuel management perspective, it will be important to understand how many digging animals are required to create fuel load reductions that significantly alter fire behaviour; presumably, this will vary depending upon the animal and habitat in question. Knowledge on pre-European densities of Australian mammals is lacking, but historical accounts indicate many species, including quenda, were abundant (Abbott, 2008). Previous research has documented similar quenda densities (~2 quenda ha⁻¹) in both fenced and unfenced areas (Ramalho *et al.*, 2018).

At Craigie Bushland, quenda density was slightly higher in the months prior to the fuel loads assessment ($N = \sim 95$ individuals in February 2017; ~ 2.2 quenda ha^{-1}). This indicates that although quenda are absent from many bushlands, they may be able to support populations large enough to significantly reduce fuel loads in urban areas once a population has re-established. However, reintroducing mammals into small urban reserves will come with many management challenges, including proximity to urban infrastructure and isolation from other habitat patches. Furthermore, most urban bushlands contain alien plant species and it is unclear how the digging actions of native animals will interact with such plants. As alien plants can influence fuel loads and fire regimes (D'Antonio and Vitousek, 1992), further research examining the impact of digging animals on alien plants, especially invasive species, is warranted. Finally, the quenda population at Craigie Bushland has reduced in size since the surface fuel loads estimates were conducted and it is unclear whether the reduced density of quenda at Craigie Bushland ($N = \sim 69$ individuals in November 2019; ~ 1.6 quenda ha^{-1}) would alter fuel loads to the same extent.

Conclusion

The quenda population at Craigie Bushland reserve has increased markedly since the first animals were introduced in 2013. The most recent estimates indicate there are likely to be a total of 69 animals within the reserve. Based on multi-session decline events, it appears the carrying capacity of Craigie Bushland is sex-specific, likely because male quenda use larger home-ranges than females. We found the body mass of male quenda declined over time (as quenda density increased) potentially as a result of intraspecific competition limiting access to important resources. Similar to other studies, rainfall was an important indicator of female breeding, with the amount of precipitation recorded in the three months prior to trapping a strong indicator of the proportion of female carrying pouch young. The capacity of the monitoring program at Craigie Bushland to detect changes in quenda abundance is strongly influenced by the number of trapping sessions conducted per year. Trapping once a year will detect a population decline of 50% with at least 80% power.

There is clearly capacity for translocated quenda to alter surface microhabitat, including estimated fuel loads, and some soil nutrients. Evidence of quenda foraging was prolific throughout Craigie Bushland. The digging activities of quenda have increased two types of bare ground measurements: the measured area of bare ground and the visually estimated percent cover of bare ground. This has subsequently reduced litter cover and litter depth; and the composition of litter is obviously patchy where quenda are actively foraging. When quenda occurred at high density, their foraging also reduced estimated surface fuel loads. Although translocation programs face many challenges, they are an increasingly utilised management practice, not just for the conservation of species, but also for the ecosystem services a species provides (Seddon *et al.*, 2014). Translocation of digging mammals into urban reserves are fairly rare; but represent an excellent opportunity to examine some of the changes that digging mammals can create on microhabitat. In Australia, the threat of predation from introduced cats and foxes (Woinarski *et al.*, 2015) restricts the broad-scale reintroduction of mammals outside of fenced reserves or islands; however, the reintroduction of mammals may still be feasible in some urban areas where it is deemed suitable (e.g. Craigie Bushland, this project; and Mulligan's Flat, Batson *et al.*, 2016). However, such translocations encounter many challenges. For example, the population of quenda at Craigie Bushland are quite likely isolated and may be susceptible to a number of risks, including genetic constraints (Ottewell *et al.*, 2014) and stochastic events (Ramalho *et al.*, 2018). Further research on the long-term likely persistence of quenda at Craigie Bushland; as well as their impacts on broader habitat attributes, is warranted.

Datasets

Data for the published paper Ryan *et al.* 2020 can be found at the UWA repository (DOI: 10.26182/5d77392100f4d) via the following link: <https://research-repository.uwa.edu.au/en/datasets/bioturbation-by-a-reintroduced-digging-mammal-reduces-fuel-loads->.

Additional data can be accessed upon request to the authors.

Ethics statement

Animal ethics for this project was coordinated under the Department of Biodiversity, Conservation and Attractions (Animal Ethics Approval numbers: DEC – AEC: 2013/31; DPaW AEC: 2015/29; DPaW AEC: 2016/46 and DBCA AEC: 2019-45C).

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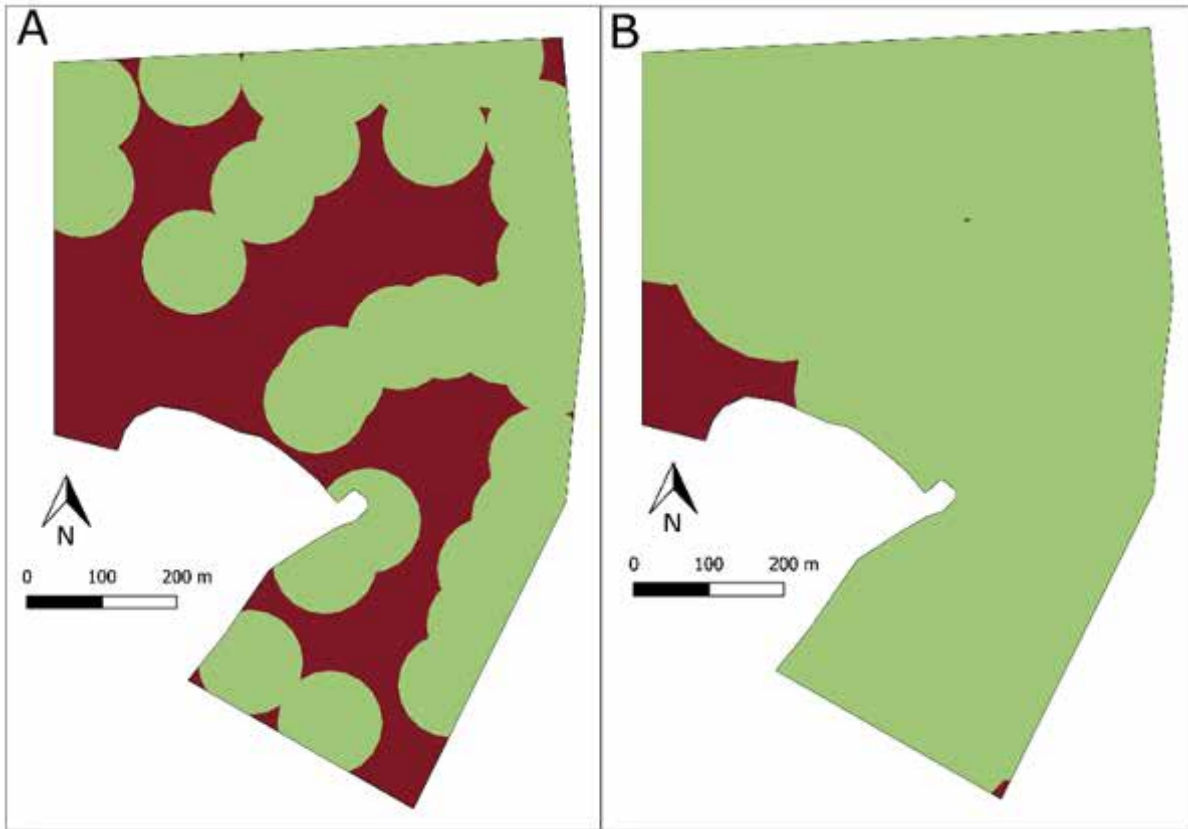
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Appendices



Appendix 1. Effective trapping area of a) female and b) male quenda from the monitoring program at Craigie Bushland. This calculation is based on the assumption that quenda with a home range centre further than the radius of an average circular home range estimate away from traps, are not able to be trapped. Home range estimates were derived from the best fitting 'secr' model. Green area represents trappable regions, red represents un-trappable area.



Quenda being released by Harry Moore. Image: B.Palmer

Appendix 2. The back-transformed estimated marginal means ($\pm$ standard error) and statistics (F -value and degrees of freedom, where appropriate) from generalised linear mixed models and linear mixed models on microhabitat response variables collected in fenced (quenda-excluded) and open (quenda accessible) paired-plots during different survey sessions (April 2014, October 2015, April 2017, October 2018) in Craigie Bushland. Fences were initially erected in April 2014, ~ 11 months following the first translocation of quenda into Craigie Bushland. Statistically significant differences from linear mixed models at $p < 0.05$ are shown in bold. Statistics are not shown for the number of quenda digs (which was analysed using generalised linear mixed model with results reported elsewhere) or for percentage cover of coarse woody debris and bare ground (due to infrequent observations these variables were excluded from analyses). Number of quenda digs, litter depth and percentage cover estimates were rounded to the nearest whole unit.

Response variable	April 2014		October 2015		April 2017		October 2018		Treatment	Survey	Interaction
	Fenced	Open	Fenced	Open	Fenced	Open	Fenced	Open	$F_{df=1,35}$	$F_{df=3,35}$	$F_{df=3,35}$
# of quenda digs	49 (3)	51 (3)	14 (2)	74 (4)	9 (1)	161 (7)	24 (2)	92 (5)	-	-	-
Area of quenda digs (m ²)	0.73 (0.12)	0.80 (0.13)	0.14 (0.08)	0.88 (0.13)	0.10 (0.07)	2.41 (0.24)	0.22 (0.08)	1.34 (0.16)	171.7***	7.1***	25.8***
Litter depth (mm)	21 (2)	19 (2)	17 (2)	13 (1)	25 (2)	11 (1)	25 (2)	9 (1)	112.9***	6.6**	18.3***
Litter cover (%)	85 (3)	82 (3)	86 (3)	68 (5)	82 (3)	49 (5)	67 (5)	45 (5)	65.2***	28.0***	5.6**
Quenda bare ground (%)	10 (2)	11 (2)	4 (1)	18 (3)	4 (1)	37 (5)	4 (1)	21 (4)	161.1***	2.25	17.5***
Vegetation (%)	6 (1)	6 (1)	9 (2)	10 (2)	11 (2)	11 (2)	24 (4)	27 (4)	0.3	40.0***	0.2
Non-quenda Bare ground (%)	3 (1)	4 (1)	3 (1)	4 (1)	4 (1)	3 (1)	5 (1)	5 (1)	-	-	-
CWD (%)	3 (1)	3 (1)	4 (1)	4 (1)	5 (1)	4 (1)	6 (1)	5 (1)	-	-	-

Asterisks indicate significant (** $P < 0.01$, *** $P < 0.001$) result.

Appendix 3. Results of linear mixed-effect models for point intercept sampling response variables between paired fenced (quenda excluded) and open (quenda accessible) treatments at Craigie Bushland, Western Australia. Significantly difference P-values are highlighted in bold.

Variable	Treatment	Coefficient	95 % CI	t-value	chisq	Pr(>Chisq)
Bare ground (%)	Fenced	0.03	0.000 - 0.065	1.32	80.26	< 0.0001
	Open	0.25	0.216 - 0.419	8.99		
Litter cover (%)	Fenced	0.79	0.728 - 0.857	24.13	15.98	< 0.001
	Open	0.61	0.453 - 0.763	- 4.0		
Vegetation (%)	Fenced	0.27	0.233 - 0.302	6.10	9.44	< 0.05
	Open	0.19	0.108 - 0.275	15.15		
CWD (%)	Fenced	0.05	0.013 - 0.088	4.06	2.55	> 0.05
	Open	0.01	-0.080 - 0.079	- 1.31		
Litter depth (mm)	Fenced	25.04	23.46 - 26.62	31.06	42.65	< 0.0001
	Open	17.64	13.83 - 21.46	- 6.53		



Quenda dig. Image: Leonie Valentine

Further information:

<http://www.nespthreatenedspecies.edu.au>

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