Development and application of survey methods to determine habitat use in relation to forest management and habitat characteristics of the endangered numbat (*Myrmecobius fasciatus*) in the Upper Warren region, Western Australia

by

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BSc (Hons)

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Author’s declaration

I declare that (a) the thesis is my own account of my research, except where other sources are acknowledged, (b) all co-authors, where stated and certified by my principal supervisor or executive author, have agreed that the works presented in this thesis represent substantial contributions from myself and (c) the thesis contains as its main content work that has not been previously submitted for a degree at any other university.

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Anke Seidlitz

Abstract

Effective detection methods and knowledge on habitat requirements is key for successful wildlife monitoring and management. The numbat (*Myrmecobius fasciatus*) is an endangered, Australian-endemic marsupial that has experienced major population declines since European settlement. The Upper Warren region (UWR) in south-western Australia contains one of the two remaining natural populations. A lack of effective survey methods has caused a paucity of information regarding this population. This PhD project aimed to develop robust survey methods and determine habitat requirements for the numbat in the UWR.

Given the perceived advantages of camera trap technology in wildlife research, camera trap trials were conducted to optimise camera methodologies for numbat detection. Swift 3C wide-angle camera traps positioned at ~25 cm above ground increased numbat detections by 140% compared to commonly used Reconyx PC900 camera traps. Elevated, angled cameras were suitable for numbat individual identification.

As numbats are difficult to catch, three non-invasive methods for numbat detection were field tested (Sign surveys, driven surveys and camera trapping). Sign surveys (searching for diggings and scats) were more successful and cost effective than driven observational surveys or camera trapping. Sign surveys are appropriate to investigate changes in occupancy rates over time, which could serve as a metric for long-term numbat monitoring.

Since camera traps are an attractive tool for wildlife detection, field trials were conducted to increase camera trap detection rates for numbats and other species. Detection rates from stationary Reconyx PC900/HC600 (40° detection angle) were compared to paired, periodically repositioned Reconyx PC900/HC600 and Swift 3C wide-angle camera traps (110° detection angle). Swift 3C wide-angle camera traps had significantly higher animal detection rates compared to Reconyx PC900/HC600 camera trap models. Stationary and periodically repositioned Reconyx camera traps performed similarly, although there were significant differences for some species including the numbat.

Sign surveys that were conducted concurrently with the above camera trap field trials showed that autumn sign surveys detected significantly more numbat signs than spring sign surveys. Even though numbat detection rates by camera traps improved by using periodically repositioned Swift 3C wide-angle camera traps, sign surveys that were conducted at the same time and at the same sites were more successful.

Sign surveys were applied to determine how forest management activities (prescribed fuel-reduction burns, timber harvesting, introduced predator control intensity) affect numbat habitat use. Numbat signs were found at 83% of 78 survey sites, indicating that numbats in the UWR are habitat generalists. Log number was the only important determinant of numbat habitat use. Logs provide numbats with refuge from predators and hollows for resting and nesting.

This project made valuable discoveries on monitoring and protecting numbats in the UWR. At present, the most effective method for monitoring the species is by using sign surveys to estimate occupancy. High habitat occupancy rates indicate that current forest management is suitable for this species provided that sufficient logs are retained in managed areas.

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List of publications and work in progress arising from this project

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Chapter 6 (in preparation for publication)

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Statement of authorship

This thesis consists of seven chapters including one published paper (Chapter 2), two papers that are in review for publication (Chapters 3 and 4) and a draft paper intended for journal submission (Chapter 6). A. Seidlitz has written all chapters, has developed all tables and figures, and is the first author of all papers already published or intended for publication. For all chapters, A. Seidlitz has received comments and editorial guidance of K. A. Bryant (principal supervisor) and A. F. Wayne and N. J. Armstrong (co-supervisors). M. Calver was a co-supervisor during the first year. He provided comments and editorial guidance for the initial proposal and literature review of which parts were used in Chapter 1. All supervisors co-authored two or more data chapters as detailed below.

Chapter 2: A. Seidlitz, K. A. Bryant, A. F. Wayne and M. Calver conceived the ideas and designed methodology; A. Seidlitz collected the data; A. Seidlitz analysed and interpreted the data under the direction of N. J. Armstrong; A. Seidlitz led the writing of the manuscript and addressed and coordinated the editorial changes suggested by journal reviewers. All authors contributed critically to the drafts and gave final approval for publication.

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Pam Firth (Detail Devil Editing Services) provided professional copyediting (Chapters 1, 5, 6 and 7) and formatting services (entire thesis) according to the guidelines laid out in the university-endorsed national Guidelines for Editing Research Theses (Institute of Professional Editors, 2019).

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Kate A. Bryant

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List of abbreviations

DBCA	Department of Biodiversity, Conservation and Attractions
EPBC Act	Australian Environment Protection and Biodiversity Conservation Act 1999
GLM	Generalised linear model
IUCN	International Union for Conservation of Nature
PIR	Passive infra-red
NSW	New South Wales
RAI	Relative abundance index
SA	South Australia
SECR	Spatially explicit capture–recapture
UWR	Upper Warren region
WA	Western Australia

Chapter 1 Introduction

The numbat (*Myrmecobius fasciatus*), Western Australian animal emblem since 1973 (image source: Department of the Premier and Cabinet 2020).

The numbat (*Myrmecobius fasciatus*) in the Upper Warren region, south-western Australia.

1.1 Context and reasons for this research

Species are declining rapidly worldwide, and their protection has become increasingly important (Dirzo and Raven 2003; Barnosky *et al.* 2011; Ceballos *et al.* 2017). In Australia, during the past 200 years and since European settlement, 30 endemic terrestrial mammal species have become extinct at a rate of about one to two extinctions every decade (Woinarski *et al.* 2014; Woinarski *et al.* 2015). Today, 55 Australian terrestrial mammal species are listed as threatened on the IUCN Red List of Threatened Species™ (IUCN 2020). Under the Australian Environment Protection and Biodiversity Conservation Act 1999 (*Cth*: EPBC Act), 10 mammals are currently listed as critically endangered and 38 mammals as endangered (Department of Agriculture Water and the Environment 2020).

In Australia, the top three threatening processes to mammals are habitat loss, introduced species and inappropriate fire regimes (Evans *et al.* 2011). This study was conducted in the Upper Warren region (UWR), which is part of the South West Australia Global Biodiversity Hotspot (Myers *et al.* 2000) and has a high conservation value as it supports a large number of rare and endangered species (Burrows and Christensen 2002; Wayne and Moore 2011). Within the UWR, open sclerophyll forests and woodlands cover more than 140 000 ha of publicly managed land (Yeatman *et al.* 2016). Forest management activities within this region are conducted by state authorities and include prescribed fuel-reduction burns, timber harvesting and control of introduced predators using poisoned bait (Wayne *et al.* 2013a). To implement species management and conservation strategies, dedicated studies are essential for individual species to investigate habitat preferences and to determine responses to forest management activities. This, however, can only be achieved when effective survey methods are established for target species, and that has not yet been accomplished for the numbat (*Myrmecobius fasciatus*; Waterhouse 1836), in the UWR.

The UWR is home to one of the last two natural populations of the endangered numbat, an Australian-endemic marsupial. The numbat has been a protected species since 1973 (Department of Parks and Wildlife 2015). Regular monitoring of numbat populations started in the early 1980s (Friend 1990), with a focus on the populations at Dryandra Woodland and at reintroduction sites (Friend and Thomas 1994a; Friend and Thomas 2003). The numbat population in the UWR has not been monitored, and there is no reliable information on the population's size and distribution or on how forest management activities affect the population. Therefore, this thesis set out to meet the objectives of the current Numbat Recovery Plan: to develop robust survey methods and to determine habitat preferences to facilitate effective species management and protection (Department of Parks and Wildlife 2017).

1.2 Chapter overview

This chapter introduces the overall problem addressed and the aims and scope of this study. It gives an overview of the study species' historic and present distribution and conservation status. This chapter also provides information on the numbat's biology and ecology, which is important as it helps to highlight the challenges involved in developing detection methods. Following this, detection methods and derived population parameters are reviewed. Survey methods used for species similar to numbats are considered and their applicability to the study species in the UWR. Traditional numbat survey methods are presented with emphasis on their limitations and possible improvements. As this study used an adaptive approach, a flow chart is provided presenting possible scenarios for the directions of this study.

1.3 The numbat

1.3.1 Past and present

The numbat is an iconic, Australian-endemic marsupial, which became the Western Australian animal emblem in 1973 (Department of the Premier and Cabinet 2020). Numbats are small (500–700 g adult body mass) and especially vulnerable to decline and extinction (Burbidge and McKenzie 1989). Accounts of the species' decline reach back to as early as 1923 (Harper 1945). Prior to European settlement, numbats occurred over much of the southern half of the Australian continent (Friend 1990, Figure 1.1). Today, the numbat occurs in only 1% of its former range (Department of Parks and Wildlife 2017). The decline is attributed to factors such as habitat loss, introduction of feral predators, changes in fire regimes, disease, drought and climate change (Friend 1987b; Friend 1990; Maxwell *et al.* 1996; Friend and Thomas 2003; Vaughan-Higgins *et al.* 2013). Two natural numbat populations remain, located at the Dryandra Woodland and the UWR in Western Australia

(WA) (Department of Parks and Wildlife 2017). A program to reintroduce numbats into areas of its former range was established in 1985 (Friend and Thomas 1994a), and a Numbat Recovery Plan has been in place since 1995 (Friend 1994). A total of 14 reintroductions to previously occupied areas have been conducted to date (Figure 1.1). Six reintroductions have not been successful; numbats are either no longer present or the status is unknown (Karroun Hill Nature Reserve [WA], Karakamia Sanctuary [WA], Mount Dale Area [WA], Stirling Range National Park [WA], Arid Recovery [South Australia (SA)] and Cocanarup Timber Reserve [WA]). Seven reintroduced populations persist (Mount Gibson [WA], Boyagin Nature Reserve [WA], Tutanning Nature Reserve [WA], Batalling forest block [WA], Dragon Rocks Nature Reserve [WA], Yookamurra Sanctuary [SA] and Scotia Sanctuary [New South Wales (NSW)] (Department of Parks and Wildlife 2017; Australian Wildlife Conservancy 2020). The most recent reintroduction was conducted at Newhaven in central Australia within the Northern Territory (Australian Wildlife Conservancy 2019, Figure 1.1). Nevertheless, numbats are still considered threatened (Woinarski *et al.* 2014) and are listed as endangered under the IUCN Red List of Threatened Species™ (Friend and Burbidge 2008) and as 'Fauna that is rare or is likely to become extinct as endangered fauna' under the Biodiversity Conservation Act 2016 (WA). In February 2018, the numbat's conservation status was elevated from vulnerable to endangered under the EPBC Act (*Cth*).

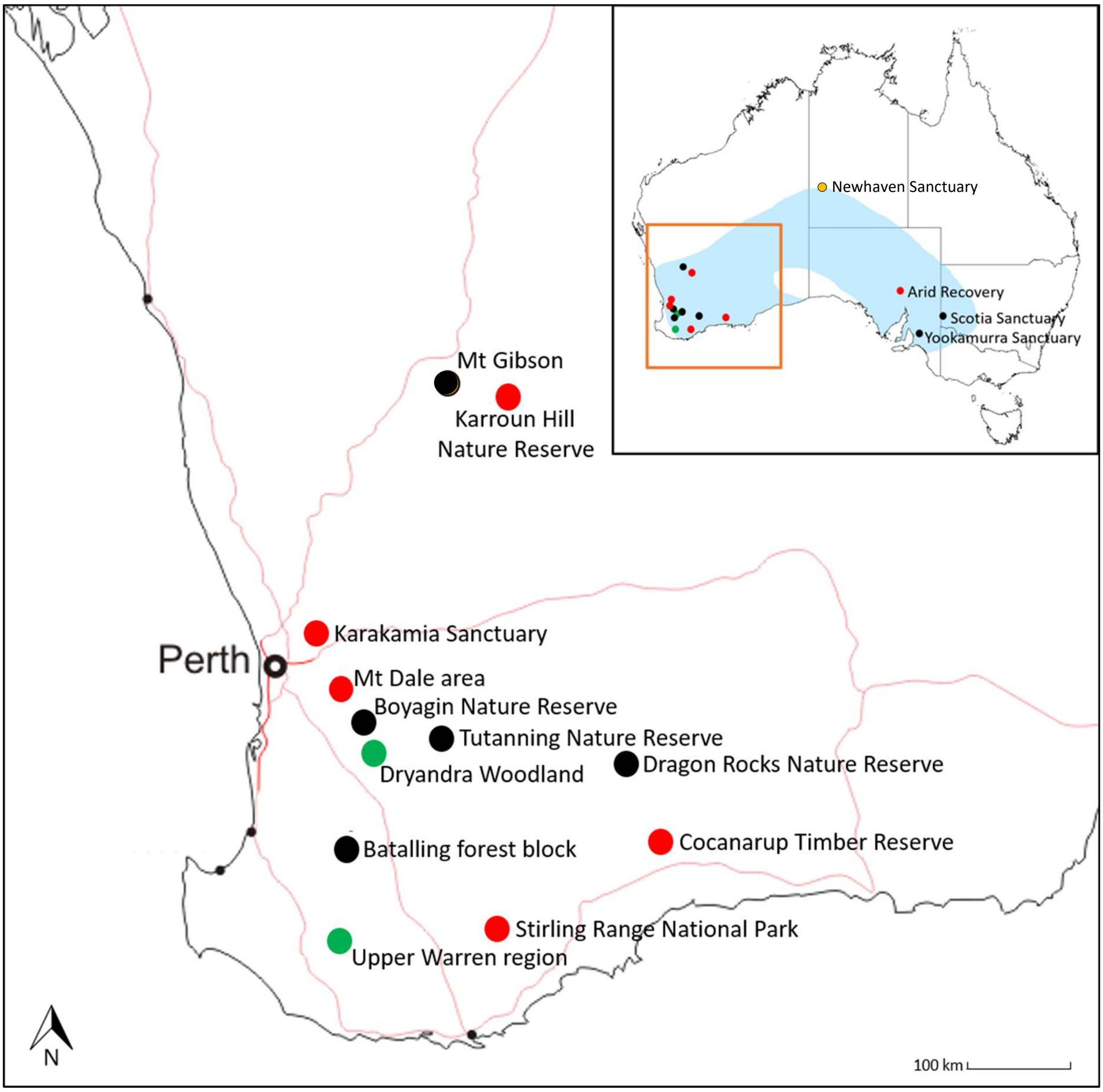

Figure 1.1. Natural numbat populations (green dot), unsuccessful reintroduction sites (red dot), reintroduced self-sustaining sub-populations (black dot) and most recent reintroduction site in central Australia (orange dot). The blue area in the insert depicts estimated historic numbat distribution, before European settlement in Australia. (Map modified from Friend and Burbidge 2008; and Department of Parks and Wildlife 2017.)

1.3.2 Biology and ecology

The numbat is the only member of the family Myrmecobiidae (Gill 1872) and is placed as sister group to all living Dasyuridae, within the order Dasyuromorphia (Zemann *et al.* 2013). Adults may reach a head–body length of 270 mm, a tail length of 200 mm and weigh between 500 and 700 g, with females being slightly smaller than males (Friend 1989). Life expectancy rarely exceeds 5 years in the wild (Friend 2008a) and 7 years in captivity (Hogan *et al.* 2012).

The fur of a numbat is largely reddish brown, becoming dark towards the tail (Figure 1.2). The species has horizontal black eye-stripes and numerous white bands across its rump. The number of white bands can differ between individuals, and bands are often interrupted on the midline, forming characteristic patterns suitable to identify individuals (Department of Parks and Wildlife 2017, Figure 1.2a).

Figure 1.2. Numbats at the Dryandra Woodland nature conservation area near Narrogin, Western Australia. (a) Adult, showing individually characteristic pelage patterns (image taken May 2017); (b) young numbat (image taken Nov 2016). Images courtesy of John Lawson (ldwv@westnet.com.au).

Numbats are myrmecophagous, feeding almost exclusively on termites (Calaby 1960b). Adults consume up to 15,000 to 20,000 termites each day, corresponding to about 10% of their body weight (Friend 1997). Ants, which can often be found in their scats, may be ingested accidentally (Friend 1989). In contrast to other myrmecophagous mammals, such as the echidna, numbats are not strong enough to break termite mounds (Friend 2008a).

Instead, they detect and uncover shallow underground termite runways and find termites in and under dead plant matter (Calaby 1960b).

As is characteristic in many myrmecophagous animals, numbats have a long, thin tongue that can extend several centimetres beyond their elongated snout (Friend 1989). Sticky saliva, produced by greatly enlarged salivary glands, coats the tongue to facilitate gathering of termites, which are scraped off the tongue by several transverse palatal ridges (Cooper 2011). The number of teeth range from 47 to 52, are variable between individuals and may even differ on opposite sides of the same jaw (Friend 1989), although at least eight post-canine teeth are always present in the lower jaw (Friend 1989). Teeth are not considered to be used extensively for feeding as they are generally small, show modest wear and some may not protrude the gum (Calaby 1960b).

Following their prey's habits, numbats are diurnal with seasonal and daily variations in foraging activity (Christensen *et al.* 1984; Friend 1989). In winter, foraging occurs during warmer midday hours (Friend 1989). During summer, numbats retreat to hollow logs or burrows during the heat of the day and concentrate their activity in cooler daytime hours (Friend and Burrows 1983).

Natural predators include several birds of prey (e.g. the little eagle [*Hieraaetus morphnoides*] and brown goshawk [*Accipiter fasciatus*]) and carpet python (*Morelia spilota*). Since their introduction to Australia, cats (*Felis catus*) and foxes (*Vulpes vulpes*) have significantly increased predation pressure on numbats (Friend 1987a).

Hollow logs are an important habitat feature for numbats (Calaby 1960b). Hollow logs are used for overnight and daytime resting, nesting and as a refuge from predators (Christensen *et al.* 1984). In addition to hollow logs, numbats use multiple self-constructed burrows. Burrows consist of a concealed entrance, a 1 to 2 m long tunnel wide enough for a numbat to pass through, and a chamber (approx. 250 mm in diameter) lined with shredded bark and dry leaves (Christensen *et al.* 1984). Burrows have better insulation properties than logs (Friend and Burrows 1983) and appear to be preferred for nursing small young (Christensen *et al.* 1984).

Numbat breeding is highly seasonal. Mating has been found to typically occur in January, during summer (Friend 2008b), but may extend from late November to early April as observed in a captive breeding colony at Perth Zoo, WA (Hogan *et al.* 2012). Females breed in their first year (Calaby 1960b) and commonly give birth to four young after a gestation period of about 2 weeks (Friend and Whitford 1993; Hogan *et al.* 2012). Each young orally attaches to one of the four teats, within the female's pouchless mammary gland area (Calaby 1960a; Friend 1989), where it remains for about 6 months (Calaby 1960b). Thereafter, young are left and cared for within nest burrows for another 4 to 5 months (Christensen *et al.* 1984). During this time, the young numbats start exploring the surrounding area and become gradually more independent until final dispersal around December (Friend and Burrows 1983).

Young numbats disperse to establish their own territory in which they normally spend the rest of their life solitarily (Friend 2008a). At Dryandra Woodland, dispersing distance was found to average 3.5 km but reached up to 10.9 km (Friend 1987a). While same-sex territories are hypothesised not to overlap, males and females may share areas of their territories (Friend and Burrows 1983; Friend 1987a). During the mating season, males roam beyond their home range in search of females (Friend 1987a).

Home-range size is uncertain and may vary considerably between individuals and habitat. Radio-telemetry studies were conducted in the UWR (Christensen *et al.* 1984), in Arid Recovery (Bester and Rusten 2009) and in Scotia and Yookamurra Sanctuaries (Hayward *et al.* 2015). Female home ranges were found to be 0.41 ha (Christensen *et al.* 1984), 16 to 53 ha (Bester and Rusten 2009) and 28 to 39 ha (Hayward *et al.* 2015). For male numbats, ranges measured 123 ha (Christensen *et al.* 1984), 18 to 133 ha (Bester and Rusten 2009) and 51 to 97 ha (Hayward *et al.* 2015). It has been hypothesised that female home ranges contract in summer, while male home ranges contract in winter (Friend 1997); however, Bester and Rusten (2009) found that both male and female home ranges contract in winter, and Hayward *et al.* (2015) found no significant differences in female seasonal home-range size (data for male seasonal home-range size was insufficient).

Habitat preferences for numbats may vary and depend on availability. Before their reduction in geographical distribution (see Section 1.4.1), numbats used to occur in a range of habitats, including areas covered by arid and semi-arid vegetation, as well as woodlands and forests as they do today (Friend *et al.* 1982; Christensen *et al.* 1984; Burbidge *et al.* 1988). It is postulated that numbats prefer habitats with a mixture of areas that provide protection from predators in the form of shrubs or log hollows, combined with areas of open understory for feeding activities (Department of Parks and Wildlife 2017). Natural numbat populations persist in the Dryandra Woodland and the UWR in the South West WA. Little is known of numbat habitat preferences in these areas as published accounts from radio-collared numbats mainly refer to home-range sizes and activity patterns (see Friend and Burrows 1983; Christensen *et al.* 1984). Habitat preferences have been reported from reintroduced numbat populations confined to introduced predator-free, fenced reserves in south-eastern Australia where numbats show a strong preference for shrub vegetation communities (Berry *et al.* 2019). Limited knowledge is available on numbat habitat preferences in relation to anthropogenic disturbances such as prescribed fuel-reduction burns and timber harvesting, as well as the effects of ongoing introduced predator control.

1.3.3 Impacts of anthropogenic disturbances and fox predation

Threats specific to numbats, as outlined in the Numbat Recovery Plan 2017, include inappropriate fire regimes, habitat disturbance and fragmentation, predation by introduced predators (cats and foxes) and climate change (Department of Parks and Wildlife 2017). Forest management actions that may affect numbats in the UWR relate to three of the mentioned threatening processes and are discussed below.

Prescribed fuel-reduction burns are implemented in Australian fire-prone environments to reduce the impact of wildfires and to protect human lives and assets as well as promote the conservation of biodiversity (Burrows 2008). Fire may negatively affect numbats directly during a burn or indirectly by increasing predation through reduction of plant cover and hollow logs (Friend 1994). However, in the Dryandra Woodland, some numbats have been observed to survive fire and to remain within their home range after burns (Department of Parks and Wildlife 2017). Other native species, especially arboreal mammals, have been shown to be negatively affected by fire, leading to declines in abundance (e.g. western ringtail possums [Inions *et al.* 1989; Wayne *et al.* 2006] and brushtail possums [Inions *et al.*

1989]). However, the effect of fire on mammals largely depends on the fire's intensity, spatial magnitude and patchiness (Friend and Wayne 2003).

It has been postulated that timber harvesting may have positively affected numbats in the jarrah (*Eucalyptus marginata*) forest of the UWR, by reducing timber density and increasing the abundance of fallen timber logs suitable as refuges (Christensen *et al.* 1984). A large, 5-year study, exploring the impacts of timber harvesting in jarrah forests, was conducted from 2001 to 2006 (Abbott and Williams 2011). During this study that included 11 mammals (but not the numbat), impacts of silvicultural treatments were assessed to be minor for 41 terrestrial vertebrate taxa (Wayne *et al.* 2011), while other studies have shown negative effects of timber harvest on local mammals (e.g. the western ringtail possum [*Pseudocheirus occidentalis*; Morris *et al.* 2000; Wayne *et al.* 2006] and bush rat [*Rattus fuscipes*; Lunney *et al.* 2009]), as well as positive effects (e.g. brush-tailed bettong [*Bettongia penicillatal*; Wayne *et al.* 2011; Wayne *et al.* 2016]). At Dryandra Woodland, numbats were observed to leave affected areas during timber harvesting operations but to return soon after (Department of Parks and Wildlife 2017). However, there are no published studies on the effect of timber harvesting on numbat populations.

European settlers introduced European red foxes (*Vulpes vulpes*) to Australia in the 19th century, and fox predation is hypothesised to have caused the decline and extinction of a large number of small- and medium-sized native mammals (Kinnear *et al.* 1988; Burbidge and McKenzie 1989; Short and Smith 1994). Fox control using 1080 (sodium fluoroacetate) poisoned baits was shown to effectively reduce fox abundance (e.g. Saunders *et al.* 2010; Marlow *et al.* 2015) and positively affect native species (e.g. Kinnear *et al.* 1998; Morris *et al.* 2000; Dexter and Murray 2009). Fox control was also shown to have a positive effect on the numbat population at Dryandra Woodland and reintroduction sites (Friend and Thomas 2003). Fox control is carried out in the entire UWR with different baiting intensities in some areas (discussed in detail in Chapter 6). It is unknown how these baiting regimes affect native mammals including the numbat.

1.3.4 The numbat in the Upper Warren region

Published, peer-reviewed information on numbats in the UWR is restricted to a single study of four radio-collared numbats conducted by Christensen *et al.* (1984). The study mainly reported the animals' home-range sizes, feeding activities and use of log hollows and burrows. Consequently, there is a lack of information about the UWR numbat population regarding its distribution, size and trends. Furthermore, there is no published literature on the effects of fire, timber harvesting or fox-baiting intensity on numbats (two fox-baiting intensities are applied to areas in the UWR, as described in Chapter 6). This lack of knowledge hinders the management and protection of the UWR numbat population. It is therefore timely to conduct numbat surveys to address these knowledge gaps and to implement a monitoring program to detect population changes over space and time. However, effective survey methods for the numbat population in the UWR have not yet been established.

1.4 Survey methods and population parameters

Numbats are difficult to detect due to their solitary lifestyle with large home ranges (Hayward *et al*. 2015). Furthermore, there are no known lures or baits to attract numbats to traps (Burrows and Christensen 2002). At the Dryandra Woodland and some reintroduction sites, numbats have traditionally been surveyed using observational driven transects and numbat sign searches (scats and diggings) (see Friend and Thomas 2003; Hayward *et al.* 2015; Berry *et al.* 2019). These methods may not be applicable in the UWR as dense vegetation may restrict observations during driven transects, and diggings and scats from other marsupials common in the region may make numbat signs indistinguishable. Yet, these and other survey methods have not been tested for their suitability for detecting numbats in the UWR. The following sections review a range of survey methods and population parameters that may be appropriate for numbat detection and monitoring. The aim of this review is to establish survey method options and related metrics that are worth testing for their effectiveness for detecting and monitoring numbats in the UWR.

1.4.1 Past and future survey methods

The type of survey method used for mammal population estimates depends largely on the biology and ecology of the target species, the habitat characteristics within which the species occurs, the study objectives and available funds (Sutherland 2006, Chapter 1). Like other myrmecophagous species, numbats occur at low densities, are difficult to detect and are not attracted to lures. Table 1.1 details survey methods used for other mammals, including those used for other myrmecophagous species (anteaters, armadillos and pangolins), and determines suitability for numbats. References to more detailed sections that describe methods already applied to numbats or that may be suitable for numbats are provided within Table 1.1.

Table 1.1. Methods commonly used to estimate population parameters for terrestrial, ground-dwelling mammals and their applicability for numbats. Examples of other ant- or termite-eating species are given where applicable.

Survey method	Examples of other termite-/ant-eating species detected	Requirements	Suitability for numbats
Live trapping	n/a	Commonly requires lures or baits for mobile species or drift fences for less mobile species.	Not suitable. Numbats are highly mobile and do not respond to known attractants.
Direct counts from points or line transects	Giant anteater (*Myrmecophaga tridactyla*) (de Miranda *et al.* 2006)	Skilled observers who can detect target animals in sufficient numbers on foot or from vehicles.	Driven transects have been conducted for numbats. See Sections 1.4.3 and 1.4.7 for further details.
Aerial surveys	Giant anteater (*Myrmecophaga tridactyla*) (de Miranda *et al.* 2006)	Use of low-flying planes or drones. Highly visible species.	Not suitable. Numbats are not highly visible.
Camera traps	Giant armadillo (*Priodontes maximus*) (Noss *et al.* 2004)	Suitable cameras and set-up techniques for detecting target species.	Numbats have been detected by camera traps. See Section 1.4.8 for further details.
Nests/ burrows	Indian pangolin (*Manis crassicaudata*) (Mahmood *et al.* 2014)	Easily detectable nests or burrows.	Not suitable. Entrances of numbat burrows or log retreat sites are well concealed.
Calls	n/a	Species that communicate by vocalisation.	Not suitable. Numbats do not commonly vocalise.
Scats/ feeding signs	Indian pangolin (*Manis crassicaudata*) (Mahmood *et al.* 2018)	Skilled observers who can detect and identify animal signs.	Numbats have been detected by scats and feeding signs. See Section 1.4.4 for further details.
Runways	n/a	Species that use tracks repetitively.	Not suitable. Numbat movements do not appear to follow regular paths.

Survey method	**Examples of other termite-/ant-eating species detected**	**Requirements**	**Suitability for numbats**
Tracks	Yellow armadillo (*Euphractus sexcinctus*) (Prada and Marinho-Filho 2004)	Soft substrate and skilled observers who can identify tracks.	Numbats have been detected by tracks. See Section 1.4.4 for further details.
Hair traps	n/a	Hair traps commonly require attractants or regularly pathways.	Not suitable. Numbats do not respond to known attractants or use tracks repeatedly.
Detection dogs	Giant anteater (*Myrmecophaga tridactyla*) and giant armadillo (*Triodontes maximus*) (Vynne *et al.* 2011)	Dogs trained to detect target species signs.	May be suitable. Has not been tested. See Section 1.4.4 for further details.

1.4.2 Absolute and relative abundance

Two types of estimates are commonly used to monitor animal populations: absolute and relative abundance. Absolute abundance measures the total number of individuals within a population, while relative abundance is an index relating to absolute abundance without actually estimating it (Sutherland 2006, p. 156). Relative abundance estimates may be less costly and easier to obtain than estimates of absolute abundance and are therefore often used to monitor population trends over time (Williams *et al.* 2002, p. 257).

Wildlife surveys have commonly relied on simple counts within a defined area (Conn *et al.* 2004). Simple counts can be converted to relative abundance indices (RAIs) – for example, by determining the number of animals detected per unit effort (Sutherland 2006, p. 156). To monitor population trends, RAIs are compared over time and space, and that relies on the assumption that detectability – the probability of actually detecting animals within the population – is constant (Conn *et al.* 2004). This assumption is often violated since animal detectability can be affected by many factors such as weather, season, time of day, vegetation density, animal behaviour and observer ability (Gese 2001). Some studies have found RAIs to reliably represent population trends (e.g. Hopkins and Kennedy 2004; Güthlin *et al.* 2014), while others have shown that indices did not reflect changes in population size (e.g. Matthews *et al.* 2011) and may even have inverse relationships (Sollmann *et al.* 2013).

Besides assuming constant detectability, RAIs are often assumed to have a linear relationship with absolute population abundance (Williams *et al.* 2002, p. 258). In some cases, however, relationships between RAIs and absolute abundance may be non-linear or even non-monotonic (Williams *et al.* 2002, p. 258). Such patterns may derive from a change in animal behaviour – for example, with changing density (Diefenbach *et al.* 1994). Inferences from RAIs that do not behave proportionally to absolute abundance, or that are unreliable due to heterogenous detection probabilities, may result in biased population estimates and lead to poor management decisions (MacKenzie *et al.* 2005a, pp. 7-12).

The relationship of RAIs, which have been used for monitoring numbat populations, to absolute abundance has not been established (see Friend and Thomas 2003). Absolute abundance was recently estimated for the eastern states sub-populations at Scotia and

Yookamurra Sanctuaries (see Hayward *et al.* 2015 and Section 1.4.7 below); however, distance sampling assumptions were violated, rendering results unreliable. A reliable estimator for numbat population parameters has yet to be found.

1.4.3 Relative abundance indices from driven transects

RAIs from driven transects, standardised to sightings per 100 km, have been used for numbats for many years and are the main indicator used at Dryandra Woodland to monitor changes in numbat populations (Figure 1.3). At Dryandra Woodland, indices are obtained by driving a 61 km standard route (established in 1989) six times within 4 to 5 days at a constant speed of 15 to 20 km h^{-1}. Surveys are conducted on non-rainy days, during hours of peak numbat activity (pers. comm. Tony Friend). The resulting index is compared to that of previous years to detect trends in numbat populations (Figure 1.3). Population increases after 1982 and 1989 result from the commencement of fox-baiting programs (Friend and Thomas 1994a). Population decline thereafter may be due to an increase in predation from cats (Department of Parks and Wildlife 2015).

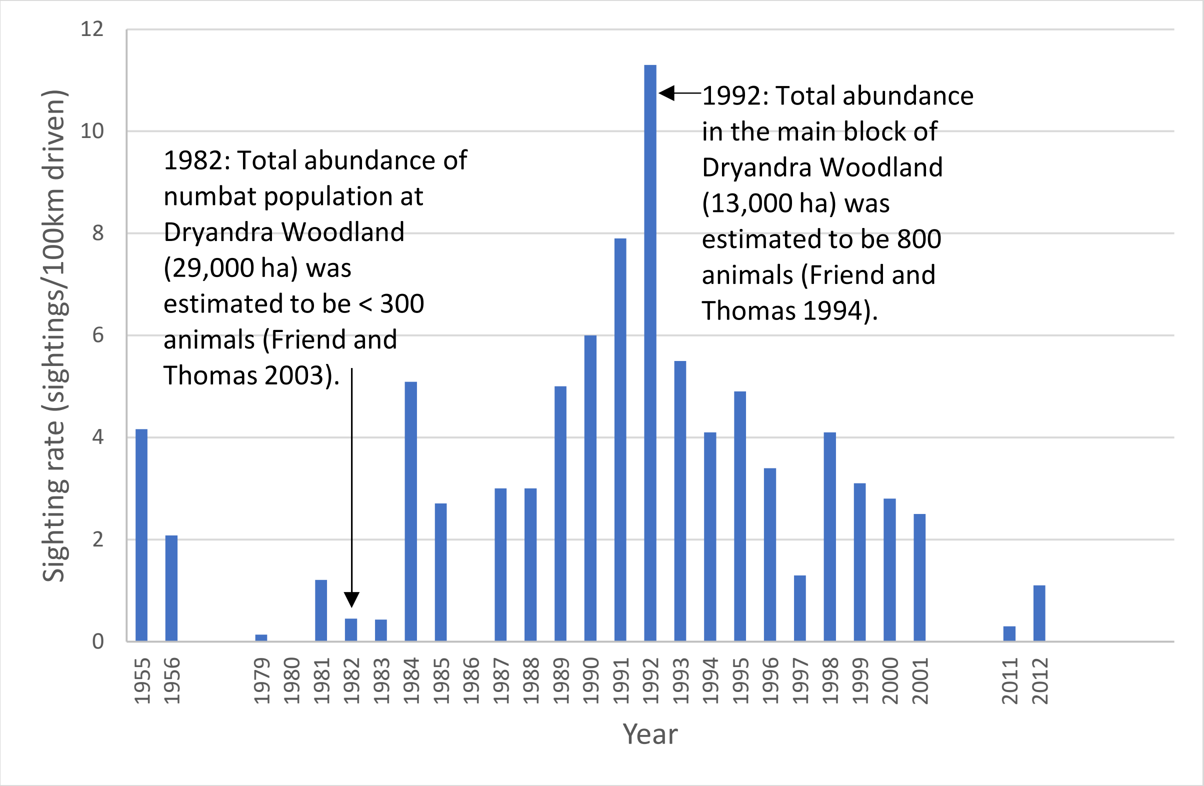

Figure 1.3. Numbat sighting rates per 100 km of driven transect at Dryandra Woodland, Western Australia. Graph adapted from Friend and Thomas (2003), with additional data (2011, 2012) from Department of Parks and Wildlife (2015). Route length (384 km) standardised since 1989 (Friend and Thomas 2003). Route length varied (200–1103 km) in earlier years (Friend and Thomas 1994a). Fox baiting commenced in part of Dryandra in 1982 and in all Dryandra in 1989 (Friend 1994).

For the numbat population at Dryandra Woodland, total abundance estimates of 300 animals in 1982 and 800 animals in 1992 were made by Friend and Thomas (2003) and Friend and Thomas (1994a), respectively. Methodologies for these estimates were not provided except for a mention of extrapolation of unpublished line transect data for the 1992 estimate (Friend and Thomas 1994a).

RAIs from driven transects were also obtained for the numbat population in the UWR and the reintroduction site Boyagin Nature Reserve. Indices from the UWR were very low, with 0.3 sightings and 1.45 sightings per 100 km in 1995 and 1996, respectively (Department of Parks and Wildlife 2015). Driven transects conducted in 2005 and 2014 returned with no sightings (pers. comm. Julia Wayne); however, numerous opportunistic sightings and records from camera traps show that the species persisted within the UWR (Figure 1.4). Low sighting rates during driven transects at the UWR may result from dense jarrah forest vegetation restricting visibility and, therefore, numbat detectability (Department of Parks and Wildlife 2015). The wandoo woodlands at Boyagin Nature Reserve, similar to Dryandra, have a more open understorey, making numbats more detectable (Friend and Thomas 2003). Driven transect indices from Boyagin remained relatively constant between approximately 1.0 and 3.0 sightings per 100 km from 1994 to 2000 except with zero sightings in 1998 and 1999 (Friend and Thomas 2003). The zero sightings were, however, not reflected in indices derived from searches for numbat signs conducted at Boyagin at the same time (Friend and Thomas 2003), which may indicate that indices from searches for signs may be a more sensitive indicator of numbat abundance than driven transects. One limitation of driven transects is the requirement for the observer and the numbat to be at the same place at the same time, while searches for signs (see Table 1.1 and Section 1.4.4) may detect animal signs that can be several days old. This extended period of time may increase detection probabilities for sign searches, making them a detection method worthy of investigation for numbats in the UWR.

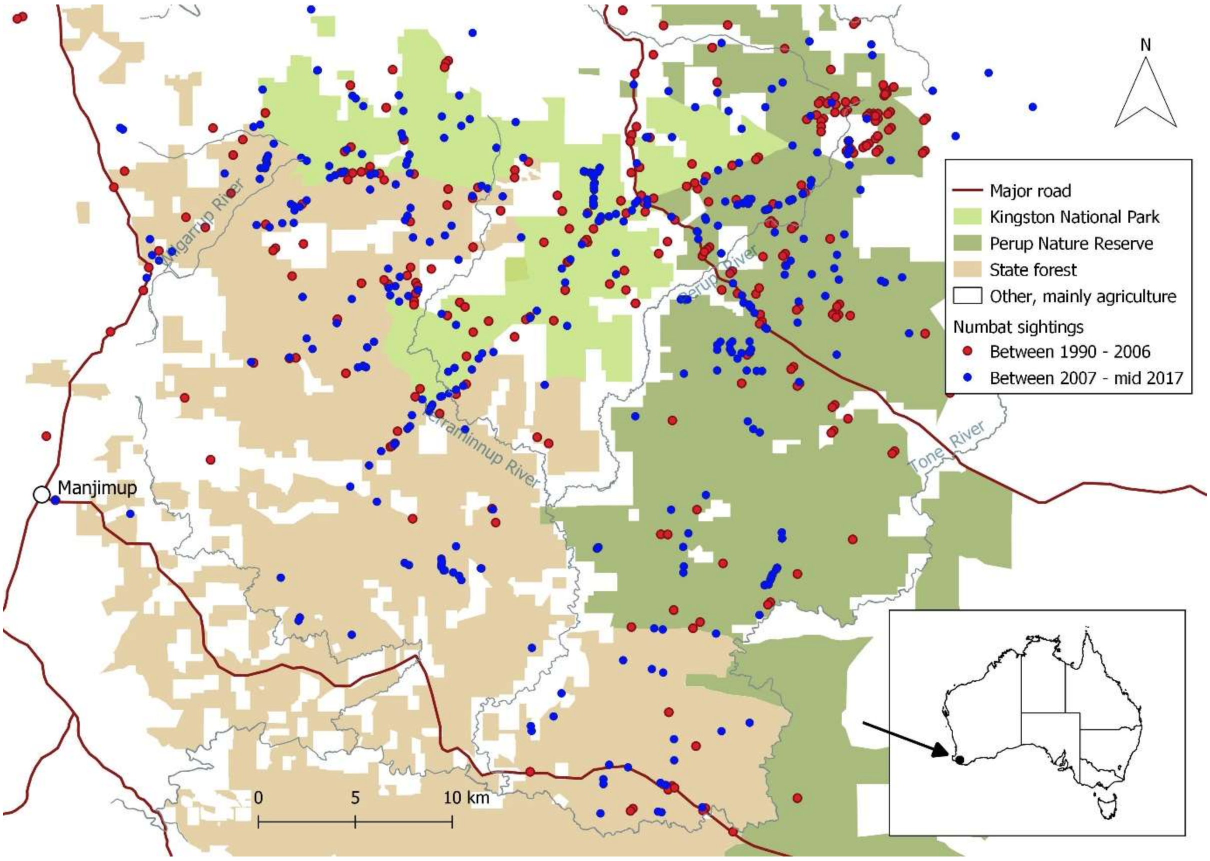

Figure 1.4. Sighting records of numbats in the Upper Warren region and surrounding areas (1990–mid-2017), Western Australia. Data courtesy of the Department of Biodiversity, Conservation and Attractions, Manjimup, Western Australia.

1.4.4 Relative abundance indices from animal signs

Species presence within an area can be determined not only from direct sightings but also from animal signs, including scats, tracks, runways, feeding signs, nests and burrows (Sutherland 2006, Chapter 10). The proportion of plots found with animal signs may then be used as an RAI. A track index was produced by Hayward *et al.* (2015) for numbats at Scotia Sanctuary. The method was loosely explained as 'dusting a series of dirt tracks for several kilometres in the early morning' (p. 260) and counting tracks over 4 days in the afternoon (Hayward *et al.* 2015). The authors acknowledge this method is limited as it ignores detectability. Patterns from the track index were not reflected by sighting rates from driven transects conducted during the same years: between 2011 and 2012, sighting rates from driven transects decreased (from approximately 17 to 5 sightings per 100 km), while the track index increased (from approximately 4.8 to 5.5 tracks per 1 km; Hayward *et al.* 2015).

Searches for scats and diggings have been conducted for numbat sub-populations – for example, at Boyagin Nature Reserve (Friend and Thomas 2003). Within the eastern part of Boyagin Nature Reserve, 44 marked sites were established along tracks, of which an average of approximately 70% were found to have numbat signs during annual surveys between 1989 and 2000 (Friend and Thomas 2003). At the UWR, searches for signs were conducted in 2015 and 2016, and numbat signs were found in 34% (Oct 2015, 49 sites), 24% (Sep 2016, 50 sites), and 36% (Dec 2016, 50 sites) of the sites (Julia Wayne, unpublished data). Since juvenile numbats leave their nests for extended excursions from mid-October and permanently disperse from November (Friend and Burrows 1983), sign searches during these months may yield higher numbat sign detections.

Animal sign-detection rates can also be increased by using detection dogs. Scat-detection dogs were used successfully to detect the presence of elusive species such as the bush dog (*Speothos venaticus*), Franklin's ground squirrel (*Poliocitellus franklinii*), puma (*Puma concolor*) and giant anteater (*Myrmecophaga tridactyla*) (Dematteo *et al.* 2009; Duggan *et al.* 2011; Vynne *et al.* 2011). Dogs were shown to reliably detect target animal signs at consistent rates over several years (Vynne *et al.* 2011), for multiple species (Long *et al.* 2007) and across a diversity of habitats (Leigh *et al.* 2015). Dog training, efficiency testing, general handling and care, and the employment of experienced dog handlers are costly

(Duggan *et al.* 2011), but if sufficient funds are available, scat-detection dogs may provide a means to increase detection of numbat signs and may therefore lead to more reliable data. Due to fund and time restrictions, the use of detection dogs was not explored during this PhD project.

1.4.5 Improving relative abundance indices

To improve the validity of RAIs, covariate information of factors that affect detectability should be collected and included in models for data analysis (MacKenzie and Kendall 2002). Covariates relating to numbat detectability include time of day and year, observer experience in spotting numbats and vegetation structure. Such covariates can be integrated for model adjustment to make detectability constant over space and time (Pollock *et al.* 2002). Yet, improving RAIs by model adjustment is limited. Factors such as vegetation density may affect detectability but also relate to numbat abundance, and it would therefore be inappropriate to be used for such model adjustment (MacKenzie *et al.* 2005a, p. 10). Furthermore, it needs to be acknowledged that not all factors affecting detectability can be recorded or even perceived (Pollock *et al.* 2002), and that RAIs may still be biased unless calibrated with robust measures of population parameters.

1.4.6 Population occupancy modelling

Similar to the percentage index used by Friend and Thomas (2003) at Boyagin Nature Reserve, occupancy describes the fraction of locations where a species was detected (MacKenzie and Royle 2005). The important difference between a percent index and occupancy modelling is that occupancy modelling accounts for the probability of detection of the target species, making it a more robust method (MacKenzie *et al.* 2005b). When the probability of detection is accounted for, occupied sites include the fraction of locations where the target species was detected, as well as an estimate for the sites where the species remained undetected but is likely to be present (MacKenzie *et al.* 2005a, p. 86). Estimates of the fraction of a landscape occupied are often closely related to population abundance, especially when animals (such as numbats) are territorial (MacKenzie *et al.* 2005b).

Occupancy modelling has been applied as an index of abundance (Diefenbach *et al.* 1994), to determine species habitat selection (Cotner and Schooley 2011) and to establish species distribution (Davies *et al.* 2017). The sampling approach (determining species presence) for occupancy estimates is versatile and can be applied to a wide range of species, including elusive species occurring at low densities (MacKenzie *et al.* 2004). Information for occupancy estimates and related detection probabilities can be collected by either temporal or spatial replication (Hines *et al.* 2010). Temporal replication refers to multiple surveys during which sampling units are visited multiple times over a short period of time, while spatial replication can be conducted during a single survey by visiting multiple sites selected from each sampling unit (MacKenzie *et al.* 2005a, pp. 67-70). As well as detecting target species through indirect methods, such as searches for animal signs (Stanley and Royle 2005), detections by direct observations are suitable either by a person or the use of cameras (MacKenzie *et al.* 2005a, p. 26). To increase detections in elusive species, a combination of methods can be used (MacKenzie *et al.* 2005a). For numbats, a combination of detections from searches for signs and camera traps (discussed in Section 1.4.8) could be suitable.

1.4.7 Absolute abundance estimates from driven transects

Absolute abundance estimates from driven transects can be derived from distance sampling methods (Sutherland 2006, p. 141). Distance sampling involves measuring perpendicular distances from animals to transects. It is assumed that all animals present on the transect are detected, while detection probability decreases with increasing distance of the animal from the observer (Buckland *et al.* 2015, pp. 10-13). Other assumptions include that transects are positioned at random in relation to animal distribution, that detections of individuals are independent, and that animal density is constant within the sampled area surrounding the transect (Buckland *et al.* 2015, pp. 12-13). A sample size of 10 to 20 replicate lines (Buckland *et al.* 2001, p. 232) and sightings of 60 to 80 animals is recommended in order to obtain robust population estimates (Buckland *et al.* 2001, pp. 240-241).

In a study of large vertebrate species, de Thoisy *et al.* (2008) demonstrated that reliable population estimates, through distance sampling, can be obtained using driven transects with moderate sampling effort (40–90 km transect length) when target species are sufficiently abundant and detectable. Estimates become unreliable, however, when detection rates are insufficient, in which case, more sampling effort needs to be implemented to improve reliability (Seddon *et al.* 2003).

To acquire sufficient sighting rates for numbats, distance sampling requires considerably long distances to be sampled. Numbats occur at low densities and may easily be overlooked because of their small size and possible concealment by vegetation, fallen trees and undulating ground surfaces. Furthermore, numbats are likely to hide in log hollows or burrows when vehicles approach. Distance sampling has been attempted, however, by Hayward *et al.* (2015) at Scotia and Yookamurra Sanctuaries. The authors established 77 transects on existing tracks, an amount considerably higher than the 10 to 20 replicates recommended by Buckland *et al.* (2001), p. 232. However, non-random placement along existing tracks may introduce substantial bias – for example, by animals avoiding tracks (Buckland *et al.* 2015, p. 37). Hayward *et al.* (2015) acknowledged violation of assumptions and explained that it is difficult to derive robust abundance and/or density estimates from distance sampling methods for numbats. If animals avoid road features and are therefore mostly detected away from transects, detection probability will be overestimated, which leads to an underestimation of density and relating abundance (Marques *et al.* 2013). Including auxiliary information describing numbat distribution in relation to tracks, from radio-collared animals, could rectify such bias (see Marques *et al.* 2013). Such data could have possibly been used by Hayward *et al.* (2015) as there were 13 numbats radio-tracked for home-range analysis within the Scotia Sanctuary during the study.

To derive a reliable estimation of detection function, needed for distance sampling, the detection of 60 to 80 animals is recommended (Buckland *et al.* 2001, pp. 240-241). Typically, detection rates of numbats during driven surveys are low (Friend and Thomas 2003). In producing an RAI by driven transects, Hayward *et al.* (2015) reported daily sighting rates of five to 18 animals per 100 km at Scotia Sanctuary. For absolute abundance estimates from distance sampling, however, the authors failed to provide the number of animals sighted; nor did they provide a confidence interval for population estimates (as seen in Seddon *et al.* 2003; Marques *et al.* 2013). It is therefore difficult to approximate the reliability of results. If numbat sighting rates from driven transects in the UWR do not increase, compared to past results (discussed above), distance sampling is not considered to be a viable survey method for this region.

1.4.8 Camera traps as survey tools

Wildlife detecting cameras (camera traps) have become an increasingly popular tool in wildlife studies since the 1990s (Burton *et al.* 2015). Today's camera traps are typically equipped with a passive infra-red (PIR) sensor that triggers the camera when objects (e.g. animals) with different surface temperatures from the background move within the detection zone (Welbourne *et al.* 2016). Depending on model specifications, camera traps can be set to record one or multiple images per trigger, videos (Scheibe *et al.* 2008) and time-lapse sequences (Weinstein 2015) and are therefore versatile in their application.

Camera traps have been used widely in animal studies, including research on species presence and richness (Pittet and Bennett 2014), animal behaviour (Story *et al.* 2014), estimation of home-range size (Gil-Sanchez *et al.* 2011) and determination of population parameters such as occupancy (Keane *et al.* 2012). Even though camera traps were originally developed for hunters targeting larger animals (Meek and Pittet 2014), technological advances, permitting the detection of smaller heat signatures (Swann and Perkins 2014), have made camera traps successful in detecting smaller mammals (Glen *et al.* 2013; McCleery *et al.* 2014; Nelson *et al.* 2014). Images from camera traps are not only useful for recording the presence of animal species at a certain place and time, they may also allow identification of individuals, when specific characteristics can be recognised (Karanth *et al.* 2006).

Camera traps can be left in the field unattended for extended periods of time (Meek and Pittet 2014) and are therefore suitable to study elusive species in difficult and remote habitat (Trolliet *et al.* 2014). Camera traps are considered to be non-invasive, causing minimal disturbance to the animal and environment (Gillespie *et al*. 2015). Compared with animal signs, species are easily identifiable on pictures, especially diurnal species such as numbats. Cameras can produce large amounts of data, and analysis can be time consuming. The sorting of images, however, requires limited experience and can be accomplished, for example, by volunteers using programs, such as the citizen science web portal Zooniverse (https://www.zooniverse.org/). Software that allows the identification of wildlife in camera trap images could also be used to simplify the image-sorting process (Falzon *et al.* 2020). Even though initial investment for camera trap surveys is high, in the long run, monitoring programs using camera traps have been shown to be cost effective (Welbourne *et al.* 2015).

Even though camera traps have many advantages, drawbacks must not be overlooked when designing camera surveys. Camera trap PIR sensors are pyroelectric devices that respond to changes in infrared energy – for example, when an object moves in front of a background with a different heat signature (Meek *et al.* 2014b). The camera may not trigger (miss animals) when the temperature difference between animal and background is low (Lerone *et al.* 2015), which may stay unnoticed in field studies. Cameras may also affect detection probabilities by attracting or repelling individual animals that notice the camera(s) (Meek *et al.* 2016a). Furthermore, technical difficulties may result in camera failure and loss of valuable data (Cutler and Swann 1999).

In a review of camera trap studies, animal detection rates were found to increase when camera studies were designed specifically for target species (Anile and Devillard 2016). A target species-specific camera trap study design includes, for example, that camera traps are set up at optimal heights depending on the size of the target species. In the UWR, numbats have been detected with the use of camera traps opportunistically (Wayne *et al.* 2013b) and during preliminary trials (pers. comm. Julia Wayne). However, there have never been any studies dedicated to optimising the use of cameras for this species. Thus, although cameras show promise as a detection tool, it is important to first optimise camera set up for numbats during trial studies to increase detection rates. Results from camera trapping surveys could be used as RAIs (detections per trap effort; Rovero and Marshall 2009) or for occupancy modelling (MacKenzie *et al.* 2005a, p. 26).

In addition, with a suitable camera set-up, it may be possible to distinguish numbat individuals by their characteristic stripe patterns. Individual animal recognition may allow the use of capture–recapture methods to estimate population abundance (Simcharoen *et al.* 2007). For camera traps in general, it is recommended to aim the camera's integrated PIR motion detector horizontally at the centre of the target species' body mass (Meek *et al.*

2014b). Such positioning seldom produces images of the animal's back, making individual recognition impossible for numbats. Positioning cameras at an elevated height (e.g. 2 m), angled slightly downwards towards the animal, may allow image capture depicting the numbat's back, providing increased identification opportunities. However, such a position is not optimal for the camera's integrated PIR sensor, which operates best horizontally near the ground (Meek *et al.* 2016b). For individual numbat identification, a special camera set-up, with PIR sensor near the ground and elevated camera position, would be required.

1.4.9 Absolute abundance estimates from capture–recapture

Capture–recapture methods arose as early as 1930 (Lincoln 1930) and have gained momentum since the 1950s (Norman 1951; Leslie 1952; Gulland 1955; Darroch 1958). Commonly, capture–recapture methods involve sampling an animal population, marking and releasing captured individuals and estimating abundance from information derived through recapture of marked individuals in subsequent samples (Otis *et al.* 1978; Efford 2004). Since the emergence of camera traps, capture–recapture methods have been applied to animals with natural, individual characteristic markings (e.g. pelage patterns, scale patterns, scars, etc.), which can be recognised on images (Karanth 1995; Noss *et al.* 2004; Zheng *et al.* 2016).

A problem frequently occurring with converting capture–recapture abundance estimates to density estimates is the estimation of effective trapping area: how far do captured individuals roam beyond the sampled area (Foster and Harmsen 2012)? The use of buffer areas around sampling areas may be used to estimate density. However, the size of the buffer can greatly influence density estimates and, therefore, needs to be estimated correctly (Efford 2004). The use of telemetry data from home-range studies can help with estimating the size of buffer strips but is often not available (Foster and Harmsen 2012).

Spatially explicit capture–recapture (SECR) is a recent development in capture–recapture methods (Efford 2004; Royle *et al.* 2014). SECR solves the effective trapping area problem by linking the individual animal's encounter location to its home range (Efford and Fewster 2013). This advance, together with rigorous accounting for capture probabilities, makes

SECR a robust population density estimation tool when general assumptions (discussed in Lindberg 2012) are met and sample size is sufficient (Foster and Harmsen 2012).

Efford *et al.* (2009) showed, by use of empirical data, that SECR methods can be feasible for relatively small data sets (total of 20 individuals, with 10 recaptures of seven individuals). Precision, however, increases with the number of recaptures and is recommended to approximately 20 (Efford *et al.* 2009). SECR has not yet been tested as a method for estimating numbat density. If a large enough numbat sample size cannot be achieved to warrant robust SECR estimates, an alternative is to report the minimum number of animals known to be alive and possibly an approximate estimate of density derived from the area sampled plus buffer area and the minimum number of animals known to be alive (see e.g. Lynam *et al.* 2007).

1.5 Aims and scope

This study was conducted in the UWR, approximately 300 km south of Perth, WA. The study area stretched over open sclerophyll forests and woodlands that cover more than 140 000 ha of publicly managed land (study area maps are provided in Chapters 3, 4 and 6).

The main aim of this research project was to develop robust numbat detection methods to estimate population parameters in order to facilitate monitoring population trends over space and time. Then, with the use of the developed detection methods, the aim was to improve the understanding of the numbat’s habitat requirements in the UWR, especially those relating to prescribed fuel-reduction burns, timber harvesting and control of introduced foxes using 1080-poisoned bait at differing intensities. More specific aims are presented within each data chapter outlining the goals of individual study components.

1.6 Adaptive approach

This PhD project took an adaptive approach, where initial results informed subsequent steps and final survey design (Figure 1.5). Live trapping was not explored as there are no known baits or lures to attract numbats to traps (Burrows and Christensen 2002). The first step was to optimise camera traps for numbat detection. Camera trap trials took place in numbat enclosures at Perth Zoo and later in the field to optimise camera trap set-up for numbat detection. Survey methods (camera trapping, sign surveys and driven transects) were compared to determine which survey method would be most efficient and cost effective for numbat detection in the UWR. Finally, the most successful survey method was used to determine if factors such as timber harvesting, time since fire and introduced predator control intensity affected numbat population parameters in the UWR.

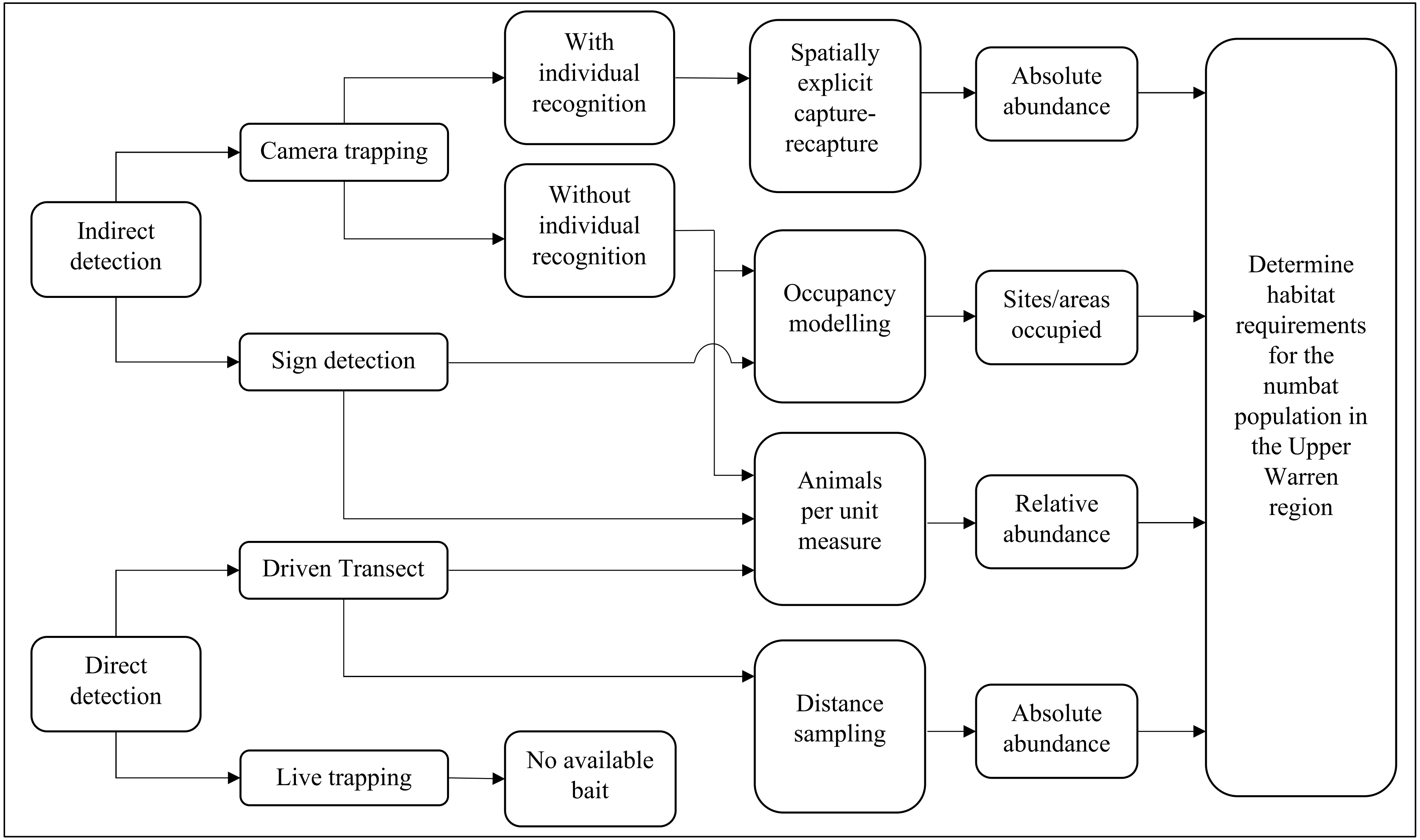

Figure 1.5. Overview of possible pathways for an adaptive approach to develop survey methods and determine factors that may affect the numbat population at the Upper Warren region near Manjimup, Western Australia.

1.7 Thesis overview

Following the adaptive approach of this thesis, chapters are presented in chronological order. Some repetition of information between chapters was unavoidable because four chapters (Chapters 2, 3, 4, 6) were prepared as standalone manuscripts for publication in scientific journals. There are also some stylistic differences between these chapters as they were prepared according to the styles of respective journals. For example, the abstract of Chapter 3 was divided into sections (context, aims, methods, key results, conclusion, implications) to follow the style of the journal *Wildlife Research*, while the others were not. While the content of these thesis chapters stayed the same as the versions to be published, some differences were unavoidable to follow the formatting of this thesis (e.g. table and figure numbers differ).

Chapter 2 describes three camera trap optimisation trials that were conducted with numbats in zoo enclosures. These trials helped to determine the appropriate camera trap height above ground for numbat detection and tested different camera trap models to increase numbat detection rates. Furthermore, it was determined if elevated, downward-angled time-lapse cameras were suitable for obtaining images of the numbat’s dorsal pelage patterns for individual numbat identification.

Chapter 3 details the first field study that compared the efficiency and effectiveness of three numbat detection methods: driven transects, sign surveys and camera trap surveys. This study helped to determine the most useful detection method that could be applied to the numbat population in the UWR and facilitate investigation of numbat habitat preferences as well as the long-term monitoring of the species.

Chapter 4 describes a field survey that was conducted to improve numbat detection rates from camera traps. Detection rates were compared from camera traps in stationary positions to those that were repositioned within sites on a weekly basis. Detection rates from standard camera traps (40° detection angle) and wide-angle camera traps (110° detection angle) were also compared.

Chapter 5 helped to determine if numbat sign surveys conducted in autumn were as effective as those conducted in spring (in spring the numbat population is assumed to peak as juvenile numbats disperse). Results from the autumn sign survey were then compared to the best-performing camera trap from Chapter 4. This led to the choice of methods used for the final field study described in Chapter 6.

Chapter 6 explored numbat habitat preferences, particularly those related to the forest management practices conducted in the UWR: prescribed fuel-reduction burns, timber harvesting and the control of foxes using 1080-poisoned bait at differing intensities. This chapter helped to develop knowledge critical for the protection and management of the numbat in the UWR.

Chapter 7 is a general discussion of the findings of the previous five data chapters and considers these findings in the context of the aims of this thesis. Also discussed are the limitations of the methods found most useful for numbat detection in the UWR, the implications for the species' management and protection, and suggestions for future research that may reduce those limitations.

Chapter 2 Optimising camera trap height and model increases detection and individual identification rates for a small mammal, the numbat (*Myrmecobius fasciatus*)

This chapter has been published:

Seidlitz, A., Bryant, K. A., Armstrong, N. J., Calver, M., and Wayne, A. F. (2020). Optimising camera trap height and model increases detection and individual identification rates for a small mammal, the numbat (*Myrmecobius fasciatus*). *Australian Mammalogy*. doi: https://doi.org/10.1071/AM20020.

2.1 Preface

This chapter comprises original research undertaken to optimise camera traps for numbat detection prior to field studies. As numbats are difficult to detect in the wild, the following trials were conducted at Perth Zoo, Western Australia where there is a breeding colony of numbats for reintroduction purposes. The adaptive approach of this PhD study began with these first trials. Initially, two trials were planned: (1) Optimising camera trap height above ground for numbat detection, and (2) determining if elevated, angled cameras can provide images suitable for the identification of numbat individuals. After the first trial, it became apparent that the camera trap model used missed a substantial amount of numbat detections. To rectify this, a different camera trap model was sourced and tested to increase numbat detection rates.

2.2 Abstract

Camera traps are widely used to collect data for wildlife management, but species-specific testing is crucial. We conducted three trials to optimise camera traps for detecting numbats (*Myrmecobius fasciatus*), a 500 – 700-g mammal. We compared detection rates from (1) Reconyx PC900 camera traps installed at heights ranging from 10 – 45 cm, and (2) Reconyx PC900, Swift 3C standard and wide-angle camera traps with differing detection zone widths. Finally, we compared elevated, downward-angled time-lapse cameras installed at heights ranging from 1 – 2 m to obtain dorsal images for individual numbat identification. Camera traps set at 25 cm had the highest detection rates but missed 40% of known events. During model comparison, Swift 3C wide-angle camera traps recorded 89%, Swift 3C standard 51%, and Reconyx PC900 37% of known events. The number of suitable images from elevated, downward-angled cameras, depicting dorsal fur patterns, increased with increasing camera height. The use of well-regarded camera trap brands and generic recommendations for set-up techniques cannot replace rigorous, species-specific testing. For numbat detection, we recommend the Swift 3C wide-angle model installed at 25-cm height. For individual numbat identification, elevated, downward-angled time-lapse cameras were useful; however, more research is needed to optimise this technique.

2.3 Introduction

Camera traps are increasingly used to detect and monitor wildlife for conservation management (Rowcliffe and Carbone 2008; Burton *et al.* 2015; Meek *et al.* 2015b) so it is important to position cameras carefully to obtain robust data and to choose the most effective camera model (Meek and Vernes 2016; Apps and McNutt 2018a; Jacobs and Ausband 2018). Motion-activated camera traps with passive infrared (PIR) sensors trigger when objects (e.g. animals) with a different surface temperature from the background move within the detection zone (Welbourne *et al.* 2016). Missed detections (where animals are present but not detected) and missed opportunities to identify individual animals should be minimised. This is best achieved by species-specific optimisation of camera trap models and set-up techniques (e.g. Glen *et al.* 2013; Taylor *et al.* 2014; Meek and Vernes 2016).

The height of camera traps above the ground can influence animal detection rates. The effects of large differences in camera trap height have been investigated in several studies. Camera traps in lower positions were found to be more effective by Meek *et al.* (2016b), who investigated instalment heights of 90 cm and 350 cm (targeting wild dogs, foxes and feral cats), and by Swann *et al.* (2004), testing camera traps at 20 cm and 120 cm (targeting warm water bottles and a human mimicking animal movement). While Newey *et al.* (2015) had better detection rates from camera traps in higher positions (120 cm versus 60 cm) when targeting sheep, Jacobs and Ausband (2018) had inconclusive results from camera traps at 60 cm and 300 cm targeting a variety of species native to Idaho, USA. In these studies, at least one of the compared camera heights substantially exceeded 50 cm, making results inconclusive for small species. The only study that tested camera-trap heights in small increments between 20 cm and 122 cm was conducted by Apps and McNutt (2018a). However, a large dog (68-cm shoulder height) was used to trigger camera-traps, making results again uninformative for small species.

As well as height, the choice of camera model is important. Detection rates may be affected by technological differences, such as trigger speed (time between animal detection and image recording) (Swann *et al.* 2004; Wellington *et al.* 2014; Meek *et al.* 2015a) and width of the PIR sensor detection zone (Meek *et al.* 2014b; Wellington *et al.* 2014; Fancourt *et al.* 2018). When comparing Reconyx PC900 camera traps (40° PIR detection zone, 0.2-s trigger speed) with Ltl Acorn Ltl-5310A wide-angle camera traps (120° PIR detection zone, 0.8-s trigger speed), Fancourt *et al.* (2018) found that the wide-angle camera trap models had higher detection rates for rabbits. This demonstrates that it is important to consider the combination of model specifications (e.g. width of the sensor detection zone, and trigger speed) together, rather than separately. Other model differences that may affect animal detection include the camera's software, pyroelectric sensor, and type and arrangement of Fresnel lenses (Welbourne *et al.* 2016; Apps and McNutt 2018b).

Camera traps may also be useful in identifying individual animals for species with unique markings, such as tigers, leopards, hyenas and Australian dasyurids (Karanth 1995; Jackson *et al.* 2006; Hohnen *et al.* 2013; Tichon *et al.* 2017; Rowland *et al.* 2020). Identifying individuals allows the use of capture–recapture modelling to obtain robust population size estimates (Royle *et al.* 2014). To obtain images for individual animal identification, camera traps are commonly set up horizontally with the camera facing parallel to the ground (Karanth 2017). Images from camera traps set up horizontally typically show characteristic markings from animal flanks or heads. But some species have characteristic markings only on their backs. A vertical camera trap set-up with the camera pointing vertically downward from 1.5 – 2 m above ground was found useful for individual recognition of skunks using dorsal fur patterns (Theimer *et al.* 2017). However, a vertical set-up reduces the field of view and detection zone of PIR-triggered camera traps (Smith and Coulson 2012) and therefore decreases detection rates (Taylor *et al.* 2014; Nichols *et al.* 2017). To overcome these limitations, typically baits are used to attract target species to a vertical camera station (De Bondi *et al.* 2010; Smith and Coulson 2012; Taylor *et al.* 2014; Nichols *et al.* 2017). However, baiting may be undesirable (e.g. if it biases study results: McCoy *et al.* 2011; Stewart *et al.* 2019) or impossible (e.g. when species are not attracted by bait: Burrows and Christensen 2002). In such cases, a different approach could be useful to overcome the field of view and detection zone restrictions experienced in a vertical camera set-up. One solution worthy of testing is to use elevated cameras (e.g. at 2 m height) with a moderate downward angle (e.g. 45°) to obtain images for individual animal identification of a species with characteristic markings on their backs.

Questions of the optimum height, appropriate camera trap model, and opportunities for individual identification were all prominent in developing monitoring strategies for the endangered Australian endemic marsupial, the numbat (*Myrmecobius fasciatus*), in one of its two remaining natural populations – the Upper Warren region in south-west Western Australia. Camera traps may detect elusive animals occurring in low densities (Kucera and Barrett 2011) but have not been tested for their efficiency for numbats. Since camera trap optimisation before field studies can reduce missed detections (Taylor *et al.* 2014), we designed this study to optimise camera traps with the use of captive numbats in zoo enclosures.

We had three aims. First, we aimed to determine the optimal PIR sensor height for camera traps to maximise detection rates for numbats, which have a body-core height of ~10 – 15 cm. We used the widely available Reconyx PC900 camera trap model (Glover-Kapfer *et al.*

2019) commonly used by researchers and government agencies in Australia (Meek *et al.* 2015b). Second, we aimed to determine whether detection rates from Reconyx PC900 camera traps (42° PIR detection zone) can be increased by using two models with wider detection zones (the Swift 3C standard and wide-angle; 52° and 110° PIR detection zone respectively). Third, we aimed to establish whether elevated, downward-angled cameras can be used to obtain suitable images for individual numbat identification from dorsal patterns, and to test which camera height (1 m, 1.25 m, 1.5 m, 1.75 m, or 2 m above ground) would be most appropriate. Because the main interest regarded image suitability and camera height for individual numbat identification, we used cameras set to time-lapse mode for this part of the study to avoid problems faced with restricted detection zones from downward-angled PIR motion sensors (see Apps and McNutt 2018a). We are unaware of any other publication that evaluated optimal camera trap height for a small mammal such as the numbat, that compared detection rates from Reconyx PC900 and Swift 3C models, or that tested elevated, angled time-lapse cameras for identification of individual animals.

2.4 Materials and methods

2.4.1 Study species

The numbat is a small (500 – 700 g) diurnal marsupial (Cooper 2011) that can be individually recognised by characteristic dorsal pelage patterns. It is difficult to detect numbats because of their solitary lifestyle, large home range (which can exceed 100 ha) (Christensen *et al.* 1984), and elusive nature. Numbats feed exclusively on termites (Calaby 1960b); there are no known lures or baits to attract numbats to traps (Burrows and Christensen 2002) so this remains a topic for investigation. The numbat occurs today in only 1% of its former range, which covered much of the southern half of the Australian continent before European settlement (Friend 1990). It is listed as endangered nationally under Australia's Environment Protection and Biodiversity Conservation Act 1999, and internationally under the IUCN Red List of Threatened Species (Friend and Burbidge 2008). Numbats occur now in several reintroduction sites (see details in Department of Parks and Wildlife 2017), and in two remaining natural populations at Dryandra Woodland and the Upper Warren region, both located in south-west Western Australia. There is a paucity of adequate monitoring information for the Upper Warren region (see Wayne 2018 for area map) due to the lack of effective survey methods to assess population trends or size.

2.4.2 Perth Zoo and enclosures

This study was conducted in a zoo environment to ensure numbat detection rates were sufficiently high for a comparative camera study. Perth Zoo in Western Australia keeps numbats in captivity as part of a breeding program for reintroduction purposes (Mawson and Lambert 2017). Two types of fenced numbat enclosures were used: one L-shaped enclosure (8 m x 11 m plus 4 m x 6 m) and three rectangular enclosures (5 m x 3 m). Each enclosure contained areas of bare ground, native vegetation, nesting and climbing structures, feeding areas, and a single adult numbat. The three trials (PIR motion-sensor height; comparison of camera trap models; time-lapse camera height for identification of individual animals) took place between June and September 2017.

2.4.3 Cameras tested

We tested the following camera models: Reconyx PC900 and HC600 (RECONYX, LLP, Holmen, WI, USA), and the Swift 3C standard and wide-angle model (Outdoor Cameras Australia, Toowoomba, Qld, Australia). We had intended to use only the Reconyx PC900 model, but one HC600 model was provided by error in the delivery of cameras. It was included in the trial because differences between PC900 and HC600 models are predominantly related to software functions, with the camera trap settings required for our trials available in both models. PC900 and HC600 models could therefore be treated as equivalent. The Swift 3C is a distributer-branded, low-cost camera trap. Details on camera specifications and accessories are given in Table 2.1. We chose Reconyx PC900 and HC600 camera traps because they are commonly used by the Department of Biodiversity, Conservation and Attractions (DBCA) for camera trap studies in the Upper Warren region. Swift 3C camera traps were chosen because they are available as standard and wide-angle models. The Reconyx cameras were used in assessments of the ideal height of camera traps for numbat detection and for identifying individual numbats, while both Reconyx and Swift cameras were used to determine differences in detection rates between camera trap models.

Table 2.1 Specifications for camera-trap models and accessories used during this comparative camera trap trial conducted at Perth Zoo, Western Australia.

Camera trap models used have zonal detection areas, resulting from an arrangement of multiple Fresnel lenses in two horizontal bands (refer to Welbourne *et al.* (2016) for more information on camera trap functionality). We used rechargeable Fujitsu LSD (HR-3UTK) batteries and 16 GB Delkin Devices (SD163X) SD cards in all cameras

Specifications	Reconyx PC900/HC600	Swift 3C standard	Swift 3C wide-angle
Trigger speed (s)	0.2	0.35~0.45	0.35~0.45
Lens angle (°)	40	52	100
Detection angle (°)	40	52	110
Image resolution (MP)	3.1^{A}	5^{B}	5^{B}
No. of batteries	12	8	8
Manufacturing dates	2013-14	05/2017	05/2017

^{A} Highest possible, ^{B} Lowest possible.

2.4.4 PIR motion-sensor height

To determine which camera trap height is most suitable for numbat detection, 12 Reconyx camera traps (11 x PC900 and 1 x HC600 models) were deployed in three rectangular numbat enclosures. In each of the three enclosures, four camera traps were stacked vertically, and firmly secured to a wooden plank with their PIR motion-sensors positioned at 10 cm, 25 cm, 40 cm, and 55 cm height above ground (Figure 2.1a). Camera trap lenses were aimed horizontally, facing the 2.5-m-distant central area of the enclosures. Camera traps were set to take three pictures at 1-s intervals per trigger, and with a quiet period of 15 s between triggers. The 15-s quiet period was used to reduce multiple detections of numbats remaining in their favourite areas for an extended time within the detection zone. PIR motion-sensor sensitivity was set to high. Camera traps were set to operate for four days; however, one Reconyx PC900 camera, positioned at 10 cm, failed to operate from the morning of Day 4 for unknown reasons. The Day-4 data of all camera traps in that enclosure were therefore excluded from the analysis.

In addition to the above, a time-lapse camera taking a picture every 2 s, was installed in each enclosure (Reconyx PC900 model set to time-lapse mode with motion detection disabled). The time-lapse cameras were aimed in the same general direction as the camera traps and were installed at 20 cm height. Time-lapse cameras were used to approximate numbat movements in front of cameras that were missed by the camera traps because they take pictures at set intervals independent of animal movements.

Figure 2.1. Camera set-up used during this study conducted in numbat enclosures at Perth Zoo, Western Australia. Trial one and two aimed to determine best (a) PIR motion-sensor height (Reconyx PC900), and (b) camera trap model (from left: Swift 3C standard, Reconyx PC900, Swift 3C wide-angle) for numbat detection. Trial three (c) explored time-lapse camera height for numbat individual identification.

2.4.5 Comparison of camera trap models

To determine differences in detection rates between camera trap models with different detection zone width, three different camera traps (see Table 2.1) were attached side-by-side to a wooden plank ~2 cm apart (Figure 2.1b). Initial camera positions were determined randomly, and, within enclosures, camera traps were rotated daily so that each camera occupied each position twice. The camera traps operated over six days in three rectangular numbat enclosures. Camera traps were aimed horizontally towards the centre of the enclosure with their PIR motion-sensors positioned at ~25 cm above ground. Cameras were set to take three images per trigger with no quiet period between triggers to maximise possible detections. We did not use a 15-s quiet period between triggers because camera trap positions were rotated daily. Therefore, cameras were not advantaged/disadvantaged by constant positioning (e.g. facing to/away from frequently visited enclosure areas). Reconyx PC900 models were set to have a 1-s interval between pictures. This was to approximate the picture recording interval time of Swift 3C models, which do not allow adjustment of this function. PIR motion-sensor sensitivity of all camera traps was set to high.

As with the height trial, a Reconyx PC900 camera in time-lapse mode was installed in each enclosure. The time-lapse cameras, taking an image every 2 s, were positioned centrally, below the camera traps at ~10 cm above ground, and aimed in the same general direction. This time-lapse camera placement was chosen to maximise the field-of-view overlap with the camera traps. A positioning above the camera traps would have caused the time-lapse camera to miss numbats moving close to the camera station.

2.4.6 Time-lapse camera height for identification of individual animals

To determine the most suitable camera height above ground for obtaining dorsal images for identification of individual numbats, five Reconyx PC900 cameras were installed in each of three numbat enclosures (two rectangular and one L-shaped enclosure). Cameras were stacked vertically and attached to wooden poles, at 1 m, 1.25 m, 1.5 m, 1.75 m, and 2 m height (Figure 2.1c). Maximum camera height was limited by enclosure height. Cameras were angled to centre images on a point on the ground, 2 m from the attachment pole. Cameras were set to time-lapse mode, taking one image every 2 s from 0800 to 1600 hours, which is when numbat activity was most likely. Batteries and SD cards were changed daily before 0800 hours. The time-lapse cameras operated over eight days to allow enough time to collect sufficient identifications of individual numbats for comparison.

2.4.7 Data collection and analysis

For the ‘PIR motion-sensor height’ and the ‘Comparison of camera trap models’ trials, camera time and date settings were synchronised during camera set-up and were visible on each image. This allowed direct comparison of images from camera traps and time-lapse cameras. Images were viewed using FastStone Image Viewer 6.2 (FastStone Soft 2019). We compared the number of numbat detections (also referred to as detection rates) from camera traps and time-lapse cameras. We defined a numbat movement in front of cameras as an event, and a detection as an event being recorded, resulting in an image depicting a numbat partially or wholly, by at least one camera trap or time-lapse camera. Using the synchronised time and date stamps on images for reference, each event was recorded as either detected or missed for each camera. Since all detections in each enclosure were of the same animal, we did not use a quiet period between successive detections which is often used in camera trap studies to ensure independence of animal detections (see Meek *et al.* 2014b for detailed definition).

For the third trial, time-lapse images were scanned for images depicting numbats. A numbat movement within the field-of view of time-lapse cameras resulted in sequences of images depicting that numbat. If a sequence of images contained one or more clear images of the numbat's dorsal pelage pattern, and thereby allowing individual numbat identification, then the identification was recorded and counted. Identifications were totalled for each time-lapse camera per day.

Numbat detections or individual identifications from the different trials were analysed using linear regression models in R 3.5.0 (R Core Team 2018). Diagnostic plots for linear regression analysis were inspected to ensure model fit. Models were fitted with the main covariates of interest 'camera trap height', 'camera trap model', and 'time-lapse camera height' for the three trials respectively. Since different numbats in each enclosure may have had different activity levels, and because those activity levels may have changed each day, the covariates 'enclosure' and 'day' were used to improve model fit. Both covariates were treated as fixed to investigate the effects of the different days while the number of enclosures was too small to allow it to be random. Predictive models were compared using the AIC (Akaike Information Criterion: Akaike 1974) function in R, and models with the lowest values were selected for further analysis (Burnham and Anderson 2002, p. 62). Further analysis comprised a Tukey's HSD test to compare all possible pairs of means (Tukey 1949).

2.5 Results

2.5.1 PIR motion-sensor height trial

There were 1869 events where at least one camera trap or time-lapse camera recorded a numbat moving in front of a camera. Most observed events were recorded by cameras at

25 cm above ground (60%), while cameras installed at 55 cm recorded the lowest percentage (16%) of events (Figure 2.2). The time-lapse cameras in each enclosure recorded 95% of observed events. Camera trap height was found to be significantly associated with the number of detections ($F_{14,40} = 18.91$, $P < 0.0001$). The initial model, which included the explanatory variable 'camera height' only, had an AIC of 564.08. The value of the AIC decreased with the addition of 'enclosure' (546.49), 'day' (551.96), 'enclosure+day' (534.22), and 'enclosure*day' (526.20). Therefore, the model with the interaction term was used as the final model. Camera traps installed at 10 cm and 25 cm had significantly higher mean detection rates than those at 40 cm and 55 cm (Figure 2.2).

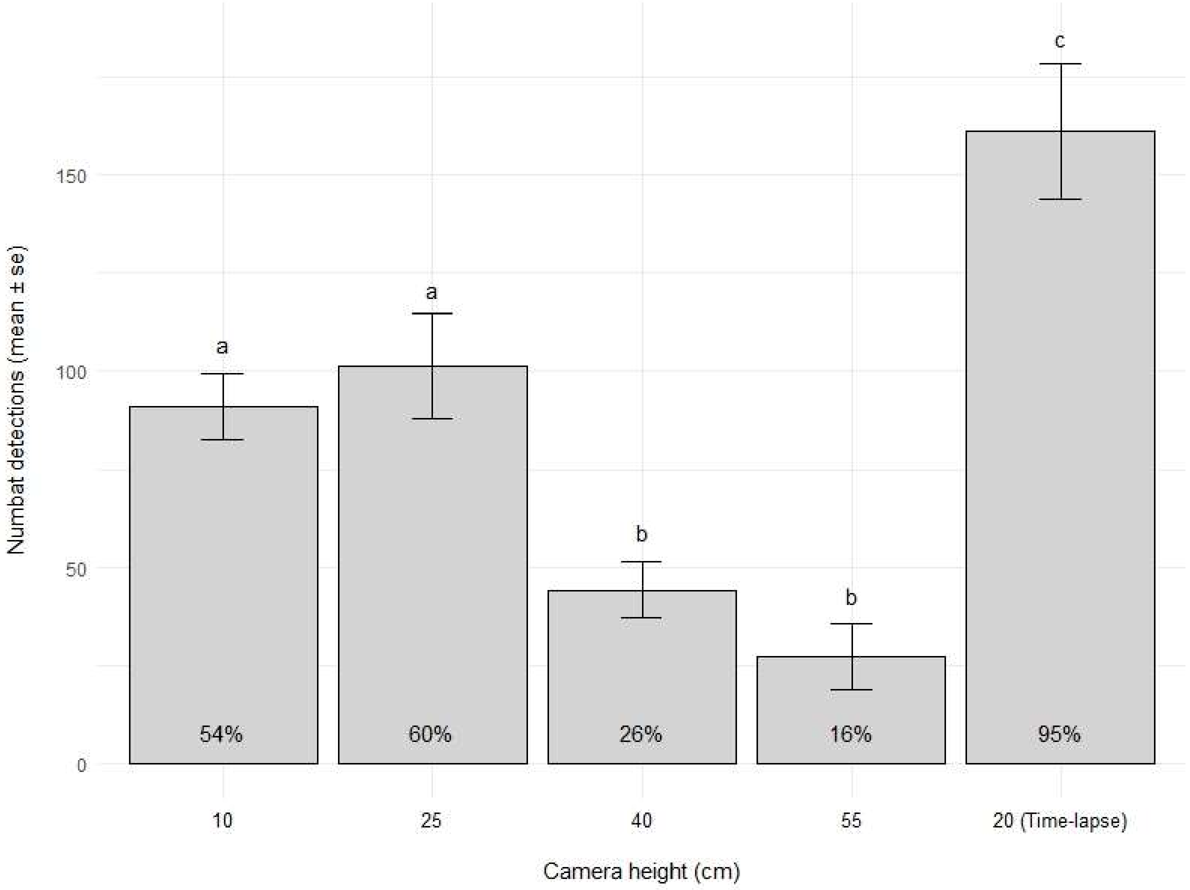

Figure 2.2. Observed mean detections per day, and percentages of total events detected (n = 1869) from Reconyx camera-traps and a time-lapse camera installed at different heights in numbat enclosures at Perth Zoo, Western Australia. Different letters indicate significant difference between groups using Tukey HSD pairwise comparisons (P-values for significant differences were all < 0.001, all other P-values were > 0.5).

2.5.2 Comparison of camera trap models

Cameras recorded a total of 3703 events where at least one camera trap or time-lapse camera recorded a numbat movement. The Swift 3C wide-angle model recorded the highest percentage (89%), and the Reconyx PC900 model the lowest percentage (37%) of observed events (Figure 2.3). Detection rates of time-lapse cameras were lower than those of the Swift 3C wide-angle camera trap model. Camera trap models were significantly associated with the number of detections ($F_{20,51} = 17.53$, $P < 0.0001$). The initial model, which included the explanatory variable ‘camera model’ only, had an AIC of 766.45. The value of the AIC decreased with the addition of ‘enclosure’ (750.49), ‘day’ (753.90), ‘enclosure+day’ (728.93), and ‘enclosure*day’ (691.93). Therefore, the model with the interaction term was used as the final model. Swift 3C wide-angle camera traps had significantly more mean detections than time-lapse cameras, Swift 3C standard camera traps and Reconyx PC900 camera traps (Figure 2.3).

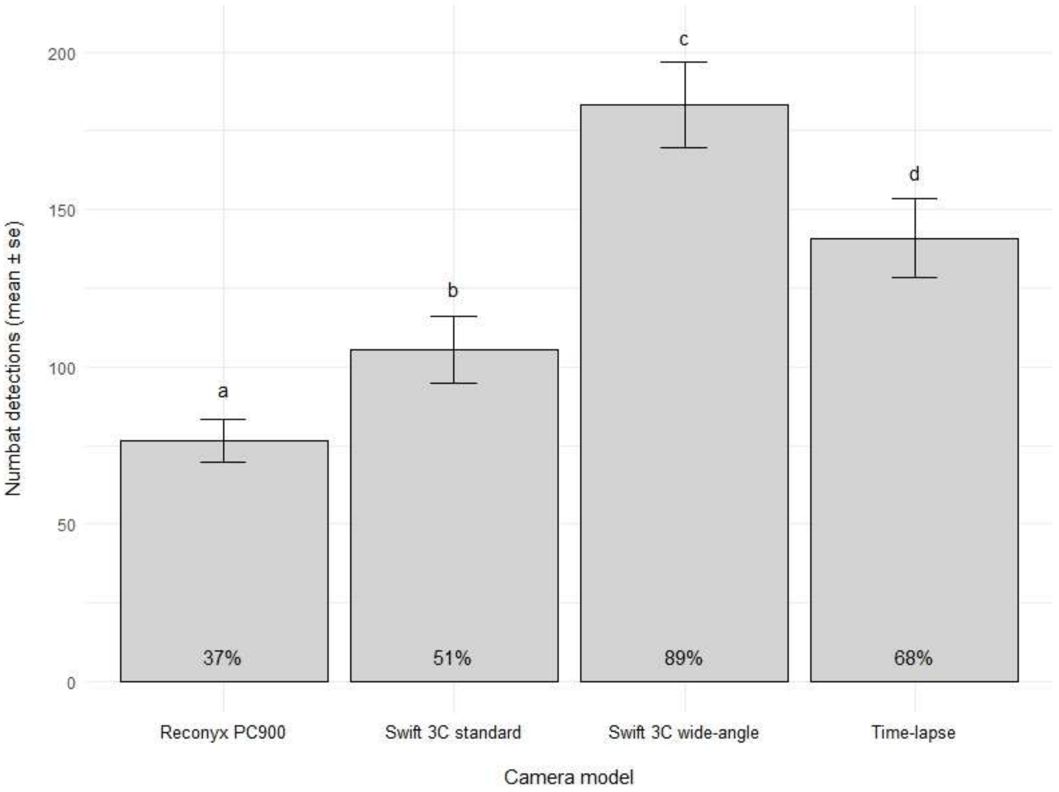

Figure 2.3. Observed mean detections per day, and percentages of total events detected (n = 3703) from a side-by-side comparison of camera-trap models with different detection angles (Reconyx PC900 = 40°, Swift 3C standard = 52°, Swift 3C wide-angle = 110°). Camera traps and a time-lapse camera (Reconyx PC900 model) were installed in numbat enclosures at Perth Zoo, Western Australia. Different letters indicate significant difference between groups using Tukey HSD pairwise comparisons (*P*-values for significant differences were all < 0.01).

2.5.3 Time-lapse camera height for identification of individual animals

Images depicting the numbats' characteristic dorsal pelage patterns (Figure 2.4) allowed a total of 2102 individual identifications. Camera height was found to be significantly associated with the number of individual identifications ($F_{18,56} = 26.14$, $P < 0.0001$). The initial model, which included the explanatory variable 'camera height' only, had an AIC of 587.67. The value of the AIC decreased with the addition of 'enclosure' (582.41), 'day' (541.83), 'enclosure+day' (525.45), and 'enclosure*day' (452.11). As with the other trials, the model with the interaction term was used as the final model. The number of individual numbat identifications differed between camera trap heights, with cameras at 2 m height having significantly more identifications than those at 1 m and 1.25 m height (Figure 2.5).

Figure 2.4. Images show the characteristic dorsal pelage patterns of numbats that allow identification of individuals. Images derived from downward angled time-lapse cameras installed at (a) 1.5 m, and (b) 2.0 m height within numbat enclosures at Perth Zoo, Western Australia.

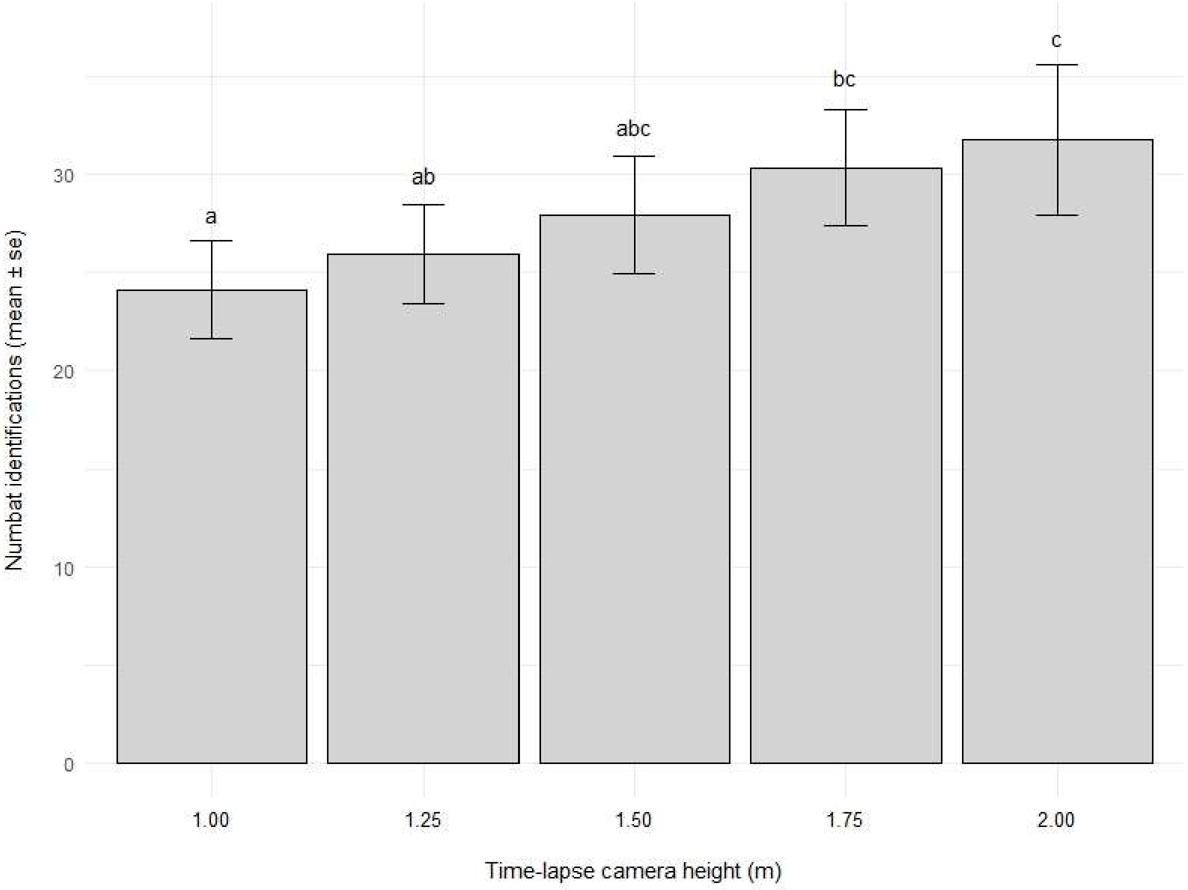

Figure 2.5. Mean numbat identifications per day (with standard error bars) derived from images of time-lapse cameras (Reconyx PC900 model) attached at different heights within numbat enclosures at Perth Zoo, Western Australia. Different letters indicate significant difference between groups using Tukey HSD pairwise comparisons (*P*-values for significant differences were all <0.01 and for non-significant differences were >0.1, except for the difference between height 1.25 m and 1.75 m which was 0.058.

2.6 Discussion

Camera trap optimisation trials are a critical preparational step for camera trap studies. Such trials can help optimise camera trap set-up techniques and identify limitations. If convenient situations (such as captive colonies) exist and the trial is well designed, informative results can be obtained within a few days. If camera trap limitations are overlooked, animal population data from camera trap studies may be inadequate or flawed and misinform management. We conducted three trials that (1) identified a suitable camera trap height for numbat detection, (2) allowed us to significantly improve detection rates by choosing a better camera trap model, and (3) informed us that elevated, downward-angled time-lapse cameras can be used to identify numbat individuals. The results are applicable to monitoring of the numbat and potentially to other small mammals.

2.6.1 PIR motion-sensor height trial

It is recommended that PIR sensors of camera traps be aimed at the body core of target species to best detect the animal's surface heat and movement (Meek *et al.* 2012; Wearn and Glover-Kapfer 2017). The detection rates observed in this study (where the target had a core body height of ~10 – 15 cm) confirmed this and show that a slightly higher sensor placement (e.g. 25 cm) is acceptable and possibly advantageous, especially when near-ground obstructions such as vegetation and undulating ground may restrict the camera's detection zone. Vegetation (grass) was found to obscure camera traps set close to the ground in a trial by Glen *et al.* (2013) of stoats (*Mustela erminea*), cats (*Felis catus*) and hedgehogs (*Erinaceus europaeus*). Even though we used Reconyx PC900 camera traps in this study, we are confident our results are applicable to other camera trap models with the same PIR sensor types, such as Swift 3C camera traps. While near-ground obstructions are likely to be encountered in the field, possibly obstructing camera traps installed in low positions (e.g. 10 cm), an installation above 25 cm could allow small mammals to stay undetected when passing close to the camera traps, below the detection zone. We therefore recommend an installation height of ~25 cm above ground for small mammals similar to numbats, when using camera traps with PIR sensors similar to those of Reconyx PC900 camera traps.

While exploring differences in detection rates of camera traps at different heights, it was established that the Reconyx PC900 camera traps at the preferred height (25 cm) missed 40% of all known events. This highlights the importance of testing camera traps on target species to learn about potential limitations such as missed events. The number of missed events in camera trap studies is seldom known, and can only be approximated when animal movements are controlled (e.g. Apps and McNutt 2018a used monitored dog movements to trigger camera traps) or when time-lapse or continuously recording video cameras are used concurrently with camera traps (Glen *et al.* 2013; Newey *et al.* 2015; Jumeau *et al.* 2017; Urbanek *et al.* 2019). Similar to this study, Jumeau *et al.* (2017) found in a study of small animals at wildlife underpasses that Reconyx HC600 camera traps missed 43% of mammal movements recorded from continuous video cameras. In contrast, Reconyx RC55 camera trap models missed only 5% of animal visits recorded by a video camera (Dixon *et al.* 2009). The Reconyx RC55 has the same fast trigger speed (0.2 s) and PIR motion-sensor zone (40°) as the PC900 and HC600, so these models should perform similarly. Perhaps the higher detection rates in the study by Dixon *et al.* (2009) are due to a difference in environment or study animal. One clear difference in that study is that cameras were aimed at bait stations that attract animals to stay for extended periods within the camera trap's detection zone, increasing the probability of detection (Gil-Sánchez *et al.* 2011; Monterroso *et al.* 2011; Rovero *et al.* 2013). As there are no known baits for numbats (Burrows and Christensen 2002), it was important to seek a more suitable camera trap model to reduce the missed detections experienced in this study with Reconyx PC900 models.

2.6.2 Comparison of camera trap models

By using camera traps with wider detection zones, we significantly increased the detection rates achieved by the Reconyx PC900 camera trap model. The Swift 3C wide-angle camera traps (110° detection zone) detected 89%, while the Reconyx PC900 models (40° detection zone) detected 37% of all known events. With a wider detection zone, the Swift 3C wide-angle camera traps also detected more events than the time-lapse camera (Reconyx PC900 model) set to 2-s intervals. These results are important to future camera trap studies of numbats and similar species. Without this preparatory camera trap trial, unawareness of camera trap limitations could have resulted in missing >60% of possible numbat detections.

These limitations would likely substantially compromise the power and confidence in the inferences, and therefore the value of subsequent studies. While performance differences between camera traps of different brands may derive from multiple factors other than detection zone width (e.g. type and arrangement of Fresnel lenses), the detection rates of the Swift 3C standard model (52° detection zone), which detected 51% of known events show that the width of PIR sensor detection zones alone can significantly affect detection rates as the Swift 3C standard and wide-angle models are otherwise identical. While increased detection rates of Swift 3C wide-angle cameras were expected, given that this camera model effectively samples a larger area, we acknowledge that differences may have been inflated by possible non-random movements of numbats in enclosures. For example, this might occur if numbats preferred running along fence lines, which were within the wide-angle cameras' but outside the standard cameras' field of view. The results of this trial and the study of Fancourt *et al.* (2018), who also found improved detection rates from camera traps with wider detection zones, may increase awareness of the importance of PIR detection-zone width in wildlife studies. Camera traps with wider detection zones than standard models may increase detection rates in a wide range of animals.

2.6.3 Time-lapse camera height for identification of individual animals

We have shown that individual numbat identification is possible from images derived from elevated, downward-angled cameras, and that the number of images depicting dorsal fur patterns increased with increasing camera height. Optimal camera height depends on several factors. Increasing camera height increases the field of view, and therefore the sample area. There will also be an optimal distance range for cameras to detect animals within the field of view. An increased height also may allow a better perspective of the animal's dorsal area, improving identification opportunities. However, an increase in camera height results in a decrease in image quality as the distance to the animal increases. These factors must be understood and balanced for optimal results, with due consideration for operator safety if climbing is required to position cameras. Here we have used time-lapse cameras but there are challenges with the use of time-lapse cameras in the field. When set to take images at very short intervals (e.g. 5 s or less), time-lapse cameras have high energy and image storage requirements, necessitating either frequent servicing and/or large battery and data storage capacities. Both may not be practical and/or feasible for field studies. Using motion-activated camera traps with inbuilt PIR sensors would solve these problems; however, aiming PIR sensors downward reduces the camera's detection zone (see fig. 3 in Apps and McNutt 2018b). A reduced detection zone may not hinder animal detection when cameras can be aimed at areas they visit (e.g. recurrently used pathways) or when animals can be attracted to the camera station by bait. Neither option applies to numbats. A possible solution for numbat studies could be the use of external PIR motion sensors that trigger cameras wirelessly or by cable attachment. This would allow setting camera traps at elevated, angled positions while placing the motion sensor at 25-cm height facing parallel to the ground. Further trials are required to explore this option.

2.6.4 Limitations

This study did not evaluate the long-term reliability, usability and cost-effectiveness of camera trap models. Also, image quality was not formally assessed; however, we deem image quality of models tested here generally as suitable for identifying animals to species level in field situations, as well as identifying numbat individuals from pelage patterns when the animal is captured within 2 m from the camera. It is likely that numbat individuals could be identified at further distances; however, we were not able to reliably determine this due to limitations in the zoo enclosures (e.g. vegetation restricting the view of camera traps). Additionally, the number of false triggers (images that did not result from numbat presence, e.g. detection of moving vegetation) per camera trap was not evaluated. False triggers were low in this study in zoo enclosures. Further trials are needed to validate and extend results from this study under field conditions.

2.7 Conclusions

Animal population data from camera trap studies are widely used to inform wildlife management. Camera trap studies can be substantially improved by first conducting well designed, short camera trap trials. Such trials help to optimise camera trap setup, identify limitations, improve animal detection rates, and therefore lead to better outcomes from camera studies for species management. Positioning camera traps with their PIR sensor 25 cm above ground and using Swift 3C wide-angle camera traps has been shown here to significantly increase the detection of numbats. Further research is recommended to enhance techniques for individual numbat identification using their characteristic dorsal pelage patterns, which was shown to be possible by using elevated, downward-angled time-lapse cameras.

2.8 Conflicts of interest

The authors declare no conflicts of interest.

2.9 Acknowledgements

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Chapter 3 Sign surveys can be more efficient and cost effective than driven transects and camera trapping: a comparison of detection methods for a small elusive mammal, the numbat (*Myrmecobius fasciatus*)

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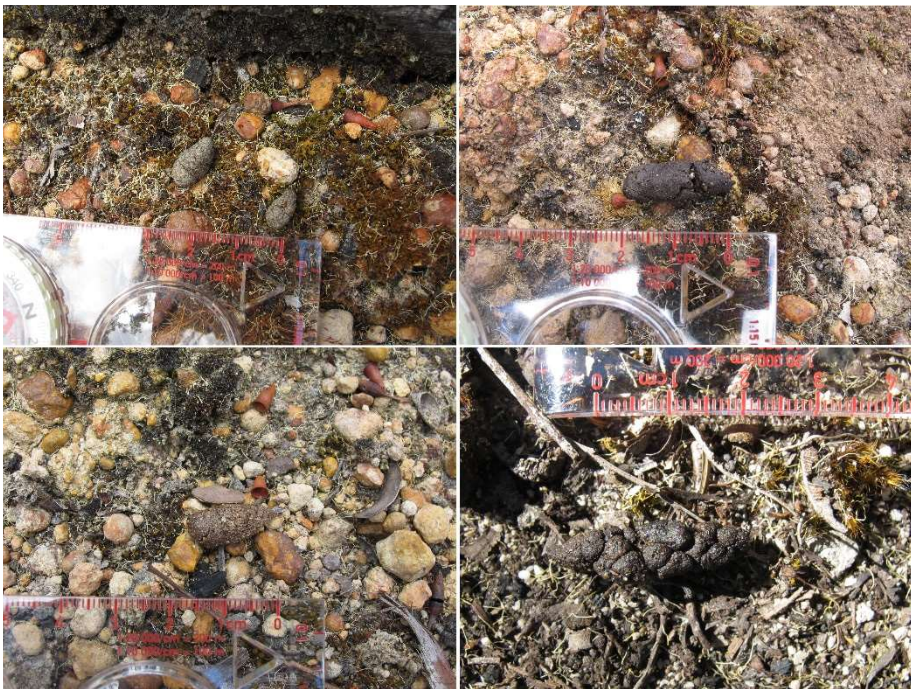
3.1 Preface

This chapter comprises original research undertaken to test driven transects, sign surveys and camera trapping for numbat detection in the UWR. These three methods were chosen because driven transects and sign surveys have traditionally been used for numbat detection at other locations and camera trapping is commonly used for mammal studies in the UWR by the DBCA. The aim of this study was to recommend a suitable method for long-term numbat monitoring in the UWR to the DBCA. Therefore, it was important to evaluate the effectiveness as well as the cost-efficiency of these methods.

Fieldwork preparations including camera trap set-up at 50 survey sites was completed before the camera trap model trial of the previous chapter was undertaken. Therefore, Reconyx PC900/HC600 camera trap models were used for this study instead of the Swift 3C wide-angle camera trap model that had increased numbat detection rates in the zoo environment. Reconyx PC900/HC600 were compared to Swift 3C wide-angle camera traps later as described in Chapter 5.

Examples of fresh numbat (*Myrmecobius fasciatus*) diggings with termite gallery breaches found during sign surveys in the Upper Warren region, south-western Australia.



Examples of numbat scats found during sign surveys in the Upper Warren region, south-western Australia.

3.2 Abstract

Context. Determining the most efficient detection method for a target species is key for successful wildlife monitoring and management. Driven transects and sign surveys are commonly used to monitor populations of the endangered numbat (*Myrmecobius fasciatus*). Camera trapping is being explored as a new method. These methods are unevaluated for efficacy and cost for numbat detection.

Aims. To compare efficacy and costing of driven transects, sign surveys and camera trapping for detecting numbats in the Upper Warren region, Western Australia.

Methods. Seven repeat sign surveys and driven transects, as well as 4 months of camera trapping, were conducted concurrently at 50 sites along three transects. Numbat detection rates and costing of the three techniques were compared, and detection probabilities were compared between sign surveys and camera trapping.

Key results. Numbat signs were detected during 88 surveys at 39 sites, exceeding camera trapping (26 detections at 13 sites) and driven transects (seven detections near five sites). The estimated probability for detecting a numbat or a sign thereof (at a site where numbats were present) ranged from 0.21 to 0.35 for a sign survey, and 0.02 to 0.06 for 7 days of camera trapping. Total survey costs were lowest for driven transects, followed by camera trapping and sign surveys. When expressed as cost per numbat detection, sign surveys were cheapest.

Conclusions. Comparative studies of survey methods are essential for optimal, cost-effective wildlife monitoring. Sign surveys were more successful and cost effective than camera trapping or driven transects for detecting numbats in the Upper Warren region. Together with occupancy modelling sign surveys are appropriate to investigate changes in occupancy rates over time, which could serve as a metric for long-term numbat monitoring.

Implications. There is no ‘best’ method for wildlife surveys. Case specific comparison of animal detection methods is recommended to ensure optimal methods. For the numbat population in the Upper Warren region, further studies are needed to improve numbat detection rates from camera trapping, and to test sign surveys in autumn (March to May)

when surviving juvenile numbats have established their own territory and assumptions regarding population closure are less likely to be violated.

3.3 Introduction

Determining efficient detection methods for a target animal is key for successful monitoring and management. This is because management goals are commonly stated in terms of population size or distribution, such as increases for threatened species, reductions for pest species or stable numbers for harvested species (Lancia *et al.* 2005). Given that few animal species are sufficiently visible and concentrated for accurate, complete counts, population parameters are commonly estimated using a range of methods (Boyce 1995). Finding the most appropriate detection method for a target animal is important because detection success can differ among methods (Silveira *et al.* 2003; Wayne *et al.* 2005; Vine *et al.* 2009; Croose *et al.* 2019). Inappropriate detection methods may cause inaccurate estimates of species distribution and abundance, inefficient use of resources, incorrect management decisions and possibly legal challenges (Caughley and Gunn 1996; Thompson *et al.* 1998).

One species difficult to detect is the endangered numbat (*Myrmecobius fasciatus*), a small (500–700 g), obligate termitophagous, diurnal, Australian endemic marsupial (Cooper 2011; Hayward *et al.* 2015). Since European settlement, habitat loss, exotic predators and inappropriate fire regimes have contributed to the numbat's decline (Department of Parks and Wildlife 2017). It now occurs in only 1% of its former range, with an estimated <1000 mature animals in ongoing decline (Woinarski *et al.* 2014; Department of Parks and Wildlife 2017). Conservation efforts successfully reintroduced numbats in several areas (Western Australia: Batalling forest block, Boyagin Nature Reserve, Dragon Rocks Nature Reserve, Mount Gibson, Tutanning Nature Reserve; New South Wales: Scotia Sanctuary; South Australia: Yookamurra Sanctuary (Department of Parks and Wildlife 2017; Australian Wildlife Conservancy, https://www.australianwildlife.org/where-we-work/mt-gibson, accessed 20 December 2019)). The only two remaining natural populations are in southwest Western Australia, in the Dryandra Woodland and the Upper Warren region (UWR) (Woinarski *et al.* 2014). The present study focuses on the UWR numbat population, where adequate monitoring information is lacking due to the absence of effective detection methods to assess population trends, size or distribution.

Obtaining reliable population estimates for numbats is challenging (Hayward *et al.* 2015). Conventional mammal trapping is unsuitable because no known lures or baits attract numbats (Burrows and Christensen 2002). In limited soil types tracking may be possible (Hayward *et al.* 2015), but it lacks wide potential. Instead, numbats are commonly monitored directly by sightings from driven transects (Calaby 1960b; Friend 1990), with results reported as relative abundance indices (sightings per 100 km) (Friend 1990; Friend and Thomas 2003; Hayward *et al.* 2015). However, comparisons of relative abundance indices to monitor population trends may be problematic because the assumption of constant detectability – the probability of actually detecting an animal when present – is often violated (Gese 2001). Animal detectability can be affected by factors such as weather, time of day, vegetation density, and observer ability (Gese 2001). More robust numbat population estimates from driven transects were achieved by Berry *et al.* (2019) within the fenced Scotia Sanctuary by applying distance sampling methods accounting for detection probabilities. Because distance sampling techniques require relatively high animal sighting rates (60–80 animals) (Buckland *et al.* 2001, pp. 240-241), they are impractical for monitoring numbats in the UWR where sighting rates from driven transects were very low, with 0.3 and 1.45 sightings per 100 km in 1995 and 1996 respectively (Department of Parks and Wildlife 2017).

Alternatively to direct sightings, animals can be detected indirectly from feeding signs and scats (Gese 2001; Sutherland 2006, Chapter 10). Numbats leave typical signs (diggings and scats), suitable for indirect detection (Calaby 1960b; Friend and Thomas 2003), and results of numbat sign surveys have previously been presented as presence–absence indices (percentage of sites occupied) (Friend 1990; Friend and Thomas 2003). Because abundance-occupancy relationships are commonly positively related (Gaston *et al.* 2000), these indices have been used to monitor numbat population trends in some subpopulations (Friend and Thomas 2003). However, the indices used were the naïve count of sites occupied, ignoring detection probability. Using raw counts without accounting for detection probability biases population estimates (Tyre *et al.* 2003; Gu and Swihart 2004; Guillera-Arroita *et al.* 2014).

More reliable estimates of numbat occupancy can be derived from repeat sign surveys in conjunction with occupancy modelling developed by MacKenzie *et al.* (2002). Occupancy modelling accounts for imperfect detection of a species, avoiding the negative bias of naïve presence–absence indices (MacKenzie *et al.* 2018).

More recently, technology has provided another option in the form of camera traps (Swann and Perkins 2014), used to monitor a range of fauna in varied settings (see examples in Meek *et al.* 2014a). Although camera traps were originally deemed unsuccessful for numbats because the species occurs at low densities (Hayward *et al.* 2015), they have successfully detected other elusive mammals that occur at low densities (Gompper *et al.* 2006; Vine *et al.* 2009; Paull *et al.* 2012), and unpublished pilot studies in the UWR confirm that numbats can be detected by camera traps (Julia Wayne, unpubl. data). Although camera trap data may be used in various ways to estimate animal population parameters (O'Brien 2011), they were used in our study to derive numbat detection probabilities from occupancy modelling to allow direct comparison with the results from sign surveys. If numbat detections by camera traps prove sufficiently high for capture–recapture methods, camera-trapping techniques could be developed further to allow individual numbat identification using dorsal pelage patterns.

There are no published assessments of the relative effectiveness of driven transects, sign searches and camera trapping for numbat detection. We therefore aimed to compare the efficacy and costing of the three different methods for detecting numbats within the UWR. Specifically, we compared the overall number of detections, and the cumulative increase of sites with detections as a function of effort of the three methods. Furthermore, we compared numbat detection probabilities from sign searches and camera trapping. Finally, we compared the costing of the three methods to evaluate method feasibility. Determining the most efficient detection method for numbats in the UWR will help develop appropriate survey methods to robustly assess population parameters, and therefore improve monitoring and management of the species. The findings may be relevant for monitoring other small mammals elsewhere.

3.4 Materials and methods

3.4.1 Study area, transects and sites

The present study was conducted in areas of national park, nature reserve and state forest within the UWR, ~ 300 km south of Perth, Western Australia (Figure 3.1). The UWR is part of the South West Australia Global Biodiversity Hotspot (Myers *et al*. 2000), and has a high conservation value because it supports many rare and endangered species (Burrows and Christensen 2002). Within the UWR, open sclerophyll forests and woodlands cover more than 140 000 ha of publicly managed land, with jarrah (*Eucalyptus marginata*), marri (*Corymbia calophylla*), and in some places wandoo (*Eucalyptus wandoo*) as dominant tree species (Yeatman *et al.* 2016). Forest management activities conducted by state authorities include prescribed fuel-reduction burns, timber harvesting and feral predator control using 1080-poisoned bait (Wayne *et al.* 2013a). Forested areas are surrounded by privately owned land predominantly used for agriculture and forestry (Figure 3.1). The region experiences a temperate mediterranean climate, with an annual average rainfall of ~ 650–900 mm (Zosky *et al.* 2017).

Three existing transects, consisting of transect A (~54 km), B (~56 km) and C (~61 km), and 50 existing survey sites were utilised (Figure 3.1). Transects were established in 1992 to monitor numbats (Julia Wayne pers. comm.) and followed unsealed double-lane roads (~34 km) and single-lane tracks (~138 km). Survey sites (16–17 sites per transect) were established in 2015 for numbat camera-trapping trials (Julia Wayne pers. comm.). Survey sites were, on average, 2.38 km apart (min 1.88 km, max 2.88 km; Figure 3.1). We deemed distances between sites sufficiently large to avoid redetection of numbats at different sites (the largest home-range size of a numbat in the UWR was measured to be 123.51 ha (Christensen *et al.* 1984)).

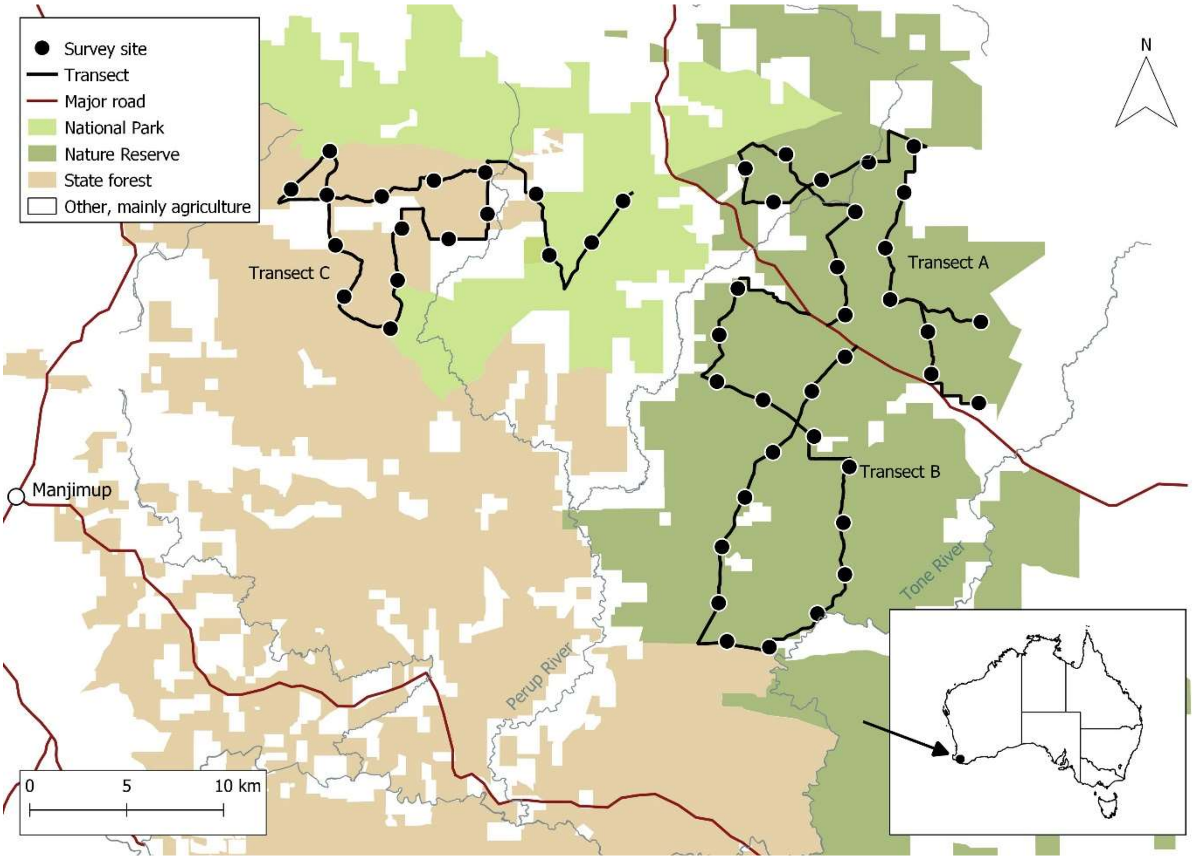


Figure 3.1. Location of transects with 50 survey sites used for this comparative study of three survey methods (driven transects, sign surveys, camera trapping) for the detection of numbats or signs thereof in the Upper Warren region, Western Australia. Shapefiles for the development of this map were provided by the Department of Biodiversity, Conservation and Attractions, Western Australia.

At each survey site, 10 plots (40 m x 100 m) were established for this study, with five plots located on each side of the transect. When the transect was bordered by private property all plots were situated on the same side (n=6 sites). Plots were placed adjacent to each other with the long edge perpendicular to the transect. A central plot was reserved for camera trapping at each survey site. The remaining nine plots were used for sign surveys.

3.4.2 Survey period and team

Camera trapping was conducted from September to December 2017 (4 months), and sign surveys and driven transects from mid-September to mid-December (3 months). This coincides with the Australian spring and early summer, and is immediately before and during juvenile numbat dispersal (Friend 1989), and therefore the time of maximum numbat abundance when detection is most likely. The three survey types (driven transects, sign surveys and camera trapping) were tested simultaneously to avoid seasonal or yearly differences in numbat abundance. All surveys were conducted by the first author, and 23 volunteers helped with driven transects and sign surveys. Volunteers received training before surveys to become familiar with required tasks. Nineteen volunteers participated in one, three volunteers in two, and one in three driven transects and sign surveys.

3.4.3 Driven transects

Driven transects were repeated seven times along the entire length of transect A, B and C. Periods between repeat driven transects were weather dependent and on average were 7 days (min. 6 days, max. 10 days). The driven transect procedure was adapted from Calaby (1960b) and Friend (1990). A car was driven at ~ 15 km h^{-1} on warm, calm days. Numbats rest during midday hours on hot days (Christensen *et al.* 1984), so we interrupted driven transects when temperatures exceeded 28°C (thermometer used: Kestrel 3000 Pocket Weather Meter). During driven transects, one driver (first author) and three additional observers (volunteers) scanned the areas in front and on either side for numbats. A numbat detection from driven transects is here defined as a sighting by any observer. When a numbat was detected, date, time, GPS location, and the numbat's distance to the road were recorded. Distance to road was estimated by counting steps. We did not use a more exact measurement because the main aim was to assess driven transects as a detection method, not to calculate population estimates by, for example, distance sampling techniques. For logistical reasons, driven transects were usually interrupted for sign searches when a survey site was reached.

3.4.4 Sign surveys

Seven repeat 5-day sign surveys were conducted during the same time as driven transects. They consisted of searches for numbat signs (fresh diggings and scats as described by Calaby (1960b), Christensen *et al.* (1984) and Connell and Friend (1985)). During each sign survey, one plot was searched at each of the 50 survey sites. Within each site, plots were chosen randomly with no plot searched twice. Searches were conducted by the first author and three volunteers. To maximise search area within each plot, team members walked ~ 5 m apart in a straight line, searching for numbat signs 2.5 m left and right. The search team walked up one side of the plot (covering half of the plot width) and down the other side (covering the other half of the plot width). When numbat diggings and/or scats were found, the signs were recorded and searching stopped. The finding of one or more numbat signs (i.e. an indirect detection of a numbat/s) on a plot during a survey was defined as a single numbat detection at that survey site. All numbat scats found were inspected by the first author using a 20x magnifying glass in the field, and later verified under a dissecting microscope. When no signs were found, searches ended after the team searched the entire plot (~ 20 min).

3.4.5 Camera trapping

We used Reconyx PC900 camera traps because they are highly regarded (Glover-Kapfer *et al.* 2019), and commonly used by the DBCA for camera trap studies in the UWR. At each of the 50 sites, one camera trap was attached to a tree ~ 25 cm above ground. This camera height was found to be most suitable for numbats (Seidlitz *et al.* 2020). Camera traps faced southwards to avoid direct sun-glare, and towards forest clearings to minimise vegetation obstructing the field of view within the first ~5 m from the camera. A 5-m detection distance and a detection angle of 40° (see manufacturer's specifications: Reconyx 2017) result in a detection area of ~ 8.7 m^2 (calculated as a circle sector). Vegetation near cameras was minimally pruned to reduce false-trigger events. Camera traps were located centrally on a plot with a minimum distance of 30 m to the transect. Cameras were set to take 10 images when triggered using the 'rapid fire' function, with no delay between triggers. The camera's passive infrared sensor sensitivity was set to high. Batteries and SD-cards were changed monthly. Images from camera trapping were scanned for numbats using FastStone Image Viewer version 6.2 (FastStone Soft 2019). A numbat detection was defined as a camera trap trigger resulting in one or more images depicting a numbat partially or wholly. To take into account that a numbat may stay in front of a camera trap for an extended period, detections were only counted when there was a minimum of 60 min between subsequent detections (Tobler *et al.* 2008; Rovero and Marshall 2009).

3.4.6 Cost analysis

Costs for each survey method were calculated independently. We considered travel, personnel, volunteer and material costs as main expense categories. Travel cost was based on the DBCA vehicle hire charge of AU$ 1.00 per km, and covered distances travelled to, from and between sites. Personnel cost was determined at an hourly rate of AU$ 69.17, the 2020 rate of a well qualified fauna conservation officer at DBCA (including 45% payroll overheads). Personnel cost included time for equipment preparation, installation, maintenance, and post-survey care, for survey planning, volunteer-related communication and preparation, scat validation under the microscope, data digitalisation and time in the field. Volunteer cost covered a daily food and accommodation allowance of AU$ 30.00. Material cost included AU$ 10.00 for stationery for all survey methods, and AU$ 20.00 for scat collection and storage materials for sign surveys. For camera trap related material costs, we divided the 2017 purchase price of camera traps, rechargeable batteries, and SD cards by 5, assuming that these materials can be used for further numbat surveys over 5 years (20% depreciation rate).

3.4.7 Evaluating method effectiveness and data analysis

To compare method efficacy, several performance metrics were evaluated, including raw numbat detections (e.g. total number of detections as defined above for each method), and the accumulation of sites with numbat detections over time using the Vegan Community Ecology package version 2.5–5 (Oksanen *et al.* 2019) in R version 3.5.0 (R Core Team 2020). Site accumulation was estimated from indirect numbat detections from sign surveys (scats and diggings), and from direct detections from camera trapping (numbat images) and driven transects (sightings). Sightings of numbats from driven transects were assigned to the nearest survey site to allow site accumulation for this transect-based method. To compare sighting rates from driven transects with historic sighting rates (sightings per 100 km) in the UWR, the number of sightings was divided by the number of kilometres driven and multiplied by 100. Additionally, efficacy of sign surveys and camera trapping were compared by determining the probability of numbat detection for each method. Probability of detection (the probability that a species is detected at a site given its presence) was calculated using the single season occupancy modelling framework (MacKenzie *et al.* 2018). A matrix with detections (1) and non-detections (0) was established from spatial replicates (50 sites) and temporal repeats (repeat surveys). Similar to other studies, temporal repeats for camera trapping were established by dividing camera trap data into 7-day periods

(Gálvez *et al.* 2016). Models to estimate detection probabilities were fitted using the RPresence package version 2.12.33 (MacKenzie and Hines 2018) in R. We acknowledge that the assumption of population closure may have been violated by conducting surveys during juvenile numbat dispersal. This may have affected occupancy estimates (the probability that a species is present at a site), which were not the aim for the present study. Because we aimed to compare the efficiency of detection methods, we consider the survey timing reasonable, and because occupancy was not estimated, the occupancy component of models was kept constant (psi~1), and habitat covariates were not used. Models were fitted with combinations of the covariates 'method', 'transect' and 'site' before comparison using the Akaike Information Criterion (Akaike 1974; Burnham and Anderson 2002, pp. 60-64), The model with the lowest value was chosen to determine numbat detection probabilities for the different methods. Model fit was examined by computing the c-hat value using the goodness of fit test in the RPresence package on the global model (the most complex model with the greatest number of parameters), as described in MacKenzie *et al.* (2018), Chapter 4. Further software used were Microsoft Excel for data digitalisation and survey cost calculations, and QGIS 3.2 Bonn for computing maps and distance travelled (QGIS Development Team 2019).

3.5 Results

During seven repeat surveys, we drove ~ 1198 km along transects A, B and C, and searched 350 plots at 50 survey sites for sign searches. Camera trapping at 50 survey sites resulted in 156 966 images. Forty-seven camera traps operated between 124 and 127 days (mean 126.5 days). Three camera traps had reduced periods (92, 95 and 103 days) due to either unknown or operator errors (code lock not entered correctly). Days on which camera traps failed to operate were included in the occupancy modelling data matrix as missing observations. In total, camera traps operated on 6235 full days.

Numbat detections were highest from sign surveys and lowest from driven transects (Figure 3.2). The first author (skilled observer) found 63% of numbat signs detected during sign surveys. Numbats were detected on transects A, B and C with the use of sign surveys and camera trapping. No numbats were detected on transect B during driven transects (Figure

3.2). Seven numbat detections were recorded from driven transects, giving a relative abundance index of 0.584 detections per 100 km.

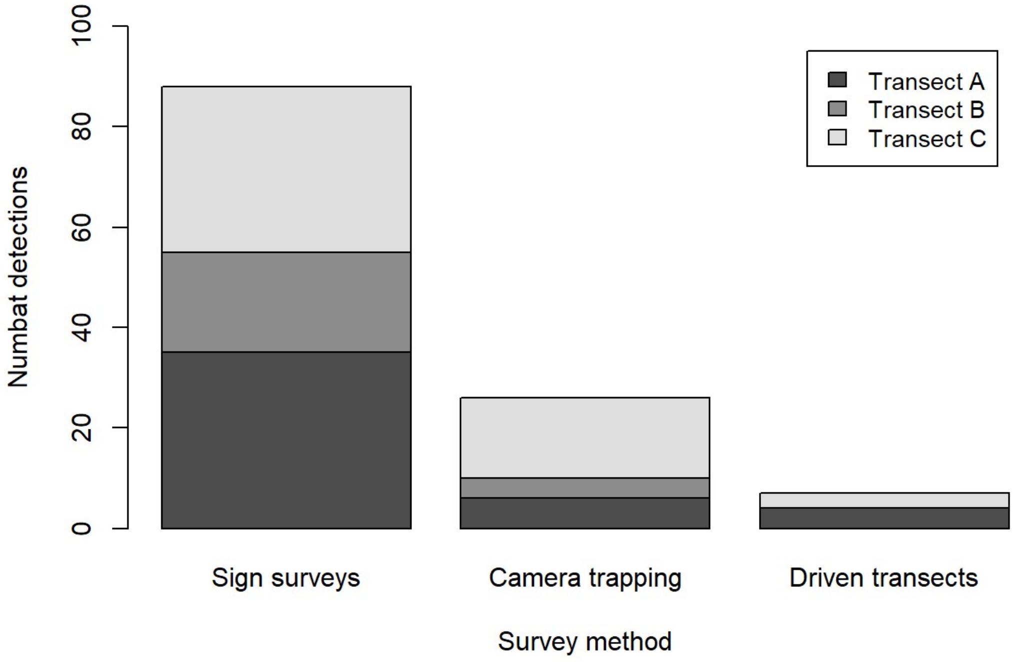


Figure 3.2. Number of numbat detections compared from three survey methods tested along three transects (each between 50–60 km long) with 50 survey sites: seven repeat surveys of driven transects (~1198 km), seven repeat sign surveys at 50 sites (350 surveys in total); and 4 months of continuous camera trapping at 50 sites (number of numbat detections more than 60 min apart). Detections refer to numbat sightings from driven transects, the number of searches in which numbat scats or diggings were detected from sign surveys and numbat detections from camera trapping. All surveys were conducted during September-December 2017 in the Upper Warren region, Western Australia.

The increase in the number of survey sites with numbat detections as a function of effort was greatest from sign surveys (Figure 3.3). After seven repeat surveys, numbats were detected at 39 (78%) and 13 (26%) survey sites from sign surveys and camera trapping respectively. During driven transects, numbats were detected near five (10%) survey sites. Numbats were detected at 40 sites (80%) from all three detection methods combined.

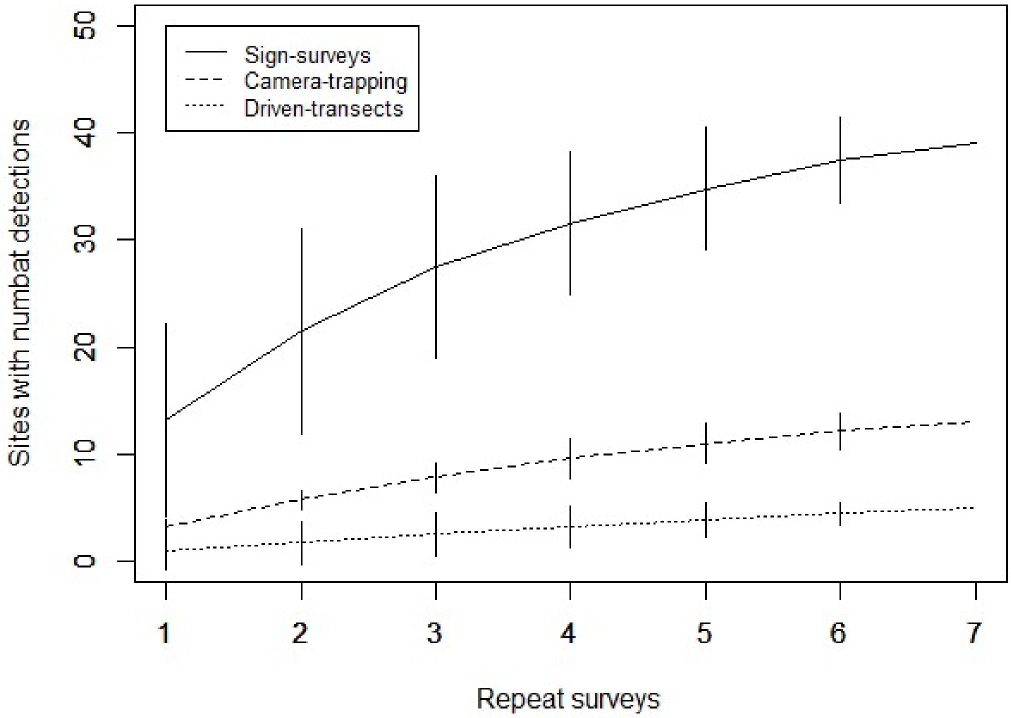


Figure 3.3. Cumulative number of survey sites (n = 50, CI = 95%) situated along three transects (each between 50–60 km long) with numbat and/or sign detections from three survey types: seven repeat surveys of driven transects (~1198 km; sightings of numbats were assigned to nearest survey site); seven repeat sign surveys at 50 sites (350 surveys in total); and 4 months of continuous camera trapping at 50 survey sites (camera-trapping data were divided into seven 18-day periods). All surveys were conducted during September–December 2017 in the Upper Warren region, Western Australia.

Using occupancy modelling, the initial model (including the explanatory variable of interest ‘method’ only), had an AIC of 624.21. The AIC did not improve when ‘method + site’ (AIC=624.78) or ‘method * site’ (AIC=655.46) were included. The AIC did improve by adding ‘method + transect’ (AIC=623.62) and ‘method * transect’ (AIC=622.98). Expanding the model further with ‘method + transect + site’ or ‘method * transect + site’ did not improve it (AIC=628.78 and 628.11 respectively). Therefore, the final model included ‘method’ and ‘transect’, and the interaction thereof as covariates to estimate detection probabilities. The estimated c-hat for the global model, after 3000 bootstrap iterations, was 1.93, noting that values greater than 1 suggest there is more variation in the observed data than expected by the model (MacKenzie and Bailey 2004). Detection probability estimates for finding a numbat or a sign thereof (at a site where numbats were present) ranged from 0.02 to 0.06 for camera trapping (per 7-day period), and from 0.21 to 0.35 for sign surveys (per survey; Table 3.1).

Table 3.1. Numbat detection probabilities from sign surveys and camera trapping

Detection probability estimates (*P*) and confidence intervals (CI) for finding a numbat or a sign thereof (at a site where numbats were present) for sign surveys (per survey) and camera trapping (per 7-day period) at 50 survey sites. Continuous camera trapping was conducted over 4 months, and sign surveys were repeated seven times (350 searches). All surveys were conducted during September–December 2017 in the Upper Warren region, Western Australia.

	Sign surveys		**Camera trapping**	
	P	95% CI	*P*	95% CI
Transect A	0.35	0.26 – 0.45	0.02	0.01 – 0.05
Transect B	0.21	0.12 – 0.33	0.02	0.01 – 0.04
Transect C	0.29	0.21 – 0.38	0.06	0.03 – 0.09

Total survey cost was lowest for driven transects and highest for sign surveys (Table 3.2). Sign surveys were 13% (AUS$3780) more expensive than camera trapping and 44% (AU$8896) more expensive than driven transects. The lowest cost per numbat detection was achieved by sign surveys, with driven transects being 8.7 times, and camera trapping 2.9 times more expensive (Table 3.2).

Table 3.2. Cost breakdown for three numbat survey methods and resulting cost per numbat detection

Driven transects (total ~1198 km) and sign surveys (seven repeat surveys of 50 sites) were conducted over 3 months. Camera traps (50 sites) operated over 4 months. Costs include travel to and from survey areas. The number of numbat detections – number of direct sightings from driven transects, number of surveys in which numbat scats or diggings were detected from sign surveys or number of numbat detections more than 60 min apart from camera trapping – are provided in parentheses. Cost per detection was calculated by dividing the total survey cost by detections. All surveys were conducted between September – December 2017 in the Upper Warren region, Western Australia.

Cost details	**Driven transects (AU$)**	**Sign surveys (AU$)**	**Camera trap surveys (AU$)**
Personnel cost (AU$69.17 h^{-1})	14 944	21 017	12 432
Vehicle cost (AU$1.00 km^{-1})	3423	4966	2413
Volunteer cost	1890	3150	0
Materials cost	10	30	10 538
Total cost	**20 267**	**29 163**	**25 383**
Cost per numbat detection	2895 (7)	331 (88)	976 (26)

3.6 Discussion

The present study broadens the understanding of the usefulness and cost efficiency of three survey methods for detecting elusive species occurring at low densities. Under the conditions reported here, sign surveys were considerably more successful and cost effective at detecting numbats in the UWR than driven transects or camera trapping. Even though the true numbat occupancy rate is unknown, we conclude here that sign surveys were more accurate than camera trapping or driven transects because the number of sites with numbat detections from sign surveys (39) was closest to the total number of sites with evidence of numbat presence from all three methods combined (40). Several studies found camera trapping to be more successful for animal detection when compared with other methods including sign surveys (see Wearn and Glover-Kapfer (2019) for an analysis of 104 method-comparing studies). Even though some of these studies compared methods similar to the ones compared here, results are not directly comparable because bait was used to attract target species (Di Cerbo and Biancardi 2013; Greene *et al.* 2016), target species were considerably larger than numbats (Bartolommei *et al.* 2012; Anile *et al.* 2014), or camera traps were set at burrow entrances (Ellis *et al.* 2017).

The success of sign surveys for detecting numbats during the present study may be attributed to several factors. Animal signs persist for a long time (Heinemeyer *et al.* 2008). Depending on weather, we assume that numbat diggings last for several days, and scats possibly for several weeks. Therefore, evidence of numbats can be detected long after the animal has left. Although advantageous for sign detection, the long persistence of animal signs also poses a disadvantage: it remains unknown when exactly the animal was present in the area (contrasting direct sightings or captures on camera, which provide information on the time of capture). To resolve this, it needs to be determined how long numbat scats persist in the natural environment – a subject for future studies. Numbat diggings are typically shallow, numerous and spread over several square meters, making them conspicuous and characteristic. Even though numbat scats are small, they are easily recognisable (Connell and Friend 1985), are often found with diggings or on logs and termite mounds (Calaby 1960b), and therefore can be detected by skilful observers. Finally, the search area was large (40 x 100 m) and searched by four observers (the lead author and three volunteers), resulting in a detection probability of 21–35%. As the lead author found most numbat signs (63%), the efficacy and precision of sign surveys could be improved by using fewer but more skilled observers. This might allow a reduction of observers as well as repeat surveys needed to achieve the same numbat detection probability, reducing survey cost.

Sign surveys could be applied for future numbat studies in the UWR in various ways. Sign surveys in conjunction with occupancy modelling could investigate changes in occupancy over time. Additionally, differences in occupancy rates among study sites with varying habitat types or management activities could be explored by including the relevant covariates in model design (Okes and O'Riain 2017; Romano *et al.* 2018; Silveira *et al.* 2018). Furthermore, DNA analysis from scat samples could open new avenues, including numbat population abundance estimates from capture–recapture analysis (Piggott *et al.* 2006; Mondol *et al.* 2009; Kindberg *et al.* 2011).

Camera trapping detected fewer numbats than sign surveys. Even though camera traps operated continuously (contrasting the shorter duration of sign surveys), and over an extended survey period (4 instead of 3 months for sign surveys), their numbat detection probability was low (2–6%). Multiple factors may have contributed. Camera traps sample a small detection zone (Apps and McNutt 2018b), so a numbat could pass near a camera trap and stay undetected (Gillespie *et al.* 2015; Pease *et al.* 2016). Some animals may even avoid camera traps (Meek *et al.* 2016a). Additionally, camera traps may not always trigger when an animal moves within the detection zone (Jumeau *et al.* 2017; Urbanek *et al.* 2019; Seidlitz *et al.* 2020), especially when the animal is small (Gompper *et al.* 2006; Damm *et al.* 2010; Rowcliffe *et al.* 2011; Urbanek *et al.* 2019).

However, camera traps have many advantages warranting further trials to improve numbat detection probabilities and reduce costs. Evaluation of camera trap images does not need highly skilled personnel, so cost could be reduced by using volunteers. Furthermore, camera traps take images of multiple species. Therefore, cameras could collect data for multiple projects, allowing cost sharing. Whereas sign surveys result in presence–absence data (unless scat DNA analysis is possible), images from camera traps may identify individuals, breeding status, age, and other demographic attributes (Jędrzejewski *et al.* 2017). Numbat detection rates from camera traps could be improved by, for example: targeted camera placement in preferred habitat areas (Harris *et al.* 2013), increasing the number of camera traps per site (Pease *et al.* 2016; O'Connor *et al.* 2017) or by using different camera trap models (e.g. Fancourt *et al.* (2018) who increased rabbit detection rates by using camera traps with wider detection angles). Some of these options may not be cost–effective because they may increase survey cost. However, significant cost savings can be achieved by using a cheaper, potentially more effective camera trap model (Driessen *et al.* 2017; Apps and McNutt 2018a; Fancourt *et al.* 2018). To be able to use camera trapping as a survey method for numbat monitoring in the UWR in conjunction with occupancy modelling, the numbat detection probabilities would need to be increased. If detection probabilities are as low as 10% (2–4% in this study), more than 26 repeat surveys (here a repeat survey was defined as a 7-day camera trap interval) are necessary to allow reliable estimates of occupancy (MacKenzie *et al.* 2018, p. 461).

The least successful method was driven transects. Similar to results from driven transects conducted in the UWR during the mid-1990s, few numbats were sighted (0.584 detections per 100 km). This low detection rate is unsuitable for population estimates from distance sampling. Furthermore, relative abundance indices from driven transects with such low detection rates are unlikely to facilitate confident detection of real temporal changes in the numbat population. Numbats are small and well camouflaged, and therefore they are difficult to spot. Additionally, numbats are likely to hide when a car is approaching, and the typically dense vegetation of the UWR restricts observations. It is thus unlikely that driven transects can be improved to become a successful numbat detection method for long-term monitoring in the UWR.

Sign surveys were more successful and cost effective than camera trapping or driven transects for detecting numbats in the UWR. During sign surveys, special attention must be paid to ensure that animal signs are correctly identified to avoid false negative or false positive identifications. We are confident here that signs were identified correctly because diggings were only counted when numerous fresh diggings were found, and scats were validated by microscopy. We recommend sign surveys for numbat monitoring in the UWR, with the use of occupancy modelling. Until reliable numbat abundance estimates can be developed, for example, from capture–recapture methods using numbat scat DNA analysis or camera trapping, it remains unknown how occupancy estimates relate to true numbat abundance. However, they do provide information on areas occupied by numbats. Such knowledge can help identify habitat preferences, informing management of the species' responses to timber harvesting, prescribed fuel reduction burns and introduced predator control. To improve occupancy estimates we recommend investigating factors affecting the production, persistence and detection of numbat signs, because these factors influence detection probabilities. Furthermore, although the occupancy models used in this study were adequate for comparing survey methods, they could be improved by including additional covariates (e.g. habitat and environmental variables) that may better account for the variation in observed numbat detections. We further recommend exploring the improvement of numbat detection rates from camera trapping, and to test sign surveys in autumn when surviving juvenile numbats have established their own territory, and assumptions regarding closure (e.g. occupancy status at each site does not chance over the survey season) are less likely to be violated. Even though occupancy model extensions are available to assess the robustness of results when closure assumption violations are suspected (MacKenzie *et al.* 2018, Chapter 6), it is best to sample populations when closure assumptions can be met (MacKenzie *et al.* 2018, p. 149). The present study highlights the importance of finding the most appropriate detection method for a target animal, but it needs to be acknowledged that a method's success depends on many factors, including species' characteristics, population density, habitat type, personnel skills, type of equipment used and survey timing. We recommend case-specific comparison of animal detection methods to ensure optimal methods are used for successful and cost-effective monitoring.

3.7 Acknowledgements

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3.8 Conflicts of interest

The authors declare no conflicts of interest.

Chapter 4 Animal detections increase by using a wide-angle camera trap model but not by periodically repositioning camera traps within study sites

This chapter has been published:

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4.1 Preface

Reconyx PC900/HC600 camera traps were less successful in detecting numbats than sign surveys (Chapter 3). However, since camera traps are an attractive tool for wildlife detection, and since the DBCA commonly uses camera traps for mammal detection in the UWR, it was worth testing if camera trap detection rates of wild numbats could be improved. Different camera trap set-up techniques and models (Swift 3C wide-angle camera traps were now available for testing) were used to achieve this goal. Even though this PhD focused on numbats, all wildlife species were considered to make this study applicable to a wider audience and for those undertaking multi-species studies.

A western quoll (*Dasyurus geoffroii*), captured by a Swift 3C wide-angle camera trap in the Upper Warren region, south-western Australia.

A brush-tailed bettong (*Bettongia penicillate*, left) and common brushtail possum (*Trichosurus vulpecula*), captured by a Reconyx PC900 camera trap in the Upper Warren region, south-western Australia.

4.2 Abstract

When using camera traps for wildlife studies, determining suitable camera models and deployment methods is essential for achieving study objectives. We aimed to determine if camera trap performance can be increased by 1) using cameras with wider detection angles, and 2) by periodically repositioning cameras within sites. We compared three camera trap groups: stationary Reconyx PC900/HC600 (40° detection angle), and paired, periodically-repositioned Reconyx PC900/HC600 and Swift 3C wide-angle camera traps (110° detection angle). Cameras operated simultaneously at 17 sites over nine weeks within the Upper Warren region, Western Australia. Swift cameras had significantly higher detection rates, leading to better performance, especially for species <1 kg and >10 kg body mass. Reconyx cameras missed 54% of known events, with most being animals that moved within the cameras' detection zones. Stationary and periodically-repositioned Reconyx camera traps performed similarly, although there were notable differences for some species. The better performance of Swift 3C wide-angle camera traps makes it more useful for community-level and species-level studies. The increased sensitivity of the Swift's passive infrared (PIR) sensor along with the wider detection zone played an important role in its success. When choosing camera trap models, detection angle and sensor sensitivity should be considered to produce reliable study results. Periodically repositioning cameras within sites is a technique that warrants further investigation as it may reduce camera placement bias, animal avoidance of camera traps, and increase spatial/habitat information when a limited number of cameras are deployed.

4.3 Introduction

Passive infrared (PIR) triggered camera traps are increasingly used in wildlife investigations for a wide range of applications (Rowcliffe and Carbone 2008; Burton *et al.* 2015; Meek *et al.* 2015b). Before deploying camera traps, researchers need to consider which camera trap model, and deployment techniques (e.g. camera height and position) are most suitable for achieving study objectives. The right choice is important because camera trap model, and deployment technique can affect study results, and therefore the inferences made, for example, on species richness and occurrence (Swan *et al.* 2014a). As new camera trap models and ideas for different set up techniques emerge, they need to be tested for their applicability in wildlife research. To expand the camera trapping body of knowledge we present here results from a comparative camera trap study that explored if animal detections could be increased (1) by using Swift 3C wide-angle instead of the commonly used Reconyx PC900/HC600 standard-lens camera trap models, and (2) by periodically repositioning camera traps within study sites.

4.3.1 Camera trap detection angle

The size of the detection area of camera traps can greatly affect animal detection rates (Rowcliffe *et al.* 2011), which should be maximised for reliable animal population estimates of either abundance or occupancy. When unobstructed, the detection area of a camera trap is determined by the width of the PIR sensor's detection angle, and the distance up to which a PIR sensor can detect animals (specifically, objects that move within the detection zone with a surface temperature that differs from background objects). Detection distance of camera traps is commonly > 10 m according to manufacturer's specifications, although that may depend on animal size. In field conditions, vegetation or landscape features may also constrain detection distance. Rowcliffe *et al.* (2011) found that Central American agouti (*Dasyprocta punctata*), a small to medium sized mammal (~3.5 kg body mass), were detected mostly within the nearest 4-5 m of Reconyx RC55 camera traps. Therefore, it may be important to maximise the detection angle of camera traps for increased detection zone size. Detection angles of commonly used standard camera traps lie between 40°- 60° but can exceed 100° in wide-angle camera traps (Meek *et al.* 2012; Trolliet *et al.* 2014; Wearn and Glover-Kapfer 2017). Even though wide-angle camera traps can perform well when compared to standard camera traps (Swann *et al.* 2004; Fancourt *et al.* 2018), they are not widely used for wildlife studies. To our knowledge, animal detection rates of wide-angle camera traps have not yet been compared to standard camera traps in a field setting with a variety of wildlife species of different size classes.

4.3.2 Periodic repositioning of camera traps

During wildlife studies, camera traps are typically stationary within sites for the entire study period (e.g. Jacobs and Ausband 2018; Moore *et al.* 2020). If animals move through the habitat randomly, without giving preference to any particular features, animal detection rates should not be affected by camera trap location. However, habitats are heterogeneous and contain a mosaic of more or less preferred areas for animals (Barraquand and Benhamou 2008). Therefore, their movements are likely to be non-random. Non-baited, random camera trap placement (often desired to meet assumptions of population statistics) may cause some cameras to be located in areas less preferred by target species. Those cameras may have reduced detection rates (Kolowski and Forrester 2017): an unwanted situation especially for rarely detected species. One way to overcome this problem is to use multiple camera traps per site to increase detections (Kolowski and Forrester 2017; O'Connor *et al.* 2017; Evans *et al.* 2019). This increase in detections may result in part from sampling additional areas within the same site, that might be more frequently used by the animals of interest. We asked, could the same be achieved by periodically repositioning single camera traps within study sites?

4.3.3 Aims and hypotheses

Our aim was to compare animal detection rates, detection probabilities, and site accumulation rates for individual species, as well as species accumulation rates for three camera trap groups: stationary Reconyx PC900/HC600 (40° detection/lens angle), and paired, periodically-repositioned Reconyx PC900/HC600 and Swift 3C wide-angle camera traps (110° detection- and 100° lens angle). Swift 3C wide-angle camera traps were shown to detect numbats (*Myrmecobius fasciatus*) more effectively than Reconyx PC900 camera traps in zoo enclosures (Seidlitz *et al.* 2020). We therefore hypothesised that Swift camera traps would generally perform better in field conditions than the Reconyx camera traps for the above-mentioned metrics. Since periodically repositioning camera traps may allow camera traps to sample a wider range of habitat features, we hypothesised that repositioned Reconyx camera traps would generally perform better than stationary Reconyx camera traps.

4.4 Materials and Methods

4.4.1 Study area

This study was conducted within the Kingston National Park and adjacent state forest of the Upper Warren region in south-western Australia, 300 km south of Perth (Fig. 1). South-western Australia is a global biodiversity hotspot (Myers *et al.* 2000). The region's publicly managed forests cover an area of more than 140 000 ha, which support several mammalian species classed as threatened under the *Western Australian Biodiversity Conservation Act 2016*. These species include the numbat (endangered), western ringtail possum (*Pseudocheirus occidentalis*; critically endangered), western quoll (*Dasyurus geoffroii*; vulnerable), and brush-tailed bettong (*Bettongia penicillata*; critically endangered). The region's forests consist mainly of open sclerophyll forests and woodlands dominated by three tree species: the jarrah (*Eucalyptus marginata*), marri (*Corymbia calophylla*) and wandoo (*Eucalyptus wandoo*) (Yeatman *et al.* 2016). Forest management activities are carried out by state authorities, and include prescribed fuel-reduction burns, timber harvesting, and feral predator control using 1080-poisoned bait (Wayne *et al.* 2013a). The region has a Mediterranean type climate with an annual average rainfall of approximately 650-900 mm (Zosky *et al.* 2017).

4.4.2 Study period and weather

This study was conducted over nine weeks from mid-March to mid-May 2018, coinciding with the Australian autumn. During this period, the average temperature was 15.5°C (min 8.4°C; max 32.2°C), and the average relative humidity was 68.8% (min 19.2%; max 92.2%). During the study there were 11 rainy days with a total of 24.6 mm precipitation (min 0.1 mm/day; max 13.5 mm/day). Weather data were obtained online from the Yerramin weather station located approximately 15 km from the study area (https://weather.agric.wa.gov.au/station/YERR).

4.4.3 Study sites

Camera stations were set at 17 existing sites (Figure 4.1). These sites are a subset of 50 study sites established in 2015 for the purpose of numbat monitoring (Julia Wayne pers.

comm.). This subset was chosen because numbats and other mammal species were frequently detected here (Anke Seidlitz, unpublished data). The average distance between sites was 2.34 km (min 1.88 km; max 2.73 km). Sites were located adjacent to unsealed roads and tracks. At each of the 17 sites, ten plots (40 m x 100 m) were established with five plots on either side of the track (unless the track was bordered by private property in which case all plots were located on the forested side, n=3). Plots were placed adjacent to each other with the short edge parallel to the track.

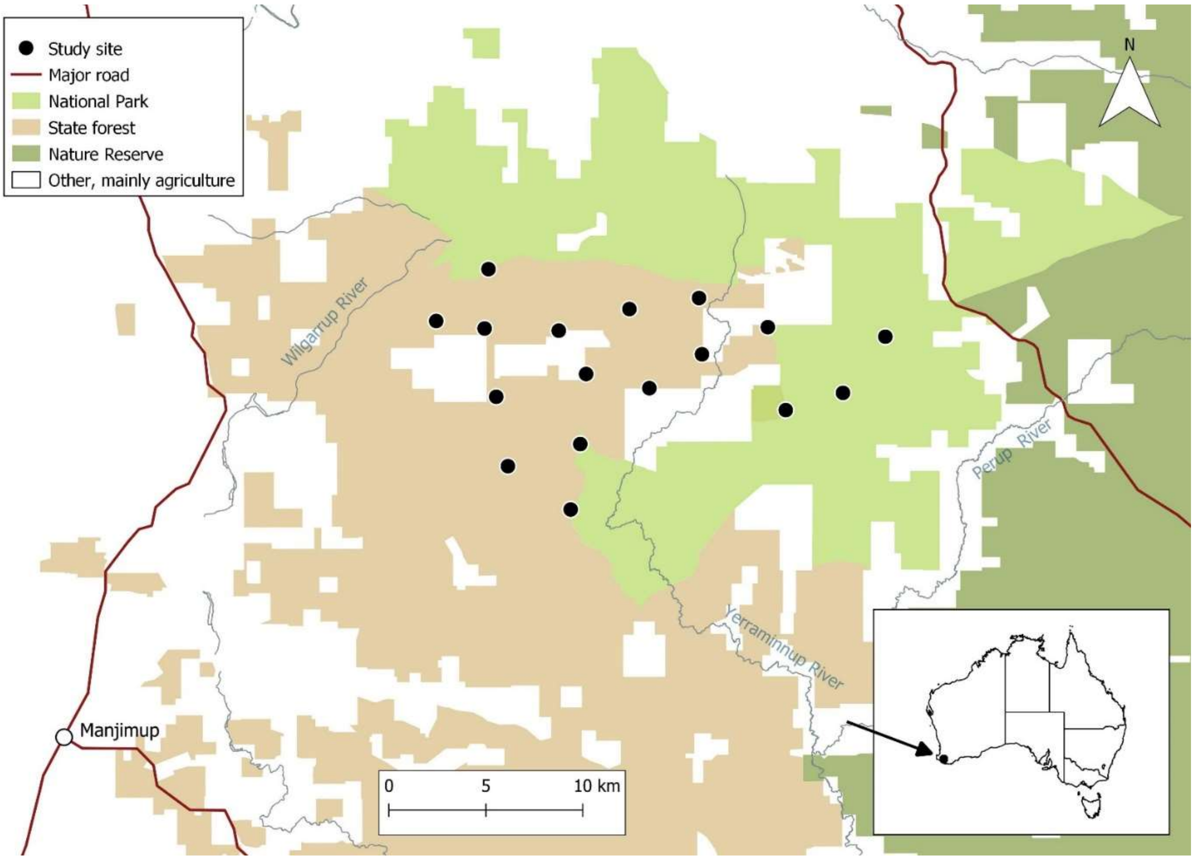

Figure 4.1. Location of survey sites (n=17) used for this comparative study of stationary (n=17) and periodically repositioned (n=17) Reconyx PC900/HC600, and periodically repositioned Swift 3C wide-angle (n=17) camera traps for the detection of wildlife in the Upper Warren region, Western Australia.

4.4.4 Camera trapping

We used 51 camera traps consisting of 17 Swift 3C wide-angle cameras (Outdoor Cameras Australia, Toowoomba, QLD, Australia), and 34 Reconyx cameras (17 of each model: PC900 and HC600; RECONYX, LLP, Holmen, WI, USA). Reconyx PC900 and HC600 models were here treated as equivalent because differences between the two models are predominantly related to software functions. Camera settings used during this study were available in both models; specifications and accessories are detailed in Table 4.1.

Table 4.1. Specifications for camera trap models and accessories used during this comparative study conducted within the Upper Warren region, Western Australia.

Camera trap models used have zonal detection areas, resulting from an arrangement of multiple Fresnel lenses in two horizontal bands. We used rechargeable Fujitsu LSD (HR-3UTK) batteries and 16 GB Delkin Devices (SD163X) SD cards in all cameras. Camera trap specifications as described in user manuals. Theoretical detection area was calculated as a circle sector using the formula: Area = π r2 (C/360) where C = lens angle, and r = detection distance.

Specifications	Reconyx PC900/HC600	Swift 3C wide-angle
Trigger speed (s)	0.2	0.35~0.45
Lens angle (°)	40	100
Detection angle (°)	40	110
Detection distance (m)	up to 30	up to 15
Theoretical detection area (m^2)	~314.2	~196
Image resolution	1080P HD, 3.1 MP	5,8,12 MP
Number of batteries	12	8
Manufacturing date	~2013-14	05/2017

At each site, three camera traps were deployed, one of each model. On a central plot, one Reconyx camera trap (PC900 or HC600 model randomly chosen) was attached to a tree for the entire study period (sticks wedged between cameras and tree trunks were used to make fine-scale adjustments to camera positioning). We refer to this camera deployment as ‘stationary’. The second Reconyx, and a Swift 3C wide-angle camera trap were mounted separately to wooden plates which in turn were attached side-by-side to a wooden board using small right-angle brackets. The use of brackets and wooden plates allowed small up/down/left/right adjustments to fine-tune individual camera trap positioning. The wooden board with the cameras (left/right position randomly chosen, cameras approximately 1.5 cm apart) was mounted to a metal stake (Figure 4.2). This camera set up was repositioned approximately weekly (eight times) to a different plot (randomly chosen) within the same site. We refer to this camera deployment as ‘repositioned’ camera traps. We therefore had three camera trap groups: stationary Reconyx, repositioned Reconyx, and repositioned Swift camera traps. No bait or lures were used at camera trap stations to avoid possible bias associated with attractants. All camera traps were set central within plots with a minimum distance of 30 m to roads/tracks. Cameras were oriented towards south to avoid direct solar interference. To minimise obstruction, camera traps were aimed towards natural clearings. Vegetation was minimally trimmed in front of cameras (within the first 5 m), to reduce unintended camera activation by moving vegetation. Camera traps were set with their PIR motion sensor at approximately 25 cm above ground. To ensure that cameras were aiming parallel to the ground, we placed a square 15 x 15 cm white card at approximately 25cm above ground at 5 m distance to the cameras. We then attached a laser pointer to the bottom of the camera housings, pointing straight forward. Cameras were adjusted until the laser pointed to an appropriate height on the white card. Additional walk-tests were performed to ensure that cameras were detecting movement in front and beyond the 5 m distance. After adjustments were completed, the white card and laser pointer were removed, test images were retained, and the cameras were activated to operate 24 h/day.

Figure 4.2. Paired, side-by-side camera trap set up of periodically repositioned Reconyx (PC900, left), and Swift 3C wide-angle camera traps for the detection of wildlife in the Upper Warren region, Western Australia. Branches and leaves were used to conceal and disrupt the shape outline of camera traps and attachment structures.

All camera traps were set to high sensitivity with no delay between triggers. Reconyx camera traps were set to take 10 images per trigger in ‘Rapidfire’ mode. This setting was chosen to allow the comparison of results from stationary Reconyx cameras to another, unrelated camera trap study in the Upper Warren region (Anke Seidlitz unpublished data). Swift 3C camera traps were set to take three images per trigger. From pilot studies (Anke Seidlitz, unpublished data), we were aware that Swift 3C camera traps may have high false trigger rates, for example, caused by moving vegetation. We therefore chose the three-image-per-trigger setting to conserve battery life and data storage space. We acknowledge that this setting difference may disadvantage Swift 3C camera traps by having a smaller chance to ‘capture’ animals on fewer given images per trigger. Time and date settings were synchronised during set up to allow direct comparison of animal detections from cameras set side-by-side. Sites were visited approximately weekly to reposition camera traps to a new plot. During those visits, batteries and SD cards were checked, and replaced when necessary.

When an animal moves within a camera’s detection zone (defined here as an event), the PIR sensor may detect that animal, trigger the camera and result in one or more images depicting the animal partially or wholly. We defined this as a detection. For our repositioned camera traps, set side-by-side, an event may have resulted in an animal detection for one but not the other camera. We defined this as a missed detection for the camera which did not record the animal. Additional animal detections were counted only when detections of the same species were separated by at least three minutes. A three-minute quiet-time may not warrant independent animal detections which may be important when determining population parameters. As we were evaluating camera trap performance, we kept the quiet-time interval short to maximise detections, yet not too short to avoid excessive re-detection of single animals. For the comparison of paired, repositioned Reconyx and Swift camera traps, we used the white card position seen in test images to categorise detected animals’ distance from the camera as either more or less than 5 m. To determine if missed detections of repositioned Reconyx cameras were in- or outside the camera’s detection zone, we compared animal detection images of paired Swift and Reconyx cameras. Features seen within those images allowed us to approximate the lens angle (field of view) and detection zone of Reconyx camera traps (lens angle and detection zone overlap in Reconyx PC900/HC600 camera traps; Table 4.1).

At 16 sites, camera traps operated between 63 and 65 days (mean 64 days). At one site, the stationary Reconyx HC600 camera trap operated for 48 days; it failed to record images for 16 days for unknown reasons. Data recorded by repositioned Reconyx and Swift camera traps from the same site and time were excluded from analysis for unbiased comparison of camera trap groups. The remaining data, used for the analysis detailed below, consists of the number of detections from 3216 camera days (1072 camera days for each camera trap group).

4.4.5 Evaluating camera performance and data analysis

We chose to display results using three commonly used metrics (detection rates, detection probabilities, and species/site accumulation plots) to be useful to a wide audience with differing objectives. These metrics may display results differently. For example, detection rates are insensitive to the number of sites where species were detected, whereas detection probabilities relate to re-detections of species at individual sites. For species accumulation, only a single detection of a species is necessary, while site accumulation for individual species provides information on that species' spatial distribution within the study area. The following species were detected <5 times by all camera trap groups and were excluded from analyses: European rabbit (*Oryctolagus cuniculus*), Rosenberg's monitor (*Varanus rosenbergi)*, short-beaked echidna (*Tachyglossus aculeatus)*, domestic sheep (*Ovis aries)*, and feral pig (*Sus scrofa*). We were unable to identify small, mouse-sized mammals (e.g. house mouse, several dunnart species) to species level on some occasions, therefore they were grouped into one category (mouse sized).

All statistical analysis was conducted in R version 3.5.0 (R Core Team 2018). Likelihood-ratio tests in combination with generalised linear models (GLMs) were used to test if the number of animal detections were affected by camera trap group. We used the glm.nb function (negative binomial regression) of the R package MASS 7.3-51.4 (Venables and Ripley 2002) due to overdispersion issues with Poisson models. For this part of the analysis, cats (*Felis catus*) and foxes (*Vulpes Vulpes*) were grouped as feral predators as they had low detection rates. For the same reason, birds were grouped into two categories, consisting of large birds (>30 cm) and small birds (<30 cm), however, emus (*Dromaius novaehollandiae*) were listed separately due to their exceptionally large size. Birds and mammals for which species identification was not possible were excluded from analysis.

We first fitted GLMs with the number of animal detections as the response variable, and 'sites' (17 study sites) and 'species' as explanatory variables. The interaction between 'sites' and 'species' was also included as species abundance varied between study sites. Prior to assessing camera trap groups, we included the different Reconyx models (Reconyx PC900 and HC600) into GLMs and used likelihood-ratio tests to verify that these models did not statistically differ in their ability to detect animals. Thereafter, data from the two Reconyx models were combined. We then fitted GLMs with 'camera trap group' as an additional explanatory variable to determine if there was a difference in camera trap group performance. We first compared repositioned Reconyx and Swift camera traps, and finally stationary Reconyx and repositioned Reconyx camera traps. When exploring differences between animal detection rates from camera trap groups for single species, we used the above described procedure for likelihood-ratio tests except that the covariate 'species' became redundant.

To determine the probability of detection of mammal species and bird groups for each camera trap group, we used the single season occupancy modelling framework (specified in MacKenzie *et al.* 2018). For each species/group, a matrix with detections (1) and non-detections (0) was established from spatial replicates (17 sites) and temporal repeats (camera days). Days on which camera traps did not operate were included in the data matrix as missing observations. Models to estimate detection probabilities were fitted using the RPresence package 2.12.33 (MacKenzie and Hines 2018). The occupancy component of models was kept constant (psi~1), and the detection probability for each species was determined for each camera trap group. We accessed model fit by estimating c-hat and χ^2 goodness of fit tests from 5000 bootstrap iterations and found no issues (data not shown). To graphically display detection probabilities, we used the R package 'forestplot' version 1.9. (Gordon and Lumley 2019).

We used the Vegan Community Ecology package version 2.5-5 (Oksanen *et al.* 2019) to compute the accumulation of species detected, and the accumulation of sites where selected species were detected over time by each camera trap group. For site accumulation, the site which experienced camera trap failure was excluded. The method "random" was applied which finds the mean accumulation curve and its standard deviation from 100 random permutations of the data.

4.5 Results

Across all camera deployments, there were 6095 animal detections (repositioned Reconyx=1468, repositioned Swift=3201, stationary Reconyx=1426). We identified 16 different mammals, one reptile, and 17 bird taxa (grouped as birds <30 cm and >30 cm). Animal groups, species and respective detection rates per 1000 camera trap days are listed in Table 4.2.

Table 4.2. Animal detection rates of three camera trap groups

Observed animal detection rates of three camera trap groups: Stationary Reconyx (PC900/HC600), and periodically repositioned paired Reconyx (PC900/HC600) and Swift 3C wide-angle camera traps (n=17 each). Cameras operated simultaneously at 17 sites over approx. nine weeks in the Upper Warren region, Western Australia. Significance codes (sig. codes) relate to adjacent detection rates, indicating a difference in species detection rates between camera trap groups: '***'= <0.001, '**'= <0.01, '*'= <0.05, '.' = <0.10. Animals were ordered by maximum body weight.

		Detections per 1000 trap days				
Categories	**Scientific name**	**Reconyx stationary**	**versus (sig. code)**	**Reconyx repositioned**	**versus (sig. code)**	**Swift repositioned**
Animals <1 kg		**81.9**		**76.4**		**362.2**
birds (<30 cm)		6.5		8.4	***	111.7
Mouse sized		0.0		0.9	***	39.1
Brush-tailed phascogale	*Phascogale tapoatafa*	0.0		0.0	***	11.2
birds (>30 cm)		67.0		38.2	***	75.4
birds (no ID possible)		1.9		0.0		56.8
Numbat	*Myrmecobius fasciatus*	6.5	***	28.9	***	68.0
Animals 1-10 kg		**1123.8**		**1144.3**		**2290.5**
Western ringtail possum	*Pseudocheirus occidentalis*	5.6		4.7	.	11.2
Brush-tailed bettong	*Bettongia penicillata*	463.7	*	578.2	***	1026.1
Quenda	*Isoodon fusciventer*	20.5		29.8		32.6
Western Quoll	*Dasyurus geoffroii*	21.4		22.3	.	35.4
Common brushtail possum	*Trichosurus vulpecula*	478.6	*	367.8	***	740.2
Tammar wallaby	*Macropus eugenii*	39.1		54.9		68.0
Feral cat	*Felis catus*	5.6		4.7		5.6
European red fox	*Vulpes vulpes*	5.6		6.5		12.1
Western brush wallaby	*Macropus irma*	27.0		27.0	***	55.9

		Detections per 1000 trap days				
Categories	**Scientific name**	**Reconyx stationary**	**versus (sig. code)**	**Reconyx repositioned**	**versus (sig. code)**	**Swift repositioned**
Mammals (no ID possible)		56.8		48.4		300.7
Animals >10 kg		**118.2**		**143.4**		**324.0**
Emu	*Dromaius novaehollandiae*	42.8	**	77.3	***	136.9
Western grey kangaroo	*Macropus fuliginosus*	75.4		66.1	***	186.2
False triggers (e.g. wind)		**160.1**		**208.6**		**7196.5**

From repositioned Reconyx and Swift camera traps set side-by-side, we determined that there were 3218 known events of which Swift cameras did not detect 17 (0.5%), and Reconyx cameras did not detect 1750 (54%). Of the events missed by Reconyx cameras, most (76%) lay within the detection zone of the cameras (Figure 4.3). A total of 436 unidentifiable mammals were detected, with repositioned Swift cameras recording 323 (248 at night at > 5m distance), repositioned Reconyx cameras 52 (48 at night at < 5 m distance), and stationary Reconyx cameras 61 (58 at night at < 5m distance). There were 8125 cases where camera traps triggered for unknown reasons, including suspected triggers due to moving vegetation (repositioned Reconyx=224, repositioned Swift=7729, stationary Reconyx=172).

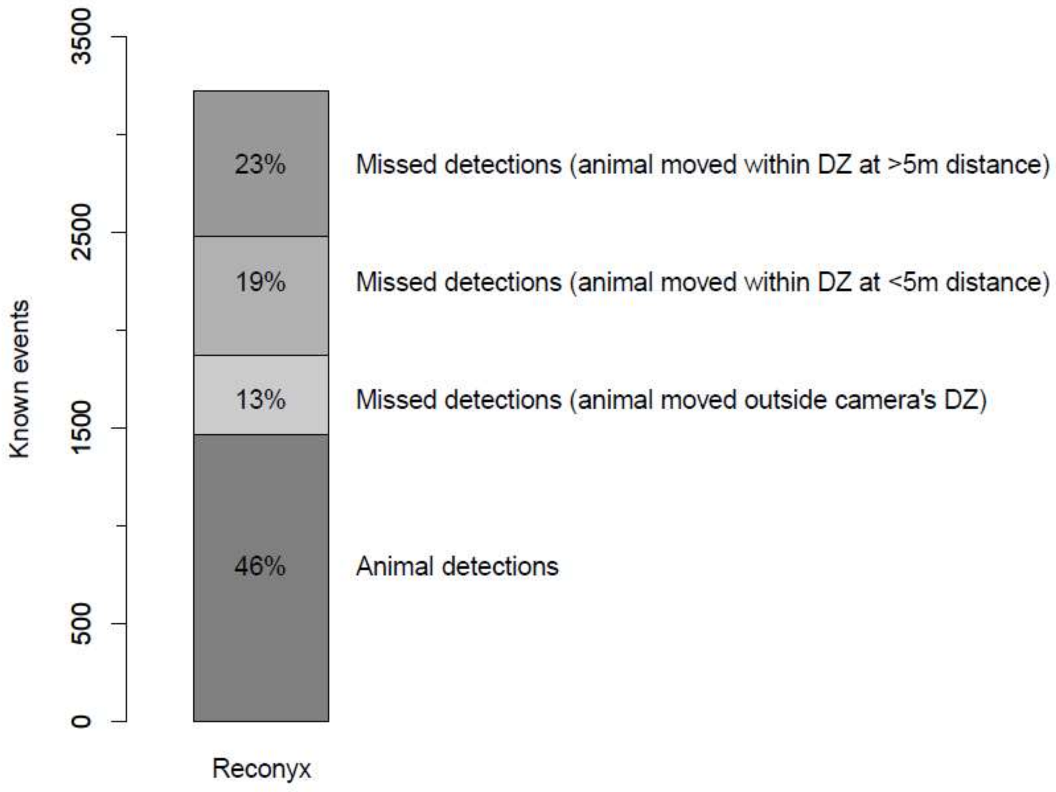

Figure 4.3. Percentage of known events (3218) recorded and missed by periodically repositioned Reconyx PC900/HC600 camera traps (n=17) from a side-by-side comparison with Swift 3C wide-angle camera traps (n=17). Swift camera traps missed 17 events (not displayed). Camera traps operated at 17 sites over approximately nine weeks within the Upper Warren Region, Western Australia. Percentages were rounded to nearest whole number for display purposes. DZ=detection zone.

4.5.1 Detection rates

Using overall animal detections, the likelihood-ratio test to determine if including camera model (Reconyx HC600 and PC900) as a factor resulted in an improved model was not significant (χ^2=0.477, *df*=1, *p*= 0.490). Conversely, when periodically-repositioned Reconyx and Swift camera traps were included, the likelihood-ratio test indicated a significant difference between the camera trap group's ability to detect animals (χ^2=239.486, *df*=1, *p*=<0.001). There was no improvement to model fit when repositioned and stationary Reconyx camera traps were included in models (χ^2=0.610, *df*=1, *p*=0.435). Significance codes are displayed in Table 4.2, indicating differences between camera groups in their ability to detect single species. When compared to Reconyx camera traps, detections from Swift 3C cameras were significantly higher for 3 out of 9 species of the 1-10 kg group, and for all species of the <1 kg and >10 kg groups. In no instance did Reconyx camera traps have higher detection rates than Swift camera traps. There were four significant differences between detections from stationary and repositioned Reconyx camera traps with repositioned cameras having higher detections for numbats, brush-tailed bettongs and emus, and stationary cameras for common brushtail possums.

4.5.2 Detection probabilities

Swift 3C wide-angle camera traps had the highest detection probabilities for all species <1 kg and >10kg body weight, with confidence intervals noticeably overlapping only for numbats and birds >30cm (Figure 4.4). Swift cameras detected mouse-sized mammals and brush-tailed phascogales at eight and nine sites respectively (data not displayed). For those species, detection probabilities could not be computed for Reconyx camera traps as there was only one detection of mouse-sized mammals and none of brush-tailed phascogales. For numbats, brush-tailed bettongs, and emus, repositioned camera traps achieved higher detection probabilities than stationary camera traps. Differences between detection probabilities of camera trap groups were less marked for animals with a body weight between 1-10 kg and most confidence intervals overlapped (Figure 4.4). However, for most species in this size group, Swift camera traps tended to have slightly, but insignificantly, higher detection probabilities than Reconyx camera traps. For the brush-tailed bettong and common brushtail possum, Swift detection probabilities were clearly higher, however, for the quenda (*Isoodon fusciventer*) and feral cat there was no discernible difference.

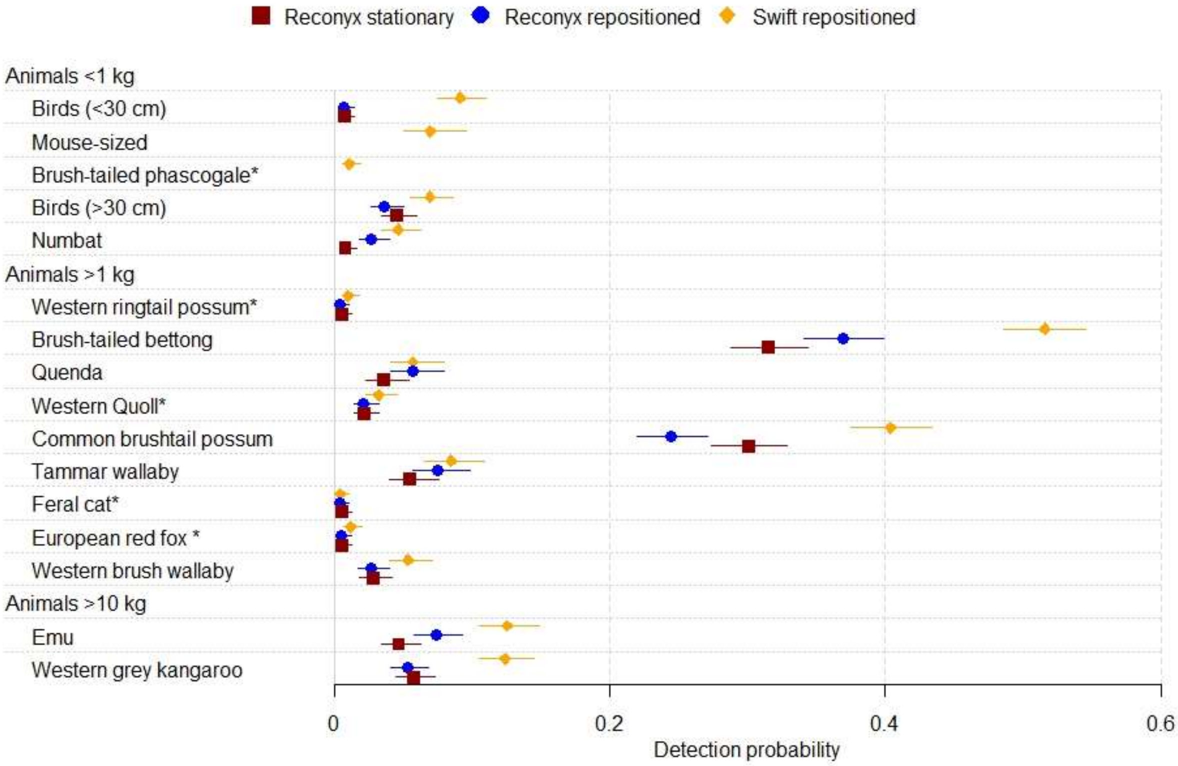

Figure 4.4. Animal detection probabilities with 95% confidence intervals of three camera trap groups: Stationary Reconyx (PC900/HC600), and paired, periodically repositioned Reconyx (PC900/HC600) and Swift 3C wide-angle camera traps (n=17 each). Cameras operated simultaneously at 17 sites over approximately nine weeks within the Upper Warren Region, Western Australia. Animals were ordered by maximum body weight. Detection probabilities for indicated species (*) may be unreliable due to low detection rates.

4.5.3 Species and site accumulation rates

Repositioned Swift 3C camera traps detected animal species at a faster and higher rate than stationary, and repositioned Reconyx camera traps (Figure 4.5a). This difference mainly derived from animals with a body weight of <1 kg (Figure 4.5b-c), which included 17 bird species (Table S4.1). Site accumulation rates for mammal species were never lower for Swift 3C wide-angle camera traps than for stationary and repositioned Reconyx camera traps. Examples of site accumulation curves are given in Figure 4.6a-f. Site accumulation for Quenda and brush-tailed bettong was similar for all camera trap groups (Figure 4.6a-b), while western grey kangaroos were detected at a faster rate and at more sites by Swift 3C wide-angle camera traps. Site accumulation rates for numbats differed between all camera trap groups, with Swift 3C wide-angle cameras traps showing markedly higher accumulation at faster rates than the others (Figure 4.6d). Numbats, western quolls and tammar wallabies were detected at more sites by repositioned Reconyx and Swift camera traps compared to stationary Reconyx cameras (Figure 4.6d-f).

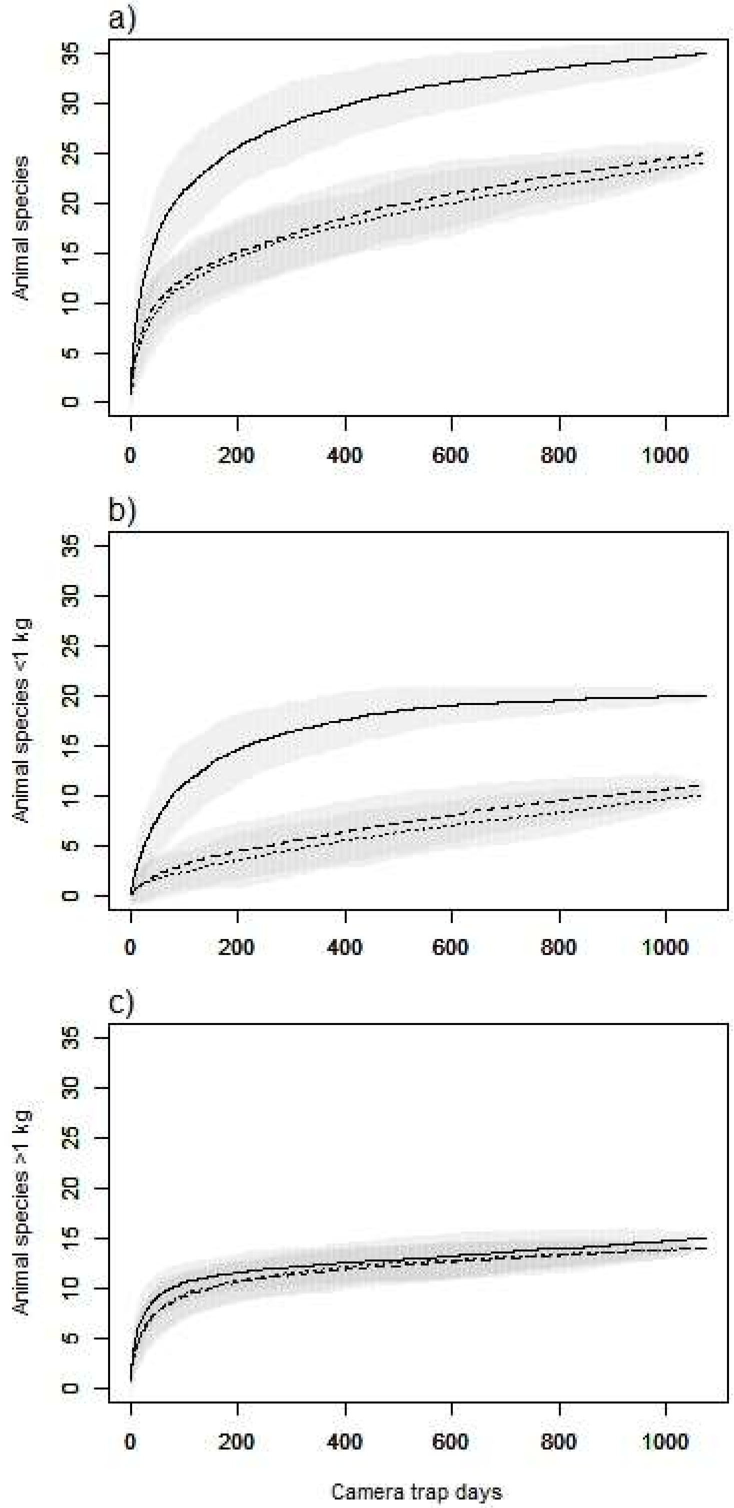

Figure 4.5. Cumulative number of a) all species, b) species <1 kg, c) species >1 kg detected by three camera trap groups that operated simultaneously at 17 sites within the Upper Warren Region, Western Australia. Stationary Reconyx PC900/HC600 camera traps (n=17, dotted line), and periodically repositioned Reconyx PC900/HC600 (dashed line) and Swift 3C wide-angle (solid line) camera traps set side-by-side (n=17 each).

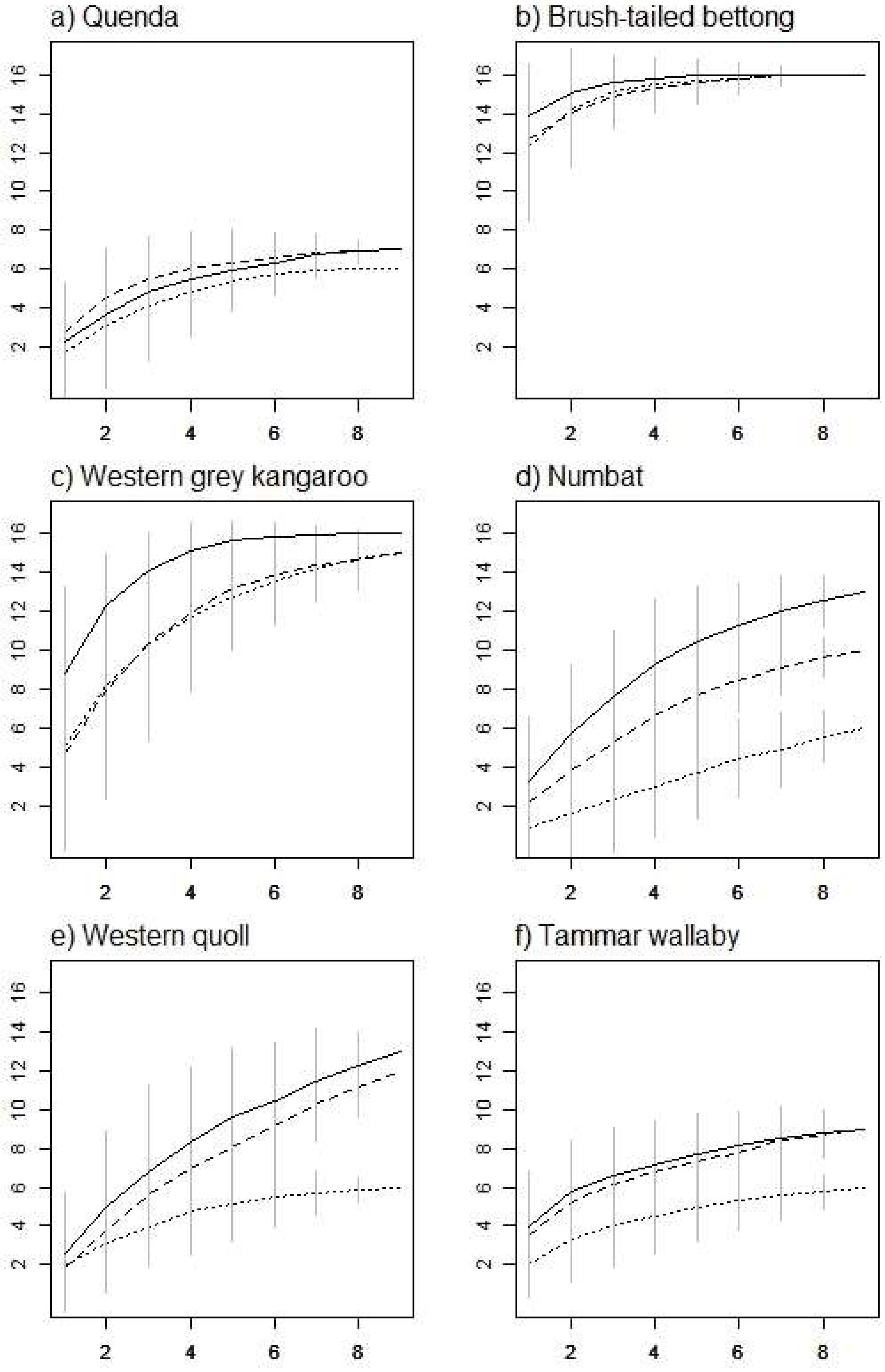

Figure 4.6. Cumulative number of sites at which species were detected by three camera trap groups which operated simultaneously at 16 sites over nine weeks within the Upper Warren Region, Western Australia. Stationary Reconyx PC900/HC600 camera traps (dotted line), and periodically repositioned Reconyx PC900/HC600 (dashed line) and Swift 3C wide-angle (solid line) camera traps set side-by-side (n=16 each).

4.6 Discussion

This study compared the performance of stationary Reconyx PC900/HC600, and paired, periodically-repositioned Reconyx PC900/HC600 and Swift 3C wide-angle camera traps. Compared to Reconyx PC900/HC600 camera traps, Swift 3C wide-angle camera traps had higher species accumulation rates. Swift camera traps had also significantly higher animal detection rates that lead to higher detection probabilities and site accumulation for many species, particularly within the <1kg and >10kg categories. Reconyx camera traps missed 54% of detections recorded by paired Swift camera traps. Of those, only 24% were caused by the smaller detection angle of Reconyx cameras. Contrary to our expectations, stationary and periodically-repositioned Reconyx PC900/HC600 camera traps performed similarly except for some species. This study shows that some camera traps (here set without lures or baits) may miss a high percentage of detectable animal movements and highlights the importance of selecting an appropriate camera trap model for wildlife detection studies.

4.6.1 Camera trap detection angle

Detection angle size was hypothesised to be an important determinant for higher detection rates of rabbits (*Oryctolagus cuniculus*) from Ltl Acorn Ltl-5310A camera traps when compared to Reconyx PC900 cameras (Fancourt *et al.* 2018). However, the authors did not report if missed detections derived from rabbit movement within or outside the detection zone of Reconyx cameras, so it remains unclear if missed detections were caused by a smaller detection angle or other differences between the camera models. When comparing Swift 3C camera trap models (wide-angle versus standard), wide-angle cameras were shown to have higher numbat detection rates in a trial conducted in zoo enclosures (Seidlitz *et al.* 2020). Yet the increase in detections may have been amplified by non-random movements of numbats in zoo enclosures (Seidlitz *et al.* 2020). During this study, the smaller detection angle of Reconyx PC900/HC600 caused missed detections, however, it was not the main determinant of the cameras' lower detection rates. The majority of missed detections from Reconyx cameras derived from animal movement events within the cameras' detection zone. Therefore, other model differences must be considered. Faster trigger speed did not cause higher detection rates from Swift camera traps as they have slower trigger speeds than Reconyx PC900/HC600 camera traps. Other factors possibly causing performance differences between Swift 3C wide-angle and Reconyx PC900/HC600 camera traps are the temperature differential threshold of PIR sensors, and the number and characteristics of Fresnel lenses that condense infrared radiation onto the sensor (see Welbourne *et al.* 2016 for information on camera trap functionality). Both affect the sensitivity of camera traps. High camera trap sensor sensitivity was also found to improve animal detection rates in a study comparing customised high-sensitivity Reconyx PC850 models to their unmodified counterparts (Heiniger and Gillespie 2018). One disadvantage of higher PIR sensitivity are the increased occurrence of false triggers caused, for example, by moving vegetation. During this study, the number of false triggers from Swift camera traps were manageable, and the importance of improved data accuracy outweighed this disadvantage. Artificial intelligence technologies may also render false triggers easily excluded (e.g. Yu *et al.* 2013; Gomez Villa *et al.* 2017; Falzon *et al.* 2020).

4.6.2 Animal size

With increasing body size, animals are more easily detected by camera traps (Wearn and Glover-Kapfer 2017). During comparative studies, it was found that differences between camera trap model performance typically reduced with increasing animal size (Swan *et al.* 2014a; Urlus *et al.* 2014; but see Damm *et al.* 2010). During this study, differences between Swift and Reconyx camera traps were more pronounced for animals <1 kg and >10 kg, and this was evident from detection rates, detection probabilities and accumulation curves. The PIR sensor of Reconyx PC900/HC600 models may potentially be less sensitive than that of Swift 3C camera traps, causing the reduced detection of animals <1 kg. Reduced detection of this weight class may not occur with a targeted camera trap set up using bait (see e.g. Meek and Vernes 2016). However, this finding is still important, as some small animals, such as the numbat, can't be attracted by bait (Burrows and Christensen 2002), and a targeted camera trap set up may not be suitable when exploring multiple species of different size classes. The performance difference between Reconyx and Swift camera models for animals >10 kg weight may derive from the Swift camera trap's ability to detect large animals at a greater distance than the tested Reconyx models. Although present, differences in detection rates between the tested camera models weren't as pronounced for animals between 1-10 kg body weight, and significantly greater detection by Swift cameras appeared to be restricted to a few species only. Without further, targeted studies it is difficult to speculate on why some species in this size class were detected differently by the camera models and others weren't.

4.6.3 Periodic repositioning of camera traps

Contrary to our expectations, periodically repositioning of camera traps within sites did not increase overall animal detection rates. This may be due to the choice of always aiming camera traps at areas of natural clearings to reduce false triggers from moving vegetation. Sampling more randomly across more heterogenic habitat features may increase the detection of some species (Swan *et al.* 2014b; Kolowski and Forrester 2017; Hofmeester *et al.* 2019). Therefore, to truly avoid camera trap placement bias, cameras need to be genuinely placed randomly, even at the micro-habitat scale to include features such as logs, dense vegetation, and water bodies. Species which may have occurred (see Wayne *et al.* 2017 for list of species), but weren't detected during this study were the rakali or water rat (*Hydromys chrysogaster*), southern bush rat (*Rattus fuscipes*), and introduced black rat (*Rattus rattus*). Mammals smaller than rats were detected (grouped as 'mouse sized') but since it was not possible to identify these species, we are unsure if species such as the western pygmy possum (*Cercartetus concinnus*) and mardo (*Antechinus flavipes*) were detected. To improve small mammal species identification, a more targeted camera trap placement with bait may be required (see e.g. Gray *et al.* 2017; Gracanin *et al.* 2019).

Periodic repositioning of camera traps improved detection rates for numbats, brush-tailed bettongs, and emus. Furthermore, the repositioning of camera traps improved detection probabilities for numbats, brush-tailed bettong and emus, and site accumulation for numbats, western quoll, and tammar wallabies. The sampling of a wider range of habitat features may have caused this increase. However, animal behaviour could also have been a reason for this increase (or decrease in the detections of common brushtail possums, and other taxa). Some animals are known to be repelled or attracted by camera traps (Séquin *et al.* 2003; Meek *et al.* 2016a). Animals repelled by camera traps may, after detecting the device (which does not necessitate the detection of the animal by the camera), avoid the camera station area. Further investigation of periodically-repositioned camera traps could reveal if detection rates of camera trap-shy animals can be increased. If animals actively avoid/seek camera traps, assumptions of animal detections being random and independent may be violated: a concern raised by Meek *et al.* (2016a) and Larrucea *et al.* (2007).

While not directly investigated in this study, our results show that, for most species, there is no disadvantage to periodically relocating cameras. By using this technique, additional spatial and micro-habitat information can be obtained. Therefore, for studies where this additional information may be important (e.g. Spatial Capture-Recapture and habitat use studies), periodically moving cameras may be advantageous and worth further investigation.

4.6.4 Camera trap model choice

This study did not investigate the durability or longevity of camera trap models. It is our impression that Swift 3C wide-angle camera traps are not as robustly built as Reconyx PC900/HC600 camera traps. However, Swift 3C camera traps operated reliably during this study (9-week deployment) without failure. Which camera trap model is chosen for a project depends on many factors such as research objectives, camera trap detection efficiencies, occurrence of false triggers, model durability and longevity, camera purchase cost, and operating and servicing times.

4.7 Conclusion

Swift 3C wide-angle camera traps detected animals of differing sizes, particularly within the <1kg and >10kg categories, more successfully than Reconyx (PC900/HC600) camera traps. This led to higher species accumulation rates, improved detection probabilities and site accumulation rates for many species. It is the increased sensitivity of the Swift 3C PIR sensor that plays an important role in its success, along with the wide-angle lens and detection zone. These are important outcomes for studies with a focus on species-level as well as community-level questions. Repositioning camera traps periodically within sites did not increase overall detection rates, however, for some species this technique appeared to be beneficial. When choosing camera trap models for wildlife detection, detection angle and PIR sensor sensitivity need to be considered to produce reliable study results. Periodically repositioning cameras within sites is a technique that needs further investigation as it may reduce camera placement bias, animal avoidance of camera traps, and increase spatial/habitat information when a limited number of cameras are deployed.

4.8 Acknowledgements

We acknowledge the Nyoongar people as the Traditional Custodians of the lands and animals on which this study was conducted. This study was conducted using permit O2968/17 from the Murdoch University Animal Ethics Committee. We thank all volunteers and co-workers for their invaluable help with this project. We are also thankful to the Department of Biodiversity, Conservation and Attractions for providing camera traps, accessories, shapefiles for area maps, and other in-kind contributions. Furthermore, we would like to thank WWF-Australia for obtaining and managing project funds. We are grateful to two anonymous reviewers who kindly reviewed and helped to improve the manuscript. This project has received funding from the Western Australian Government's State NRM Program (A16002), Murdoch University, and Coles Supermarkets Australia Pty Ltd. Many thanks also to Leeanne and Neil Chapman for providing a home away from home to the field team.

4.9 Conflicts of interest

The authors declare no conflicts of interest.

4.10 Supporting information

Table S4.1. List of bird species confidently identified

Common name	Scientific name
Yellow-rumped thornbill	*Acanthiza chrysorrhoa*
Australian owlet-nightjar	*Aegotheles cristatus*
Australian ringneck parrot	*Barnardius zonarius*
Rufous treecreeper	*Climacteris rufus*
Grey shrike-thrush	*Colluricincla harmonica*
Australian raven	*Corvus coronoides*
Laughing kookaburra	*Dacelo novaeguineae*
Western yellow robin	*Eopsaltria griseogularis*
Australian magpie	*Gymnorhina tibicen*
Fairy wren	*Malurus sp.*
Golden whistler	*Pachycephala pectoralis*
Scarlet robin	*Petroica boodang*
Common bronzewing	*Phaps chalcoptera*
Western rosella	*Platycercus icterotis*
Tawny frogmouth	*Podargus strigoides*
Grey fantail	*Rhipidura albiscapa*
Grey currawong	*Strepera versicolor*

Chapter 5 Comparison of numbat detections from sign surveys conducted in spring and autumn and Swift 3C wide-angle camera traps

5.1 Preface

This chapter comprises original research undertaken to compare the effectiveness of (1) sign surveys conducted during different seasons (spring vs autumn) and (2) sign surveys and Swift 3C wide-angle camera traps. In Chapter 3, detection rates from sign surveys were compared to those from Reconyx PC900/HC600 camera trap models. Since Chapters 2 and 4 found that Swift 3C wide-angle camera traps detected numbats more effectively than Reconyx PC900 camera traps, an additional comparison was warranted. This chapter is kept short to avoid excessive repetition since much of the information was already covered in previous chapters.

A numbat (*Myrmecobius fasciatus*), captured by a Swift 3C wide-angle camera trap in the Upper Warren region, south-western Australia.

5.2 Introduction

Important considerations when designing wildlife monitoring programs are survey timing (season) (Read *et al.* 2015) and the selection of optimal survey methods (MacKenzie 2005). Juvenile numbats disperse around December, the start of summer (Friend and Burrows 1983). Numbat surveys are often conducted during spring and summer (see Friend and Thomas 2003) to increase detection of this rare species. However, when repeat surveys for occupancy studies are conducted during juvenile dispersal, the assumption that the population is closed – that is, the site occupancy status remains the same during the survey season – may be violated. The best season for conducting numbat sign surveys for the purpose of occupancy modelling may therefore be autumn before the start of winter rains that may rapidly erode numbat signs.

Camera traps are an attractive survey tool (advantages of camera traps were listed in Chapters 1 and 3), and since it is already established that Swift 3C wide-angle camera traps detect numbats more effectively than Reconyx PC900 camera traps (see Chapter 2), it was important to compare numbat detection rates from Swift 3C wide-angle camera traps to those of sign surveys.

This study aimed to determine if numbat sign surveys conducted in autumn were as successful as those conducted in spring. The second aim was to compare numbat detection rates of periodically repositioned Swift 3C wide-angle camera traps to those of autumn sign surveys.

5.3 Methods

Five sign surveys were conducted during the Australian autumn from mid-March to mid-April 2018. Methods used for sign surveys were as described in Chapter 3 (except for survey period and number of repeat surveys, as mentioned above) including the use of the 50 sites across three transects. Results from autumn sign surveys were compared to (1) the results of spring sign surveys (seven repeat surveys in September to December 2017; Chapter 3) and (2) the results from periodically repositioned Swift 3C wide-angle camera traps set at 17 sites for 9 weeks (mid-March to mid-May 2018; Chapter 4). For the second comparison, a subset of the 50 sites for autumn sign surveys was used, using the same 17 sites that contained the Swift 3C wide-angle camera traps.

5.3.1 Data analysis

For data analysis, R version 3.5.0 (R Core Team 2020) was used. A binomial GLM was fitted with the covariates 'season', 'site' and 'survey' to determine if the number of numbat sign detections (as defined in Chapter 3) differed between seasons (spring and autumn). To determine if the inclusion of the covariate of interest 'season' was improving model fit, an ANOVA chi-square test was used. The Vegan Community Ecology package version 2.5-5 (Oksanen *et al.* 2019) was used for the accumulation of sites from spring and autumn surveys at 50 sites with numbat sign detections. The method 'random' was applied, which finds the mean accumulation curve and its standard deviation from 100 random permutations of the data.

5.4 Results

During five repeat autumn sign surveys, 250 plots were searched for numbat signs at 50 survey sites. Numbat scats and/or diggings were found on 128 plots (51%) at 41 sites (82%). Sign surveys conducted in autumn resulted in more numbat sign detections compared to those conducted in spring (Figure 5.1), and this difference was significant (*P*-value < 0.001).

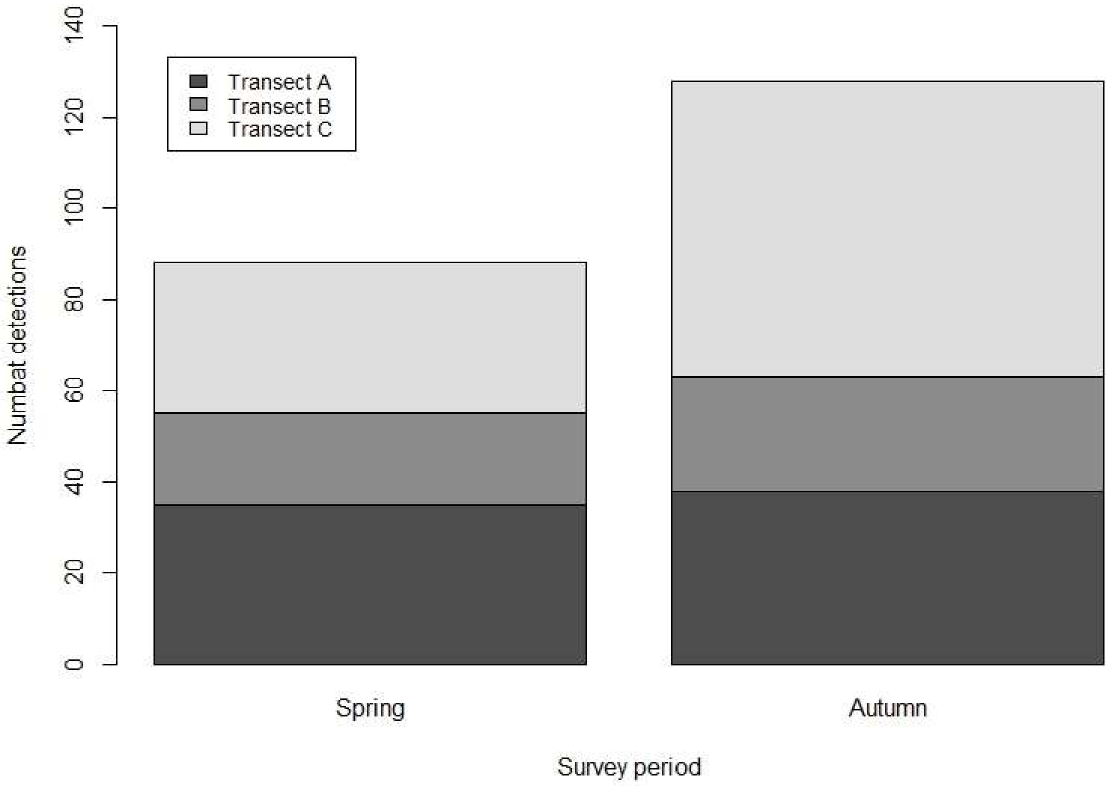

Figure 5.1. Number of numbat sign detections from sign surveys conducted in spring 2017 (seven repeat surveys) and autumn 2018 (five repeat surveys) at 50 survey sites along three transects. Detections refer to the number of sign searches during which numbat scats or diggings were found. Surveys were conducted in the Upper Warren region, Western Australia.

The accumulation of sites with numbat signs found was faster and slightly higher from sign surveys conducted in autumn than spring, during which signs were found at 39 sites (78%; Figure 5.2).

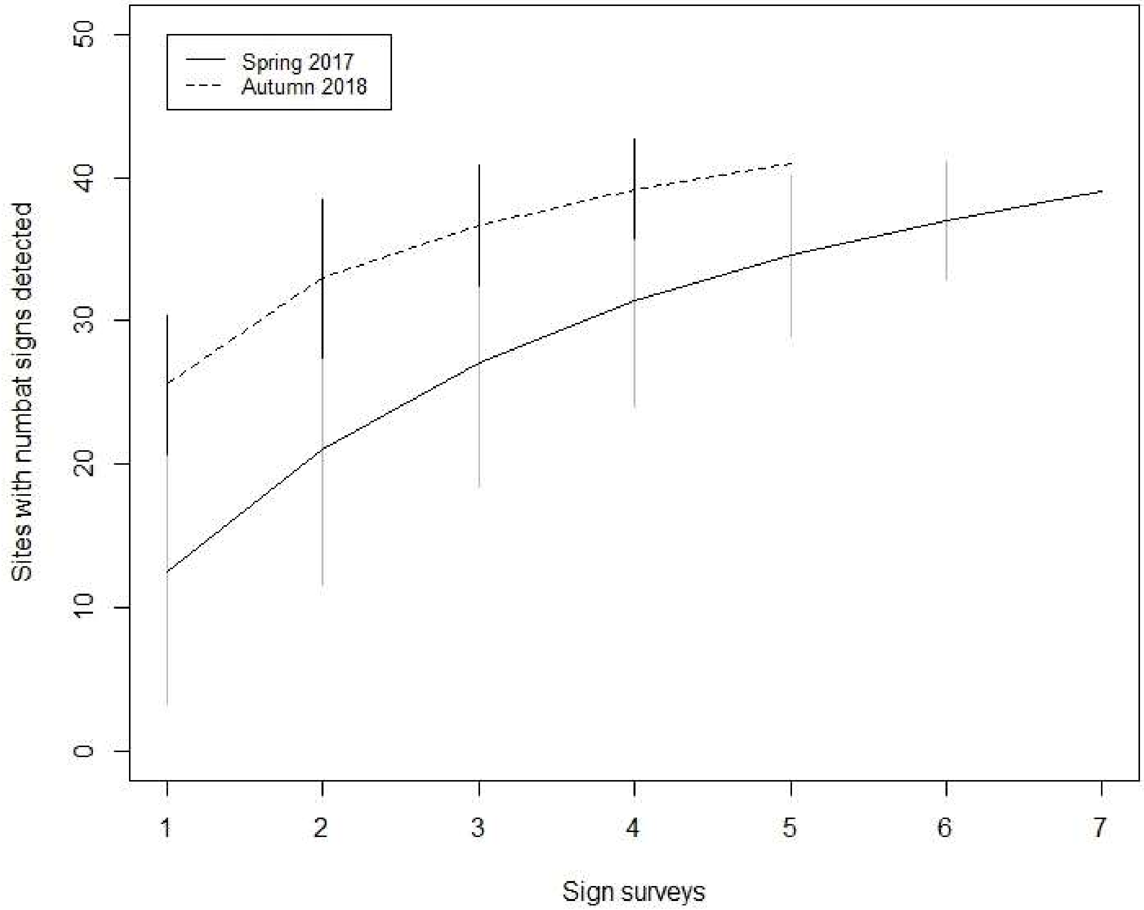

Figure 5.2. Cumulative number of sites, with 95% confidence interval, at which numbat signs were detected by sign surveys conducted in different seasons. Surveys were carried out at 50 sites within the Upper Warren region, Western Australia.

Site accumulation rates for numbats differed between autumn sign surveys and Swift 3C wide-angle camera traps, with sign surveys showing markedly higher site accumulation at faster rates than camera traps (Figure 5.3). Numbat signs were found at 16 sites after three sign surveys, while camera traps detected numbats at 13 sites after 9 weeks (Figure 5.3). Note: It took 1.5 days to complete one sign survey at each of the 17 sites.

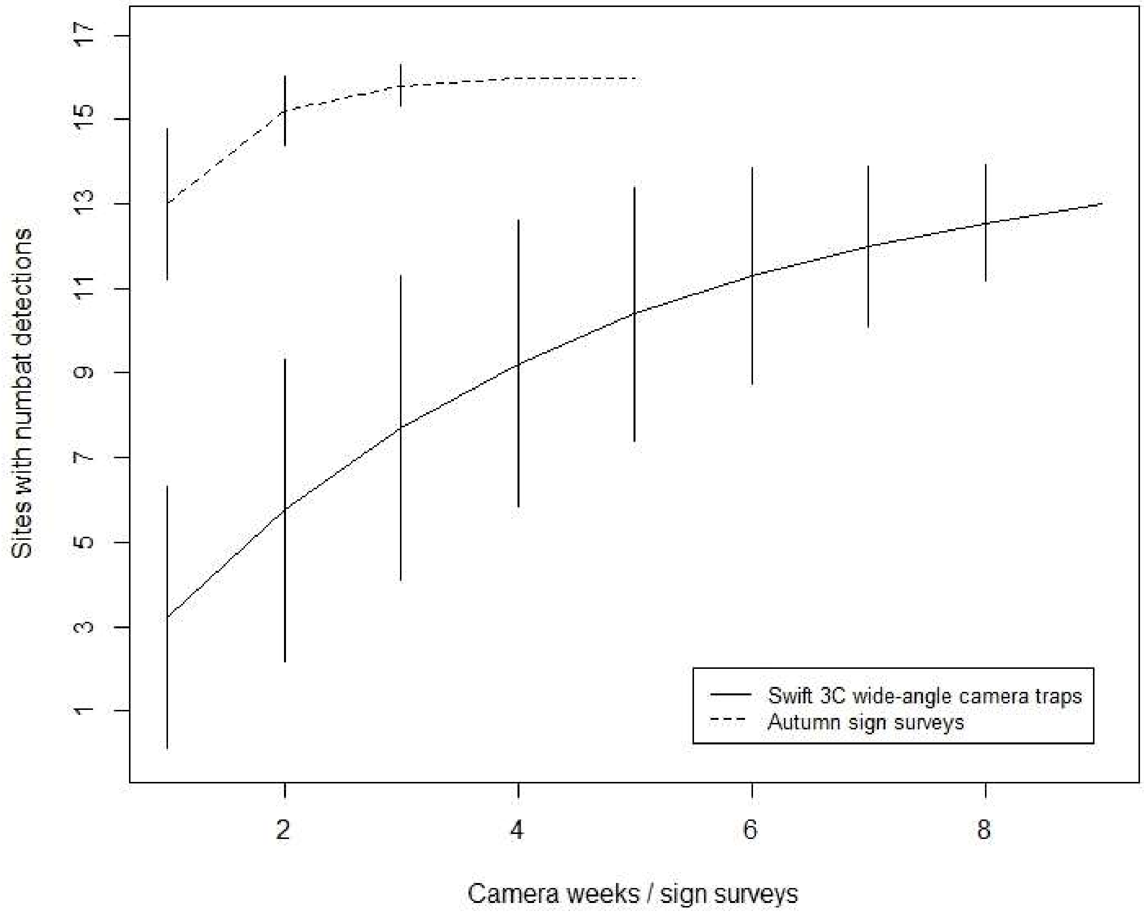

Figure 5.3. Cumulative number of sites, with 95% confidence interval, at which numbat signs were detected by autumn sign surveys, and numbat images were recorded by Swift 3C wide-angle camera traps. Sign surveys (consisting of five repeat surveys at each site) were conducted from mid-March to mid-April 2018, and camera trapping (continuous) from mid-March to mid-May 2018. Surveys were carried out at 17 sites within the Upper Warren region, Western Australia.

5.5 Discussion

Autumn surveys detected significantly more numbat signs than spring surveys. Sign surveys conducted in autumn may be more effective than those conducted in spring due to the dry weather over summer/autumn months that may allow numbat signs to persist longer and accumulate in the environment than during wetter winter/spring months. An accumulation of numbat signs would make them more detectable to observers. However, there are a number of other possible reasons for the higher detection rates as discussed below.

Observer ability to detect numbat signs may also have played a role in the success of autumn surveys. With increasing practice from repeat surveys, the author, who conducted all surveys, may have become more familiar with the range of numbat signs (across spring 2017 surveys followed by autumn 2018 surveys) and would therefore have had an increased ability in autumn to detect them. This increased ability would positively affect volunteer training that was conducted by the author, resulting in a more skilled survey team.

It may also be by chance that autumn sign surveys were more successful than spring sign surveys. Since there was no repetition of surveys over several autumn and spring seasons, no solid conclusions can be drawn regarding survey success. However, results from this trial may indicate that there is no disadvantage of conducting numbat sign surveys during autumn seasons. One advantage of conducting sign surveys in autumn is that closure assumptions are less likely to be violated than in spring when juvenile numbats disperse.

Similar to the results from Chapter 3, sign surveys resulted in a faster and higher accumulation of sites found with evidence of numbat presence than camera trapping. Even though numbat detection rates significantly improved by using Swift 3C wide-angle camera traps compared to Reconyx PC900/HC600 camera traps (see Chapter 2), autumn sign surveys were markedly more successful than camera traps during this study at 17 survey sites.

5.6 Conclusion

This study did not find evidence that numbat sign surveys conducted in autumn are less successful than those conducted in spring. Since population closure assumptions are less likely to be violated when using autumn surveys for numbat studies using occupancy modelling, we recommend conducting numbat sign surveys in autumn rather than spring. Furthermore, autumn sign surveys detected numbat presence more successfully than periodically repositioned Swift 3C wide-angle camera traps. This reaffirms the conclusions drawn in Chapter 3, and we continue to recommend numbat sign surveys over camera trapping for long-term monitoring of numbats in the UWR.

Chapter 6 Habitat use of the numbat (*Myrmecobius fasciatus*) in relation to habitat characteristics and forest management practices

6.1 Preface

This final research chapter applies sign surveys with occupancy modelling to determine habitat preferences of numbats in the UWR. This chapter aims to inform the management concerned with the conservation of the UWR numbat population how current forest management actions affect the species.

In the previous chapter, it was established that an autumn timing for sign surveys is advantageous; fieldwork for this chapter was conducted in spring/summer due to logistic reasons.

Example of jarrah (*Eucalyptus marginata*) forest habitat in the Upper Warren region, south-western Australia.

Example of wandoo (*Eucalyptus wandoo*) woodland habitat in the Upper Warren region, south-western Australia.

6.2 Abstract

Understanding how habitat characteristics and forest management practices affect wildlife is key for threatened species conservation. We investigated how habitat characteristics such as ground cover, number of log hollows, forest type, timber harvesting, time since last fire and intensity of fox control affect numbat (*Myrmecobius fasciatus*) habitat use in the Upper Warren region (UWR), south-western Australia. We used numbat signs (scats and diggings) together with occupancy modelling to determine important habitat factors. Numbat signs were found at 65/78 sites, resulting in a naïve occupancy of 83%. Numbat signs were found in various habitat categories, indicating that in the UWR, the species is a habitat generalist. Numbat habitat use was positively related to the number of logs, which was higher at timber-harvested sites. The probability of detecting numbat signs increased during later surveys. Results from this study indicate that the UWR numbat population is larger than previously thought, potentially exceeding 2000 numbats. We recommend further studies to determine the population's distribution south of the study area and to ensure the retention of logs during forest management activities to improve the conservation of this species.

6.3 Introduction

Disturbances are critical components of many ecosystems, affecting ecosystem structure and function and the dynamics of wildlife communities and populations within (Pickett and White 1985). Species have adapted to natural disturbance regimes; however, when anthropogenic activities alter or add disturbances, ecosystem function may temporarily or permanently be disrupted, frequently causing wildlife populations to decline. How anthropogenic activities affect wildlife populations is often of interest to land managers concerned with wildlife conservation. In some areas, disturbances co-occur (Burton *et al.* 2020), and this can be observed in jarrah forests of the UWR in South West WA that are harvested for timber, burned to reduce fuel loads and have a population of introduced foxes (*Vulpes vulpes*) (Wayne *et al.* 2013a).

Disturbance from timber-harvest operations include soil compaction (Whitford and Mellican 2011) and the felling and removal of trees (Abbott and Williams 2011). The effects of timber harvest on mammals have received considerable attention worldwide (Zwolak 2009;

Bogdziewicz and Zwolak 2014; Costantini *et al.* 2016; Lavery *et al.* 2020) and in Australia (Norton and May 1994; Calver and Wardell-Johnson 2004; Lunney *et al.* 2009). Several studies have explored the effects of timber harvest in the jarrah forest in South West WA, with tree-hollow-dependent species often being negatively affected (e.g. the brush-tailed phascogale *Phascogale tapoatafa* (Rhind 2004), western ringtail possum *Pseudocheirus occidentalis* (Wayne *et al.* 2006) and brushtail possum *Trichosurus vulpecula* (Morris *et al.* 2000), while ground-dwelling mammals remain largely unaffected (e.g. brush-tailed bettong *Bettongia penicillata* [Morris *et al.* 2000; Wayne *et al.* 2016], quenda *Isoodon obesulus* and western quoll *Dasyurus geoffroii* [Morris *et al.* 2000]). Following timber harvest, silvicultural burns are commonly applied to recently harvested coupes to encourage seed germination and reduce fuel loads (Morris *et al.* 2000; Abbott and Williams 2011). Prescribed fuel-reduction burns are also conducted generally in Australian fire-prone forests, aiming to reduce the impact of wildfires, protect human lives and assets, as well as promote the conservation of biodiversity (Burrows 2008). How species are affected by fire depends largely on the fire's intensity, spatial magnitude and patchiness (Friend and Wayne 2003). Any fire that burns a substantial part of the forest is likely to cause some reduction in small- to medium-sized mammal abundance (Christensen and Abbott 1989). The survival and recovery of mammals within burned habitat is linked to the availability of shelter and food (Friend 1993; Friend and Wayne 2003). A number of studies have been conducted in the jarrah forest in South West WA, exploring the direct or indirect effects of fire on small- and medium-sized mammals (e.g. western ringtail possum [Inions *et al.* 1989; Wayne *et al.* 2006], brushtail possum [Inions *et al.* 1989], tammar wallaby *Macropus eugenii* [Christensen 1980] and brush-tailed bettong [Christensen 1980]). The recovery of native fauna following fires may also be linked to fox predation (Christensen 1980). Foxes were introduced to Australia in the 19th century and have been held responsible for the decline and extinction of a large number of small- and medium-sized native mammals (Kinnear *et al.* 1988; Burbidge and McKenzie 1989; Short and Smith 1994). At many sites in Australia, including the south-west of Western Australia, foxes are controlled with baits containing 1080 poison (sodium fluoroacetate) (Marlow *et al.* 2015). This poison is deemed suitable for the control of introduced species, as many south-west Australian native species have evolved a tolerance for this toxin that occurs in some local plant species (King *et al.* 1978; King *et al.*

1981; Twigg and King 1991). A reduction in fox abundance and an increase in native mammal abundance following 1080 baiting was found – for example, for the rock-wallaby (*Petrogale lateralis*) in the south-west Australian Wheatbelt region (Kinnear *et al.* 1998) and for brush-tailed bettong, quenda, brushtail possum and western quoll in the UWR (Morris *et al.* 2000). The UWR is part of the Southwest Australia Global Biodiversity Hotspot (Myers *et al.* 2000), and its jarrah forests and woodlands are home to several threatened mammal species (Wayne *et al.* 2017), including the numbat (*Myrmecobius fasciatus*).

6.3.1 The numbat

The numbat is a small (~500–700 g), Australian-endemic, diurnal marsupial. Numbats are listed as endangered under Australia's *Environment Protection and Biodiversity Conservation Act 1999* and internationally under the IUCN Red List of Threatened Species (Friend and Burbidge 2008). Prior to European settlement, the numbat occurred over much of the southern half of the Australian continent; however, following habitat destruction and the introduction of exotic predators, the numbat's range has contracted to about 1% of its former extent (Friend 1990). Today, numbats can be found in two remaining natural populations in south-west Western Australia (Dryandra Woodland and UWR) and in several reintroduction sites (Department of Parks and Wildlife 2017). The total estimated population size was 880 to 1280 adult individuals in 2018 (Threatened Species Scientific Committee 2018). The population in the UWR was proposed to be the largest, with an estimated 200 to 500 adult individuals (Threatened Species Scientific Committee 2018); however, these estimates were not based on empirical data as there were no numbat population studies conducted in the UWR. Numbats are difficult to detect in the UWR, partially due to dense vegetation (Friend and Thomas 2003) but also because numbats have a solitary lifestyle with a large home-range size that can exceed 100 ha (Christensen *et al.* 1984). The species also cannot be attracted to traps as there are no known effective lures or bait (Burrows and Christensen 2002). Numbats feed on termites, which they extract from shallow, subterranean galleries and dead plant matter (Calaby 1960b). Even though individuals are difficult to find, numbat activity can be detected by characteristic signs (diggings and scats, described in Calaby 1960b; Friend and Thomas 2003) that are left behind during feeding excursions (Seidlitz *et al.* 2021). Numbat signs have been commonly used to determine their presence and distribution (Friend 1990; Friend and Thomas 2003) but have not yet been used to link numbat activity to habitat characteristics such as ground cover or forest management activities. From field observations (e.g. Calaby 1960b) and studies of radio-collared numbats (Maisey and Bradbury 1982; Christensen *et al.* 1984), it is known that numbats use hollows in fallen trees (logs) for refuge from predators and for overnight or daytime resting and nesting. Numbats use multiple logs with hollows within their home range (Maisey and Bradbury 1982). However, it is not known how important the availability of logs and hollows are to numbats as they also use self-constructed burrows (see Maisey and Bradbury 1982 for description of burrow characteristics).

6.3.2 Numbats and forest management activities

The effects of forest management activities on the UWR numbat population have not yet been studied. Inappropriate fire regimes, which may arise from frequent prescribed fuel-reduction burns (Clarke 2008), were postulated to threaten numbat populations (Department of Parks and Wildlife 2017), although no evidence was found that altered fire regimes were related to the numbat population decline that occurred in the late 1970s at the Dryandra Woodland, the second remaining natural numbat population (Friend 1990; Friend and Thomas 2003). Timber is harvested in the UWR within state forests, while national park and nature reserve areas are preserved. There are no published empirical studies on the effects of timber harvest on numbats. However, numbats were observed to use recently harvested areas soon after operations were completed (Department of Parks and Wildlife 2017). Introduced predator control, primarily targeting foxes, is an important native fauna conservation tool in the UWR (Orell 2004). Numbat population response to predator control in the UWR is not monitored; however, at the Dryandra Woodland, a sharp increase in numbat sightings was recorded within a few years after fox baiting commenced (Friend and Thomas 2003).

6.3.3 Aims

This study aimed to increase the ecological knowledge of the endangered numbat to aid the conservation of this species. We investigated associations between numbat habitat use, determined by the presence/absence of numbat signs (scats and diggings) and forest management activities and habitat characteristics. Specifically, we aimed to determine if the time since last fire, forest type (including jarrah and wandoo forest), timber-harvest history and the intensity of fox baiting affects numbat habitat use. The results of this study will provide valuable information to natural resource management concerned with forest productivity, wildfire suppression and biodiversity conservation.

6.4 Materials and methods

6.4.1 Study area

This study was conducted in areas of state forest, nature reserve and national park within the UWR in south-west Western Australia (Figure 6.1). The study area extends over approximately 140 000 ha of publicly managed land, characterised by a subdued undulating landscape with sclerophyll forests and woodlands, dominated by jarrah (*Eucalyptus marginata*), marri (*Corymbia calophylla*) and in some valleys, wandoo (*Eucalyptus wandoo*). The climate is of Mediterranean type, with warm dry summers and cool wet winters. Annual precipitation across the region ranges from 650 to 950 mm (Zosky *et al.* 2017).

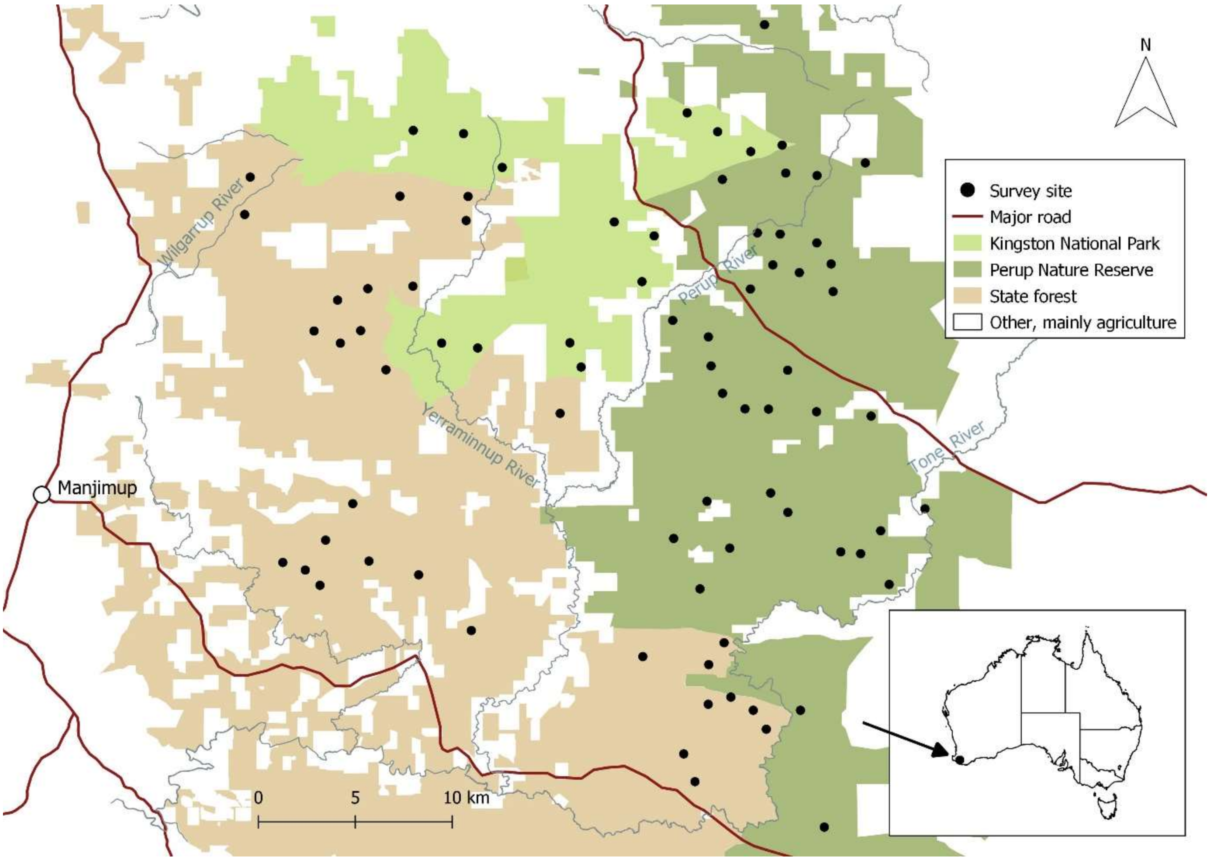

Figure 6.1. Location of study sites (n=78) in the Upper Warren region, Western Australia. Shapefiles for the development of this map were provided by the Department of Biodiversity, Conservation and Attractions, Western Australia.

6.4.2 Fire history

Fire has been part of the Australian landscape for millions of years (Bowman 2003; Hopper 2003), with estimated natural fire intervals of 30 to 100 years for jarrah forest (Enright and Thomas 2008). The natural fire regime has been adapted to human needs, first by Aboriginal peoples (Hassell and Dodson 2003) who arrived in the south-west about 50 000 years ago (Turney *et al.* 2001) and thereafter by European settlers who arrived about 200 years ago. Overall, it appears that Aboriginal peoples used fire frequently to burn predominantly small areas, resulting in a mosaic of patches at different post-fire stages (Abbott 2003; Bowman 2003). Aboriginal burning practices are not well documented, and thus it is difficult to determine how frequently jarrah forest was burned before and during early European settlement (Christensen and Abbott 1989; Abbott 2003). Enright and Thomas (2008) suggested that jarrah forests of south-western Australia were used intermittently by Aboriginal peoples and were not actively managed with fire. Though, from evidence found on grasstrees (*Xanthorrhoea preissii*) Ward *et al.* (2001) suggested that the jarrah forest experienced fire as often as three times per decade during pre- and early European settlement. However, caution was called for to not infer landscape scale burning from fire scars on individual grasstrees (Hopper 2003; Wardell-Johnston *et al.* 2004), and errors were found when validating evidence from grasstrees against known fire histories (Enright *et al.* 2005; Miller *et al.* 2007; Miller *et al.* 2012). Since the 1960s, prescribed fuel-reduction burns have been applied to south-western forests, including the UWR (Burrows and McCaw 2013). Today, burns are applied approximately every 6 to 12 years, with some areas being excluded for longer to create a mosaic of forest habitat with a diversity of post-fire stages (Burrows and McCaw 2013).

6.4.3 Timber-harvest history

Significant silvicultural harvest of the jarrah forest began soon after European settlement in Western Australia in the 1870s, and became regulated with the implementation of the Forests Act 1918 (Bradshaw 1999). Within the UWR, current national park and nature reserve areas are now protected but have historically been timber harvested (for a detailed timber-harvest history, see Bradshaw 1999). Past harvest techniques and years since last harvest vary greatly for these areas (Wayne *et al.* 2006). For the purpose of this study, areas that have not been harvested for >33 years were used to represent ‘mature’ forest types. Within state forests, two major harvest techniques are currently used: gap creation and shelterwood. Gap creation is characterised by the removal of most trees within an area of ≤10 ha (some habitat trees are retained), while shelterwood involves a partial harvest of overstorey trees, typically over larger areas (Conservation Commission of Western Australia 2013). Silvicultural burns are applied to recently harvested areas to reduce logging debris and promote regeneration.

6.4.4 Fox-baiting history

Intermittent distribution of 1080-poisoned bait for the control of introduced foxes started in some areas of the UWR during the 1970s (Burrows and Christensen 2002). With the implementation of the broad scale ‘Western Shield’ conservation program in 1996, baits are now distributed four times a year by aircraft (5 baits/km^2) to most forest areas of the UWR

(Orell 2004). We refer to this treatment as ‘standard’ baiting. A supplementary baiting regime (from here on referred to as supplemented baiting) has been applied to an area within the core of the Perup Nature Reserve since 2010, where the standard aerial baiting (four times per year) is supplemented by ground baiting deployed by hand from a vehicle along selected tracks with an emphasis on tracks adjacent to or close to agriculture (12 times a year, one bait every 200 m along selected tracks) (Wayne *et al.* 2013b).

6.4.5 Study sites and plots

Seventy-eight study sites were established using a stratified random design. Site categories are detailed in Table 6.1. We used the ‘random points inside polygons’ function in QGIS 3.2 Bonn (QGIS Development Team 2019) to place study sites within selected areas. The minimum distance between sites was set to 1 km to minimise the chance of redetection of signs of individual numbats at different sites. Additionally, sites were placed at a minimum distance of 100 m from private properties, 200 m from a different fire history (see exceptions in Table 6.1) and 30 m from streams. Subsequently, study sites were ground-truthed and moved (n=4) to ensure that site characteristics matched the desired categories. The final distance between nearest sites was on average 2.1 km (min 1.1 km, max 6.6 km). This study was designed for single species, single season occupancy modelling (see MacKenzie *et al.* 2018, Part 2). At each site, eight 40 x 100 m plots were established to allow repeated site surveys within different areas at each site. Where possible (36 sites), plots were placed adjacent to each other in two rows of four plots. Where landscape features did not allow this placement, plots were distributed arbitrarily near each other in a way that ensured plots lay within the desired habitat category.

Table 6.1. Survey site categories of 78 study sites to determine the effect of forest management activities and habitat characteristics on numbat habitat use in the Upper Warren region, Western Australia.

Standard and supplemented introduced predator control refers to quarterly aerial baiting only and quarterly aerial baiting plus monthly hand baiting respectively.

Forest type (dominant tree species)	Timber-harvest history (years since last harvest)	Years since last fire	Fox baiting	Number of sites
Wandoo	Not harvested	0–2	Standard	4
			Supplemented	4[1]
		5–10	Standard	4
			Supplemented	3
		>20	Standard	1
			Supplemented	4[2]
Jarrah	Mature (>33)	0–2	Standard	6
			Supplemented	6
		5–10	Standard	6
			Supplemented	6
		>20	Standard	4
			Supplemented	6
	Gap creation (<6)	0–2	Standard	2
		3–4	Standard	3
		5–10	Standard	1
	Gap creation (20-25)	0–2	Standard	3
		3–4	Standard	3
	Shelterwood (<6)	3–4	Standard	6
	Shelterwood (20–25)	0–2	Standard	2
		3–4	Standard	2
		>20	Standard	2[3]
Total number of sites				**78**

[1] Minimum distance of 200 m from a different fire history was not achieved. One site was located within 50m, [2] 90 m, [3] 130 m from a different fire history.

6.4.6 Sign surveys

Sign surveys were shown to be an effective survey method for the detection of numbat presence (Seidlitz *et al.* 2021). Four sign surveys were conducted between 8 October and 29 November 2018. Sign surveys were conducted by two people: the first author (all surveys) and a trained field assistant (all surveys but 6 days) or an experienced volunteer (6 days). Surveys were only conducted on dry days because numbat scats are more difficult to find when the scats and ground are wet. At each site, one randomly selected plot was searched for numbat signs for 30 min. Plots were divided longitudinally, and two observers walked (~10 m apart) the length of a plot on one side and returned on the other side to cover as much ground as possible. Because of the 30 min time limitation, plots were not always searched entirely. When scats or diggings were found, the search time was paused to record findings. Found scats were examined on the spot with a 30x magnifying glass and, at the end of each day, with a dissecting microscope to ensure validity (numbat scats characteristically contain termite exoskeleton parts). The finding of one or more scats on a plot during a survey was defined as a sign detection (whenever numbat diggings were found, scats were also found).

6.4.7 Habitat surveys

Habitat surveys were conducted to collect data on variables that may affect numbat habitat use or the detection of numbat signs. Habitat surveys were conducted on 32 days between 6 December 2018 and 10 January 2019. Within each plot that was searched for numbat signs, a 50 m transect line was placed along the midline and from the start to halfway into the 100 m plot. Habitat variables collected are detailed in Table 6.2.

Table 6.2. Description of numerical habitat variables that may affect numbat habitat use or the detection of numbat signs where applicable.

Variables were collected along a 50 m transect line within each of four survey plots at 78 sites in the Upper Warren region, Western Australia. Ground cover % was calculated from 100 data points recorded every 50 cm along transect line.

Variable (unit)	Rationale	Data collection/details
Ground cover: bare ground (%)	Affects ground available for numbats to dig/find termites and detection of numbat signs.	Including gravel/small rocks (<5 cm Ø).
Ground cover: fine woody debris (%)	Affects ground available for numbats to dig/find termites and detection of numbat signs.	Leaf litter and wood <25 mm Ø).
Ground cover: plants	Affects ground available for numbats to dig/find termites and detection of numbat signs.	Plants <50 cm tall that typically covered the soil surface.
Ground cover: coarse woody debris (%)	Attracts termites.	Wood 25–200 mm Ø.
Favourable ground cover: bare ground + fine woody debris + coarse woody debris (%)	Affects ground available for numbats to dig/find termites and detection of numbat signs.	Recorded every 50 cm along transect line.
Flat wood fragments (number)	Numbat scats are often found on these.	Wood fragments >20 x 20 cm roughly flat in shape. Any partially or wholly within 2 m from transect line.
Termite mounds (m^3)	May relate to mound-building termite abundance. Numbat scats are often found on these.	Any partially or wholly within 2 m from transect. Above ground volume (V), was calculated from the prolate spheroid volume formula: $\frac{V}{2} = \left(\frac{4}{3}\right)\pi b^2 \cdot a$, where a=diameter and b=height.
Termite mound (number)	May relate to mound-building termite abundance. Numbat scats are often found on these.	Any partially or wholly within 2 m from transect.

Variable (unit)	Rationale	Data collection/details
Logs (number)	Provide refuge. Numbat scats are often found on logs.	Any logs with a large-end diameter > 200 mm, partially or wholly within 2 m from transect.
Log hollows (number)	Provide refuge.	Any with an entrance size ≥4 cm and depth of ≥20 cm in logs partially or wholly within 2m from transect.
Tree basal area (m^2)	Habitat characteristic describing stand density.	All trees partially or wholly within 2 m from transect. Basal area (BA) was calculated based on the formula for the area of a circle: $BA = \pi r^2$, where r=radius at breast height (≈1.3 m above ground).
Cut stumps (number)	Habitat characteristic describing timber-harvest intensity.	Number of cut stumps partially or wholly within 2 m from transect.
Soil type (ribbon length mm)	Describes soil texture. Termites may have soil preferences.	Methods adapted from McDonald and Isbell (2009). Surface soil (top 5 cm) was sieved (2 mm sieve), moistened, formed into a ball, and pressed out between thumb and forefinger to form a ribbon. Higher soil clay content relates to increased ribbon length.

6.4.8 Data analysis

All statistical analysis was conducted in R version 4.0.2 (R Core Team 2020). A matrix with numbat sign detections (1) and non-detections (0) was established from spatial replicates (78 sites) and temporal repeats (four surveys). Data were analysed with single species, single season occupancy modelling (MacKenzie *et al.* 2018, Chapter 4) using the RPresence package 2.12.33 (MacKenzie and Hines 2018). Occupancy estimates were interpreted here as the probability of habitat use and not the probability of site occupancy because it is not known how long numbat signs persist in the natural environment. Even though numbat signs are evidence that the animal has used the respective habitat in some way, detected signs (especially scats) may be weeks old, and the animal may have left the area.

We examined numerical habitat covariates (Table 6.2) and categorical covariates (covariates of main interest: forest type and harvest history, time since fire and fox-baiting intensity). To determine which numerical habitat covariates are important for numbat habitat use and the probability of detecting numbat signs, univariate occupancy models were fitted by first holding the detection probability (p) constant while including site-specific covariates, and second holding occupancy (psi) constant while including survey-specific covariates. Wald tests were applied to univariate models, and covariates with a p-value <0.05 were carried forward for inclusion in multivariate model selection processes, with the AIC criterion used to determine the best model (Burnham and Anderson 2002, pp. 60-64). A Pearson chi-square statistic in combination with parametric bootstrapping (n=999) was used to assess model fit (MacKenzie and Bailey 2004).

6.5 Results

Four numbat sign surveys were conducted at each of the 78 sites (312 searches). Numbat scats were detected at 65 sites (83%). No numbat signs were detected in 4/20 wandoo sites, 8/34 jarrah mature sites and 1/12 shelterwood sites. Of the sites without sign detections, 4/13 were situated in recently burned areas (0–2 years category). There was a wide range of variation in the data of many habitat variables used for univariate model selection, especially for termite mound number/volume, log hollow number and cut stumps number (Table 6.3). The number of logs was the only numerical covariate significantly associated with numbat habitat use (Table 6.3). No numerical covariate was found to be significantly associated with the detection of numbat signs (Table 6.3).

Table 6.3. Covariates, their mean values with standard deviation (SD), and *p*-values from Wald tests applied to univariate models (where appropriate) to determine if covariates affect numbat habitat use (models with detection probability [p] held constant) or the probability of detecting numbat signs (models with occupancy [psi] held constant).

Significant *p*-values are denoted in bold.

Covariate	Mean (SD)	psi (covariate), p (.)	psi (.), p (covariate)
Ground cover: bare ground (%)	0.275 (0.173)	0.304	0.822
Ground cover: fine woody debris (%)	0.547 (0.162)	0.128	0.684
Ground cover: plants <50 cm (%)	0.108 (0.084)	0.103	0.496
Ground cover: coarse woody debris (%)	0.033(0.025)	0.084	0.523
Favourable ground cover (bare + fine woody debris + coarse woody debris (%)	0.861 (0.091)	0.201	0.243
Flat wood fragments (number)	35.750 (15.028)		0.314
Termite mounds (m^3)	0.512 (1.136)	0.107	
Termite mounds (number)	2.971 (3.186)	0.911	0.154
Logs (number)	4.721 (3.480)	**0.046**	0.618
Log hollows (number)	1.189 (1.676)	0.257	
Tree basal area (m^2)	1.097 (0.465)	0.512	0.416
Cut stumps (number)	0.740 (0.931)	0.154	
Soil type (ribbon length mm)	15.801 (7.206)	0.382	0.266

We found numbat signs at 100% of the gap creation sites. Since occupancy models do not perform well when encountering boundary estimates (i.e. values close to 0% or 100%) (Welsh *et al.* 2013), we combined gap creation with shelterwood sites to form the category 'jarrah harvested' for further analysis. Numbat signs were found at 23/24 jarrah-harvested sites (96%). The covariate 'forest type' (jarrah mature, jarrah harvested and wandoo not harvested) was excluded from the AIC-based model selection process for numbat habitat use because the number of logs per site is associated with forest type, with jarrah-harvested sites having a higher number of logs than unharvested sites (Figure 6.2).

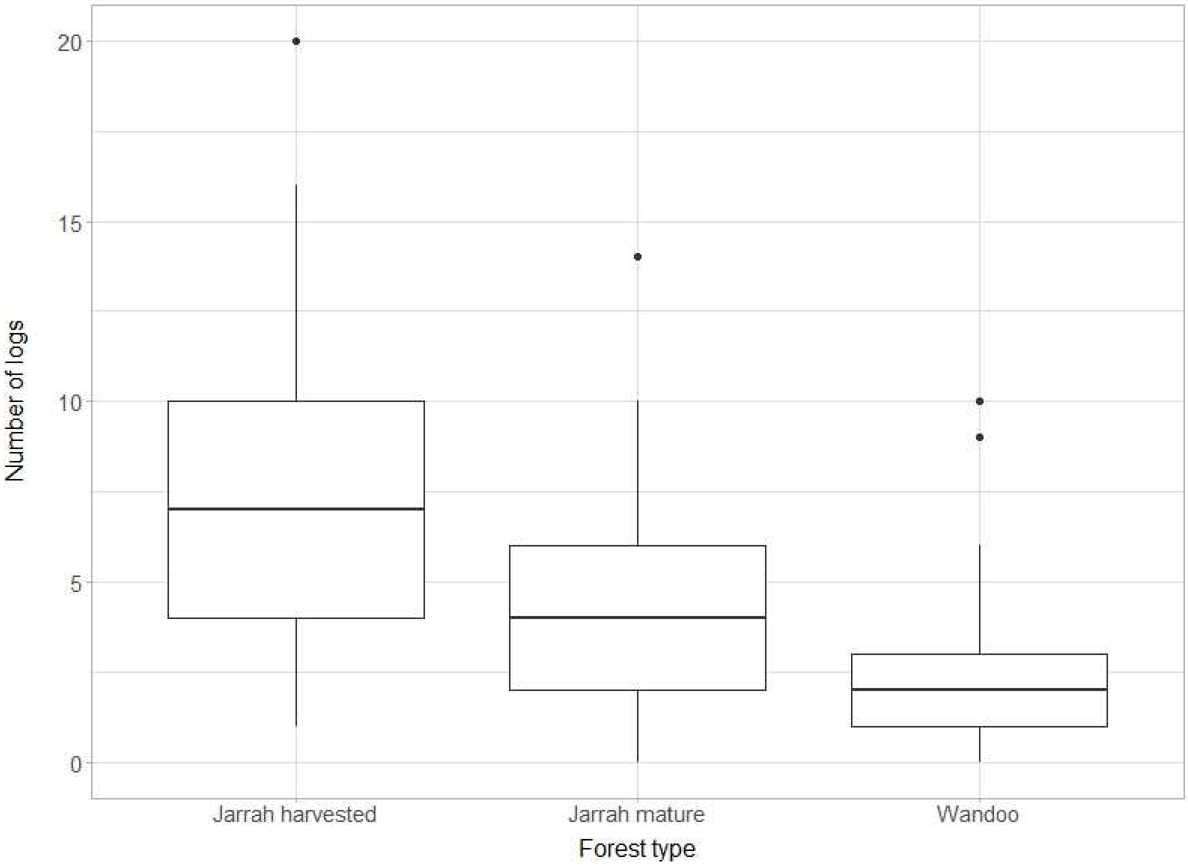

Figure 6.2. Boxplot of the number of logs partially or wholly within a 50 x 4 m transect per survey plot (n=312) in relation to forest type. The bold horizontal lines correspond to the median in each group. The top and bottom of the box represent the third and first quartiles (interquartile range). The upper whisker corresponds to the third quartile plus 1.5 times the interquartile range, and the lower whisker corresponds to the first quartile minus 1.5 times the interquartile range. Asterisks represent data points outside this range.

AIC-based model selection of site-specific covariates classed the univariate model including ‘log number’ as top ranked (Table 6.4). All models that include the covariate ‘log number’, were ranked higher than models excluding this covariate, providing evidence that the number of logs was an important factor. Support for the top ranked model was also indicated by the greater model weight (Table 6.4). When exploring models for factors affecting detection probabilities, the highest ranked model included the covariate ‘survey’ (Table 6.5). The final model therefore included ‘log number’ as the most important determinant of numbat habitat use and ‘survey’ as the most important determinant for the probability of detecting numbat signs. Using this model, log number was positively related to numbat habitat use (Figure 6.3), and there was an increasing trend of detecting numbat signs during later surveys (Figure 6.4).

Table 6.4. Summary of model selection procedure for the probability of numbat habitat use (Ψ).

ΔAIC is the relative difference in AIC values from the model with the smallest AIC value; w is the AIC model weight; −2l is twice the negative log-likelihood, and K is the number of parameters.

Model	ΔAIC	w	−2l	K
Ψ (log number), p (.)	0	0.4072	359.3263	3
Ψ (log number + fox baiting), p (.)	1.8195	0.1639	359.1458	4
Ψ (log number + time since fire), p (.)	2.0351	0.1472	355.3614	6
Ψ (time since fire), p (.)	2.3673	0.1247	357.6936	5
Ψ (.), p (.)	2.9975	0.091	364.3238	2
Ψ (fox baiting), p (.)	3.6371	0.0661	362.9634	3

Table 6.5. Summary of model selection procedure for the probability of detecting numbat signs (p).

ΔAIC is the relative difference in AIC values from the model with the smallest AIC value; w is the AIC model weight; −2l is twice the negative log-likelihood, and K is the number of parameters.

Model	ΔAIC	w	−2l	K
Ψ (.), p (survey)	0	0.6781	351.0303	5
Ψ (.), p (survey + time since fire)	2.5539	0.1891	347.5842	8
Ψ (.), p (survey + forest type)	3.6841	0.1075	350.7144	7
Ψ (.), p (.)	7.2935	0.0177	364.3238	2
Ψ (.), p (time since fire)	9.8986	0.0048	360.9289	5
Ψ (.), p (forest type)	10.9728	0.0028	364.0031	4

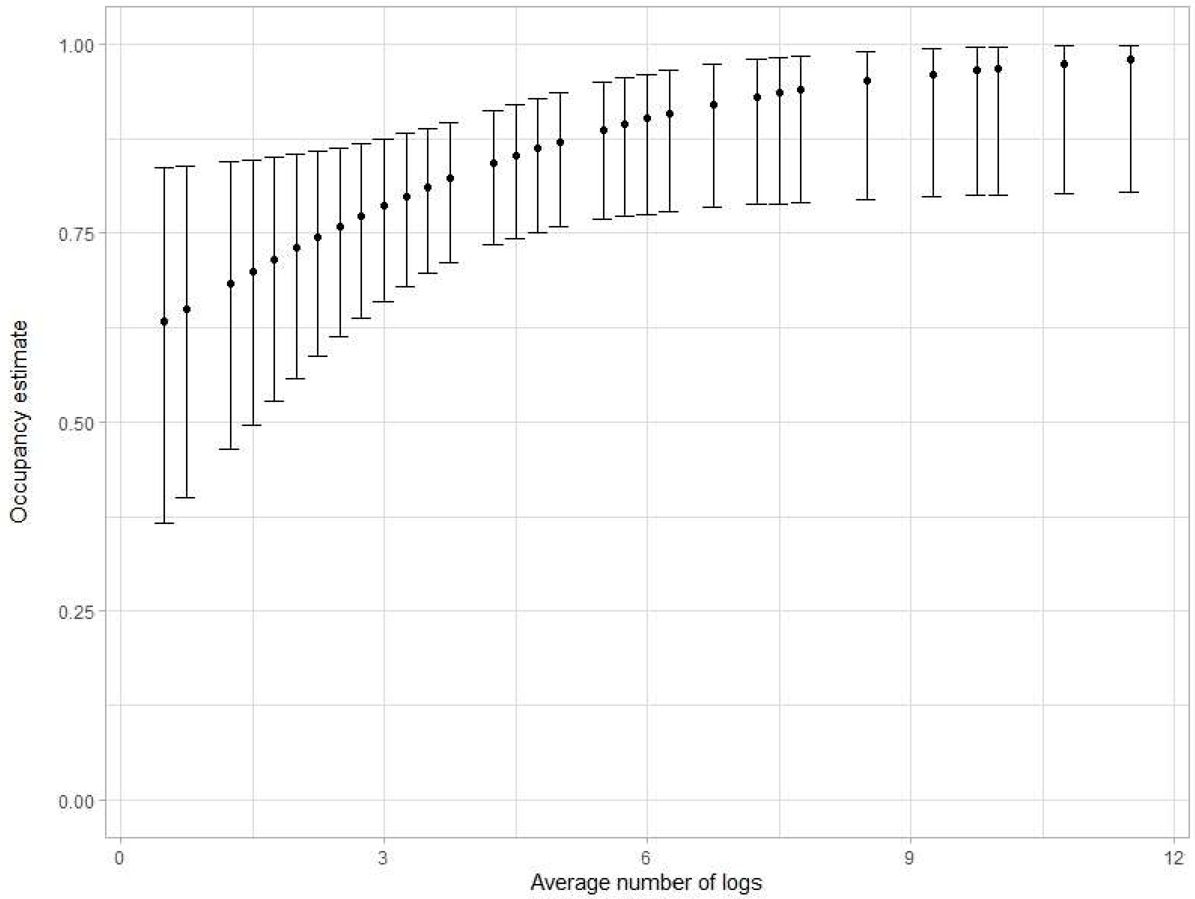

Figure 6.3. Occupancy estimate with 95% confidence interval, here interpreted as the probability of numbat habitat use, in relation to the average number of logs counted per survey site (logs that were partially or wholly within a 50 x 4 m transect). Data derive from four repeat sign surveys at 78 sites in the Upper Warren region, Western Australia.

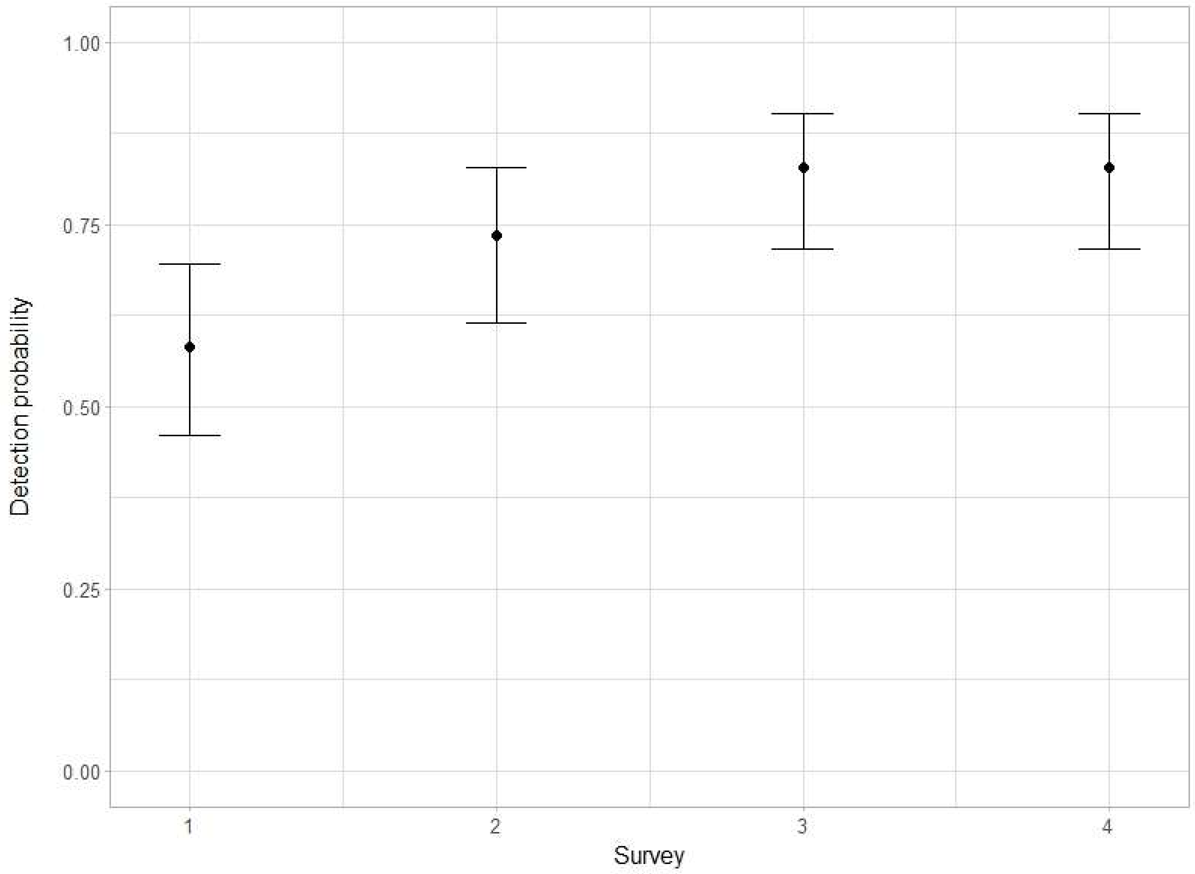

Figure 6.4. Estimates of detection probability with 95% confidence interval for finding numbat signs during four repeat sign surveys at 78 sites in the Upper Warren region, Western Australia.

Model fit was generally poor with c-hat values around 4 to 5 for all models. C-hat values of this magnitude indicate poor model fit rather than overdispersion (Mazerolle 2020). Model fit did not improve (and conclusions did not change) by standardizing variables (results not shown). We therefore did not standardise variables for data analysis. Poor model fit was likely caused by the unusually large number of sites where numbat signs were detected. There were few sites of absences, making it difficult to determine what drives numbat habitat selection and potentially indicating a degree of flexibility in habitat preference across a range of variables. The use of QAIC-based model selection is recommended when overdispersion is present but not when poor model fit is caused by other inadequacies in the model’s structure (MacKenzie *et al.* 2018, pg. 106). We therefore used the results from the AIC-based model selection.

6.6 Discussion

This study provides valuable information on habitat use of the endangered numbat in the UWR, Western Australia. A large proportion (83%) of study sites, situated in various habitat categories, were found to contain evidence of numbat activity. While numbat habitat use appeared to be related to the number of logs present at sites, the probability of detecting numbat signs increased during later surveys. The number of logs was related to forest type and harvest history, with most logs in recently harvested jarrah forest sites and least in unharvested wandoo sites. This study found no evidence that the time since last fire, fox-baiting intensity (standard versus supplemented) or other habitat characteristics were associated with the probability of numbat habitat use or the probability of detecting numbat signs.

6.6.1 Numbat habitat requirements

The detection of numbat signs in habitats with differing characteristics indicates that numbats in the Upper Warren region are habitat generalists. This is not surprising as the species used to occur in a wide variety of habitats before major range contractions, including mulga (*Acacia aneura*) woodland, sand-plain and dune areas dominated by spinifex (*Triodia* spp.) hummock grasslands and eucalypt woodlands and forests (Friend *et al.* 1982; Christensen *et al.* 1984; Burbidge *et al.* 1988). Besides a sufficient supply of termites (not measured here) (Friend 1989), the other key habitat requirement for numbats appears to be the availability of refuges, providing protection from predators in addition to resting and nesting opportunities. Despite model limitations due to high rates of habitat use, logs were found to be the only important variable for numbat habitat use during this study. This makes biological sense and is consistent with frequent observations of numbats using logs as refuge, both within log hollows (if available) and in burrows under logs. Numbat burrow entrances are often found under logs (Maisey and Bradbury 1982) that help conceal and protect burrow entrances from digging predators such as foxes and monitor lizards (*Varanus* spp.). The two remaining natural numbat populations are located in forested areas (Department of Parks and Wildlife 2017) where the opportunity to find refuge in or under logs may have aided the survival of the numbat as a species. In sand-plain and dune areas, where numbats became extinct, logs would be scarce, possibly exposing numbats to predation pressure from introduced species such as foxes and cats. We recommend ensuring that an average of six to 12 logs (number of logs partially or wholly within a 4 x 50 m transect) is retained after forest management activities (e.g. timber harvesting and prescribed burning) to achieve a 0.9 or higher probability of numbat habitat use. The method of counting logs from this study does not allow a conversion of results to a log density-per-hectare estimate that would be more practical for management recommendations. We therefore recommend to also explore field techniques that would allow such conversion.

6.6.2 Forest type

Forest type was associated with log numbers and is therefore related to numbat habitat use. During timber harvest, the merchantable parts of trees are removed, yet upper parts of the trunk and branches are left behind (Abbott and Williams 2011), increasing the amount of logs available and therefore promoting numbat habitat use in harvested areas. Since numbat signs were found at all but one harvested site, we did not investigate if the time since harvest affects numbat habitat use. Numbats have been postulated to prefer wandoo over jarrah forest ((Calaby 1960b). In agreement with Christensen *et al.* (1984), we found no such preference. This study found a trend of higher log availability in jarrah compared to wandoo forest and therefore an increased numbat habitat use of jarrah forest.

6.6.3 Time since fire and fox control

This study found no evidence that the time since fire or the intensity of fox baiting affected numbat habitat use. Fire may negatively impact numbats by directly killing them or by reducing important resources such as food and shelter. In an unpublished (not peer reviewed and no methodological details available) experimental study at the Dryandra Woodland, fire was not considered a significant threat to numbats, and surviving individuals continued using their pre-fire home ranges. Yet, the loss of logs and vegetation cover appeared to increase predation pressures (Department of Parks and Wildlife 2017). A study of medium-sized mammals in a temperate eucalypt forest in south-eastern Australia found that prescribed fire caused an immediate reduction of understorey cover by more than 80%, and that predation by cats and foxes increased after fire (Hradsky *et al.* 2017). In this study, we found numbat signs in areas that were recently burned (0–2 years ago) by prescribed fuel-reduction burns, indicating that these areas are used. More detailed work is needed to more fully understand how season and intensity of fire affect numbats (e.g. density, survivorship, behaviour, habitat use and occupancy), their important resources such as food and shelter (hollow logs) and their susceptibility to predation.

In the Dryandra Woodland, numbat detections increased following the commencement of fox control using 1080-poisoned baits (Friend 1990; Friend and Thomas 1994b; Friend and Thomas 2003). In the UWR, a comparison of numbat detections before and after commencement of fox control (1970s–1990s; Wayne *et al.* 2017) or in areas fox baited versus unbaited (no or limited comparable areas) has not been investigated. However, we found no difference in numbat habitat use between areas that received standard and supplemented fox-baiting regimes. Interpreting these results is, however, confounded due to DBCA management conducting the supplemented fox baiting in areas considered to have a greater conservation need. An experimental or other well-designed scientific study is needed to better demonstrate the effects of introduced predator control on numbat populations in the UWR or elsewhere.

6.6.4 How many numbats are there in the Upper Warren Region?

Even though numbat habitat use is not closely related to numbat abundance, we can make careful assumptions about the UWR numbat population size while considering the species' home-range size and the proportion of sites found with numbat signs (naïve occupancy). Even though the naïve occupancy rate commonly underestimates true occupancy (MacKenzie *et al.* 2002), we are using the naïve occupancy rate here to estimate the UWR numbat population size most conservatively. While numbat home-range size varies between individuals, ranging from 0.41 to 133 ha (Christensen *et al.* 1984; Bester and Rusten 2009; Hayward *et al.* 2015), it is estimated to average ~50 ha per adult numbat (Friend 1987a). Male and female territories may overlap, while same-sex home ranges are exclusive (Friend 1987a). The study area contains ~140,000 ha of forested land, predominantly consisting of jarrah forest. Assuming 83% of this habitat is used by numbats, and conservatively assuming a single numbat occupies an exclusive home range of ~50 ha (even though male and female home ranges do overlap (Friend 1987a), then the study area could contain an estimate of

2324 numbats. Similar and other potentially suitable numbat habitat extends further south than the scope of this study area. It is therefore plausible that the UWR numbat population could exceed this estimate.

6.6.5 Limitations

Although this study found no evidence that the forest management activities investigated here affect numbat habitat use negatively, the results need to be considered carefully. Occupancy modelling, here interpreted as numbat habitat use, does not provide information on numbat abundance at study sites. Even though numbat activity was evident, forest management activities such as timber-harvest operations or prescribed fuel-reduction burns may reduce numbat abundance at sites. Care must be taken, for example, to not conclude that numbats are resilient to a wide range of fire regimes as this study did not evaluate fire intensity, seasonal timing or spatial magnitude. Furthermore, this was an observational study, which limits the power of conclusions drawn compared to experimental studies with replication and randomly/independently assigned treatment and control sites. Yet, experimental studies are difficult to conduct for a species such as the numbat with low densities and restricted distribution that limit the amount of area that can be allocated to different treatments.

Juvenile numbats disperse from November to December (Friend and Burrows 1983). During this study, two surveys were completed prior, and the final two surveys during, juvenile dispersal. Therefore, it is possible that population closure assumptions were violated (e.g. some survey sites may be occupied only during later surveys due to immigration-only movements). The increase of detection probabilities for the final two surveys may indicate that immigration-only movements were present. However, we believe that our conclusions about numbat habitat use still hold because (1) the increase of detection probabilities for later surveys may also be caused by progressively dryer weather (allowing numbat signs to accumulate, making them more detectable) and/or by the increased skill of the observers to find numbat signs in later surveys, (2) allowing detection probabilities to vary between surveys (as we have done) absorbs some of the effects of immigration-only movements (MacKenzie *et al.* 2018, p. 148) and (3) there were only 11/78 survey sites with detection histories that may indicate immigration-only movements (e.g. 0011, 0001, 0010). We have re-analysed the data while excluding the 11 sites that possibly experienced immigration-only movements (data not shown), and inference from results did not change.

6.7 Conclusions

This study expands the ecological knowledge of the endangered numbat to aid the conservation of this species and the management of its habitat. In the UWR, the numbat appears to be a habitat generalist with logs being a key habitat requirement to provide refuge from predators as well as resting and nesting opportunities. While timber-harvested sites with associated increased log availability positively influenced numbat habitat use, there was no evidence that time since fire or the intensity of fox baiting affected numbat habitat use. On the basis of the data we currently have, and following the precautionary principle, forest management should ensure the retention of six to 12 logs (number of logs partially or wholly within a 4 x 50 m transect) during management activities to aid the species' conservation. Localised increases in firewood collection, which targets logs but also involves illegal removal of standing trees, may also negatively affect numbat habitat quality and therefore numbat conservation and recovery. Since there was a high occurrence of numbat habitat use in jarrah and wandoo forest areas, both forest types should be considered as potential numbat habitat. The high percentage of sites found with numbat scats indicates that the UWR numbat population could be greater than previously thought, possibly exceeding 2000 numbats. We recommend further studies to determine the extent of the population in forests south of the study area.

6.8 Acknowledgements

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6.9 Conflicts of interest

The authors declare no conflict of interest.

Chapter 7 General discussion

The numbat (*Myrmecobius fasciatus*) in the Upper Warren region, south-western Australia.

Numbats were once widespread over much of the southern half of the Australian continent; however, they are now restricted to about 1% of their former range (Department of Parks and Wildlife 2017) and are listed as endangered. Two natural numbat populations remain, one of which is located in the Upper Warren region (UWR), south-west Western Australia. Due to the lack of effective survey methods, information on this population's size and distribution, or on how forest management activities affect the species, is limited. Therefore, this PhD project aimed to improve the understanding of the numbat population in the UWR to inform management for the conservation and recovery of the species. To achieve this, survey methods were first developed to establish the most efficient and effective method for numbat detection in the UWR. Thereafter, the most successful method was applied to determine numbat habitat preferences, particularly those in relation to prescribed fuel-reduction burns, timber harvesting and the control of introduced foxes using 1080-poisoned bait at differing intensities. The findings of this thesis, as discussed below, have significantly advanced our knowledge of the numbat population in the UWR and will be valuable to the management concerned with the protection of this species.

7.1 Camera trap optimisation

Camera traps are an attractive tool for detecting wildlife, including rare and elusive species (Kucera and Barrett 2011). One advantage of camera traps is that they commonly detect and take images of multiple species, thereby potentially increasing efficiencies and reducing survey and monitoring costs. The DBCA is concerned with native species management and conservation in Western Australia. In the UWR, this agency commonly studies and monitors mammals using the well-regarded Reconyx PC900 and HC600 camera trap models. Therefore, considerable effort was invested during this PhD project to test and optimise camera traps (starting with Reconyx PC900 and HC600 camera trap models) for the detection of numbats that could be studied simultaneously with other mammal species.

With the use of numbats in enclosures at Perth Zoo, it was established that, for numbat detection, the PIR sensor of camera traps is best positioned 25 cm above ground. This height is also suitable for several other threatened species that occur in the UWR (e.g. western ringtail possum, western quoll, and brush-tailed bettong). However, it was also established that Reconyx PC900 camera traps, set at optimal height, missed 63% of known numbat detection events. Such a high rate of missed detections could lead to an underestimate of numbat abundance in field studies. Swift 3C wide-angle camera traps were found to have significantly higher numbat detection rates than Reconyx PC900 camera traps, recording 89% of known events. These findings highlight the importance of camera trap testing and optimisation for target species that has already been recognised (Meek *et al.* 2015a).

An additional trial at Perth Zoo determined that elevated, downward-angled, time-lapse cameras are useful for obtaining images that depict the numbat’s dorsal area, allowing numbat individual identification. The identification of numbat individuals would allow the use of capture–recapture methods that could lead to numbat density or absolute abundance measures. In zoo enclosures, downward-angled, time-lapse cameras were easily assessable for frequent battery and memory card changes, which is required due to the high energy and image storage needed when taking pictures at short intervals (e.g. every 2 sec). The operation of time-lapse cameras in the field, however, is hampered by high energy and image storage needs as frequent servicing or the use of expensive battery and memory storage extensions may be too costly. Future technological advances may resolve these issues and make this technique for numbat detection and individual identification feasible. During this PhD project, downward-angled, time-lapse cameras were not further explored.

7.2 Numbat detection methods

Comparing detection methods for a target species is essential for determining the most efficient and cost-effective methods. The comparison of sign surveys, driven transects and camera trapping revealed that, for numbat detection, sign surveys were most successful. Even though commonly used at Dryandra Woodland and some reintroduction sites (see Friend and Thomas 2003; Hayward *et al.* 2015; Berry *et al.* 2019), driven transects were not successful during this study, most likely due to dense vegetation that restricts numbat detections in the UWR. Driven transects are therefore not recommended for numbat detection in the UWR. Sign surveys and camera trapping, however, were worthy of further investigation.

7.2.1 Sign surveys

Sign surveys were found to be useful for the detection of numbat presence in the UWR. One might expect that finding sufficient signs of a small mammal that occurs at low densities, such as the numbat, would be difficult or even impossible. As numbat density was estimated to be low (one adult per ≈50 ha; Friend 1987a), it came as a surprise to find signs of this species at more than 78% of study sites from all sign surveys conducted for this thesis. Numbat sign-detection probabilities ranged from 0.21 to 0.35 for initial surveys (Chapter 3) to 0.58 to 0.83 for final surveys (Chapter 6). These results demonstrate that sign surveys, together with occupancy modelling, are a feasible option for numbat monitoring in the UWR. There are, however, some disadvantages and limitations to this method.

A high level of skill and enthusiasm is needed from personnel that conduct numbat sign surveys. If personnel fail to detect numbat signs that are present in areas of high occupancy, the number of repeat surveys must be increased to obtain robust occupancy estimates. For example, for populations with 80% site occupancy, three and eight repeat surveys are recommended when detection probabilities are 0.6 and 0.3, respectively (MacKenzie *et al.*

2018, pg. 461). The use of unskilled personnel could therefore significantly increase survey costs.

It takes practice to become skilled at detecting numbat signs. From personal experience, the author was less confident and skilled in finding numbat signs during early surveys, resulting in lower detection probabilities compared to the final surveys. It is therefore recommended that, once skilled, the same personnel conduct numbat sign surveys for long-term monitoring, especially when the results are compared between years. Occupancy modelling can account for a change in observers (or a change in skill level); however, keeping as many variables as possible constant between surveys/years reduces problems with overfitting models with too many covariates.

A further limitation is that sign surveys with occupancy modelling only provide presence and absence data. Little information is revealed about animal abundance at sites that were found occupied (except for the information that there was at least one animal). This may make sign surveys insensitive to changes in population abundance following, for example, disturbance such as fire. Only rough absolute abundance estimates can be achieved when individual animals use exclusive territories and have a known home-range size. For example, Chapter 6 estimated that there could be more than 2000 numbats in the UWR, assuming that 83% of the habitat within the study area (~140 000 ha) was occupied. This is likely to be an underestimate of the UWR numbat population based on an average exclusive home range of ~50 ha per numbat. First, large areas of more suitable habitat extend further south of the study area that need exploring for numbat presence, and second, male and female home ranges do overlap (Friend 1987a). More robust absolute abundance estimates from sign surveys could be achieved if DNA extraction from scats would allow numbat individual identification. In this case, sign surveys could be used together with SECR to provide density and absolute abundance estimates (see e.g. Wegge *et al.* 2019). Absolute abundance or density estimates are more sensitive to changes within animal populations than RAIs or occupancy estimates and are therefore more reliable for population monitoring (Sutherland 2006, Chapter 3).

7.2.2 Camera traps for numbats and other species in the UWR

Numbat detection rates from Reconyx PC900/HC600 camera traps were found to be lower than those of sign surveys. To increase numbat detections from camera traps, Reconyx PC900/HC600 and Swift 3C wide-angle camera traps were periodically repositioned within study sites (Chapter 4). The repositioning of cameras significantly increased numbat detection rates from Reconyx PC900/HC600 camera models and, with the use of Swift 3C wide-angle cameras, detections were further doubled. Yet, sign surveys that were conducted at the same sites during the same time again had higher numbat detection rates than camera traps. This reaffirmed the recommendation to use sign surveys with occupancy modelling as the preferred survey method for numbats in the UWR – until better survey methods can be developed. If absolute abundance or density estimates are needed, and if cameras could be used in field conditions to recognise numbat individuals (e.g. if above-mentioned difficulties with downward-angled, time-lapse cameras could be resolved), then the periodic repositioning of such cameras would be worthwhile to increase numbat detection rates. The use of camera traps for numbats may also be suitable in combination with sign surveys. This would allow the detection of numbats and other wildlife at the same time, allowing the merging of studies for better economy.

Compared to the commonly used Reconyx PC900/HC600 camera traps, Swift 3C wide-angle camera traps also had significantly higher detection rates for a range of species other than numbats. This is an important finding that researchers need to consider when performing camera trap studies for wildlife generally. Reconyx camera traps were found to be the second most used brand in Australia (Meek *et al.* 2015a), and this manufacturer was also found to be one of the most frequently recommended by camera trap users worldwide (Glover-Kapfer *et al.* 2019). Missed detections from Reconyx camera traps but also from other camera trap brands have been found to be a concern (Driessen *et al.* 2017; Jumeau *et al.* 2017; Apps and McNutt 2018a; Fancourt *et al.* 2018; Heiniger and Gillespie 2018; Jacobs and Ausband 2018; Urbanek *et al.* 2019). Camera trap studies that do not consider camera trap limitations may have reduced robustness of data, possibly resulting in poor species management recommendations. We therefore recommend species-specific comparative studies of camera trap models to identify camera traps that increase detection rates. Swift 3C wide-angle camera traps were here found to increase animal detection rates compared to Reconyx PC900/HC600 models for most observed species. During this 9-week field study, Swift 3C wide-angle camera traps performed well without failure; however, this camera model does not appear to be as robustly built as Reconyx camera traps. Therefore, it is recommended to conduct longevity and durability trials as well as detection rate studies to find models that are most reliable.

7.3 Numbat habitat use

The final study of this thesis explored numbat habitat use, particularly that relating to forest management activities. Numbat signs were found at 83% of 78 study sites in various habitat categories, indicating that the species is a habitat generalist in the UWR. No evidence was found that the time since last fire or the intensity of fox baiting (standard versus supplemented) affected numbat habitat use. There was some indication that the species' habitat use related positively to the number of logs found at study sites. Therefore, it seems that timber harvesting, which increases the number of logs in harvested coupes, may positively affect the numbat. This study has not, however, investigated the importance of trees and other structural habitat components retained in harvested coupes that may also be important for numbat habitat use. These aspects therefore need exploring in future studies to obtain a more holistic knowledge of numbat habitat requirements.

With due consideration of the limitations, the findings of this thesis give no evidence that the aspects of existing forest management activities that were investigated (e.g. timber harvest history, time since fire, baiting intensity) significantly affect the numbat population in the UWR negatively. In these aspects it appears, therefore, that currently, no changes to management practices are required to ensure the conservation of this species in the region. However, there are many aspects of management activities that may affect numbats that remain to be determined (e.g. prescribed fire season and intensity, more modern timber harvesting practices, jarrah dieback, roading, recreation, public firewood collecting, etc). Based on the results of this study, to maintain the numbat population, I recommend that management must at least ensure that sufficient logs are retained after forest management activities. Based on the data we currently have; we recommend an average of six to 12 logs partially or wholly within a 4 x 50 m transect area to maintain or encourage habitat use by numbats. These recommendations need careful consideration since numbat habitat use was derived from occupancy estimates that may be insensitive to changes in numbat abundance following forest management actions. More robust population estimates, such as absolute abundance from capture–recapture methods, may lead to more reliable recommendations for numbat management in future.

7.5 Future studies

Future studies that are a continuation of the work accomplished during this PhD project as well as unrelated studies are recommended to increase the knowledge regarding the numbat population in the UWR:

1. During this PhD project, it was determined that one skilled observer can find more numbat signs than three volunteers (Chapter 3). To optimise the efficiency and cost-effectiveness of sign surveys, trials with different numbers of observers of different skill levels could be conducted.
2. To reduce survey costs, numbat sign surveys could be combined with sign surveys for other cryptic species – for example, the western ringtail possum. However, it needs to be determined if the results of numbat sign surveys are compromised when observers also search for the signs of other species.
3. Numbat signs give evidence of numbat habitat use but give no information on the timeframe of last visitation. Studies on the longevity of numbat signs in relation to factors such as season, substrate, moisture and aspect could improve inferences from numbat sign surveys.
4. Development of more accurate measure of population density/size to better understand numbat responses to threats, pressures and management activities. For example, studies exploring DNA analysis from scats combined with capture–recapture methods could be explored to derive absolute abundance and density estimates.
5. Studies exploring elevated, downward-angled, time-lapse cameras in field settings or elevated, downward-angled camera traps with external PIR sensors set at optimal heights could be explored. Both options would allow the capturing of images depicting the individually characteristic pelage patterns on numbat rumps. With high

enough detection rates and successful recognition of numbat individuals, capture–recapture methods could provide absolute abundance and density estimates.

6. More detailed studies are needed on how the intensity, season, spatial magnitude and frequency of fire and timber harvesting affect numbats directly (e.g. survival, density, occupancy, susceptibility to predation).
7. A better understanding is needed regarding the importance of trees and other structural habitat components retained in coupes that were harvested for timber.
8. Studies are also needed on how fire and timber harvesting affect resources needed by numbats (e.g. termite abundance and log availability).
9. Since logs were identified to be an important factor for numbat habitat use, studies that explore this resource are important. Information is needed on log characteristics that make them of value to numbats (diameter, length, hollows, state of decay, tree species, number of branches, etc.). To get a clearer picture on the importance of logs, an experimental approach, where the number of logs is manipulated, would give the clearest evidence of a causal link with occupancy. This would be logistically difficult but may be possible during timber harvest operations. Studies of logs should also explore the creation and destruction of logs by fire and how legal and illegal collection of firewood affects the availability of logs.
10. Studies are needed in relation to numbat population ecology including factors affecting survivorship, mortality and recruitment to better understand the effects of threats and management activities and to determine population viability.
11. Factors that may affect forest health and structure and, therefore, numbat resources (e.g. logs and termites) need exploring, including climate change, drought and plant pathogens such as dieback (*Phytophthora cinnamomi*) and canker (*Quambalaria coyrecup*).

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