Can we use camera traps to estimate population size or density of the quenda (*Isoodon obesulus fusciventer*)?

This thesis is submitted in partial fulfillment of the requirements for a Master of Biological Science, Zoology

SCIE 5511 – 13 Research Dissertation

Faculty of Science

The University of Western Australia

(May 2017)

Word count: 8037

Formatted in accordance with the journal *Australian Mammalogy*

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Abstract

Obtaining measurements of abundance is a key component of wildlife conservation yet this information is lacking for numerous species. Australian mammal populations, in particular, are not well understood because they are generally rare or exhibit cryptic behaviour, making them difficult to survey. There is consequently an increasingly urgent need for cost-efficient and accurate monitoring for these species. Recently, camera traps have received considerable attention as a survey tool with camera trapping research often focused on attempting to estimate population density, which is critical for wildlife conservation. Capture-mark-recapture methods, conventionally used to determine population size, typically require individual identification. However, using camera traps to estimate the population size of nondescript species would greatly increase their use. Few researchers have attempted to use camera traps as surrogates for traditional fauna surveys. Therefore, the extent to which we can rely on camera traps and on the resulting indices of abundance to monitor populations of nondescript species is still largely unknown. The majority of small to medium-sized mammals in Australia, such as the quenda (*Isoodon obesulus fusciventer*), do not have unique markings yet camera traps are often used to provide population information for these species. The aim of this study was to determine if camera trap hit rates of quenda could be calibrated with live trapping population and density estimates obtained through spatially explicit capture-recapture, to determine if camera traps can be used to estimate population size or density for this species. Quenda were trapped at seven sites with differing densities using both live and camera traps, and the relationship between the hit rates derived from camera trapping and robust population and density estimates derived from live trapping were investigated for this unrecognisable marsupial. Densities ranged from zero to 1.81 animals/ha, population estimates ranged from zero to 71 and hit rates ranged from zero to 3842 hits/1000 days. The relationship between population estimates and hit rates was significant with a Spearman rank correlation coefficient R of 0.89. The relationship between density and hit rate was also significant with an R of 0.77. The results indicate that camera trap surveys can provide valid abundance or density measures that would be sufficient to monitor quenda populations and that camera traps may be a viable alternative to live trapping for this species.

Contents

Acknowledgements

Firstly, I would like to gratefully acknowledge Main Roads WA for the financial support for this project and I would also like to thank my supervisors, in particular Roberta Bencini, for their support and for generously sharing their knowledge and expertise. Thank you to Mike and also Mandy for providing the camera equipment.

Thank you to Dr Charles Krebs who provided valuable guidance and assistance, as did Alice Kenney, who also let me use her camera trap program "Wild Photo Trap" to calculate the hit rates.

I would also like to thank Dr Leonie Valentine and Rebecca Campbell, who generously shared their study site and knowledge with me and taught me how to handle bandicoots. Also, thank you to the staff at Whiteman Park, in particular Dr Chris Rafferty, for allowing me to store the traps within the park and thank you to Daisy Pearse, for contacting us and allowing the bandicoot populations to be measured at Fiona Stanley Hospital.

Finally, many thanks to my family, friends and volunteers who assisted me in the field, I hope you enjoyed it.

Introduction

Knowledge and an understanding of animal populations and how they change are essential for species conservation (Engeman 2005; Bengsen *et al.* 2011). Reliable and accurate information regarding population size or abundance, density, distribution and stability, and the factors influencing these, is critical for ecological research and conservation, and this is often obtained through monitoring (Stanley and Royle 2005; Wiewel *et al.* 2007; Bengsen *et al.* 2011; Paull *et al.* 2011; Sollmann *et al.* 2012; Jareño *et al.* 2014; Taylor *et al.* 2014). Without this information, the most effective conservation and recovery actions for a species cannot be identified (Paull *et al.* 2011; Bain *et al.* 2014). Monitoring is of utmost importance, so it is disturbing that 50.8% of Australian recovery plans lack any monitoring and evaluation schemes (Ortega-Argueta *et al.* 2011). When monitoring does occur, there are often unclear objectives, inadequate sampling or inappropriate time scales (Dajun *et al*. 2006). It is unknown, therefore, whether targets have been met and if the management actions have been appropriate for the species (Ortega-Argueta *et al.* 2011). Consequently, there is an increasingly urgent need for effective and extensive monitoring, particularly for Australian mammals (King *et al.* 2007; Paull *et al.* 2012; Swan *et al.* 2014; Woinarski *et al*. 2014; Diete *et al.* 2015).

Ecological studies on animal populations require high quality data that can be collected efficiently (Barros *et al*. 2015). The methods selected to survey populations must produce the greatest detection success while minimising costs (Garden *et al.* 2007; Bain *et al.* 2014; Jareño *et al*. 2014). This is often challenging, particularly for cryptic Australian mammals as they are often secretive, occur at low densities and may occupy inaccessible habitat (Claridge *et al.* 2004; Bain *et al.* 2014). A wide variety of direct and indirect techniques are available to survey animal populations, with direct sampling often relying on the physical capture of animals (De Bondi *et al*. 2010). This is usually achieved using pitfall, Elliott or wire cage traps, depending on the target species (De Bondi *et al*. 2010). Direct sampling is ideal for research into individual characteristics and is a common sampling method used in Australia for small and medium sized mammals (Paull *et al*. 2012). Indirect methods present a practical alternative to capture-mark-recapture and can sample fauna using tracks, scats, diggings or hair. These methods are generally cheaper, faster, and easier to use but they can be subject to bias and inaccuracy (Lyra-Jorge *et al*. 2008; Bain *et al*. 2014; Jareño *et al*. 2014).

The choice of survey method is, therefore, a critical component that influences the accuracy of data collected and the subsequent interpretation of results (Garden *et al.* 2007; Lyra-Jorge *et al*. 2008; De Bondi *et al*. 2010). Determining the most appropriate method depends on the goals of the study, the target species, the survey sites and the budget and timeframe (Garden *et al.* 2007; Lyra-Jorge *et al*. 2008; Manzo *et al*. 2012; Barros *et al*. 2015).

Capture-mark-recapture based on live trapping is the traditional method used to understand population dynamics and can be used to accurately estimate density, population size, species richness and composition, conservation status and responses to recovery actions (Mills *et al*. 2002; Lemckert *et al.* 2006; De Bondi *et al*. 2010; Jareño *et al*. 2014). This method has been used extensively in recent years as detailed information can be obtained from individual animals, such as age, sex, condition and reproductive status (Wiewel *et al*. 2007; De Bondi *et al*. 2010; Jareño *et al*. 2014). The density estimates obtained via capture-mark-recapture are considered to be the most robust and accurate for small mammal species (Wiewel *et al*. 2007; De Bondi *et al*. 2010; Jareño *et al*. 2014), however, trap success depends on the species and individuals, and their morphological, ecological and behavioural characteristics, as well as weather conditions and trap mechanisms (Barros *et al*. 2015).

Live trapping, while effective, has drawbacks. Of particular concern are the welfare implications for trapped animals. Some adult female marsupials, for example, tend to eject pouch young when trapped, and may inadvertently crush them while in the trap (Paull *et al*. 2011). There is also a risk of accidental death from stress or predation, and trapped animals can be exposed to adverse environmental conditions that they could otherwise avoid (Lemckert *et al.* 2006; De Bondi *et al*. 2010; Paull *et al*. 2012). Trapping may also interfere with their normal activities and behaviour, and restrains their movement in time and space (Claridge *et al*. 2004; De Bondi *et al*. 2010). In addition, animals may learn to either avoid or seek out traps, which may affect monitoring efficiency (King *et al*. 2007). Common non-target species may also saturate traps, resulting in fewer captures of the target species (King *et al*. 2007). Capture-mark-recapture provides little data for species that are difficult to trap and rare, cryptic or elusive species so it is, therefore, impractical for such species (Stanley and Royle 2005; De Bondi *et al*. 2010; Paull *et al*. 2012). It is also a time-consuming method as researchers need to be in the field for long periods and it is labour intensive, often resulting in poor return for effort (Mills *et al*. 2002; Claridge *et al.* 2004; De Bondi *et al*. 2010; Paull *et al*. 2012). Capture-mark-recapture is a difficult and expensive method to employ that can be impractical over the large temporal and spatial scales usually required to make informed management decisions (Villette *et al*. 2016). Finally, the assumptions of capture-mark-recapture can be restrictive and, if they are violated, the population estimates produced may be of questionable quality (Engeman 2005; Krebs 2014). It is for these reasons that indirect sampling methods are becoming more common for surveying wildlife species.

Camera traps have rapidly become essential tools for wildlife research and monitoring in Australia and overseas (Manzo *et al*. 2012; Meek and Pittet 2012; Güthlin *et al*. 2014; Taylor *et al*. 2014; Diete *et al*. 2015; Meek *et al*. 2015a, 2015b). They are a promising indirect sampling method and are preferred over hair tubes, for example, as hair tubes are often grossly inefficient at detecting target mammals, they are prone to error and the high costs associated with genetic analysis often preclude their use (Catling *et al.* 1997; Lobert *et al*. 2001; Mills *et al*. 2002; Garden *et al*. 2007; Claridge *et al*. 2010a, 2015). Camera traps are remotely deployed fixed cameras that are triggered to capture images or video of passing animals at a given location and time, without the need for humans to be present (Rowcliffe *et al*. 2008; De Bondi *et al*. 2010; McCallum 2013; Meek *et al.* 2015a, 2015b). They may be passive infrared or time-lapse cameras and they allow the capture and storage of thousands of images over long periods (Smith and Coulson 2012; McCallum 2013; Meek *et al.* 2015a). They have traditionally been used to confirm the presence or absence of species; however, they are now often used to investigate population dynamics, habitat use and behaviour, to survey wildlife distributions, and are even used to produce population or density estimates (Rowcliffe *et al*. 2008; Manzo *et al*. 2012; Burton *et al*. 2015; Diete *et al*. 2015; Meek *et al.* 2015a*,* 2015b). Most camera trap research overseas has concentrated on large and medium sized mammals such as big cats. However, in Australia, small mammals have largely been the focus of camera trap research (De Bondi *et al*. 2010; Meek and Vernes 2016).

Camera traps are less time consuming and laborious than most direct methods and are more practical and cost effective (Wiewel *et al*. 2007; Rowcliffe *et al*. 2008; De Bondi *et al*. 2010; Smith and Coulson 2012; McCallum 2013; Dundas *et al*. 2014; Taylor *et al*. 2014; Welbourne *et al*. 2015; Meek and Vernes 2016; Villette *et al*. 2016). They operate in most habitats and are able to withstand extreme weather conditions (Vine *et al*. 2009; De Bondi *et al*. 2010; Manzo *et al*. 2012; Paull *et al*. 2012; McCallum 2013; Villette *et al*. 2017). These versatile tools can be left in the field for relatively long periods without checking and have greater detection efficiency relative to other methods. Therefore, they are particularly useful for cryptic or rare species and habitat specialists (Cutler and Swann 1999; Rowcliffe *et al*. 2008; Borchard and Wright 2010; Claridge *et al*. 2010a; Paull *et al*. 2012; Smith and Coulson 2012; McCallum 2013; Dundas *et al*. 2014; Taylor *et al*. 2014). Cameras can be deployed over greater spatial and temporal scales and are able to record multiple species, resulting in large data sets (De Bondi *et al*. 2010; Foster and Harmsen 2012; Paull *et al*. 2012; McCallum 2013). The major advantage of camera traps is that they remove the need to handle animals, providing ethical benefits as there is no direct contact between humans and animals (Claridge *et al*. 2004; Wiewel *et al*. 2007; Rowcliffe *et al*. 2008; Claridge *et al*. 2010a; Paull *et al*. 2012; McCallum 2013; Dundas *et al*. 2014; Meek and Vernes 2016).

Abundance or density estimates for mammalian species can be achieved using camera traps (Bengsen *et al*. 2011; Güthlin *et al*. 2014) but traditionally these methods require individual identification, which is often accomplished using unique coat patterns or other natural markings or features (Bengsen *et al*. 2011; Dundas *et al*. 2014; Güthlin *et al*. 2014; Weerakoon *et al*. 2014; Diete *et al*. 2015; Villette *et al*. 2016). Population estimates for individually recognisable species are achieved using capture-mark-recapture techniques in the same way as traditional capture-mark-recapture approaches, as they are based on the re-capture of individuals by cameras (Rowcliffe *et al*. 2008; Manzo *et al*. 2012; McCallum 2013; Sollmann *et al*. 2013; Güthlin *et al*. 2014; Weerakoon *et al*. 2014; Diete *et al*. 2015; Villette *et al*. 2016). This method is problematic for species that have a uniform morphology and lack unique markings because individuals are indistinguishable in photographs (Weerakoon *et al*. 2014; Meek *et al*. 2015a; Villette *et al*. 2016). This is typical of the majority of species, including most Australian mammals, making these capture-mark-recapture methods limited in their use (Carbone *et al*. 2001; Rowcliffe *et al*. 2008; Burton *et al.* 2015; Diete *et al*. 2015; Villette *et al*. 2016). Consequently, there is great interest in alternative methods for nondescript species, as techniques that could estimate abundance using photographic rates of these species would significantly increase the value of camera traps (Carbone *et al*. 2001; Rowcliffe *et al*. 2008; Burton *et al.* 2015). One such method is the use of photographic hit rates to calculate indices of abundance for species with uniform morphology (Rowcliffe *et al*. 2008; Rovero and Marshall 2009; Bengsen *et al*. 2011; Foster and Harmsen 2012; Manzo *et al*. 2012; Güthlin *et al.* 2014; Weerakoon *et al*. 2014; Villette *et al*. 2016). In theory, the number of images taken per unit time contains information about a species population size (Rowcliffe *et al*. 2008, 2014). If the population increases or decreases the index should reflect that and where a species is in greater abundance, its photographic hit rate should be higher (Engeman 2005; Rovero and Marshall 2009; Rowcliffe *et al*. 2014). An index is expected to vary directly with population size and is proportional to and reflects abundance, but is not an estimate of the actual population size (Engeman 2005; Stanley and Royle 2005; O'Connell *et al*. 2011). Though controversial, indices of abundance are used as surrogate measures of population size as they allow rapid assessments of abundance where other methods are not feasible, and they can be used to make relative comparisons between populations (McKelvey and Pearson 2001; Engeman 2005; Kelly and Holub 2008; Rovero and Marshall 2009; Sollmann *et al*. 2013; Weerakoon *et al*. 2014). Consequently, they have become integral to resource-constrained wildlife managers. They have become so prominent that computer programs that exist purely to calculate hit rates have also been developed, such as Wild Photo Trap created by Kenney. For nondescript animals, calibrating the relationship between hit rate and density is critical if photographic hit rates are to be used as an index of abundance, however, this rarely occurs (Kelly 2008; Rovero and Marshall 2009; Foster and Harmsen 2012; Sollmann *et al*. 2013; Villette *et al*. 2016).

A group of Australian marsupials that would greatly benefit from monitoring via the use of camera traps are the bandicoots (Marsupialia: Peramelidae) because they have experienced substantial declines in abundance and distribution with the extinction of two species, the pig-footed bandicoot (*Chaeropus ecaudatus*) and desert bandicoot (*Perameles eremiana*). An additional eight species are listed as vulnerable or endangered under the *Environment Protection and Biodiversity Conservation Act 1999* (Claridge and Barry 2000; Bilney *et al.* 2010; Diete *et al.* 2015; Valentine *et al.* 2017). Their conservation is hampered by lack of data and this can be partly attributed to the difficulty in surveying these species. Therefore, camera traps could potentially represent an ideal alternative to traditional survey methods for monitoring bandicoots (Claridge and Barry 2000; Mills *et al.* 2002; Paull *et al*. 2012).

Effective and cost-efficient survey techniques have yet to be developed and refined for the majority of wildlife populations, and there has been comparatively little research into the effectiveness of camera traps for measuring Australian mammal populations (De Bondi *et al*. 2010; Welbourne *et al*. 2015). Therefore, it is unknown the extent to which we can rely on photo-indices to monitor our native species (O'Connell *et al*. 2011; Weerakoon *et al*. 2014). Comparing the efficacy and accuracy of camera trapping relative to live trapping would determine the suitability of this method for unrecognisable mammal populations, which are under-represented in camera trap research (Rowcliffe *et al*. 2008; Meek *et al*. 2015a).

I sought, for the first time, to investigate the relationship between indices of abundance derived from camera traps and robust population and density estimates derived from live traps for the quenda (*Isoodon obesulus fusciventer*), a nondescript marsupial. The objective of this study was to determine if camera trap hit rates of quenda could be calibrated with live trapping population and density estimates obtained through spatially explicit capture-recapture, such that camera traps can be used to estimate population size and density for this species. It was expected that the hit rates and live trapping population and density estimates would be strongly correlated for the quenda. The findings of this study will help inform future researchers in their choice of sampling methods for the quenda, as it will determine if camera traps can be used as substitutes for traditional fauna surveys for this nondescript species.

Materials and methods

To test this hypothesis, capture-mark-recapture methods using conventional trapping were compared with camera trapping methods developed by Alice Kenney at the University of Canberra's Institute of Applied ecology (pers. comm) at seven sites with different quenda densities.

Study species

The southern brown bandicoot or quenda is a medium-sized omnivorous marsupial, ranging in size from 400 to 2000g (Braithwaite 1995; Valentine *et al*. 2013). Quenda occur across south-west Western Australia, preferring scrubby dense vegetation around swamps and watercourses or open jarrah forest (Braithwaite 1995; Cooper 1998; Valentine *et al*. 2013; Chambers and Bencini 2015). Quenda are ecosystem engineers, as they are capable of turning over approximately 4 tonnes of soil per individual per year (Valentine *et al*. 2013). They are nocturnal, solitary and territorial animals with home range estimates varying from 0.5 to 7ha (Lobert 1990; Braithwaite 1995). Males generally have larger home ranges than females, ~2.3ha and ~1.8ha respectively, and they may overlap in areas of high density (Broughton and Dickman 1991; Braithwaite 1995; Valentine *et al*. 2013). Densities range from 1 to 5 per ha (Lobert and Lee 1990). Unlike the endangered eastern subspecies (*I. obesulus obesulus*), the quenda is persisting in bush fragments and reserves in the peri-urban fringe surrounding Perth (Valentine *et al*. 2013; Howard *et al*. 2014). However, they are classified as Priority 4 under the Western Australian *Wildlife Conservation Act 1950.* Quenda were once common across the south-west but they are now absent from many areas or are persisting in low numbers, due to a combination of fragmentation, loss of habitat, and predation by the introduced red fox (*Vulpes vulpes*) and feral cat (*Felis catus*) (Braithwaite 1995; Driessen and Rose 2015; Valentine *et al*. 2017). The quenda is ideal for this study as they do not typically have distinguishing markings (e.g. stripes or spots) that would allow for individual identification, and there are no similar species that they could be mistaken for in photographs. They have a short conical muzzle, small rounded ears and they have coarse grey/brown fur above and white below with a short, lightly furred tail that may by shortened or lost completely due to fighting (Braithwaite 1995; Driessen and Rose 2015).

Study sites

Live and camera trapping were conducted in the Perth region of south-west Western Australia from May 2016 to March 2017. Quenda populations were surveyed with both camera traps and conventional cage traps at seven sites across Perth: Whiteman Park, Blue Poles Road, Maralla Road Nature Reserve, Aileen Plant Park, Moitch Park, Craigie Bushland and Thomsons Lake Nature Reserve (Figure 1). This region experiences a Mediterranean type climate with hot, dry summers and mild, wet winters with an average annual rainfall of 868mm (Bureau of Meteorology, station #009034). Multiple sites were chosen to maximise the range of quenda densities.

Figure 1. Map of the seven study sites across Perth, Western Australia: Craigie Bushland, Maralla Road Nature Reserve, Whiteman Park, Blue Poles Road, Aileen Plant Park, Moitch Park and Thomsons Lake Nature Reserve. The site names are located adjacent to the marker.

Whiteman Park is located ~19km northeast of Perth (31°49´34.32"S; 115°55´14.88"E) and is a ~4000ha park managed by the Department of Planning. The study was conducted along a 4km stretch on the western edge of the park (~14ha) where the dominant vegetation is a mix of *Corymbia calophylla*, *Eucalyptus marginata* and *Melaleuca* damplands. Introduced predators are controlled at these sites via 1080 bait and shooting (C. Rafferty, pers. comm).

Blue Poles Road (~4ha) is located within Whiteman Park, approximately 1.4km east of the Whiteman Park site (31°49´30.72"S; 115°56´0.96"E). These sites were considered different sites because they were too far apart for quenda to be moving between them. They share the same dominant vegetation and predator control measures.

Maralla Road Nature Reserve is located ~35km northeast of Perth (31°44′38.04"S 115°58′51.6"E) and is ~145ha. This site is dominated by *Banksia* spp. woodland including *B. attenuata* and *B. menziesii*, over sparse shrubland including *Calytrix fraseri, Verticordia nitens* and *Hibbertia hypericoides*. Open paddock with remnant *Banksia* species is also present and introduced predators are not controlled (B. Inglis, pers. comm).

Craigie Bushland is located ~21km north of Perth (31°47′39.48"S; 115°46′46.56"E) and is ~41ha. The open woodland vegetation consists of a mosaic of *E. gomphocephala, C. calophylla*, *E. marginata* with *B. attenuata* and *Allocasuarina fraseriana.* In 2013, quenda were translocated from Ellenbrook and Twin Swamps Nature Reserves into Craigie Bushland to protect the Western Swamp Tortoise (*Pseudemydura umbrina*) (Valentine *et al*. 2016). At this site, quenda are protected from introduced predators by a predator-proof fence.

Aileen Plant Park is located within the Fiona Stanley Hospital grounds, approximately 18km south of Perth. Aileen Plant Park (32°4′5.16"S; 115°50′54.96"E) is ~1ha and the vegetation is a mixture of *E. marginata* and *Banksia* spp. It is remnant bushland and predators are not controlled.

Moitch Park (32°4′19.92"S; 115°50′47.04"E) is ~2.5ha and is dominated by *Banksia* spp. woodland. It is located ~440m from Aileen Plant Park, also within the grounds of Fiona Stanley Hospital. Quenda could easily cover this distance, however, these sites are separated by the hospital's infrastructure (internal roads and buildings) so movement between these locations is unlikely, especially within the short timeframe of the survey. Moitch Park is also a remnant bushland site in which predators are not controlled.

Thomsons Lake Nature Reserve is located ~28km south of Perth (32°9′2.4"S; 115°49′42.3"E) and is ~538ha. The vegetation around the lake is dominated by rushes and sedges including *Typha* spp., *Bauma* spp., *Viminaria juncea* and *Acacia saligna* shrubs. This gives way to a belt of trees including *Eucalyptus* spp. and *Melaleuca* spp., and the shrub *Jacksonia furcellata*. This is then replaced by open woodland dominated by *Eucalyptus* spp. and *Banksia* spp. Fauna within the reserve is protected from introduced predators by a predator-proof fence and regular baiting with 1080 (CALM 2005).

Camera trapping

There was one camera trapping session for each site allocated randomly to either before or after the live trapping session. Twenty motion-activated camera traps divided among the following models were used in this study: Reconyx HC550 (n=11) and HC600 (n=9) HyperFire (Reconyx^{TM}, Reconyx Inc., Wisconsin USA) as these models are designed for small mammals (Meek and Pittet 2012). Each camera was equipped with an 8GB memory card and set to high sensitivity. The cameras were set to take three photographs over a 3 second period each time the motion sensor was triggered, with no delay between trigger events, 24 hours a day as recommended by Meek *et al*. (2012). These models use LED white flash or infrared illumination. Camera traps were left in the field for ten days, because studies targeting small mammals generally employ longer sampling periods than those used for live trapping (De Bondi *et al*. 2010; Meek *et al*. 2012). Each camera was attached to a steel pole and positioned ~1.3m above a scent lure which was a PVC pipe (200 x 40mm) filled with universal bait (a mixture of rolled oats, sardines and peanut butter) wired to the base of the steel pole. Small 2 mm holes drilled into the pipe prevented animals from consuming the bait while attracting them with the strong scent of the peanut butter and sardines. The bait was not refreshed while cameras were deployed. The cameras were positioned so that the sensor and camera lens were facing the ground as in Smith and Coulson (2012) as this orientation results in greater detection of bandicoots and potoroos, increases the ease of species identification and also reduces the amount of vegetation that needs to be cleared (Smith and Coulson 2012; Diete *et al*. 2015). In most cases, the cameras were placed in naturally clear areas; however, if understorey and/or ground-layer vegetation was present beneath a camera, it was removed to reduce false triggers.

Footage from all of the cameras operating during each session at each site was pooled together for the analyses. Data from the camera traps were managed and analysed using Wild Photo Trap 2.0 (A. Kenney, pers. comm). This program requires each animal (or lack thereof) in every image to be classified and a hit window, which is the length of time used to group consecutive photographs together as single detections or hits (Villette *et al*. 2016), must be selected. Hit rates were calculated as the number of hits per 1000 camera days, both for the full length of camera deployment (ten days) and also for the first four days, in line with the live trapping time frame, using a hit window of five minutes. Hit windows have ranged from two minutes to one hour in studies of Australian mammal species and five minutes has been shown suitable for species such as the northern hopping-mouse (*Notomys aquilo*) (Diete *et al*. 2015).

Live trapping

Live trapping was conducted at all sites, either before or after camera trapping, to ensure that the activities associated with live trapping did not impact the effectiveness of the camera traps and vice-versa (Swan *et al*. 2014). Live trapping was conducted over four consecutive nights as recommended by the Department of Parks and Wildlife (SOP #9.2, https://www.dpaw.wa.gov.au/images/documents/plants-animals/monitoring/sop/sop09.2_cagetraps_v1.1.pdf). Medium sized wire cage traps were used (220 x 220 x 450mm, Sheffield Wire Works, Welshpool, Western Australia) and were baited with universal bait. Each trap was placed under vegetation and covered with a hessian bag to protect the animals from the elements. The traps were checked and cleared each morning at first light, before being reset. In hot weather, traps were checked both in the morning and afternoon. Trapped quenda were transferred to a dark cloth handling bag where they were implanted with a passive integrated transponder or PIT tag (Trovan ID100 (1.4), Trovan, Ltd., North Humberside, UK) between the shoulder blades. Standard measurements of pes length, head length, reproductive status and body weight were also recorded. Quenda were released at the point of capture and captured animals other than quenda were released immediately. All aspects of this study were approved by The University of Western Australia's Animal Ethics Committee (approval number RA/3/100/121) and were consistent with the guidelines in the 'Australian code of practice for the care and use of animals for scientific purposes' (NHMRC 2013).

Quenda population and density estimates were calculated from live trapping data using Efford's maximum-likelihood spatially explicit capture-recapture model in the program DENSITY 5.0 (Efford 2012). The default parameters for DENSITY 5.0 were used for all estimates and the buffer width was set to 100m.

Layout

All sites had a camera trap within 20m of a cage trap location in order to sample the same individuals. Both camera and live traps were placed randomly with respect to quenda distribution. *Whiteman Park* had 16 camera traps and 32 cage traps deployed along a 2.89km transect. The spacing between camera traps varied between 80m and 740m (average 261m). Two cage traps were deployed at each camera trap location, resulting in 160 camera nights and 128 trap nights. *Blue Poles Road* had five camera traps spaced 200m apart along an 800m transect and two cage traps were deployed at each camera trap location, resulting in 50 camera nights and 40 trap nights, respectively.

Maralla Road Nature Reserve had eight camera traps that were positioned in pairs. Each pair was ~200m apart from the next pair and ~100m apart from each other. Three cage traps were placed within 20m of each camera trap location, resulting in a total of 96 trap nights and 80 camera nights.

Aileen Plant Park had 26 cage traps placed on seven parallel transects, with four traps on six transects and two on one transect, resulting in 104 trap nights. Traps were spaced ~15m apart along each transect and transects were 15m apart. Thirteen camera traps were randomly placed at a cage trap location, however, two malfunctioned, resulting in 110 camera nights.

Moitch Park had five camera traps positioned along three transects of ~100m, resulting in 150 camera nights. Camera traps were spaced 20m apart along each transect and transects were 30m apart. At each camera trap location, two cage traps were deployed, resulting in 144 trap nights.

At *Craigie Bushland*, 30 cage traps were deployed between 10 and 20m from vehicle tracks for ease of access. The traps were set to circumnavigate the fenced area and were along internal pathways. Cage traps were between 40m to 170m apart (average 100m) and 15 camera traps were placed on every second cage trap location. This resulted in 120 trap nights and 150 camera nights.

Thomsons Lake Nature Reserve had 15 camera traps placed around the lake such that they were accessible from vehicle tracks, resulting in 150 camera nights. The cameras were spaced between 70 and 750m apart (average 420m) around the lake and two traps were deployed at each camera trap location, resulting in 120 trap nights.

Statistical analyses

All analyses were conducted at the site level, so the data were pooled for traps and cameras at each site. As stated above, population and density estimates were calculated for each site using DENSITY 5.0 (Efford 2012) and hit rates (the number of hits per 1000 camera days for each species) were calculated using Wild Photo Trap 2.0 (A. Kenney, pers. comm). Spearman rank correlations were used to determine if live trapping population and density estimates could be predicted by hit rates for the quenda. Hit rates were correlated against the density estimates and also population estimates to test for a relationship and alpha of 0.05 was used for significance.

Results

In total, 18 species were recorded across all sites, with an average of 5.57 (s.e. = 0.78) species at each site. Quenda were the only targeted species, therefore, 17 non-target species were recorded from both live and camera trapping. This includes four introduced species and fourteen native species (Table 1). Each technique captured at least one species that the other technique failed to detect; however, the majority of species at each site were detected by a single sampling method. Overall, camera trapping detected a greater number of species than live trapping and all but one species detected from live trapping were captured on camera traps.

Table 1. The species detected across all sites, showing the method/s by which they were detected. The numbers in the parentheses indicate the number of sites the species was present at.

Common name	Scientific name	Live trapping	Camera trapping
Southern brown bandicoot or quenda	*Isoodon obesulus fusciventer*	*(4)	*(4)
Brush wallaby	*Macropus irma*		*(2)
Western grey kangaroo	*Macropus fuliginosus*		*(5)
Common brushtail possum	*Trichosurus vulpecula hypoleucus*	*(1)	*(2)
Black rat[a]	*Rattus rattus*	*(5)	*(5)
Rabbit[a]	*Oryctolagus cuniculus*		*(1)
Fox[a]	*Vulpes vulpes*		*(2)
Cat[a]	*Felis catus*		*(1)
Yellow-rumped thornbill	*Acanthiza chrysorrhoa*		*(1)
Rainbow bee-eater	*Merops ornatus*		*(1)
Singing honeyeater	*Gavicalis virescens*		*(1)
Splendid fairy-wren	*Malurus splendens*		*(2)
Magpie	*Cracticus tibicen*	*(1)	*(2)
Raven	*Corvus coronoides*		*(1)
Bobtail	*Tiliqua rugosa*	*(1)	*(2)
Ctenotus[b]	*Ctenotus* sp.		*(2)
Dugite	*Pseudonaja affinis*		*(1)
Tiger snake	*Notechis scutatus*	*(1)	

[a] introduced, [b] identified to genus level

Quenda

Quenda were captured at four of the seven sites and both live and camera trapping detected quenda when they were present at a site. The total number of quenda trapped across all sites was 89 (50 males, 38 females and one unknown because it escaped before being sexed) and the number of quenda caught varied among sites from zero to 56 (average 12.7 quenda per site, s.e. = 7.96, Table 2). Males were generally heavier and larger with a mean body mass of 1195g ± 32.0 (SE) (Table 3) and most individuals were in visibly good condition. Density estimates ranged from zero to 1.81 animals/ha and population estimates ranged from zero to 71 (Table 2). Over the full deployment, the camera traps recorded a maximum of 514 quenda hits at a single site, with a minimum of zero (Tables 2 and 4) and the hit rates varied from zero to 3842.11 hits/1000 days (Tables 2 and 4). When using the data from the first four days of camera deployment, a maximum of 343 quenda hits occurred at a single site, with a minimum of zero and the hit rates varied from zero to 6125 hits/1000 days (Table 2). Quenda were always alone when captured by the camera traps and they were active between the hours of 1700 and 0800, with the majority of captures occurring between the hours of 2100 and 0500.

Table 2. Number of quenda (*Isoodon obesulus fuscivente*r) trapped, estimated population size, estimated density (indivisuals/ha), number of camera hits and the hit rates (hits/1000 days) for each site and the site size (ha).

	Site size (ha)	Number of live trapped quenda	Estimated population size	Estimated density	Number of camera hits	Hit rates (10 days)	Hit rates (4 days)
Whiteman Park	14	0	0	0	0	0.00	0.00
Blue Poles Road	4	0	0	0	0	0.00	0.00
Maralla Rd	145	0	0	0	0	0.00	0.00
Moitch Park	2.5	3	4.0	0.41	38	236.02	347.22
Aileen Plant Park	1	5	5.0	1.35	85	772.73	1340.91
Thomsons Lake	538	25	60.0	1.81	24	195.12	220.34
Craigie Bushland	41	56	71.0	1.53	514	3842.11	6125.00
Average (SE)	106.5 (74.5)	12.71 (7.96)	20 (11.83)	0.73 (0.30)	94.43 (70.88)	720.85 (530.39)	1147.64 (848.85)

Table 3. Mean body weight, head length and pes length for the male and female quenda (*Isoodon obesulus fuscivente*r) caught across all seven sites (±SE).

	Males	Females
Weight (g)	1195±32.0	906±22.0
Head length (mm)	88.7±0.82	82.9±1.06
Pes length (mm)	63.0±0.52	57.7±0.50

Statistical analyses

Quenda were the only target species, so they were the only species included in the statistical analyses. Eight mystery hits where animals could not be classified to species were excluded from the analyses, as were ‘useless’ images that lacked any animals. The camera trap hit rates and live trapping population and density estimates were positively associated. The relationship between the population estimates and hit rates was significant ($R(5)$ = 0.89, 0.025>P>0.01) and is shown in Figure 2. The relationship between quenda density and hit rates was also significant but less so than population estimates ($R(5)$ = 0.77, 0.05>P>0.025) and is shown in Figure 3. The correlations were identical for hit rates calculated from both four and ten days.

Live trapping

A total of 752 trap nights across all sites produced 162 captures of six species, resulting in a success rate of 21.5%. Trap effort varied from 40 trap nights to 144 trap nights with a mean effort of 107.4 (s.e. = 12.7) trap nights. Overall, five native and one introduced species were captured (Table 1) and the most commonly detected species was the quenda followed by black rats (*Rattus rattus*). Live trapping detected between one and four species at each site with an average of 1.86 (s.e. = 0.51). This method was the only sampling method that detected the tiger snake (*Notechis scutatus*) (Table 1).

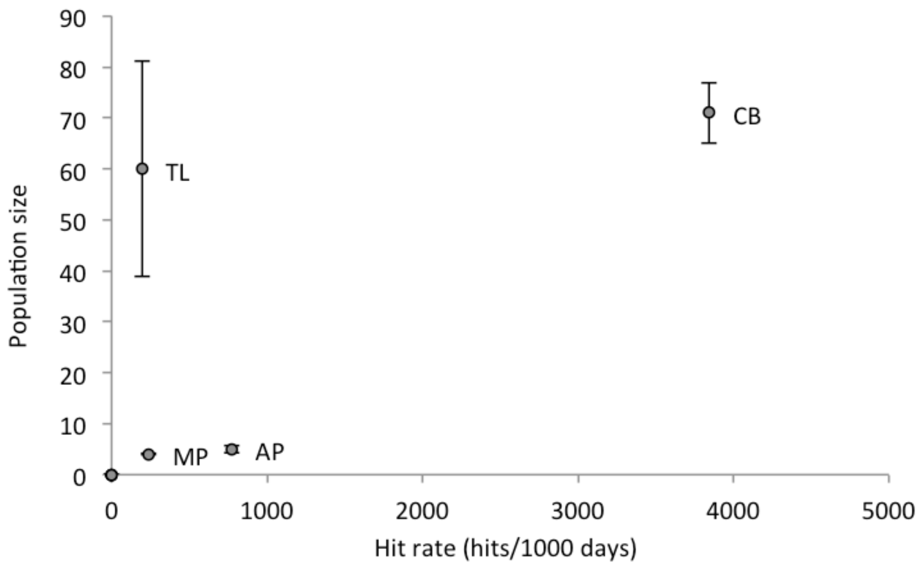

Figure 2. Population estimates obtained from live trapping versus camera trap hit rates calculated from ten days (R=0.89) for seven quenda (*Isoodon obesulus fusciventer*) populations in Perth, Western Australia. Site names are adjacent to the marker. TL denotes Thomsons Lake Nature Reserve, CB denotes Craigie Bushland, AP denotes Aileen Plant Park, and MP denotes Moitch Park. Error bars are ±1 S.E. for population estimates.

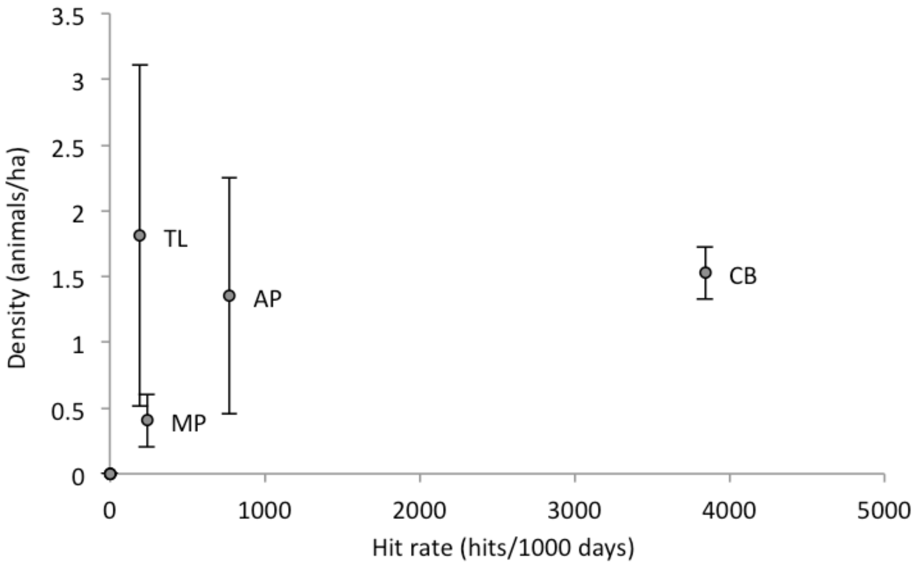

Figure 3. Density estimates obtained from live trapping versus camera trap hit rates calculated from ten days (R=0.77) for seven quenda (*Isoodon obesulus fusciventer*) populations in Perth, Western Australia. Site names are adjacent to the marker. TL denotes Thomsons Lake Nature Reserve, CB denotes Craigie Bushland, AP denotes Aileen Plant Park, and MP denotes Moitch Park. Error bars are ±1 S.E. for density estimates.

Camera trapping

Total camera trapping effort was 850 camera nights and mean effort for a single camera trapping session was 121.4 (s.e. = 16.1) but ranged from 50 to 160 camera nights. The camera traps recorded a total of 1973 hits across all sites sampled, with Craigie Bushland having the greatest number of hits (557) and Maralla Road Nature Reserve having the fewest number of hits (13, Table 4). Of the 1973 hits, eight (0.40%) captured animals that due to poor photographic quality were unable to be identified and 907 (46.0%) were triggered by vegetation, light or shadows. A total of 1058 (53.6%) hits contained animals that could be identified to species, including four introduced and thirteen native species (Table 1 and 4). Therefore, the cameras detected 16 non-target species with the most commonly detected species being the quenda, followed by black rats and the western grey kangaroo (*Macropus fuliginosus*). Camera trapping detected between two and eight species at each site (mean = 5.0, s.e. = 0.87) and was the only method that detected twelve of the eighteen species (Table 1). At the site level, camera trapping detected a greater number of species than live trapping, for the majority of sites. Animals recorded on camera traps were typically alone; however, in some cases, kangaroos were in groups of up to three individuals.

Table 4. Number of camera nights, species recorded, number of hits in parentheses and the hit rate calculated from ten days (number of hits/1000 days) for each species at each site.

Site	Number of camera nights	Fauna species recorded	Hit rate
Whiteman Park	160	Black rat [a] (32)	246.27
		Western grey kangaroo (25)	186.57
		Bobtail (5)	37.31
		Western brush wallaby (4)	29.85
		Fox [a] (3)	22.39
		Magpie (3)	22.39
		Yellow-rumped thornbill (2)	14.92
		Cat [a] (1)	7.46
Thomsons Lake Nature Reserve	150	Western grey kangaroo (62)	504.07
		Quenda (24)	195.12
		Rabbit [a] (12)	97.56
		Splendid fairy-wren (3)	24.39
		Black rat [a] (2)	16.26
		Common brushtail possum (1)	8.13
		Raven (1)	8.13
Moitch Park	150	Quenda (38)	236.02
		Ctenotus sp. [b] (9)	55.90
		Black rat [a] (3)	18.63
		Singing honeyeater (2)	12.42
		Rainbow bee-eater (2)	12.42
		Dugite (1)	6.21
Craigie Bushland	150	Quenda (514)	3842.11
		Black rat [a] (32)	240.60
		Bobtail (4)	30.08
		Common brushtail possum (3)	22.56
		Ctenotus sp.[b] (3)	22.56
		Western grey kangaroo (1)	7.52
Maralla Road Nature Reserve	80	Kangaroo (8)	109.59
		Splendid fairy-wren (4)	54.79
		Fox [a] (1)	13.70
Blue Poles Road	50	Western grey kangaroo (11)	175.00
		Magpie (2)	50.00
		Western brush wallaby (1)	25.00
Aileen Plant Park	110	Black rat [a] (154)	1400.00
		Quenda (85)	772.73

[a] introduced, [b] identified to genus level

Discussion

This study is the first to attempt to use camera traps to estimate the density and population size of quenda or any bandicoot or peramelid and the results have demonstrated that it may be possible to use camera traps alone to survey this individually unrecognisable species. Strong correlations between hit rates and live trapping population and density estimates obtained through live trapping were expected, and the results support this hypothesis. The results suggest that hit rates may not only be used to estimate the abundance of quenda, but also the density of quenda populations. The calibration and comparison of these standard operating procedures ensures that the results are realistic and practical. The ability to use cameras traps to measure aspects of quenda populations offers a rapid and more practical sampling method for resource-constrained researchers, wildlife managers or even community groups with little training.

The results suggest that population size and density can be calibrated to camera trap hit rates, but the significance of these relationships may be influenced by the decision to include the three sites where quenda were absent. Therefore, these relationships could be stronger, particularly the relationship between hit rate and density. Unfortunately quenda were present at only four of the seven study sites and it would have been preferable to have a greater number of sites with quenda, as this would ensure a more robust calibration of methods. While a range of densities was required, three sites without quenda was unexpected. They were supposedly present at Whiteman Park, Blue Poles Road and Maralla Road Nature Reserve, but they were not captured either by live trapping or camera trapping at any of these sites. It is possible that the combination of open habitat and the presence of introduced predators (foxes and feral cats) has resulted in local extinctions of the species at these sites (Claridge *et al*. 2010b; Smith and Coulson 2012) or they occur at such low densities that they were not captured by either method. Also, sampling effort was not equal among sites as three sites had fewer cameras, and this may have influenced the hit rates and subsequent statistical analyses. If sampling effort had been equal among the sites, there is a possibility that the correlation would have been stronger because, with increased sampling effort, it is possible that additional quenda would have been detected, resulting in a greater hit rate and thus a stronger correlation. Additionally, as Thomsons Lake Nature Reserve had a greater number of quenda than Moitch Park and Aileen Plant Park, there should have been a greater number of hits and therefore, a higher hit rate at this site. The lower hit rate can possibly be explained by the heavy rain and storms during the Thomsons Lake camera deployment, which would have influenced the behaviour of the quenda as their activity is suppressed during heavy or continuous rain (Read 1988). If their activity was suppressed, then they would not have been foraging and would not have come into contact with the camera traps as often, therefore, resulting in a lower hit rate and a weaker correlation. Unfortunately, the storms could not be avoided, as due to time constraints, the camera traps could not be deployed at any other time. The camera traps themselves are unlikely to be the reason why the hit rates were lower at Thomsons Lake Nature Reserve, as both models were able to detect quenda and were still functioning by the end of the camera trapping session.

Camera trapping is considered underutilised for several taxa, including individually unrecognisable mammals, despite its rapid adoption into wildlife research (Cutler and Swan 1999; De Bondi *et al*. 2010). Camera traps are able to provide non-invasive rapid and efficient surveys, as I have shown in this study and as also shown in Claridge *et al*. (2010a) and Bain *et al*. (2014). They are also cheaper and less labour intensive, however, no attempt was made to calculate the cost-effectiveness of camera traps relative to live traps in this study, because costs are typically project specific and depend on study requirements and context (Paull *et al.* 2012; Swan *et al*. 2014; Welbourne *et al*. 2015). Previous studies (e.g. Garden *et al*. 2007; De Bondi *et al*. 2010; Welbourne *et* al. 2015; Villette *et al*. 2017) have shown that camera traps are cheaper and less time consuming to deploy.

Relative abundance indices produced from camera traps, such as hit rates, have been criticised because they are rarely calibrated to independent estimates of abundance or density (Kelly and Holub 2008; Rovero and Marshall 2009; O'Connell *et al*. 2011; Weerakoon *et al*. 2014; Hofmeester *et al*. 2016). It is intuitive that hit rates obtained from camera traps are related to abundance, as encounters between animals and cameras are expected to increase with increasing density or population size (Rovero and Marshall 2009; Watkins *et al*. 2010; Hofmeester *et al*. 2016). Using non-calibrated abundance indices, particularly for threatened species, bears a strong risk of making incorrect decisions and inferences about a population (Sollman *et al.* 2013). Therefore, calibrating the relationship between hit rates and density or abundance, as done here, allows informed decisions regarding a population to be made. For example, abundance or density can firstly be estimated and then population trends can be monitored using this information. Many management and research problems are well served by abundance indices and the use of camera trapping hit rates as an index of abundance is promising for the rapid assessment of rare or elusive species, or in areas where other methods are unfeasible (Rovero and Marshall 2009). It is important to note that camera traps cannot replace studies that require data that can only be obtained from handling animals, such as reproductive status, body weight or tissue samples for genetic analysis. However, the calibration between hit rates and abundance in this study indicates that camera traps could be used to make informed management decisions for this species. This is especially true if additional sites can be included in future surveys to strengthen the relationship between live trapping and camera captures. Initially, I attempted to fit a regression line to the data but it was not significant unless it was forced through the origin, in line with the assumption that if bandicoots were present they would be photographed. I resorted to using the Spearman rank correlation because of the limited number of sites and the uncertainty about this assumption. If additional sites can be added to this study, I may be able to use a regression, that would produce a predictive equation that would allow population estimates or densities to be calculated from camera trap hit rates. Additionally, an important assumption regarding indices of abundance obtained from camera traps, and one assumed here, is that of equal detectability of a species across sites. It is reasonable to assume this for comparisons of the same species across different sites as they are roughly the same size and have the same habitat requirements (Harmsen *et al*. 2010).

While this study is the first to calibrate the relationship between hit rates, density and abundance for quenda, similar studies have been conducted on other nondescript mammals with mixed results. Bengsen *et al*. (2011) found that camera trap abundance indices could monitor changes in a feral pig population and Diete *et al*. (2015) found that camera trapping can be used as a sampling method for the northern hopping-mouse (*Notomys aquilo*). Also, Villette *et al*. (2016, 2017) found that camera trapping can be used as a robust means to estimate density of red squirrels (*Tamiasciurus hudsonicus*), red-backed voles (*Myodes rutilus*) and deer mice (*Peromyscus maniculatus*). Other researchers have also found that camera traps can be used to estimate the density or abundance of the Irish and European hare (*Lepus timidus hibernicus* and *L. europaeus*), European pine marten (*Martes martes*), red-necked wallaby (*Macropus rufogriseus*), quokka (*Setonix brachyurus*) and ungulates (Rowcliffe *et al*. 2008; Rovero and Marshall 2009; Manzo *et al*. 2012; Dundas *et al*. 2014; Caravaggi *et al*. 2016). By contrast, Weerakoon *et al*. (2014) found that camera traps were ineffective at detecting changes in black rat population size and Villette *et al*. (2017) were unable to assess camera traps as a means of estimating density for snowshoe hares (*Lepus* *americanus*). This study adds to this growing body of knowledge as it demonstrates that it is possible to survey and monitor a medium sized marsupial with camera traps.

Conservation and management implications

This research has important implications for the conservation and management of quenda in Western Australia and endangered bandicoots elsewhere. In order to detect population changes, monitoring must be effective and accurate, and precise measures of population size can impact on the efficacy of management or conservation strategies (Wayne *et al*. 2013). This study has shown that camera traps can be used to sample quenda populations, allowing a choice of sampling methods for future studies. They have proven useful tools as they can assess quenda abundance and produce rapid and quantifiable results in a non-invasive manner. As there was no difference in the statistical results when hit rates calculated from four camera nights were used, camera trap surveys used to measure quenda populations could be limited to just four days instead of ten, producing even faster results. They are inexpensive tools that can detect changes in quenda population size; however, their success is dependent on weather conditions, as there will be few hits during periods of heavy and continuous rain, as observed in this study at Thomsons Lake Nature Reserve. In this study, camera traps detected a greater number of species than live traps and were able to detect small, medium and large mammals as well as birds and reptiles. This ability to monitor multiple species at once further highlights the cost-effectiveness and efficiency of camera traps. Finally, camera traps may facilitate regular monitoring and consequently improve wildlife management by enabling informed decisions regarding populations of the quenda to be made. Camera trapping hit rates as an index of abundance may also allow us to standardise and reduce the costs of monitoring programs (Rovero and Marshall 2009) ensuring effective conservation of this species.

Limitations

This study has several limitations that should be considered. Firstly, the low number of sites and low sample size make it difficult to make overarching conclusions about the utility of camera traps for measuring quenda population size or density. A stronger conclusion could be made if this study had been conducted over a larger number of sites where quenda were present, which due to time constraints, could not be achieved in this study. Secondly, sampling effort was inconsistent between sites, with the number of camera nights varying between 50 and 160. To ensure an accurate comparison between methods and strong conclusions, it is ideal to have equal sampling effort across sites. Finally, the hit window length may have influenced the results. As hit windows are species specific, a five minute hit window may not be the most appropriate to maximise the correlation between hit rate and population size or density for this species.

Future research

While the results presented here are promising, further research is required. Firstly, it would be beneficial to conduct this study over a greater number of sites and over a wider range of quenda densities to strengthen and confirm the correlation between hit rates and population size and density. Secondly, repeating this survey with equal sampling effort at each site would be ideal to ensure accurate comparisons between methods. Thirdly, it would be desirable to determine if hit window length affects the correlation between hit rates and population size or density, and if five minutes is the most appropriate hit window for this species, as this was beyond the scope of this project. Five minutes was chosen based on previous research, however, optimal event definitions are dependent on a study's target species, site, design and intent, and there are currently no standard lengths of time a researcher could use to define an event or hit window (Meek *et al*. 2014; Diete *et al*. 2015). In addition, determining if hit rates obtained from unbaited camera traps are correlated to density or abundance would be prudent, as the use of baited cameras has been criticised as animals are attracted into the survey area, which may result in increased population estimates (Dundas *et al*. 2014). Finally, determining if the relationships found here can be applied to populations in other regions, particularly in areas where this and other species of bandicoots are rare or endangered, would be extremely useful.

Using camera traps to estimate the population size of species with uniform morphology is one of the most difficult challenges faced by wildlife researchers, and estimating abundance and density for nondescript species is an ongoing focus of wildlife research (Chandler and Royle 2013; Sollmann *et al*. 2013; Dundas *et al*. 2014). This study has shown that camera trap surveys can potentially provide valid measures of abundance that would enable researchers and practitioners to monitor quenda populations. The relationship between hit rates and population and density estimates suggests that camera traps are a potential alternative to live trapping for the quenda, however, further research is required over a greater number of sites to strengthen these relationships. Despite the need for further research, the results of this study are promising because they indicate that it may be possible to survey and monitor nondescript medium-sized marsupials with camera traps without the necessity of live trapping, which is costly, time-consuming and has welfare implications. Therefore, camera traps may provide a useful, practical and cheap alternative to traditional methods, which may lead to an improvement in the conservation and management of our native marsupials.

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