Between a rock and a hard place: Management issues for the endangered brush-tailed rock-wallaby, *Petrogale penicillata*, in north-eastern New South Wales.

Catie M. Gowen (B. Nat. Res., Hons)

Photo credit: Shane Ruming

A thesis submitted for the degree of Doctor of Philosophy of the University of New England

Ecosystem Management
School of Environmental and Rural Science
University of New England
Armidale, New South Wales, 2351
AUSTRALIA
November 2015

Declaration

I certify that the substance of this thesis has not already been submitted for any degree and is not currently being submitted for any other degree or qualification. I certify that any help received in preparing this thesis, and all sources used, have been acknowledged in this thesis.

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Catie M. Gowen

27/11/15
Date

Abstract

The endangered brush-tailed rock-wallaby (*Petrogale penicillata)* has declined significantly across its range in south-eastern Australia. Several catchments in north-eastern New South Wales (NSW) represent the last strongholds for the species in the wild. Appropriate management of these populations is of critical importance. This thesis outlines an investigation of the efficacy of current management of brush-tailed rock-wallabies in the Green Gully section of Oxley Wild Rivers National Park (OWRNP).

In response to an apparent population decline, deduced from annual aerial surveys, local managers hypothesised that behavioural responses of brush-tailed rock-wallabies to helicopter-based surveys might have negatively biased population estimates. Camera traps with time-lapse capabilities were set to record an image of rock-wallabies every minute between 0600 and 1400 h, in an attempt to document behavioural changes in response to an approaching helicopter. During the 2014 and 2015 surveys, the behavioural response of brush-tailed rock-wallabies to helicopters was not found to be a significant source of bias for population estimates of rock-wallabies at Green Gully. As a consequence, it was necessary to assess the reliability of the aerial survey by comparing it with three ground-based population estimation techniques. Cost-benefit analysis of the aerial and ground-based methods demonstrated that camera-based, individual recognition and photographic mark–recapture were the most accurate and cost effective methods of monitoring brush-tailed rock-wallaby populations. Aerial survey proved to be a poor substitute, as the data collected bore no reliable relationship to the state of the populations of rock-wallabies on the ground. Given the significant recurring economic cost and the inability to produce scientifically valid data, it is recommended that the use of aerial survey to estimate population size of brush-tailed rock-wallabies at Green Gully be abandoned, and replaced with a rigorous monitoring program aimed at a subset of the broader population.

An investigation of change in vegetation at Green Gully immediately following its inclusion into the protected area network in 2004, found that the herbaceous understorey has shifted from a grazing-tolerant, native perennial grass assemblage dominated by species common in commercially managed native pastures, to a grazing-sensitive, high-conservation-value perennial grass sward typical of pre-European conditions. In addition, a decline in the turnover of introduced species occurred between 2005 and 2012. Although overall native species density declined, the overall landscape (gamma) diversity remained the same due to an increase in turnover of native species. Percentage cover of native species also increased markedly. The return of grazing-sensitive species, such as *Themeda triandra*, resulted in a significant reduction in bare ground and litter.

The interaction of fire and grazing has impacts on food resources available to brush-trailed rock-wallabies and these factors have potential to be manipulated as part of conservation strategies. Vegetation surveys conducted every three months in paired burnt/unburnt and grazed/ungrazed plots close to brush-tailed rock-wallaby colonies revealed high seasonal variability in plant species density, likely driven by rainfall variability. Grazing exclusion did not alter species density over the 13-month study period, but burning, in conjunction with sustained macropod grazing, did significantly alter the grassy understorey to a sward of basal and rosette forming species. With respect to functional traits, burning and grazing had a significant impact on the species composition of the plots, with clonal and stoloniferous species correlated with the highly disturbed sites. Specifically, the combined impacts of burning and grazing resulted in a shift in the main mechanisms species' utilised to store their resources. Continuous monitoring of experimental plots via camera traps revealed that brush-tailed rock-wallabies exhibited a strong preference for recently burnt patches that were relatively close to their refuges. Individuals actively fed on the burnt patches of vegetation at Green Gully to a greater extent than unburnt patches for the entire monitoring period. Relatively few records of sympatric macropods on experimental plots indicated that competition was not a limiting factor in the management of food resources at Green Gully. The broad-scale observational and experimental work in this study revealed that patch burning had a strong influence on animal behaviour and that the interaction between fire and grazing itself is strong. Accordingly, small patch burning around the colonies could be a useful tool to manage food resources for rock-wallabies.

The findings within this thesis indicate that the management of brush-tailed rock-wallabies is complex, and appropriate actions are likely governed by site-specific characteristics. Moreover, this research shows that these characteristics can be identified and measured. The management recommendations outlined for Green Gully in this thesis have wider implications for brush-tailed rock-wallabies throughout their range, and the adoption of these findings into management will aid in the persistence of this endangered macropod where it remains in the landscape.

Acknowledgements

I am indebted to my supervisors, A/Prof Karl Vernes, Dr Guy Ballard, Dr John Hunter and Prof Nick Reid, for their expert guidance, valuable insights, constructive criticism, humour, and encouragement during this study, and without whom none of this would have possible. To Dr Gerhard Körtner and Stuart Green, thank you for your invaluable assistance in the field. Many thanks go to the New South Wales National Parks and Wildlife Service (NPWS) who first established a monitoring program of the brush-tailed rock-wallaby and provided access to the park. To NPWS ranger Piers Thomas for providing knowledge of the field site, and assistance within the field and donating much needed field equipment. Thanks also NPWS Walcha field staff Milo and Jason, whose assistance and sense of humour were invaluable. To my fellow post grad students at UNE who provided much needed support, friendship and advice: Sarah McDonald, Jessica Sparkes, Fran Zewe, Trent Forge, Helen Morgan, Huw Nolan, Sam Morgan, and Joshua van der Eyk. Advice and assistance with statistical analysis was generously provided by Gavin Mellville. and Remy van de Ven. Thanks to my family for their support, and to my brother, Benjamin Gowen, for his much needed help in the field. Finally, to Craig, my partner, best friend, support giver and field assistant: I couldn’t have done it without you. Thank you.

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1. General Introduction

1.1. Background to the study

The brush-tailed rock-wallaby (*Petrogale penicillata*) is a small, endangered, macropod that has suffered continual contraction of its geographical distribution throughout the last century (Menkhorst and Jarman 2004). The species' persistence is threatened by increasing isolation due to habitat fragmentation, impacts from pest species and inappropriate fire regimes (Menkhorst and Jarman 2004, Delaney et al. 2005).

This study focused on a meta-population of the brush-tailed rock-wallaby, at Green Gully, within Oxley Wild Rivers National Park (OWRNP). The location is recognised as one of the last remaining population strongholds for the species and the presence of brush-tailed rock-wallabies was a key driver in the purchase of Green Gully for the inclusion within Oxley Wild Rivers National Park.

Following acquisition in 2004, local National Parks and Wildlife Service (NPWS) staff sought to monitor the status of this species. They implemented annual, aerial surveys to monitor population trends of brush-tailed rock-wallabies within Green Gully and other parts of Oxley Wild Rivers National Park. However, the suitability of the method has subsequently been questioned (Vernes et al., 2011). As a consequence, local NPWS staff have sought to determine the 'best' method for assessing population trends, into the future.

Notably, the shift in land use within Green Gully, from cattle grazing to National Park, caused changes in fire management practices. Green Gully, as an agricultural enterprise, had been burned frequently, at low to medium intensity, for decades. This strategy sought to promote vegetation best suited to the cattle grazing enterprise, especially a grassy understorey. Although optimising livestock production was the primary focus, former owners of Green Gully also describe deliberately burning around brush-tailed rock-wallaby colonies, nominally to promote grass growth for the wallabies (P. O'Keefe pers comm). Since Green Gully was acquired by NPWS, local fire management priorities have changed. As public land managers NPWS are forced to try to balance a variety of imperatives, including asset protection and diverse conservation values. This has led, generally, to less frequent burning within the Green Gully system and raised questions about which fire strategy, might be 'better' for rock-wallabies.

1.1.1. Aerial survey at Green Gully

Unsurprisingly, when faced with a need to cost-effectively survey brush-tailed rock-wallaby populations within the rugged gorges of OWRNP, local managers opted for aerial survey, which had previously been used for yellow-footed rock-wallabies, (*Petrogale xanthopus*) in western NSW and South Australia (Lim and Giles 1987).

Since 2004, NPWS has undertaken annual aerial surveys of the same (n = ~40) rock wallaby colonies within OWRNP. By 2012, the accumulated data (Fig. 1-1) suggested an ongoing trend of population decline. Faced with both an apparent decline in brush-tailed rock-wallaby numbers, and limited funds for maintaining aerial surveys, NPWS sought to determine if aerial survey was providing reliable data on population trends. As well as uncertainty about the ability of aerial observers to reliably index the population, behavioural responses of the wallabies (e.g. flight behaviour, perhaps increasing over time), were posited as sources of error for the aerial surveys. Clearly, to inform management decisions it was vital that an assessment of the efficacy of aerial survey for brush-tailed rock-wallabies be made along with investigation of the behavioural responses of rock-wallabies to aerial survey.

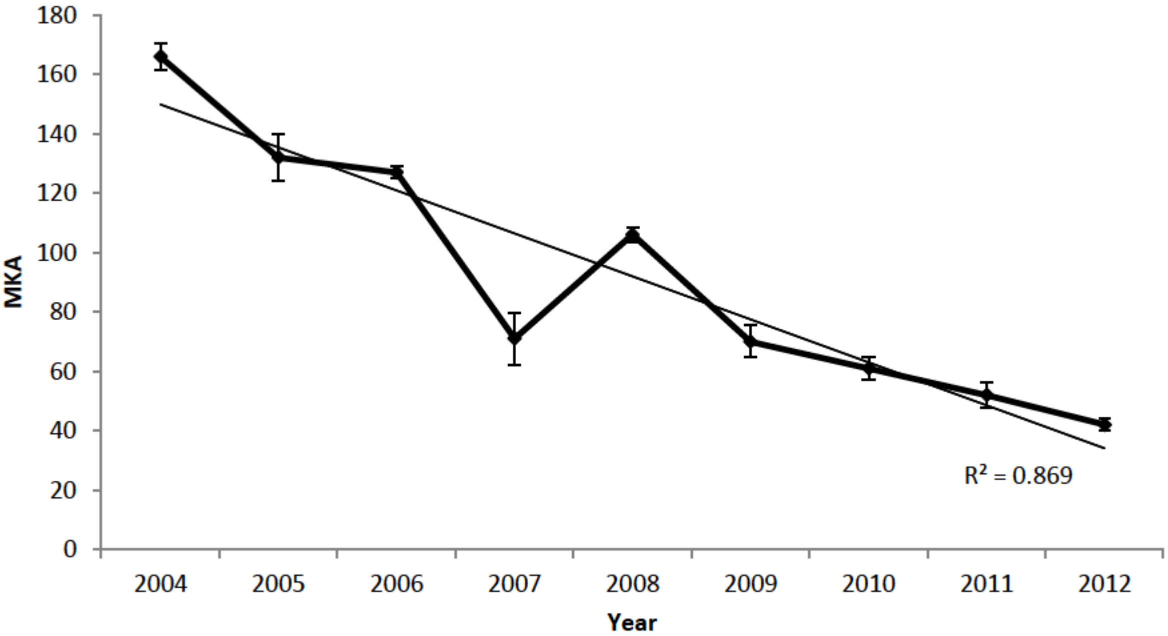


Figure 1-1 Brush-tailed rock-wallaby minimum known alive (MKA) (±1 S.E.) from experimental colonies from aerial surveys undertaken by NPWS. (Data sourced from NPWS).

1.2. Thesis aims and objectives

My goal was to build on available knowledge regarding brush-tailed rock-wallabies, in order to both inform management at a local scale, and provide additional insight for land managers across the species' current range.

Accordingly, this study had a number of key aims:

1. To establish appropriate estimation protocols for estimating population size and related trends of brush-tailed rock-wallabies

2. To examine the response of the grassy understorey to changes in tenure and associated management approaches

3. Identify the responses of refuge dependent herbivores to vegetation manipulation

1.3. Thesis structure

This thesis consists of five research chapters (Chapters 3–7) addressing issues of brush-tailed rock-wallaby management within the study site, plus a synthesis chapter (Chapter 8) summarising the findings and recommendations. The remainder of this chapter serves to introduce the thesis, outline how the sequence of chapters, and the research questions they address, were designed to build on the knowledge of the conservation of rock-wallabies at Green Gully. Chapter 2 introduces the study site and key issues pertaining to the biology and management of brush-tailed rock-wallabies.

Chapter 3 examines the behavioural response of brush-tailed rock-wallabies to aerial survey at Green Gully. It includes an analysis of the behaviour exhibited by *P. penicillata* to approaches by helicopter, and the ability of the on-board observers to enumerate rock-wallabies. As well as highlighting some important conservation challenges, it sets the scene for the subsequent research.

In order to ascertain if the population decline noted by managers was real or an artefact of poor experimental design, accurate measures of population size was required. Chapter 4 assesses the accuracy and applicability of four population census techniques for 6 brush-tailed rock-wallaby colonies at Green Gully. The study includes an analysis and appraisal of aerial survey, individual recognition, time-lapse photography and photographic mark-recapture. This chapter also includes a cost-benefit analysis of these techniques, considering their ability to estimate rock-wallaby population size. The results of this study are pivotal to the design of subsequent monitoring studies for brush-tailed rock-wallabies.

Just as it was important to ascertain whether characteristics of annual aerial survey had negatively biased managers' estimates of population trends, it was also prudent to consider whether or not a real decline could have reasonably been an artefact of changes in fire management. Chapter 5 presents a case study documenting the change in the grassy understorey at Green Gully following the cessation of livestock grazing and frequent low-intensity burning to promote forage. This chapter includes an analysis of the occurrence and frequency of understorey plant species in the study area, and the implications that the change in the grassy understorey have on the availability of food resources for the brush-tailed rock-wallaby. Sample-based rarefaction curves were used to examine turnover and determine if a change occurred between the two sampling events. This chapter demonstrates the importance of National Parks' management strategies for the conservation of the grassy understorey in north-eastern New South Wales.

Chapter 6 examines the impacts of small scale, low-intensity patch burning, in tandem with macropod grazing, on the herbaceous understorey surrounding brush-tailed rock-wallaby colonies. Further, it highlights the implications of this interaction for use of fire as a conservation management tool for rock-wallabies.

Further to the consideration of fire impacts on vegetation, Chapter 7 examines the response by *P. penicillata* to fire in greater detail by investigating fine-scale changes in foraging patterns and habitat use following low-intensity fire.

Chapter 8 provides a synthesis of the research, drawing together major findings of the study within a broad theoretical framework.

2. Study area and animal

2.1. Brush-tailed rock-wallabies

2.1.1. General description

Brush-tailed rock-wallabies are a medium-sized (5 to 11 kg) member of the family Macropodidae (Order Marsupialia). They tend to inhabit structurally complex, rocky environments such as cliffs, boulder piles and rock outcrops (Joblin 1984, Murray et al. 2008, Vernes et al. 2011). Vegetation surrounding these inhabited rocky environments varies, ranging from open woodland to dense rainforest (Eldridge and Close 2008). Authors have suggested such specialised habitat use reflect the animal’s requirements for safe refuge from predators, climatic extremes and competition from sympatric herbivores (Short 1982, Joblin 1984). It has been suggested that shelter microclimate is a major determining factor in site suitability, largely due to the role that they play in thermoregulation (Murray et al. 2008).

Their preferred habitat necessitates that brush-tailed rock-wallabies are able to move swiftly and precisely through rugged and precipitous outcrops with highly precise bounds (Joblin 1984). This capacity is assisted by their long, flexible tail (utilised for balance) and roughly textured, well-padded feet (for traction) (Jarman et al. 1989). Individuals tend to be dull brown above, reddish-brown on the rump, with the neck and shoulders usually grey (Menkhorst and Jarman (2004) (Fig. 2-1) but individuals can vary considerably in appearance, due to stripes, blazes and other distinctive colour patches, and can be used to identify individuals (Chapter 4) (Joblin 1984). Jarman et al. (1989) noted that the rufous colouration of the male may be more distinctive in some individuals, compared to females.

Figure 2-1 Brush-tailed rock-wallabies at Green Gully, Oxley Wild Rivers National Park

Few empirical data exist regarding average colony size (Vernes et al. 2011), primarily due to their inaccessibility. However, work by Piggott et al. (2006a) indicated that colonies were often small, ranging between 2–30 individuals. Total population sizes of brush-tailed rock-wallabies remaining in Australia have been estimated between 15 000 and 30 000 individuals (Menkhorst and Jarman 2004). Approximately 80% of the remnant population exists in north-eastern New South Wales, predominantly within the Macleay and Clarence River gorges (Taggart 2008). Appropriate sampling methods for brush-tailed rock-wallabies must address these particular characteristics of the species, including the areas they inhabit. Otherwise, conservation management will not have key information about population trends, into the future.

2.1.2. Conservation status and distribution

Historically, the range of the brush-tailed rock-wallaby followed roughly the line of the Great Dividing Range, extending from the Grampians in western Victoria to Nanango in south-east Queensland. Naturalised populations outside of Australia can also be found in Hawaii (Lazell et al. 1984) and New Zealand (Eldridge et al. 2001).

Although brush-tailed rock-wallabies were widespread and abundant in south-eastern Australia they have declined in many areas, including throughout western New South Wales and Victoria. The species' range is now fragmented, particularly in the south where brush-tailed rock-wallabies are now mostly found in small isolated populations dotted across their former range (Short and Milkovits 1990) (Fig. 2-2). However, populations that form the Northern Evolutionary Significant Unit (ESU) stronghold exist as an interacting metapopulation, and this forms the impetus for the work conducted in this thesis.

Several threats have contributed to the decline of this species. The arrival of Europeans led to intense agriculture and the introduction of red foxes (*Vulpes vulpes*). Joblin (1984) cites both as major causes for brush-tailed rock-wallaby decline. In addition, disturbances such as fire, floods and competition for resources with herbivores (native or introduced) have been suggested as the major drivers in the species' decline (Menkhorst and Jarman 2004). Likewise, brush-tailed rock-wallabies were hunted, both for their fur and meat, and because they were considered an agricultural pest during the late 19th and early 20th century. These activities deliberately removed half a million individuals (Eldridge and Close 1995). Consequently, the species is listed as Vulnerable under the Commonwealth Environment Protection and Biodiversity Conservation Act (1999), and Endangered under the New South Wales Threatened Species Conservation Act (1995).

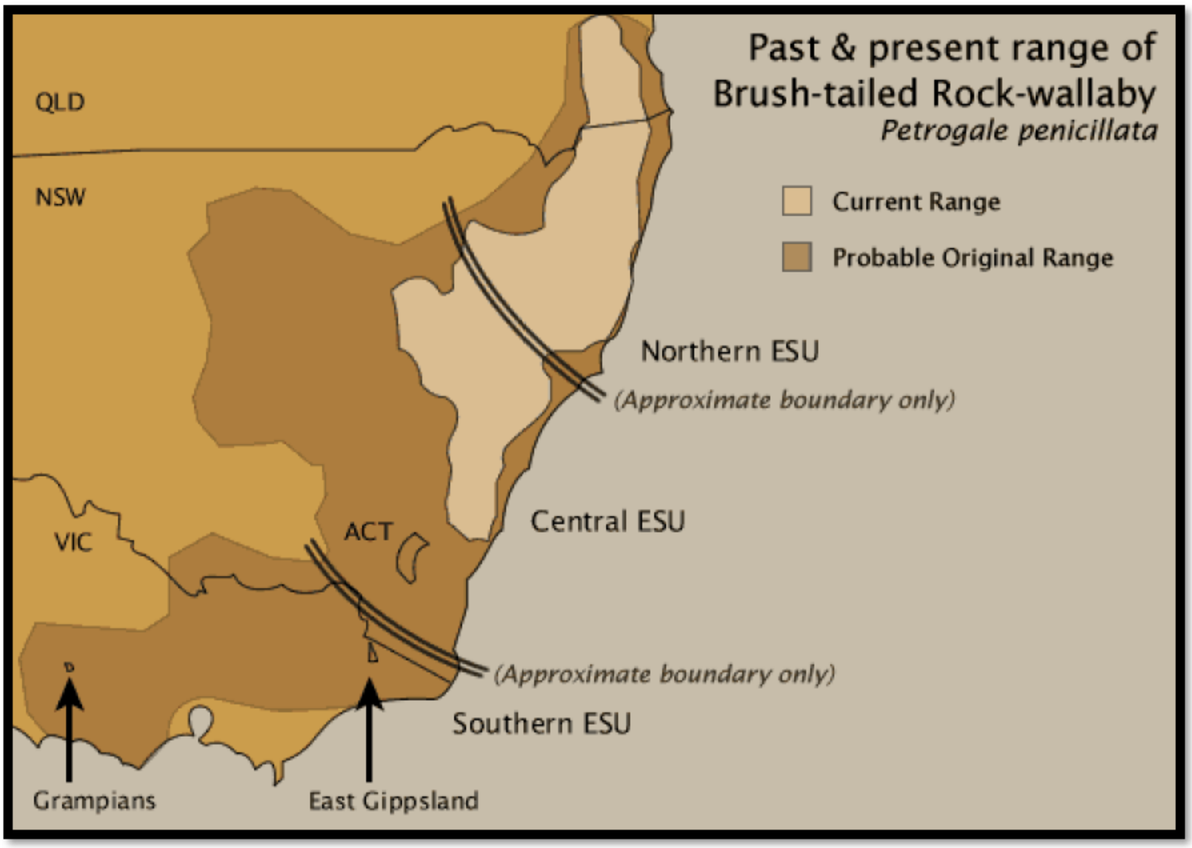


Figure 2-2 Distribution of *P. penicillata* in south-eastern Australia. Adapted from DECC (2008)

2.2. Monitoring and managing brush-tailed rock-wallabies in Australia

2.2.1. Aerial survey of macropods in Australia

Due to Australia's vast and often rugged landscape, wildlife (including macropods) are frequently monitored by aerial survey. Both fixed and rotary-winged aircraft have been utilised to census native wildlife, including kangaroos (*Macropus* spp.), along with introduced pests, such as pigs, goats, and wild horses (Hill et al. 1985, Southwell et al. 1986, Hone 1990, Choquenot 1995a, Bayne et al. 2000, Tracey and Fleming 2007, Cairns et al. 2008, Melville et al. 2008, Vernes et al. 2009). While its ability to cover large and inaccessible areas is appealing, the accuracy and precision of aerial surveys has been questioned by many authors, largely with respect to observer bias and the inability to produce consistently accurate measures in comparison to rigorous ground based estimates (Bodie et al. 1995, Clancy et al. 1997, Tracey et al. 2008, Vernes et al. 2011). Use of multiple observers, in order to develop correction factors, has been suggested to improve the overall accuracy of aerial based population estimates (Tracey et al. 2008), but the true accuracy of surveys remains unknown until rigorous ground-truthing is undertaken (Schwarz and Seber 1999).

If the overall objectives of an aerial survey involve accurate population demographic data to be collected, then it is vital that animals under survey do not develop behaviour that would be adverse to the outcomes of the census. Changed behaviour in response to repeated aerial survey has been observed in the detection of feral goats in the rangelands of north-eastern NSW (Bayne et al. 2000). Tracey and Fleming (2007) noted a learned alert response behaviour to helicopters, which involved the movement of animals frequently into areas that were perceived to be safe. This however did not result in a marked impact on their ability to detect the animals, as they were able to consistently enumerate eastern grey kangaroos on consecutive surveys (Tracey et al. 2005). Evasive behaviour in response to surveyors is common in most macropod species. Wild macropods will often react to surveyors with evasive behaviour (Southwell 1994), and due to their ability to achieve high speeds in comparison to human walking speed, the potential for movement before sightings is high. Depending on the species in question, a learned behaviour has the capacity to hinder the surveyors' ability to effectively count the animals, especially if they flee from the field of view upon the disturbance created by the approaching helicopter. Any survey that makes use of aerial components over a long period of time should therefore consider the impacts of such a reaction when evaluating the results obtained. To date, no research exists on the behavioural response of brush-tailed rock-wallabies to the sound of an approaching aircraft, however aerial survey has been utilised to survey yellow-footed rock-wallabies (Lim and Giles 1987), with no adverse behavioural responses reported.

Some merit should be afforded when considering the viability of the aerial survey technique in relation to estimating population numbers of small colonies of brush-tailed rock-wallabies within the Oxley Wild Rivers National Park. The behavioural ecology of the species renders the technique of aerial survey a potentially viable option of surveying populations, particularly within the most remote and inaccessible areas of habitat within which they occupy. Much time is afforded to basking on exposed cliff faces and ledges (Laws and Goldizen 2003) (particularly during early morning), increasing the overall detection probability of individuals during the survey period (Gowen and Vernes 2014), and thus improving the ability for the greatest number of animals to be observed during initial contact. However the probability of detecting multiple individuals at colonies is low (Gowen and Vernes 2014), and this may have major implications for the success of the aerial survey method.

2.2.2. Population estimation techniques for rock-wallabies

The effective management of wild populations of animals requires that reliable population numbers are known (Le Mar et al. 2001). For brush-tailed rock-wallabies, various ground-based techniques have been implemented in an attempt to enumerate population sizes, including genetic sampling from faecal DNA (Piggott et al. 2006a, Piggott et al. 2006b), faecal pellet counts (Jarman and Capararo 1997), individual recognition (Vernes et al. 2011), and remote camera traps (Gowen and Vernes 2014). However, all methods vary in their ability to provide vital demographic information on the size of the colonies, population structure, recruitment or dispersal.

Unique markings and morphological characteristics have been used extensively to identify individual animals in long-term observational studies in behaviour, ecology and conservation biology (Jarman et al. 1989, Hohnen et al. 2013). Such studies have used scars (Gilkinson et al. 2007), unchanging pelage patterns (Jarman et al. 1989, Karanth and Nichols 1998, Trolle and Kéry 2003, Bengsen et al. 2011), and other morphological traits (Goswami et al. 2007) to reliably identify individuals. The same principles have enabled the development of photographic mark-recapture in abundance studies (Alonso et al. 2015) and it has been variously applied to species such as tigers (*Panthera tigris*) (Karanth and Nichols 1998, Wang and Macdonald 2009), snow leopards (*Uncia uncia*) (Jackson et al. 2006), bobcats (*Lynx rufus*) (Heilbrun et al. 2006) and red foxes (*Vulpes vulpes*) (Guthlin et al. 2014) however the ability to identify individual brush-tailed rock-wallabies from remote camera traps has not been examined.

2.2.3. Managing brush-tailed rock-wallabies with fire

Fire as a management tool is now common practice throughout many areas in Australia for a variety of taxa including reptiles (Trainor and Woinarski 1994, Woinarski et al. 1999), mammals (including macropods) (Masters 1993, Meers and Adams 2003, Isaac et al. 2008, Tuft et al. 2012), birds (Woinarski 1990, Woinarski et al. 1999), invertebrates (Andersen 1991, York 2000, Andersen et al. 2006) and plants (Keith and Bradstock 1994, Bradstock et al. 1997, Russell-Smith et al. 2002). Fire is also a popular management tool in south-eastern Australia’s grassy woodlands, and, particularly for endangered ecosystems, the long term effects of contemporary vegetation management are of critical concern (Parr and Andersen 2006).

The use of small, low-intensity fires has been suggested to be beneficial for the management of brush-tailed rock-wallabies (Prober and Thiele 2005) through the provision of high quality food resources (Tuft et al. 2012). The relationships between fire and these rock-wallabies have, to date, not been studied in detail. Anecdotal reports suggest that fire has a negative impact on the persistence of rock-wallaby colonies throughout their historical range, with evidence to suggest that fire can result in rock-wallabies disappearing from a colony (Burrowa–Pine Mountain, upper Murray River), relocating (Kangaroo Valley), and disappearing and then recolonizing (Mount Wallerawang in eastern New South Wales) (NPWS 2002).

The preferential use of recently burnt vegetation has been documented in a wide variety of herbivores, for example, macropods (Meers and Adams 2003, Styger et al. 2011, Tuft et al. 2012), large African species (Klop et al. 2007, Eby et al. 2014), bison (Raynor et al. 2014, Ranglack and du Toit 2015) and white-tailed deer (Meek et al. 2008). Fire in grasslands creates an ‘ecological magnet’ for many species (Archibald et al. 2005), resulting in heavy selection and sustained use of regrowth in post-burned areas by macropods and other herbivores (Southwell and Jarman 1987, Meers and Adams 2003, Klop et al. 2007, Styger et al. 2011, Tuft et al. 2012, Eby et al. 2014, Raynor et al. 2014). For macropods, including brush-tailed rock-wallabies, the preferential use of freshly burnt areas has been shown to impact on the recovery of the ground-layer vegetation due to increased grazing pressure (Meers and Adams 2003, Tuft et al. 2012); however the extent to which individuals were selecting these areas was not quantified.

The complex interactions occurring between the grazers and the effects of the fire make it difficult to distinguish between the independent effects of the two disturbances on the ground-layer vegetation (Tremont 1994). Clearly, both fire and grazing can affect ecosystem processes independently but their combined interaction is likely greater than their effects in isolation (Allred et al. 2011).Therefore, understanding the influence of fire and grazing on ecosystems is important for conservation (Kirkpatrick et al. 2011).

2.3. Choice of study area and site selection

The study area was Green Gully (31°02’50”S, 152°05’60”E), Oxley Wild Rivers National Park. Few brush-tailed rock-wallaby sites within NSW attract intensive conservation management efforts. Green Gully is an exception, and the focus of local brush-tailed rock-wallaby management by NPWS, including annual aerial surveys.

Located in north-eastern NSW, approximately 50km east of the town of Walcha in the New England Tableland Bioregion (Fig. 2-3), Green Gully is 16 200 Ha of steep gorge country. This topography characterises the eastern edge of the Great Escarpment upon which the Oxley Wild Rivers National Park is centred (Muir 2012).Green Gully is characterised by a relatively large number of easily accessible rock-wallaby colonies, many of which are inhabited by at least five individuals.

Before its inclusion in Oxley Wild Rivers National Park (in 2004), Green Gully was part of a large grazing property. Domestic livestock were removed when Green Gully was purchased by National Parks and Wildlife with funds from the National Parks Foundation. Currently, the Park is managed to meet both recreation and conservation objectives (DEC 2005) . Planned burning is undertaken to reduce available fuel in strategic locations, limit the spread of wildfire, and to protect neighbouring properties as well as facilities, cultural assets and patches of World Heritage-listed rainforest.

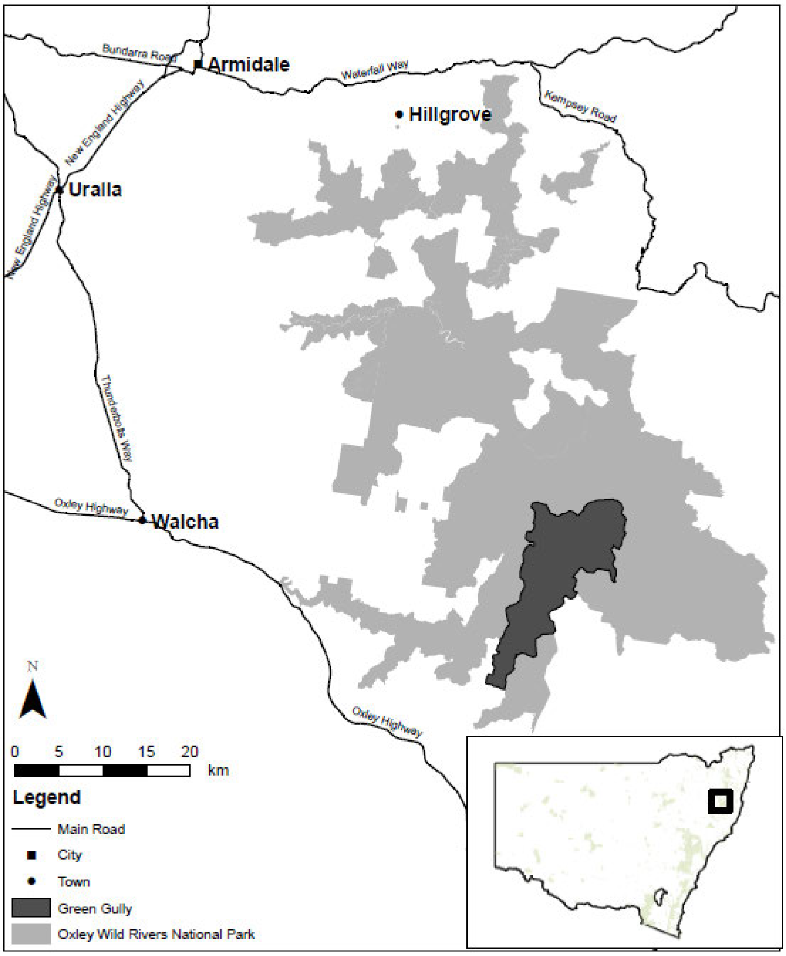


Figure 2-3 Oxley Wild Rivers National Park, in north-eastern NSW, showing the location of Green Gully

2.3.1. Distribution and characteristics of colonies

Brush-tailed rock-wallabies are known from many locations within Green Gully and the surrounding Oxley Wild Rivers National Park (Fig. 2-3). For this study, nine colonies within the Green Gully and neighbouring Yarrowitch catchments comprised the study population (Fig. 2-4). Colonies were chosen primarily due to ease of access, along with the knowledge that long-term populations of rock-wallabies were present. Distinguishing characteristics varied for each colony, however at the majority of sites, different classes of vegetation existed that provided cover, and colonies varied significantly in their structural complexity (Table 2-1; Figs 2.5-2.7).

Table 2-1 Distinguishing features of the brush-tailed rock-wallaby colonies at Green Gully.

Colony name	Location	Aspect	Altitude (m)	Distinguishing Features
E1	31°01’40”S, 152°10’86”E	E	890	• Large rock pile on the cliff line • Open site, however cracks and crevices afford some concealment
Old Camera	31°03’74”S, 152°09’09”E	E	610	• Approximately 100 m of cliff face • Open site • Large cliff face was the primary basking area, which was easily accessible for observations
Yarrowitch Lower	31°06’95”S, 152°06’44”E	SW	350	• Large rock shelf that extended for approximately 100 m • Comprised of many trees and shrubs that made it difficult to thoroughly survey the site
Fig Tree	31°03’14”S, 152°09’13”E	NE	570	• Fig trees (*Ficus spp.*) dominated the majority of the site • Large rock pile at the end of a spur • Top of the habitat easily accessible from the SW • Large NE facing cliff
Cliff	31°05’05”S, 152°08’82”E	SE	860	• Densely covered with shrubs and mature trees • Rock pile at the edge of the cliff • Entire site can be traversed on foot, though overall slope of the site makes survey difficult
Cave	31°05’04”S, 152°08’79”E	E	880	• Large rock pile with cave/tunnel running through the centre • Apex of the site dominated by a large *Ficus spp.* • Majority of the site open • Entire perimeter easily accessible
Horse Gate	31°06’54”S, 152°06’48”E	NW	490	• Relatively small boulder pile • Two well matured *Ficus spp*. • Open habitat
Yarrowitch Pinch	31°06’59”S, 152°08’04”E	SW	840	• Sweeping rock pile • Dominated by shrubs and trees • Basking sites easily identifiable
Brumby Pass	31°01’40”S, 152°10’18”E	NW	340	• Two opposing cliff faces located along the creek line • Characterised by numerous ledges, crevices and cracks that were favoured basking sites • Could not easily access NW facing cliff face, so observations were conducted form the E facing cliff line • Relatively clear of vegetation

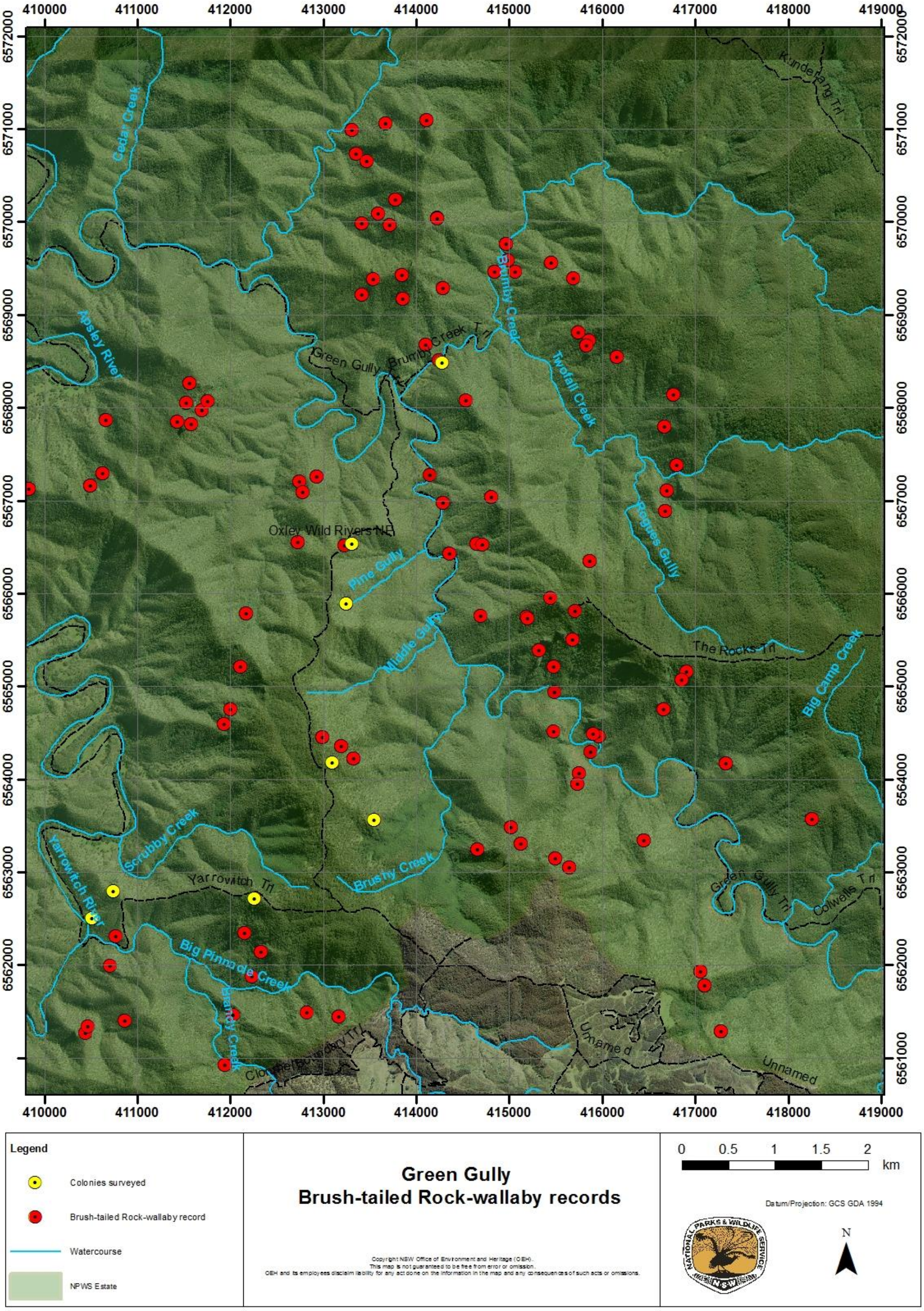


Figure 2-4 Locations of surveyed brush-tailed rock-wallaby colonies at Green Gully and Yarrowitch trail (yellow dots). Red dots indicate known rock-wallaby colonies within the greater catchment area

Figure 2-5 Brush-tailed rock-wallaby colonies at Green Gully. Sites pictured here are: a) E1 b) Old Camera c) Yarrowitch Lower

Figure 2-6 Brush-tailed rock-wallaby colonies at Green Gully. Sites pictured here are: d) Fig Tree e) Cliff f) Cave

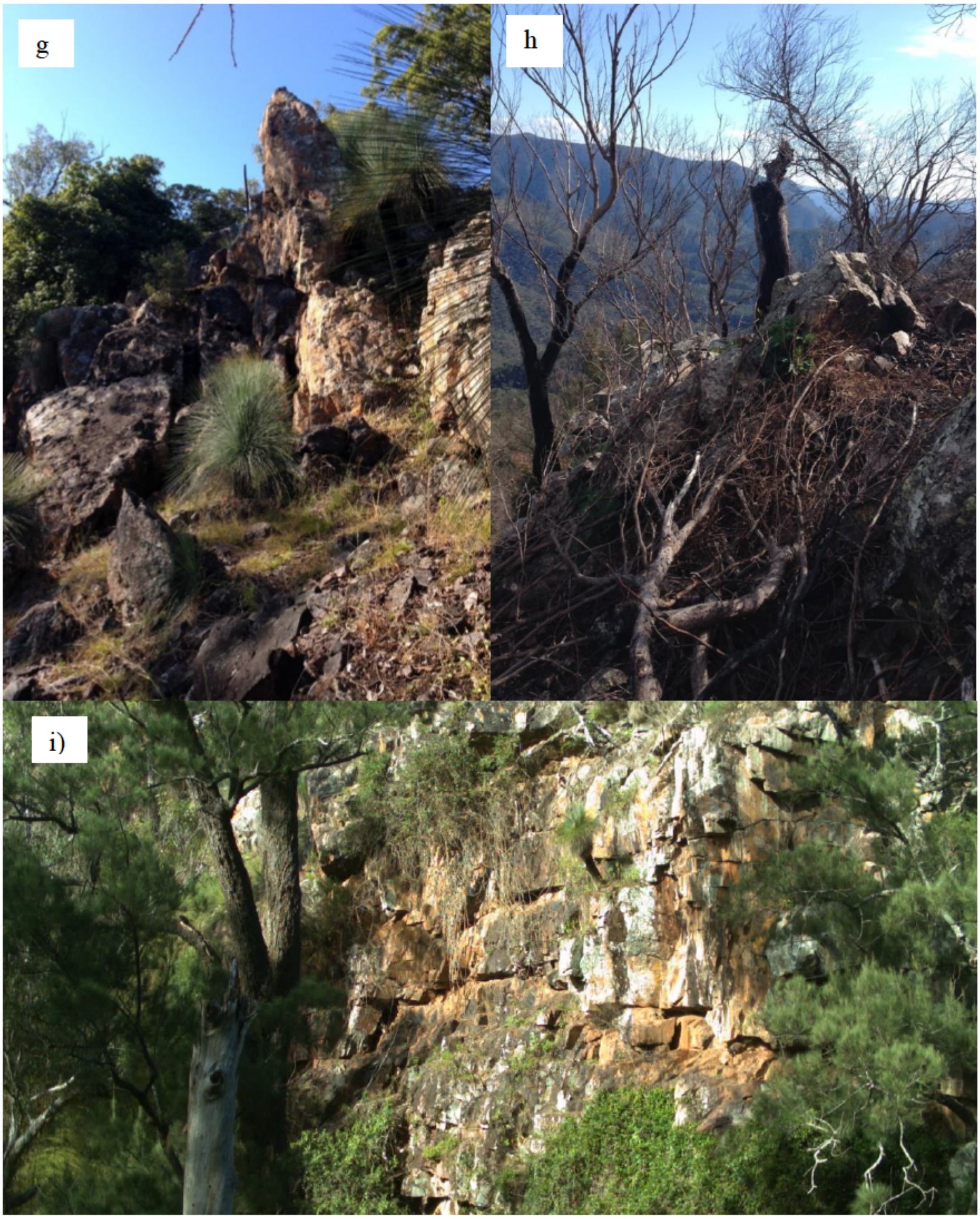


Figure 2-7 Brush-tailed rock-wallaby colonies at Green Gully. Sites pictured here are: g) Horse Gate h) Yarrowitch Pinch (post fire) i) Brumby Pass

2.3.2. Geology and geography

The majority of Oxley Wild Rivers National Park consists of sedimentary rock types dating from the Devonian and Permian Periods. The southern section lies on the sedimentary Myra Beds, whilst the northern section of the park is composed of schistose sandstone and schistose conglomerate. Along the rivers alluvial deposits from the Quaternary Period are found along the rivers and major drainage lines (Morris 2005).

Local topography is typical of that found in Oxley Wild Rivers National Park, with steep-sided valleys and gorges, and occasional valley flats and escarpments. The intersection of two large river systems, the Apsley and Yarrowitch, with the smaller Green Gully Creek and other minor drainage lines, has created a mixture of elevation and aspect inhabited by a range of vegetation. In the southern and mid-sections of Green Gully elevations range from 300 to 1000 m a.s.l. but maximum elevations occur in the more uniform, north section, and range from 1000 to 1200 m a.s.l.

2.3.3. Regional climate

The local climate is cool temperate, with distinct summer and winter seasons. The wettest period typically extends through November until February, with the remainder of the year being substantially drier. Annual rainfall averages approximately 615 mm, with monthly averages varying between 10 and 100 mm in the spring and summer. Winter monthly rainfall varies between 10 and 50 mm (BOM 2015). Average daily temperature in the summer approximates 29 ºC, and in the winter average daily temperature is approximately 20 ºC (Fig. 2-8).

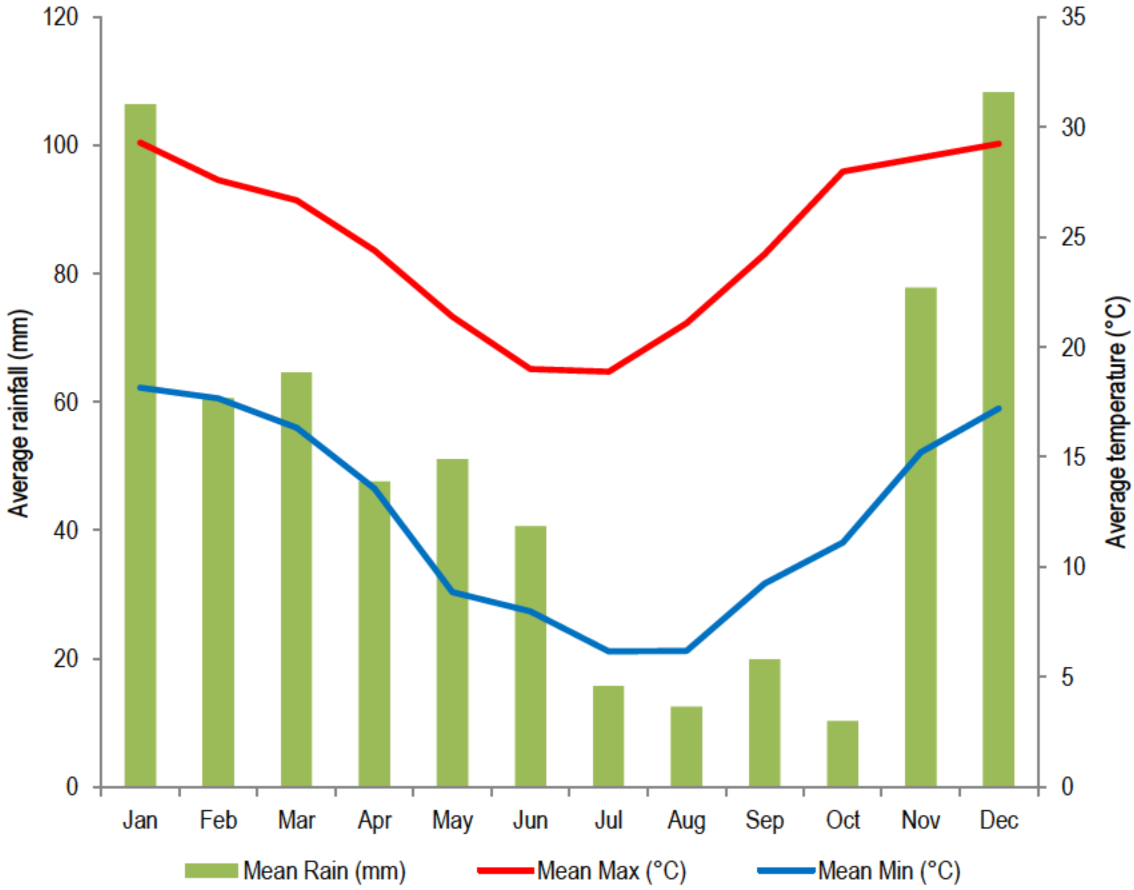


Figure 2-8 Local trends in rainfall and temperature from 2012 to 2015. Rainfall averages from Benditi, near Green Gully and temperature is from Gingers Creek, 30km away Source: (BOM 2015)

2.3.4. Vegetation

A number of vegetation communities and associations are present within the study area, including dry rainforest, gorge rim communities, dry sclerophyll woodlands, plateau remnants and riparian valley floor communities (Thomas 2003). The most widespread plant community found at Green Gully are the areas of dry sclerophyll woodlands. Classified as Northern Gorge Dry Sclerophyll Forests by Keith (2004) , these communities are dominated by species such as *Eucalyptus tereticornis, E. laevopinea, E. melliodora* and *Angophora floribunda*. The presence of a continuous grassy ground layer, along with a mixture of sclerophyllous and mesophyllous shrubs is characteristic (Keith 2004, Morris 2005).

The lower slopes encompass riparian and valley floor communities at Green Gully. Morris (2005) describes four plant communities occurring along with lower slopes and valley floors of Green Gully, with the most common vegetation type being the *Casuarina cunninghamiana* subsp. *cunninghamiana* and *Melaleuca bracteata* communities lining the major watercourses within the area. In addition, *E. conica, E. tereticornis, E. moluccana, E. albens* and *E. dealbata* also dominate the riparian communities (Keith 2004).Visible pockets of dry rainforest communities are generally restricted to the protected, steep gullies and drainage lines of the area, and are commonly interspersed within the eucalypt woodland matrix (Thomas 2003, Morris 2005). Species such as *Alectryon forsythia, Notelaea microcarpa, Olea paniculata* and *Backhousia sciadophora* are the most common canopy plants (Keith 2004, Morris 2005).

2.3.5. The mammal community at Green Gully

Along with brush-tailed rock-wallabies, several other macropodids also occur within the study site. These include the swamp wallaby (*Wallabia bicolor*), eastern grey kangaroo (*Macropus giganteus*), common wallaroo (*Macropus robustus*), red-necked wallaby (*Macropus rufogriseus*) and parma wallaby (*Macropus parma*). Many other mammal species have been recorded in the area (Table 2-2).

Table 2-2 Species of mammals observed to be present within the *Eucalyptus* woodland habitat at Green Gully

Common Name	Scientific Name
Monotremes	
Short-beaked Echidna	*Tachyglossus aculeatus*
Platypus	*Ornithorhyncus anatinus*
Dasyurids	
Spotted-tail Quoll	*Dasyurus maculatus*
Brush-tailed phascogale	*Phascogale tapoatafa*
Antechinus	*Antechinus* spp.
Possums	
Common Ringtail Possum	*Pseudocheirus peregrinus*
Common Brushtail Possum	*Trichosurus vulpecula*
Short-eared possum	*Trichosurus caninus*
Bandicoots	
Long-nosed Bandicoot	*Perameles nasuta*
Northern brown bandicoot	*Isoodon macrourus*
Macropods	
Brush-tailed rock-wallaby	*Petrogale penicillata*
Swamp Wallaby	*Wallabia bicolor*
Red-necked Wallaby	*Macropus rufogriseus*
Eastern Grey Kangaroo	*Macropus giganteus*
Common Wallaroo	*Macropus robustus*
Parma wallaby	*Macropus parma*
Canidae	
Dingo / Wild dog	*Canis lupus dingo* x *familiaris*

Introduced/Feral Species	
Red fox	*Vulpes vulpes*
Feral cat	*Felis catus*
Feral Pig	*Sus scrofa*
Domestic Cattle	*Bos taurus*
Domestic Horses	*Equus caballus*
Feral goats	*Capra hircus*
Rabbit	*Orytcolagus cuniculus*
Hare	*Lepus europaeus*

*In NSW, the term ‘wild dog’ includes dingoes. Under NSW legislation dingoes are considered both native and declared pests

3. Do behavioural responses of brush-tailed rock-wallabies to aerial surveys adversely impact population estimates?

3.1. Introduction

Aerial survey has traditionally involved aircraft bringing on-board observers to an area of interest, where they attempt to locate, identify and count the wildlife of interest as the aircraft passes overhead (Fleming and Tracey 2008). As detailed in Chapter 4, the technique is often a method of choice for land managers who need to monitor wildlife where ground-based surveys would be hindered by spatial scale, limited access, or dense vegetation (Ransom 2012). In Australia, aircraft are frequently utilised to survey native wildlife, including kangaroos (*Macropus* spp.), and introduced pests, such as pigs (*Sus* spp.), goats (*Capra aegagrus*), and wild horses (*Equus caballus*) (Hill et al. 1985, Southwell et al. 1986, Hone 1990, Choquenot 1995a, Bayne et al. 2000, Tracey and Fleming 2007, Cairns et al. 2008, Melville et al. 2008, Vernes et al. 2009). Aerial survey is also utilised for birds (Kingsford and Porter 2009) and marine mammals (Preen and Marsh 1995, Pollock et al. 2006).

The extent to which aerial survey changes the behaviour of target animals, and affects the outcomes of monitoring programs is an important question. Adverse reactions from target animals can compromise fundamental assumptions of the survey, such as an equal probability of detection for all individuals (Pollock and Kendall 1987, Choquenot 1995a, Southwell et al. 2007) and influence both the precision and accuracy of population estimates (Linklater and Cameron 2002, Tracey and Fleming 2007).

Behavioural responses to aerial survey, often assumed to be anti-predator behaviour, vary within and between species. Observed behaviours can range from increased vigilance (Harrington and Veitch 1991) to a panic response (Calef et al. 1976). Factors affecting the severity of response can also vary, for example with distance to the survey platform (Calef et al. 1976, MacArthur et al. 1982), sex and age of target animals (Lenarz 1974), cover (Krausman et al. 1986), time of day (Bayliss and Giles 1985), group size (Czech 1991) and topography (Cote 1996). The type and extent of responses have also been linked with experience of exposure to aircraft (Krausman et al. 1986), hunting (aerial shooting) (Bayne et al. 2000), sampling intensity and survey platform characteristics including altitude of the aircraft (Southwell 2005) and its type (Fleming and Tracey 2008).

Whether or not animals characteristically seek or break from cover, freeze or move, and disperse or group together in response to an aircraft may have a profound effect on the accuracy and precision of population estimates (Bleich et al. 1992, Linklater and Cameron 2002, Tracey and Fleming 2007). To this end, Southwell (2005) emphasised the importance of considering the responsiveness of target animals to aerial survey. Although there may often be an assumption of negative bias, it is important to note that responses by the target animal may also enhance species identification, with the simple movement of the organism potentially providing observers with additional information (Southwell et al. 1995, Vernes et al. 2009).

Since 2004, NSW National Parks and Wildlife Service have been conducting annual population surveys of the brush-tailed rock-wallaby colonies within Oxley Wild Rivers National Park; using aerial surveys (see Chapter 1**)**. However, no assessment of the behavioural response of the rock-wallabies to those surveys has been made. In response to an apparent population decline deduced from these aerial surveys (Chapter 1) local managers hypothesised that behavioural responses of brush-tailed rock-wallabies to the annual aerial surveys might have been negatively biasing population estimates. Characteristic anti-predator behaviour exhibited by brush-tailed rock-wallabies varies from minor responses such as foot thumping, alert body posture and fixation on the perceived threat, to more extreme responses such as fleeing to refugia (Carter and Goldizen 2003). It is therefore conceivable that behavioural responses to aerial survey could bias population counts in either a positive or negative way.

No published data has attempted to quantify the behavioural effects of aerial survey on populations of brush-tailed rock-wallabies. Consequently, I aimed to document and assess the behavioural response of brush-tailed rock-wallabies to multiple aerial surveys, in order to determine if responses to the technique were likely to bias population estimates.

3.2. Methods

3.2.1. Study site and colony selection

The behavioural response of brush-tailed rock-wallabies to helicopter-based aerial surveys was documented at eight discrete colonies (n = 5 aerially surveyed and n = 3 not aerially surveyed, ‘control’ colonies) at Green Gully in June 2014 and June 2015. Colonies varied in physical condition, ranging from cliff faces to large boulder piles (for a detailed description of each colony, see Chapter 2). Aerial surveyed colonies were allocated to colonies that had been aerially surveyed annually for the past decade. Control colonies were chosen from known localities that had had no previous exposure to aerial surveys, and although all were located to the west of the aerially surveyed colonies, allocation was made for practical reasons. Spatial interspersion and variation in basking behaviour was between the control and experimental colonies was not considered to be a limiting factor. From complementary on-ground surveys (Chapter 4), brush-tailed rock-wallaby population size at these colonies was estimated to range from 4 to 20 individuals.

Given that individuals are generally associated with particular colonies (Piggott et al. 2006b), any animals greater than a year old were assumed to have previously been exposed to aerial surveys at the treatment (surveyed) colonies. This assumption was supported by the ongoing presence of marked/known individuals at colonies over multiple years (Chapter 4).

Most aerially surveyed colonies were north-east facing, whilst all control colonies had a westerly aspect. Despite these differences in aspect, it is unlikely that the ability to document behavioural responses was affected, as it was the change in behaviour in response to the aircraft passing overhead that was being examined, rather than broad-scale activity patterns.

3.2.2. Aerial surveys

Annual aerial surveys for brush-tailed rock-wallabies within Oxley Wild Rivers National Park comprise four replicates of a daily survey undertaken from a Bell Jetranger helicopter. In 2014 these four days were consecutive but in 2015 poor weather required the survey to extend over non-consecutive days. On each survey day, the pilot flew the helicopter along the same pre-determined route at approximately 45 knots ground speed between 0800 and 1100 h each day. Colony locations were pre-loaded into the aircraft’s GPS-receiver and each flight was also logged via GPS to enable post hoc alignment with camera data (Fig 3-1). At each brush-tailed rock-wallaby colony, the pilot manoeuvred the aircraft to within 50 to 100 m from the target area, so long as it was safe to do so. Two to four observers independently recorded all individual brush-tailed rock-wallabies they sighted and data were summarised (by NPWS staff) as the maximum number of individuals sighted at colony (hereafter minimum known alive, or MKA) during any survey. From 2004 to 2011, data were recorded in hard copy form and later transcribed into digital format. From 2012 onwards, observers recorded count data using a ‘press-button’ digital data logger, connected to a GPS receiver

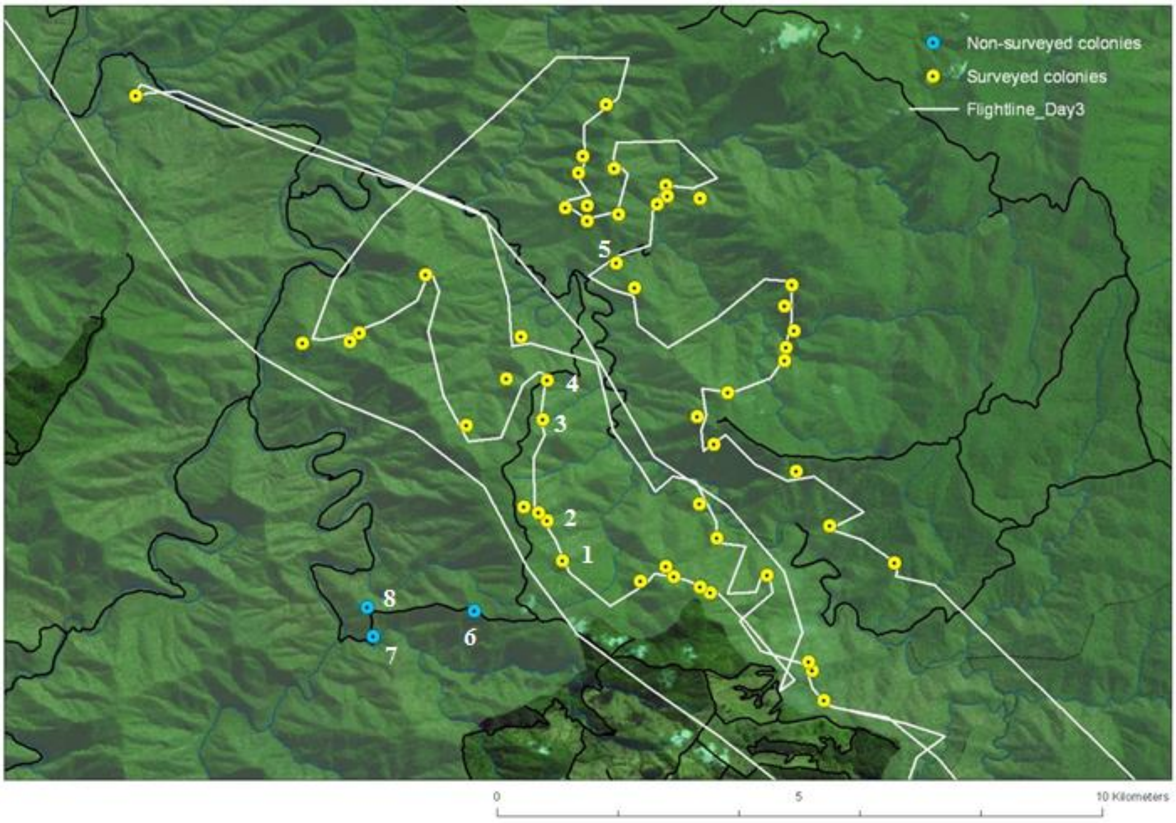


Figure3-1 Flight path of NPWS helicopter during an aerial survey of Green Gully. Colonies 1–5 were aerially surveyed, whilst colonies 6–8 were controls (not surveyed).

3.2.3. Behavioural observations

Camera traps (n = 26) were deployed across the eight study colonies prior to the annual aerial survey in order to monitor rock-wallaby behaviour at basking sites. Cameras were mounted on steel posts (at approximately 0.5 m height above ground level). A mixture of Reconyx Hyperfire HC600 and Brinno TLC 100 cameras were used. Mixing camera traps can be a legitimate source of concern in some studies, due to differential detection probabilities, but in this case they were used as time-lapse units only, negating issues associated with differences in the various models' passive infra-red detection capabilities.

Some cameras were mounted horizontally in order to adjust for differences in colony structural complexity, and to maximise the useable field of view. Where the basking area was larger than the field of view of a single camera, multiple cameras were used. Care was taken to ensure individual fields of view did not overlap. Minor modification of habitat (e.g. removal of vegetation immediately in front of the camera) was sometimes required to clear the field of view.

Every camera was programmed to record an image each minute from 0600 to 1400 h (480 images per camera, per day). This period of the day overlaps NPWS aerial surveys and is when brush-tailed rock-wallabies are commonly observable on colonies (Gowen and Vernes 2014). At one minute intervals between photos, each camera could take sufficient images to cover the daily survey period but not exhaust batteries or storage media, thereby avoiding the need for human interference at the colony until after the annual surveys were complete.

3.2.4. Processing of camera trap data

Each image was reviewed using ExifPro 2.1 software. Metadata tags for species and number of individuals were assigned to each image, in addition to the time and date metadata tags automatically added to each image by the cameras. Customised software written by Dr Gerhard Körtner (School of Environmental and Rural Science, UNE) was used to collate metadata tags and export summaries of data for each camera at each colony.

3.2.5. Modelling behavioural response data

Observations were separated into three time categories based on the approach of the helicopter to the rock-wallaby colonies, during the survey period. These were:

1) Pre (10 minutes prior to the survey period)

2) Survey (time of contact) and

3) Post (10 minutes after they survey period).

A buffer of ten minutes was used to account for overt disturbance behaviours associated with sounds of a helicopter within the survey area. Survey period for the nil-treatment colonies was allocated as per the treatment colonies in order for comparisons to be made.

The rock-wallaby count data were summarised by grouping photos into five minute intervals then calculating a mean count for each interval. Each camera was treated separately. Data were then modelled by fitting splines to the grouped count data. Those data recording during the time of aircraft passage over the colony (Period 2) were removed from the analysis. For each colony the model which was fitted was

$$\text{Lcount} \sim \text{Camera*Date*Time} + \text{spl(Time)} + \text{Date:spl(Time)} + \text{error}$$

where Lcount is calculated as log(Count+0.9). The models were used to predict a count value for Period 2 which was subsequently compared to the observed count. This was done for each camera and each date where there was an aircraft present. The predicted values represent rock-wallaby abundance, averaged over five minute intervals, immediately prior to and just the aircraft passage (Periods 1 and 3). The difference between the predicted and observed counts (Period 2) was fitted to a second model, specifically:-

$$\text{Difference} \sim \text{Date*Colony} + \text{error}$$

Behavioural response at a colony level was modelled to ascertain if population size had a significant impact on the presence of flight response. The wallaby count data was summarised by calculating a mean count for each interval during the survey period (Pre, During and Post). This approach was applied to both control and treatment colonies. The data were analysed using a Generalised Linear Model with a log link and Poisson variance, allowing for over-dispersion. Year and treatment (control, surveyed) and their interaction were fitted as fixed factor effects, with colony and date within colony fitted as random effects.

3.3. Results

3.3.1. Modelled response data

Significant differences were identified in the amount of activity between the fixed terms of Camera, Date and Time (*P*= ≤0.001). Random terms were generally significant – for spl(Time), *P*<0.05 and for Date:spl(Time), *P*≤0.001. Data from one camera was not included in the models because the counts were generally too low (this may have been due to poor camera placement relative to a predicted basking site).

Results of the GLM indicated that the behavioural response of rock-wallabies to aircraft did not differ significantly between colonies or between colonies within year (P>0.05) (Table 3-1). A significant difference in mean counts was noted between year of survey (*P* ≤0.001) (Table 3-1) and between colonies (Table 3-2).

In both years the predicted counts for the surveyed and not surveyed periods were similar (0.51 vs 0.50 in 2014 and 0.85 vs 0.88 in 2015) (Table 3-3). Averaged over both years the predicted counts were also very similar (0.66 in both periods) (Table 3-2). Predicted mean counts varied between colonies, for example a mean predicted count of 0.3 for the Horse Gate colony vs 2.0 for the Brumby Pass colony (Table 3-4).

Table 3-1 Results of the model describing the relationship between aerial survey and behavioural response of brush-tailed rock-wallabies at Green Gully between 2014 and 2015

	Pr(Chisq)
Year	<0.001
Colony	0.88906
Year*Colony	0.78013

Table 3-2 Summary table comparing surveyed and non-surveyed rock-wallaby mean predicted counts at Green Gully

Aerially Surveyed	Predicted Count	Standard Error
Y	0.66	0.16
N	0.66	0.16

Table 3-3 Summary table comparing surveyed and non-surveyed rock-wallaby mean predicted counts in 2015 and 2015 at Green Gully

Year	Aerially Surveyed	Predicted Count	Standard Error
2014	N	0.51	0.14
2014	Y	0.50	0.12
2015	N	0.85	0.22
2015	Y	0.88	0.22

Table 3-4 Summary table comparing mean predicted counts at surveyed and non-surveyed rock-wallaby colonies at Green Gully

Colony	Aerially surveyed	Predicted Count	Standard Error
Brumby Pass (5)	Y	2.04	0.17
Cave (1)	Y	0.65	0.11
Cliff (2)	Y	0.60	0.08
Fig Tree (4)	Y	0.32	0.04
Old Camera (3)	Y	1.73	0.16
Horse Gate (7)	N	0.30	0.04
Yarrowitch Lower (8)	N	0.68	0.08
Yarrowitch Pinch (6)	N	0.23	0.14

3.3.2. Behavioural observations from time-lapse images

No evidence of fleeing was observed from the images collected during Period 2 (when the aircraft was at the colony). Rock-wallabies exhibited typical basking behaviour, and remained in the same position throughout the duration of the ‘survey’ time category (Fig. 3-2).

Interestingly, a flight response was observed by brush-tailed rock-wallabies when a wedge-tailed eagle (*Aquila audax*), landed at one of the colonies and was captured on remote camera. These birds are known to kill brush-tailed rock-wallabies at the site. In that encounter, rock-wallabies immediately fled, and did not return to normal basking behaviour for a number of hours (Fig. 3-3).

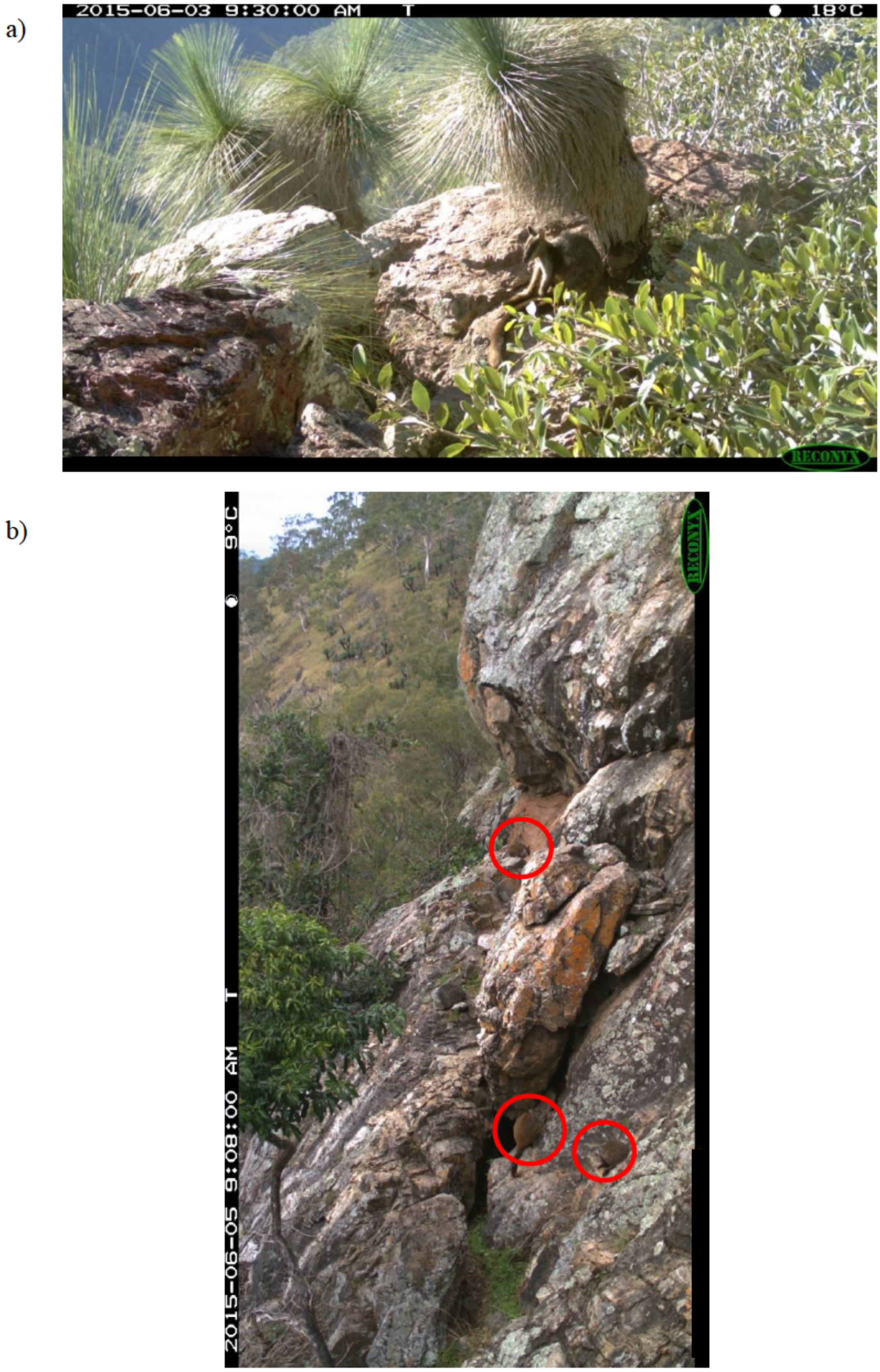


Figure 3-2 Images of brush-tailed rock-wallabies during the survey period 2, at a) Cliff colony and b) Old Camera colony

Figure 3-3 Sequence of images showing presence of rock-wallabies (10:24 am) immediately preceding the arrival of a wedge-tailed eagle at 10:25 am. Individuals do not resume basking for over an hour, returning at 12:01 pm.

3.4. Discussion

Aerial surveys at Green Gully with low and slow-flying helicopters did not appear to stimulate flight responses by brush-tailed rock-wallabies. Likewise, there were no apparent effects on animals' immediate or longer-term welfare; however it is difficult to predict if similar results would have occurred if the populations were being surveyed for the first time. Therefore, future studies aimed at examining the behavioural response of brush-tailed rock-wallabies at colonies with no history of aerial survey are required. No significant differences were observed in mean counts for the pre, during and post surveys between control and surveyed colonies. This accorded well with behavioural observations from the surveyed colonies; no brush-tailed rock-wallabies were seen fleeing during the approach of helicopters.

There was however, a significant difference in predicted mean counts between years. This may have been a function of variation in individual basking behaviours, possibly in temporal activity patterns, and the number of individuals utilising the basking sites between the two annual surveys. Minor changes to the physiological state of individuals were not measured in this study, so changes in heart rate or body temperature could not be detected. However, from an ethological perspective, no fleeing was noted. Although this finding might be considered positive from an animal welfare perspective, it might also be a decided negative for the success of the surveys; if animals did not flush, they might have been missed by observers. Furthermore, Melville et al. (2008) point out that vegetative cover will always result in an unknown proportion of the target animal being unavailable to observers irrespective of the potential flight behaviour exhibited. In the case of Green Gully, visibility due to vegetative cover varies significantly between colonies, rendering it difficult to assign an index of abundance that would allow researchers to track population trends.

A number of factors contribute to the ability of observers to recognise specific objects, including their outline, colour and contrast against a background, as well as their movements (Bruce et al. 2003). Fleming and Tracey (2008) state that rather being a hindrance, animal movement aids in the recognition of individuals as they can be more readily identified by observers than if motionless. For example, a study by Bodie et al. (1995) on aerial survey of bighorn sheep (*Ovis canadensis*) showed that 76% of moving *O. canadensis* were recorded by observers, whereas only 39% of stationary animals were seen. Similar observations have been made for the aerial survey of red kangaroos (*Macropus rufus*) (Clancy 1999)

Historically, observations made during aerial surveys of macropods suggest that kangaroos commonly move away from a helicopter, with the subsequent responsive movement posing a significant problem for the accuracy and precision for aerial surveys (Fewster et al. 2008). In contrast, Lim and Giles (1987) noted no adverse behavioural responses of yellow-footed rock-wallabies to aerial surveys. Adequate knowledge of the target animals' biology is vital when designing an aerial survey, as animal activity can influence the visibility/detection of each species. Sheltering behaviour of macropods (affected by temperature and time of day) will significantly impact on the probability of detection and must be taken into account (Bayliss and Giles 1985). Brush-tailed rock-wallabies spend the majority of the day in basking positions, often remaining in a single area for hours (Joblin 1984, Gowen and Vernes 2014). A lack of reactive movement, along with masking by overhead vegetation can have significant implications for the success of aerial surveys. Choquenot (1995b) noted a high visibility bias in relation to vegetative cover when surveying kangaroos, with 89% of individuals missed in thick riverine habitat, compared to the 23% of individuals not observed in open grassland.

Overall size of the target animal will also influence detection. Generally, the larger the animal, the greater the probability of detection (Jachmann 2002). However, even large animals such as elephants (Caughley and Goddard 1975) are missed during aerial survey. Other factors also impact upon the probability of detection of animals by aerial observers, including group size and social behaviour, the shape and colour of animals and also inherent responsive movement. For example, Jachmann (2002) noted that the contrast between the pelage colour of the target species and the background significantly affects detection from the air. This finding has implications for aerial survey of brush-tailed rock-wallabies. The most common colouration of the species is dull brown above, tending to reddish-brown on the rump, with the neck and shoulders usually grey (Menkhorst and Jarman 2004). The similarly coloured granitic outcrops which this species inhabit throughout the study site therefore provide little contrast to increase the overall probability of detection of rock-wallabies at Green Gully. This, along with the absence of reactive movement may result in substantial visibility bias for observers.

Increased exposure to aircraft has the capacity to alter flight behaviours of animals, by generating a learned alert response (Tracey and Fleming 2007). Bayne et al. (2000) described a learned alert response and a subsequent reduction in detection of feral goats after being exposed to regular aerial hunting. In addition, detection ‘shyness’ and a reduction in detection probability was observed in feral horses exposed to repeated aerial survey (Vernes et al. 2009).

Alternatively, animals may habituate to aerial survey, as demonstrated by Weisenberger et al. (1996) in a study of desert mule deer (*Odocoileus hemionus crooki*) and bighorn sheep (*Ovis canadensis*). Both species habituated to the sound of simulated jet aircraft noise and responded less frequently as the study progressed (112 simultaneous overflights per season, for 3 seasons). However, surveys at Green Gully occur so infrequently that it seems unlikely that habituation would occur. Rather, it seems reasonable that the lack of apparent response to aircraft could be due to other factors, such as a relatively low perceived risk to individuals' welfare.

Experience significantly influences the prey animals' perception of fear. The more certain an animal is that the approaching predator intends to attack, the more risk it will perceive (Stankowich and Blumstein 2005). When confronted by a potential predator, rapid assessment of the inherent risk and subsequent decision making are vital (Bouskila and Blumstein 1992). Importantly, different predator species will have a different level of risk assigned to them by prey (Walther 1969), and this may be related to predator-specific behaviour (Stankowich and Blumstein 2005).

Visual appearance, rather than the sounds, of predators has been documented by multiple authors as initiating a greater alert response in small macropods. Blumstein et al. (2002) reported that red-necked pademelons (*Thylogale thetis*) responded to the sight of common predators (such as the red fox) and not their sounds. Tammar wallabies (*M. eugenii*) have been found to responded similarly (Griffin et al. 2003).

Rock-wallabies are commonly preyed upon by wedge-tailed eagles (*Aquila audax*) (Short 1982, Menkhorst and Jarman 2004). On the one occasion that my cameras recorded the occurrence of a wedge-tailed eagle at a colony, a strong apparent behavioural response by rock-wallabies was observed. According to Carter and Goldizen (2003), brush-tailed rock-wallabies do not rely on crypsis to avoid predators, but instead rely upon early detection and a quick escape. The lack of responsive movement by these rock-wallabies in response to the approaching aircraft therefore suggests that the perceived threat from an eagle is much greater than the risk associated with a low-flying aircraft.

3.4.1. Management Implications

Accurate and repeatable estimates of population size, distribution and growth are necessary for the appropriate management of populations (Linklater and Cameron 2002). For the management of threatened species, like brush-tailed rock-wallabies, rigorous estimates of population size and change that can be defended in scientific and public fora are imperative if management is to be effective. Prior to this study, no literature addressed the perceived problem of brush-tailed rock-wallaby behavioural responses to low-flying aircraft and implications for the success of surveys.

Response to helicopters was not identified as a significant source of bias for the aerial survey of rock-wallabies at Green Gully, and thus conservation managers' concerns about declining counts of rock-wallabies from annual surveys (see Chapter 1) could not be accounted for by behavioural responses to helicopters. Therefore, alternative explanations needed to be considered. The next logical step was to consider whether the aerial survey was reliable, by seeking to compare it with other methods that would seek to obtain a total count of a subset of the known colonies (*sensu* Pollock and Kendall (1987), which is the focus of Chapter 4).

4. Cost and efficiency of population estimation techniques: a comparison of methods for the brush-tailed rock-wallaby

4.1. Introduction

Conservation managers need rapid and cost-effective means to assess the abundance of wildlife species that are cryptic or otherwise difficult to census (Silveira et al. 2003, Oliveira-Santos et al. 2010). Sampling for such fauna was once limited to detecting spoor (Bider, 1968), but techniques to estimate animal abundance have evolved to encompass approaches such as observation along line transects, aerial survey and intensive trapping and tagging programmes (Green 1985, Vernes 1999, Fashing and Cords 2000, Jones et al. 2004). Methods differ in their accuracy, precision and cost-efficiency, all of which can affect interpretation of data and the efficacy of management (De Bondi et al. 2010).

Understanding the limitations of measurement options (along with a clear understanding of the objectives of the research and characteristics of the target animal) is essential (Lyra-Jorge et al. 2008b, De Bondi et al. 2010). For example, line transects have been widely applied for the estimation of animal abundance because they are regarded as simple, economical and relatively precise (Schwarz and Seber 1999) but reactive movement, e.g. as a result of a response to an observer, can impact upon population estimates (Southwell 1994).

Aerial survey has been used internationally for a range of medium to large terrestrial mammals such as such as white-tailed deer (*Odocoileus virginianus*) (Stoll et al. 1991), goats (*Capra hircus*) (Bayne et al. 2000), caribou (*Rangifer tarandus*) (Calef et al. 1976), bighorn sheep (*Ovis canadensis*) (Bodie et al. 1995), elephants (*Loxodonta africana*) (Vermeulen et al. 2013) and zebra (*Equus grevyi*) (Parker et al. 2011). It is also utilised to survey marine mammals (Bayliss 1986) and wetland birds (Porter et al. 2006).

Aerial survey has been used extensively in Australia, particularly for macropods and feral goats (Short and Bayliss 1985, Short and Hone 1988, Grigg et al. 1997, Cairns 2003). The technique typically involves observers on-board a low-flying aircraft (fixed wing or helicopter) counting target animals along a predetermined transect (Grigg et al. 1997). Although aerial survey enables the detection of animals in areas that are relatively inaccessible and therefore often impractical for ground-based enumeration techniques, variable detection probabilities can hinder precise estimates of absolute abundance (Tracey et al. 2005). Fortunately, this issue can be accounted for with the use of correction factors (Tracey et al. 2008).

Recently, the use of indirect and relatively non-invasive monitoring techniques, e.g. camera trapping, has become a popular and increasingly viable option (Cutler and Swann 1999, Meek et al. 2015). Most camera traps are passive devices designed to take images of animals that pass by them (Meek et al. 2012, Rovero et al. 2013). In the last decade, use of camera traps in ecological studies has increased dramatically, particularly due to the relatively low cost and maintenance (once installed), assumptions of minimal disturbance and distress placed on subjects and the ability of the photos to provide information on date, time, activity patterns and individual identification if animals have individually-distinct marks.

Unique markings and morphological characteristics have been used extensively to identify individual animals in long-term observational studies in behaviour, ecology and conservation biology (Jarman et al. 1989, Hohnen et al. 2013). Such studies have used scars (Gilkinson et al. 2007), unique pelage patterns (Jarman et al. 1989, Karanth and Nichols

1998, Trolle and Kéry 2003, Bengsen et al. 2011) and other morphological traits (Goswami et al. 2007) to reliably identify individuals. The same principles have enabled the development of photographic mark-recapture in abundance studies (Alonso et al. 2015) and it has been variously applied to species such as tigers (*Panthera tigris*) (Karanth and Nichols 1998, Wang and Macdonald 2009), snow leopards (*Uncia uncia*) (Jackson et al. 2006), bobcats (*Lynx rufus*) (Heilbrun et al. 2006) and red foxes (*Vulpes vulpes*) (Guthlin et al. 2014).

Cost-benefit comparisons of population sampling methods are often overlooked, and indeed may be unfeasible for practitioners, but they are indispensable (Cutler and Swann 1999, Gaidet-Drapier et al. 2006, De Bondi et al. 2010). To date, assessments of the relative cost-benefit of methods for monitoring population trends in brush-tailed rock-wallabies have been limited. As a consequence, conservation managers have lacked key information for planning and implementing monitoring programs.

In this chapter, I present comparisons of the cost and efficacy of aerial survey, which has been used by the managers of Green Gully for ten years, with three alternative methods for estimating brush-tailed rock-wallaby population size. My aims were to 1) determine the accuracy of the aerial survey by comparing estimates with those collected via three camera-based methods 2) calculate and compare the cost and efficacy of the aerial survey with the three ground-based techniques and 3) provide managers with the information required to make informed decisions about monitoring as they seek to conserve the local population of brush-tailed rock-wallabies.

4.2. Methods

4.2.1. Colony selection and timing of fieldwork

Population size was estimated at six discrete brush-tailed rock-wallaby colonies in Green Gully, between 2013 and 2014. The colonies varied in physical condition, ranging from cliff faces to large boulder piles (for a detailed description of each colony, see Chapter 2). Cover consisted of fig trees (*Ficus* spp.) various *Eucalyptus* spp. and small shrubs (see Chapter 2).

Ground-based surveys were planned to coincide as closely as possible with annual aerial surveys of brush-tailed rock-wallabies conducted by National Parks and Wildlife Service personnel but changes in flight itineraries, and the availability of equipment prevented surveys from being coincident (Table 4-1). In addition, the little variation in population numbers that was expected to occur in the small time frame between surveys was not expected to limit the ability to accurately compare census methods. Availability of equipment necessitated that the Reconyx survey occurred in early 2013, and did not occur in 2014.

Table 4-1 Timing of population estimation techniques throughout the 2013 and 2014 sampling periods

Method	2013 Sample Dates	2014 Sample Dates
Aerial Survey	15-18/05/13	12-15/05/14
Reconyx Individual ID	21/02/13-13/03/13	No survey
SLR based Individual ID	25/05/13-09/06/13	26/05/14-12/06/14
Time-lapse	5/07/13-14/07/13	28/06-11/07/14

4.2.2. Aerial survey

Annual aerial surveys of brush-tailed rock-wallabies have been conducted in Green Gully by NSW National Parks and Wildlife since 2004 (Ranger Piers Thomas, *pers. comm.*). Details of the aerial survey are provided in Chapter 3. It should be noted however that a technical fault with the hand held data loggers was discovered after the aerial survey in 2015, and is thought to have affected estimates taken since 2012. This fault possibly affected the census by underestimating the numbers of rock-wallabies tallied versus those actually seen at each colony.

4.2.3. Ground-based survey: individual identification via single lens reflex photography

To the extent that it was possible (given rugged terrain), each colony was circumnavigated on foot by an experienced photographer armed with a Single Lens Reflex (SLR) camera (Canon 350D camera with canon EF 70-300mm f4-5.6 IS USM lens). Multiple photographs were taken of each animal encountered (Fig. 4-1), and variable characters along with distinguishing features noted on a standard form (Appendix 1). Following the work of Jarman et al. (1989), variable characters included sex, age, developmental class (e.g. adult or young-at-foot), and build. Distinguishing features often included oddity characters (e.g. spots, scars, tears or nicks in the ears), deformities from old injuries, or ear tags from concurrent studies etc. Animals were additionally located and scrutinised using binoculars (Carl Zeiss 10x40 magnification) and a spotting scope (Nikon 15-35 × 80 magnification). Identity was assigned by confirming at least two natural pelage features (e.g. size/shape of chest blaze, scars, nicks, or other facial markings) or artificial (i.e., human-made) marks (e.g., ear tags or GPS collars were present on a subset of the population due to a concurrent telemetry project) observed in an image, or consecutive series of images, similar to the methods used to identify individual rock-wallabies by Vernes et al. (2011).

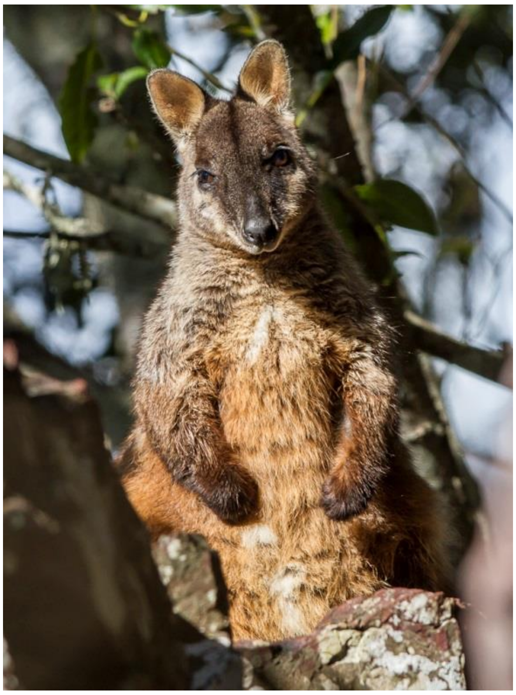


Figure 4-1 Example of ideal animal placement for individual identification using a Single Lens Reflex Camera

Due to variation in terrain, structural complexity and population size between colonies, sampling effort necessarily varied (Table 4-1, Table 4-2). Each colony was surveyed multiple (2–5) times during each 5-day site visit and the time afforded to survey at each colony was recorded. Morning and afternoon observations were made, in order to account for possible differences in individual animals’ or whole colonies’ behaviour (Table 4-1, Table 4-2). Although multiple observers have been utilised for this type of survey at Green Gully over time, for consistency the same observer was utilised for all surveys undertaken during this research

Table 4-1 Total sampling effort at each of the brush-tailed rock-wallaby colonies in 2013. Each block of time represents a separate visit to a colony over the 17 days spent in the field. Obs time refers to the effort afforded (minutes) by the observer during each sampling event. Time Period is the clock time the observer spent on each colony

	E1		**Cave**		**Cliff**		**Old Camera**		**Fig Tree**		**Brumby Pass**	
Date	**Time Period**	**Obs Time (mins)**	**Time Period**	**Obs Time (mins)**	**Time Period**	**Obs Time (mins)**	**Time Period**	**Obs Time (mins)**	**Time Period**	**Obs Time (mins)**	**Time Period**	**Obs Time (mins)**
25/05/2013	1205 – 1250	45	1336 – 1351	15	1412 – 1442	30	1503 – 1519	16	1526 – 1546	20		
26/05/2013							1333 – 1410	37	1026 – 1126	60	1155 – 1235	40
27/05/2013	953 – 1032	39	1055 – 1123	28	1125 – 1157	32	1306 – 1348	42	1511 – 1554	33	1419 – 1433	14
29/05/2013	0929 – 0956	27	1015 – 1034	19	1036 – 1109	33	1132 – 1152	20	1158 – 1252	56		
6/06/2013			1446-1515	29			1330-1415	45	1137-1247	70	1024 – 1102	38
7/06/2013	0908 – 0926	18	0945 – 1000	15	1002 – 1025	23	1055 – 1133	38			1144 – 1204	20
8/06/2013	0842 – 0924	42	0945 – 1006	21	1008 – 1030	22	1444 – 1455	11	1305 – 1444	99	1113 – 1145	32
9/06/2013	1354 – 1418	24	1255 – 1303	8	1305 – 1332	27			1054 – 1151	57		
Total time (mins)		195		135		167		209		395		144

Table 4-2 Total sampling effort at each of the brush-tailed rock-wallaby colonies in 2014. Each block of time represents a separate visit to a colony over the 17 days spent in the field. Obs time refers to the effort afforded (minutes) by the observer during each sampling event. Time Period is the clock time the observer spent on each colony

	E1		**Cave**		**Cliff**		**Old Camera**		**Fig Tree**		**Brumby Pass**	
Date	**Time Period**	**Obs Time (mins)**	**Time Period**	**Obs Time (mins)**	**Time Period**	**Obs Time (mins)**	**Time Period**	**Obs Time (mins)**	**Time Period**	**Obs Time (mins)**	**Time Period**	**Obs Time (mins)**
26/05/2014	1003 – 1030	27	1042 – 1107	25	1110 – 1135	25	1505 – 1510	5	1400 – 1450	50	1250 – 1330	40
27/05/2014	1007 – 1036	29	0848 – 0918	30	0920 – 0945	25	0817 – 0833	16	1414 – 1521	67	1115 – 1206	51
28/05/2014	0813 – 0818	5	1110 – 1133	23	1135 – 1200	25			1026 – 1042	16	0838 – 0906	28
29/05/2014	1403 – 1426	23	1247 – 1312	25	1315 – 1341	26	0815 – 0907	52	0916 – 1038	82	1103 – 1149	46
30/05/2014	0817 – 0842	15	0853 – 0934	31	0938 – 0946	8	1006 – 1048	42	1057 – 1159	62		
10/06/2014			1402 – 1421	19	1423 – 1430	7	1302 – 1352	50			1109 – 1204	55
11/06/2014	0956 – 1031	35	1051 – 1117	26	1119 – 1152	33	0846 – 0939	53	1311 – 1413	62		
12/06/2014	0923 – 1001	38	1421 – 1434	13	1436 – 1515	39	1145 – 1215	30	1228 – 1400	92	1033 – 1132	59
13/06/2014									0827 – 0948	81		
Total time (mins)		172		192		188		248		512		279

4.2.4. Passive infra–red mark–recapture

4.2.4.1. Camera use and placement

Reconyx HC600 Hyperfire camera traps (n_1 = 9, n_2= 10, and n_3 = 10) were installed at three colonies. Cameras were programmed to record ten images per activation, with no delay, when triggered by an animal passing in front of the camera, or heat-in-motion. Because Reconyx cameras permit it, each camera was also programmed to record images every 5 minutes using the time-lapse function (as per 4.2.5 below). Surveys lasted for ten days. Cameras were mounted 0.5 m above ground level on steel posts and directed at brush-tailed rock-wallaby basking sites and foraging areas immediately adjacent to colony refuges. In some cases, minor modification of habitat (e.g. removal of vegetation that would have resulted in false triggers or branches obscuring the field of view) was undertaken to optimise the performance of the camera traps.

4.2.4.2. Identifying individuals from camera trap images

Images were initially grouped into two categories, i.e. time-lapse and heat-in-motion.

Time-lapse triggers produced single images, whereas heat-in-motion triggers produced sequences of images which were subsequently grouped into ‘events.’ For this study, a sequence was defined as the series of images (n = 10) resulting from a single trigger. An event was defined as consecutive sequences less than 5 minutes apart.

Previous research has positively identified individual macropods based on their appearance (Jarman et al. 1989, Johnson 1989). A similar approach was employed to discriminate between individual brush-tailed rock-wallabies. Identity was assigned using the same techniques utilised with the single lens reflex photography (above). Most of the photos suitable for this purpose were those taken front-on, i.e. of the ventral surface, with the animal looking directly at the camera (Figure 4-2). When photographs were considered to be of insufficient quality, e.g. due to inadequate lighting, distance (i.e. too close or too far from the camera) and poor angles (e.g. individuals facing directly away from the camera) they were excluded. The number of individuals was recorded as minimum known alive (MKA) per survey, per colony.

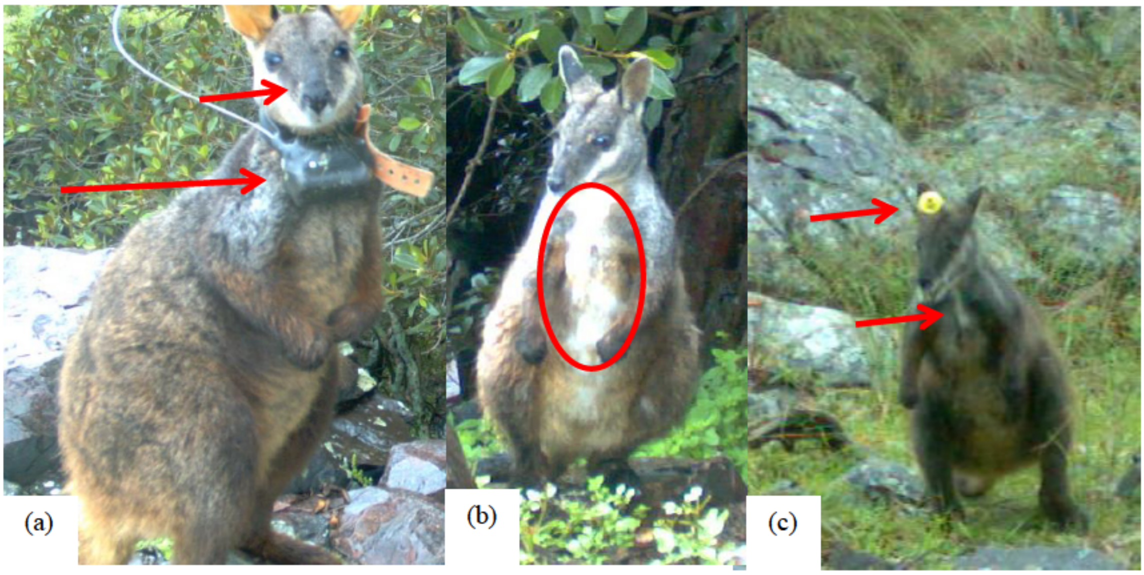


Figure 4-2: Distinguishing features used to identify brush-tailed rock-wallabies: (a) GPS collar along with scars above nose; (b) characteristic white blaze continuing down entire abdomen and (c) small longitudinal chest blaze and ear tag.

4.2.4.3. Population estimation analysis

Two mark-recapture methods (i.e. Schumacher and Schnabel) were used to estimate the number of rock wallabies present at each surveyed colony (Schnabel 1938, Schumacher and Eschmeyer 1943). Both techniques assume a closed population and this was reasonable given the close timing of surveys and spatial separation of colonies. The Schumacher estimate is derived using linear regression, i.e. the number of individuals previously marked is plotted against the proportion of marked individuals in the *t*-th sample (Schumacher and Eschmeyer, 1943).The Schnabel estimate is a weighted average of multiple Petersen estimates (Schnabel 1938, Schumacher and Eschmeyer 1943). Animals were considered to have been ‘marked’ when first recognised as an individual, and classed as ‘recaptured’ when observed on a subsequent survey day.

4.2.5. Time-lapse photography

4.2.5.1. Camera traps: use and placement

Brinno TLC100 time-lapse cameras (n = 18) and Reconyx Hyperfire HC600 (n = 10) were deployed for 10 days at six separate rock-wallaby colonies. Each colony was traversed in order to select sites for camera trap placement. The camera traps were allocated maximise monitoring of the habitat, and any individuals present, without fields-of-view overlapping between cameras. Each camera trap was programmed to record an image every five minutes. To maximise the duration of the surveys, relative to battery life, cameras operated only between 0630 and 1700 h each day. This daily schedule was planned to coincide with the characteristic basking behaviour of brush-tailed rock-wallabies.

4.2.5.2. Image analysis

Each image was reviewed manually and a count of all visible rock-wallabies recorded. Since each image was time-stamped, counts from simultaneous images were used to determine a spatially independent MKA per colony.

4.2.5.3. Probability of detection

The probability of detecting various numbers of rock-wallabies, at each colony, was calculated using PRESENCE4.0 (MacKenzie et al. 2002). Probability of detection, Psi (ψ) was estimated using a single season analysis using the one group, constant population model. Each photo was treated as an independent sampling event. Calculations were performed for morning (pre-1200 h) and afternoon (post-1200 h) periods due to known differences in rock-wallaby behaviour between these periods (Gowen and Vernes 2014).

4.2.6. Calculating cost-benefit of surveys

Costs of each survey type were calculated as the total variable costs associated with each census technique. Variable expenses were those that varied based on the activity being undertaken, and included such items as fuel, camera traps (and associated equipment), and the costs associated with field assistance and data analysis (Lyra-Jorge et al. 2008b). Costs were estimated based upon the requirement of having to purchase all equipment for an initial survey.

4.2.7. Data analysis

A regression analysis of aerial survey MKA against ground survey MKA was performed in order to determine if the data collected could provide managers with a useable index of abundance.

Chi-squared analysis was used to establish if the probability of detecting individuals before and after 1200 h differed significantly.

4.3. Results

4.3.1. Population estimation

Aerial survey consistently returned the lowest estimates of individual brush-tailed rock-wallabies per colony with estimates of just ten individuals in 2013 and 13 individuals in 2014 (Fig. 4-3). In addition, aerial survey was failed to detect individuals at two colonies where presence was detected by on-ground methods. However, due to the technical fault with the data loggers, it is possible that the abundance data collected from the aerial surveys are an artefact of this equipment failure and may be higher than estimated. (Fig. 4-4). SLR based individual identification consistently returned the highest MKA results across the six surveyed colonies (with 45 individuals in both 2013 and 2014). Reconyx based individual identification resulted in a MKA estimate of 33 in 2013, using data collected from only three rock-wallaby colonies.

With each successive visit (in both 2013 and 2014), fewer new individuals were recorded, and MKA reached an asymptote for each colony around day eight (Fig. 4-4).

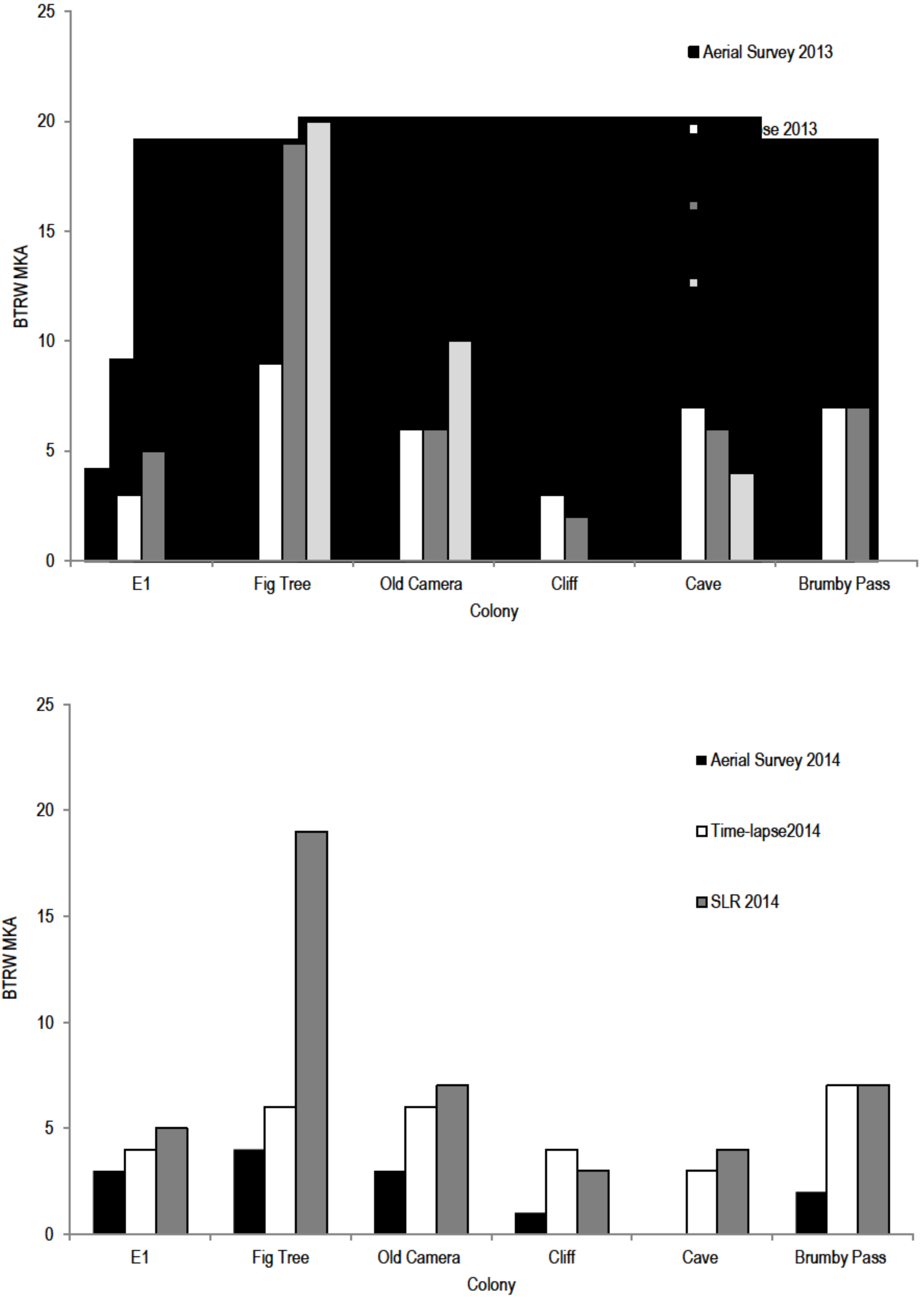


Figure 4-3 Minimum known-to-be-alive estimates from all census methods in 2013 and 2014. Note: Reconyx 2013 only surveyed three colonies, and Cave was not surveyed aerially.

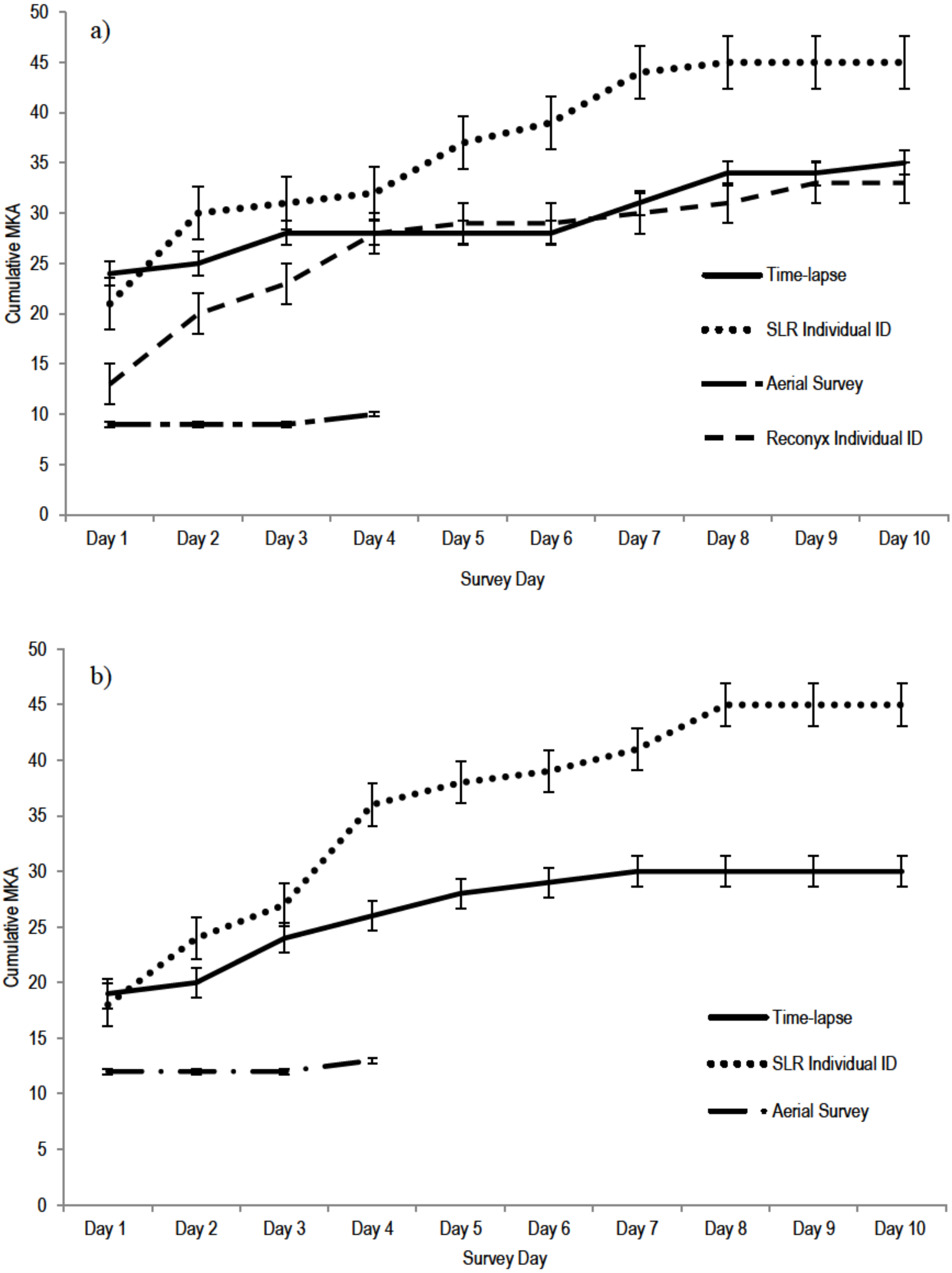


Figure 4-4 Plot of cumulative brush-tailed rock-wallaby MKA against survey day at Green Gully, Oxley Wild Rivers National Park, in a) 2013 and b) 2014.

4.3.2. Use of aerial survey MKA as an index of abundance

Linear regression of MKA (aerial surveys) versus MKA (ground surveys) revealed a non-significant relationship ($R^2 = 0.353$, $P = 0.052$) (Fig. 4-5).

represent

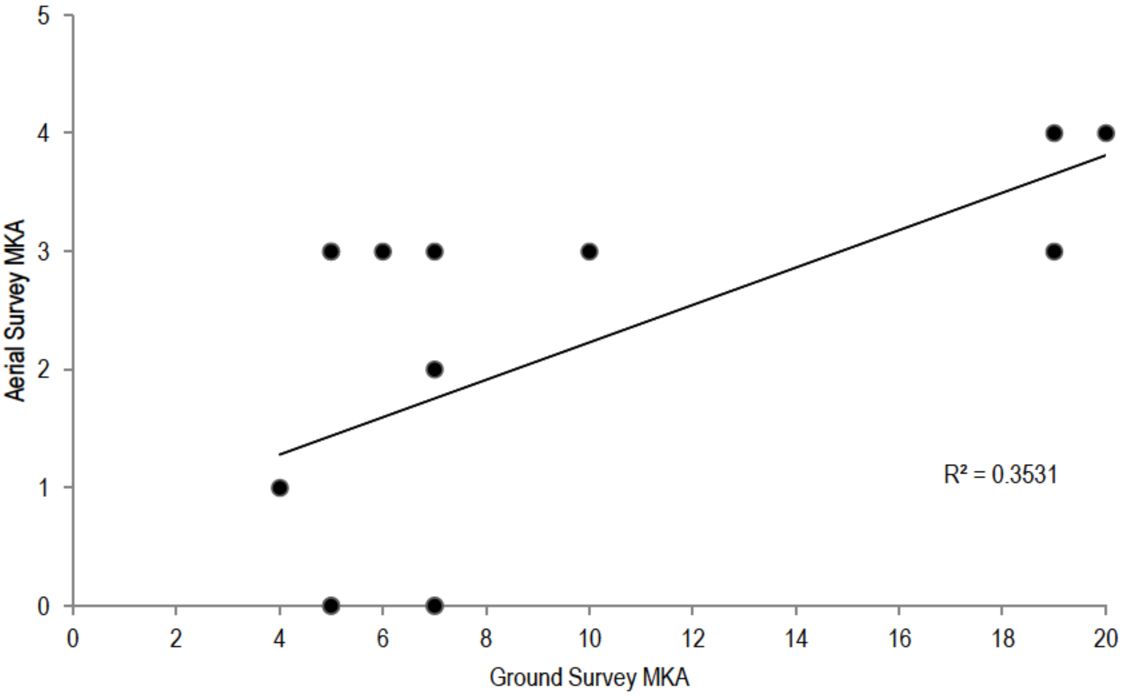


Figure 4-5 Scatterplot of aerial MKA vs ground-based MKA with ordinary least squares regression line, resulting in a R^2 of 0.353. Points represent the MKA of 5 colonies surveyed aerially and with ground-based methods. The maximum MKA from the ground-based census techniques at each colony was allocated for comparison.

4.3.3. Mark–recapture

Comparisons of the mark–recapture estimates showed little variation in the derived population numbers of colonies at Green Gully (Table 4-3). When analysing the total number of individuals identified with the SLR technique, all three ground-based methods returned very similar estimates, returning population numbers between 32 and 35 animals, with potentially 46–48 animals at the upper 95% confidence limit (Table 4-3). Confidence intervals at the Fig Tree colony were wider than the other surveyed colonies, with Brumby Pass colony having the narrowest confidence intervals (Table 4-3). Total population estimates were not calculated from the Reconyx-based mark–recapture as only three colonies were surveyed, compared to six for the other techniques. The raw estimate achieved was a cumulative score, whereas the Schumacher and Schnabel methods returned average population estimates for each sampling event (Table 4-3).

Table 4-3 Population estimates for the six rock-wallaby colonies surveyed at Green Gully during 2013 and 2014

	Raw Estimate	Schumacher	Lower 95%	Upper 95%	Schnabel	Lower 95%	Upper 95%
SLR 2013							
Brumby Pass	**7**	7.0	6.1	8.2	6.9	4.6	10.8
Cave	**4**	4.0	3.5	4.7	4.0	2.4	6.9
E1	**5**	5.3	4.2	7.1	5.3	3.2	9.1
Cliff	**3**	2.9	1.9	5.4	2.7	1.2	6.1
Old Camera	**7**	6.2	4.5	10.0	5.4	3.6	8.5
Fig Tree	**19**	17.2	14.7	20.7	16.8	13.0	23.7
Total	**42**	39.7	35.0	46.0	38.4	32.0	48.1
SLR 2014							
Brumby Pass	**7**	6.2	5.0	8.1	6.1	4.0	9.8
Cave	**6**	5.3	4.2	7.2	5.3	3.4	8.5
E1	**4**	4.6	3.8	5.7	4.5	2.9	8.4
Cliff	**2**	1.9	1.4	2.8	1.8	0.8	4.6
Old Camera	**6**	5.3	4.2	7.1	5.1	3.3	8.2
Fig Tree	**19**	16.8	13.4	22.7	16	12.1	22.4
Total	**45**	38.9	34.0	45.4	38.5	31.5	49.4
Reconyx ID							
Cave	**4**	3.8	3.2	4.7	3.6	2.4	6.5
Fig Tree	**20**	18.3	16.6	20.3	18.1	14.1	25.1
Old Camera	**9**	8.3	7.4	9.6	8.1	5.6	12.1

4.3.3.1. Time-lapse versus motion detection for individual identification

Cameras recorded 2341 images of rock wallabies via the time-lapse function and 303 separate events with the heat-in-motion setting (Table 4-4).

An individual could be positively identified in 42% of the time-lapse images, compared to 38% of the heat-in-motion events (Table 4-4). In the latter, it took an average of 12 consecutive images (or event) to achieve a positive identification (Table 4-5).

Table 4-4 Comparisons between ability to positively identify brush-tailed rock wallabies from time-lapse images versus heat-in-motion events

	Time-lapse	**%**	**Motion (Events)**	**%**
Positive Identification	984	42.0	115	38.0
Unknown	1357	58.0	188	62.1
Total	2341		303	

Table 4-5 Average number of sequences per event and the number of images required to achieve a positive ID of brush-tailed rock-wallabies from heat-in-motion images

Colony	**Average no. sequences per event**	**Average no. images for positive ID**
Old Camera	3.5 ± 1.5	12 ± 1.5
Fig Tree	2.5 ± 0.2	15 ± 2.3
Cave	1.8 ± 0.2	10 ± 0.0
Average	2.6 ± 0.2	12 ± 1.7

4.3.4. Probability of detection

Overall, the probability of detecting brush-tailed rock-wallabies varied substantially between colonies, ranging from 32.3% detection probability at E1, to a 92.5% chance of observing an individual at Old Camera (Table 4-6). Results of a Chi-squared test (*P* = 0.3784) indicated there was no significant difference in the probability of detecting more than one brush-tailed rock-wallaby before or after 1200 h each day.

Table 4-6 Probability of detecting at least one individual brush-tailed rock-wallaby each day of a survey period, broken into pre 1200 h and post 1200 h.

	Probability of Detection (%)	Lower 95%	Upper 95%
Brumby Pass			
≥1 Individuals (Pre 1200 h)	77.5	74.3	80.4
≥1 Individuals (Post 1200 h)	83.5	80.6	86.0
Fig Tree			
≥1 Individuals (Pre 1200 h)	72.4	69.0	75.5
≥1 Individuals (Post 1200 h)	58.3	55.0	61.9
Old Camera			
≥1 Individuals (Pre 1200 h)	92.5	90.3	94.2
≥1 Individuals (Post 1200 h)	81.1	78.1	83.8
Cave			
≥1 Individuals (Pre 1200 h)	32.6	29.3	36.2
≥1 Individuals (Post 1200 h)	39.7	36.2	43.4
Cliff			
≥1 Individuals (Pre 1200 h)	40.6	37.0	44.2
≥1 Individuals (Post 1200 h)	50.9	47.1	54.8
E1			
≥1 Individuals (Pre 1200 h)	32.3	28.8	35.9
≥1 Individuals (Post 1200 h)	41.0	37.1	45.1

4.3.5. Cost comparison

The cost of a single survey was greatest for Reconyx mark–recapture at $13 897, however averaged over the expected life (approximately 4 years) of the camera traps, the cost reduces to $1837 per survey.

The total costs of variable expenses for a 4-day aerial survey were AU$7580 (Table 4-7). Initial 10 day surveys utilising the SLR individual identification, or time-lapse techniques cost AU$7163 and AU$12 397 respectively (Table 4-7, 4-8). Expenditures for subsequent aerial surveys remained the same, but replicating ground-based techniques decreased on successive surveys, because capital equipment (cameras) would be expected to be functional for a number of years before requiring repair or replacement (Table 4-7, 4-8).

Table 4-7 Comparison of the estimated costs of different items (variable expenses) for Aerial survey and SLR Individual Recognition considering an initial survey and any subsequent survey. Cost of researchers and data analysis based on average wage for NPWS and University of New England, and based on a 7.5 h working day

	Variable Expenses (AUS$)	**Cost of Initial Survey**				**Cost of Subsequent Surveys**		**Ecological Benefit**
	Item	Unit Value	Quantity	Total $	Total (hours)	Total ($)	Total (hours)	Data Type
Aerial Survey	Hire of helicopter (inc pilot)	$1200 per hour	~1½ hours each day (x4)	7200	8	7200		Species incidence
	NPWS observer allowance (x3)	$40 per hour	~1½ hours each day (x4)	180	8	180		MKA
	NPWS data analysis	$40 per hour	5 h	200	5	200		Landscape scale population data
Total				**7580**	**21**	**7580**		
SLR Individual Recognition	SLR Camera + Lens	~$2300	1	2300	N/A			Species incidence
	Eneloop rechargeable batteries	$6	8	48	N/A			MKA
	Researcher's allowance	$42 per hour	10 days	3780	90	3780	90	Colony demographics (e.g. sex ratios, evidence of breeding)
	Diesel	$1.50/L	1 × 250 km return trip (10 return field trips @ ~10 km)	90	N/A	90		Total enumeration
	Data analysis	$42 per hour	3 days	945	22.5	945	22.5	Site-specific colony data
Total				**7163**	**112.5**	**4815**	**112.5**	

Table 4-8 Comparison of the estimated costs of different items (variable expenses) for time-lapse and Reconyx mark-recapture considering an initial survey and any subsequent survey. Cost of researchers and data analysis based on average wage for NPWS and University of New England, and based on a 7.5 h working day

	Variable Expenses (AUS$)	**Cost of Initial Survey**				**Cost of Subsequent Surveys**		**Ecological Benefit**
	Item	Unit Value	Quantity	Total $	Total (hours)	Total ($)	Total (hours)	Data Type
Time-lapse	Brinno TLC100 camera traps	$170	15	2550	N/A			Species incidence
	Reconyx HC600 camera traps	$800	10	8000	N/A			MKA
	Installation equipment (star pickets, brackets etc)	$10	10	100	N/A			Site specific colony data
	Diesel	$1.50/L	2 × 250km return trip	180	N/A	180		
	Researcher's allowance	$42	2 days	630	18	720	18	
	Research assistant data analysis	$25 per hour	5 days	937.5	37.5	937.50	37.5	
Total				**12397.5**	**55.5**	**1837.5**	**55.5**	
Reconxy Mark-Recapture	Reconyx HC600 camera traps	$800	15	12000	N/A			Species incidence
	Installation equipment (star pickets, brackets etc)	$10	15	150	N/A			MKA
	Diesel	$1.50/L	2 × 250km return trip	180	N/A	180		Colony demographics (e.g. Sex ratios, evidence of breeding)
	Researcher's allowance	$42	2 days	630	18	720	18	Total enumeration
	Research assistant data analysis	$25	5 days	937.5	37.5	937.5	37.5	Site specific colony data
Total				**13897.5**	**58**	**1837.5**	**55.5**	

4.4. Discussion

Aerial survey has the capacity to provide a rapid assessment of populations at a broad, even landscape-wide, scale but its suitability for enumerating populations of brush-tailed rock-wallabies is dubious (Vernes et al. 2011). In this study, ground-survey techniques consistently returned higher population estimates than aerial surveys, with the passive infra-red camera trap surveys estimating the greatest population counts per colony. A regression of aerial survey against ground surveys suggested it was not even a good index of population size. Cost-effectiveness of ground-based surveys was also consistently better than aerial surveys; they provide a cheaper alternative and greater return per unit cost over time.

The relatively low estimates of brush-tailed rock wallabies derived via aerial survey may be related to dense vegetative cover in Green Gully, along with a lack of movement response to the sound of an approaching aircraft (see Chapter 3). Although the apparent advantages of aerial survey relative to spatial coverage and access (Ransom 2012) are appealing, my data suggest that the monitored aerial surveys undertaken at Green Gully could not reliably detect brush-tailed rock-wallaby presence, nor provide an index of brush-tailed rock wallaby abundance. However, as previously noted, the technical problems associated with the collection of the aerial survey data necessitate further investigation, specifically, additional comparison now that the problems have been rectified. This will ensure that the use of the aerial survey data is not dismissed misguidedly.

4.4.1. Successes among the population estimation techniques

Although aerial survey was found to perform relatively poorly, other methods demonstrated promise for future monitoring efforts. I found that 1) reliable individual identification of brush-tailed rock-wallabies from photographs was possible, and 2) photographic mark-recapture can be a useful tool for estimating brush-tailed rock-wallaby population size. At two of the three colonies surveyed using infra-red cameras, MKA estimates were higher than those achieved by the SLR based individual identification. However, it is possible that these estimates were higher as the Reconyx mark-recapture technique may be open to greater error. Further research is required to test the suitability of this technique on colonies where total population size is known. It is possible that the disturbance created by the presence of the observer on the colonies resulted in individuals actively avoiding detection.

Although one would ideally assess the methods on populations of known size, this was not possible in the wild. Consequently, the use of the Schumacher and Schnabel approaches was used to ascertain the degree to which the MKA estimate approximated actual counts. Close approximation between the three estimates suggested that the majority of the animals present at the colonies had been observed and individually identified in most cases. Notably, this may not have been the case for the Fig Tree colony, raising some questions about the applicability of the techniques at large, structurally complex colonies. An interesting point to note was the large range in population sizes of all brush-tailed rock-wallaby colonies surveyed. To date, no work has been conducted on the ability or carrying capacity of a boulder pile, rock face, or other suitable rock-wallaby habitat and why some sites (particularly in the case of Green Gully) can harbour many individuals.

At Fig Tree, both models returned estimates with wide confidence limits (from the SLR 2013 and 2014 surveys), likely due to the continued sightings of new individuals during the latter part of each survey period. This is possibly due to the structurally complex nature of the site, along with a high vegetation density, and more work is required to determine how vegetation density affects the probability of detection. Relatively tighter confidence limits were returned based on the data collected from the Reconyx survey, suggesting that this MKA estimate is closer to the true population when compared with the SLR estimates. Although it is possible that some individuals remained undetected at the conclusion of the survey periods, on the basis of personal observations of the colony over time I believe that the majority of animals were detected. Compared to the other ground-based estimation techniques, time-lapse population estimation returned the lowest estimates of colony size; its use might be considered if only presence data is required.

Each of the methods I used had advantages and disadvantages. An advantage of mark–recapture and time-lapse population estimation over individual identification is that a researcher is not required to be on-site for the duration of the study, and instead is only required during camera deployment and data retrieval (along with battery changes, depending on the duration of the study). However, issues with the time interval between triggers and sub-optimal camera positioning can occur and may account for the fact that more individuals were identified by SLR photography at one of the colonies.

The use of passive infra-red camera traps for mark-recapture made identification of individuals difficult in some situations. Unlike other studies that yield photographs of both flanks of an animal (Heilbrun et al. 2006, Guthlin et al. 2014), identification of brush-tailed rock-wallabies is greatly assisted by a clear photograph of an individual's face and ventral surface. This increases the opportunity to identify both sex and unique pelage characteristics, such as chest blazes and ear marks. My finding, that an average of 12 images was required to positively identify a brush-tailed rock-wallaby, reflects the suggestion by Hohnen et al. (2013), that maximising the number of photographs taken upon each trigger of the camera significantly increases the amount of information available to the observer.

Utilising the heat-in-motion capability of the camera traps permitted multiple images of an individual to be successively recorded, with the continually changing angles ultimately assisting with identification. In addition, the use of multiple images from the Reconyx cameras enabled easier identification from time-lapse images, as a record of various images from an individual had already been collected. Therefore, the heat-in-motion identification informed the time-lapse individual identification and it was not possible for a single observer to do both independently. Using the two techniques together, may be useful for consecutive surveys that are conducted within a short period of time. If, for instance, heat-in-motion is utilised for an initial survey, the data collected may allow time-lapse photography to be used for subsequent censuses.

Generally, if an animal was more than five metres from the camera, identification became difficult, especially at night. The identification of brush-tailed rock-wallabies was not possible in 60% of both time-lapse images and heat-in-motion activations. For the time-lapse images, poor angles and distance from the camera were the main contributing factor for the inability to distinguish animals. For the heat-in-motion events, individuals were indistinguishable mainly due to insufficient contrast to enable the definitive identification of markings at night, or due to blurred images of animals in motion. According to Rovero et al. (2013), day-time blurred photos occur as a result of a long shutter speed, usually rendering those unusable. Some of these issues could be resolved by improving the characteristics of the camera-traps. Further, some limitations could be addressed by increasing the time that rock-wallabies spend in front of cameras, and hence the number of images available to scrutinise for distinctive features. Similarly, maximising the number of photographs taken per trigger (e.g. 10, as per this study) can provide relatively more opportunities for a diagnostic image to be recorded.

The introduction of lure or attractants arranged in front a camera can be useful when bait preferences of the target species are known and the study aims to maximise detections while minimising study duration (Bengsen et al. 2011). In this case, utilising a bait station to encourage individuals to face towards the camera end expose their ventral side may be advantageous in the identification process. However, the use of an attractant or more images per trigger will both also result in an increased volume of images, thereby possibly having a significant negative effect on the cost-effectiveness of the method.

Irrespective of the quality of the photographs collected, observer error can be a serious problem if only a few photographs of an individual are available (Hohnen et al. 2013).The rock-wallaby colonies in this study have been monitored intensively by camera traps for a number of years, and no camera avoidance behaviour was noted. This may also have been due to the use of infra-red camera traps, rather than other models, such as ‘white-flash’ units, where camera shyness over time has been documented (Lyra-Jorge et al. 2008a).

A small lag period existed between the timing of the aerial survey and the implementation of the ground-based techniques (aerial surveys occurred in May, and camera surveys occurred from late May to June). However, Joblin (1984) states that variation in colony size over short (1 -2 month) timeframes is low, with only the male juveniles dispersing after 6 months out of the pouch. Due to this, there is little reason to believe that the true colony size should have changed significantly between the period covered by the aerial and ground surveys. In addition, an external radio-tracking experiment found that brush-tailed rock-wallabies exhibited little movement to neighbouring colonies, giving rise to the treatment of each colony as a closed system. This is supported by work conducted by Piggott et al.

(2006b) who found that genetic relationships between female brush-tailed rock-wallabies at neighbouring colonies were low, and within colony dispersal was significantly male biased.

4.4.2. Cost effectiveness

Both ground-based remote camera methods were found to be relatively cost-effective sampling methods for brush-tailed rock-wallabies. Despite the high cost of camera trapping initially, the ability to reuse the equipment for subsequent surveys and other projects significantly decreases future expenditure (Lyra-Jorge et al. 2008b). SLR based individual identification requires the presence of a trained observer on-site at all times during the survey period. Logistically, the use of the camera traps was easier to employ than the SLR technique, and permitted sampling to be conducted on larger spatial and temporal scales. No loss of equipment (from theft or vandalism) was experienced in this study however if this is identified as a potential issue, the cost-effectiveness of these methods may be reduced.

In terms of initial and recurrent costs (and ignoring Consumer Price Index), recurrent aerial survey will remain constant (AU$7580), as the major costs associated with this method involve the hire of the aircraft along with the operator. In addition, this method of census is largely weather dependent, with delays, postponements and cancellations all contributing to the overall costs of the aerial survey method (Piers Thomas, *pers comm.*). Comparing overall survey costs, the results of this study suggest that time-lapse population estimation and Reconyx individual identification provide the most viable economic approach upon purchase of all the required equipment (AU$12 397 and AU$13 897 respectively), with the cost efficiency during subsequent surveys increasing significantly. Furthermore, it has been suggested that when surveying rare species more expensive methods (if proven to be more accurate), are more appropriate for long term studies, especially when equipment can be shared and multiple population estimation techniques can be used (Silveira et al. 2003, Gaidet-Drapier et al. 2006, Lyra-Jorge et al. 2008b).

Probabilities of detecting individuals on colonies were similar to that previously documented (Gowen and Vernes 2014), and this highlights the importance of correctly identifying the most appropriate survey time. This has major implications for the cost efficiency for the SLR individual identification method, as identifying when an observer is most likely to detect individuals will enable a reduction in time spent at colonies where probability of detection is low and their time can be allocated accordingly.

The presence of artificial marks (such as GPS collars and ear tags) was used as an additional form of identification when distinguishing between individuals using the Reconyx based individual identification. The physical capture and artificial marking was conducted in a separate telemetry project, and thus the costs associated with the trapping and tagging of these animals was not included here. The ability to quantify the role the GPS collars played in the identification of individuals is difficult, as an observer could not explicitly state that the rock-wallaby may or may not have been identified without the prence of the articical mark. According to (Alonso et al. 2015), the concurrent use of camera traps with capture-resight data is a viable population estimation tool, and should not be overlooked by conservation managers when establishing the most appropriate abundance estimation protocols for their target species.

4.4.3. Implications for brush-tailed rock-wallaby management

Studies attempting to determine the ‘best’ fauna-sampling technique in economic terms are problematic if they do not consider reliability in detecting all individuals of the target species (Garden et al. 2007). In this study, SLR individual recognition, and photographic mark – recapture returned the most accurate population counts of brush-tailed rock-wallaby colonies. Aerial survey proved to be a poor substitute, as the data did not reliably reflect the state of the populations of rock-wallabies on the ground. However, aerial survey at Green Gully, in the immediate sense, should be primarily used as a presence/absence measure and track colony persistence, due to the existence of a 10 years' worth of data. In addition, in order to accurately track abundance and trends, a rigorous ground based monitoring program aimed at a subset of the broader population should be implemented. In doing so, population trends can be examined, and vital demographic information collected such as breeding success and dispersal, which is useful for active management.

Clearly, the questions being asked in any survey are of key importance. Therefore, it is recommended that managers are clear about the purpose of any survey or monitoring program and assess the relative merits of each technique before committing to any particular approach or combination of methodologies. The question, or stated focus of a management initiative is of vital importance, as identifying if abundance or trend in a population is required will impact on the projects overall cost effectiveness. For example, if presence only data is required, the use of time-lapse photography could provide a cost-effective method to achieve this. However, estimates of absolute density may not always be required, and many surveys are aimed at observing population trends over time. In this case, it is not as vital that every individual is identified, and aerial survey does provide a rapid assessment tool that can cover broad areas of management, and could also be utilised to locate new colonies within the gorge system. In addition, if the question managers pose is related to population expansion or collapse, then the intimate knowledge collected from a small number of colonies within a metapopulation may not provide an answer. Additional ground-based methods, such as scat surveys, have also been documented as providing a cost-effective and accurate alternative to estimate presence and habitat usage in rock-wallabies (Telfer et al. 2006), however cannot cover as greater area as that of an aircraft.

If estimates of absolute density are required, the deployment of camera traps at target colonies offers a helpful means of obtaining baseline population demographic information, data vital to assessing the long-term effectiveness of conservation and management measures (Jackson et al. 2006). The identification of individuals from their pelage patterns and the ongoing cataloguing of all images accrued over time would provide information on the duration of their ‘residency’ within colonies. Additionally, ground-based techniques provide demographic data that permit assessment of population viability and investigation of aspects of animal ecology and behaviour that may prove vital in determining appropriate conservation approaches. Advances in remote camera technology will see advancements in camera sensors, shorter intervals between detection and trigger, improved image quality, and greater storage capacity through improvements in microchip technology (McCallum 2013). These advances have the potential to alter the cost-effectiveness of surveys, and this must be taken into account when planning for any long-term replicated projects. Developments in automatic animal biometrics will also assist with animal individual identification, and will significantly reduce the time required to analyse data (Falzon 2014).

Real time uploading of camera data to a remote computer via a satellite link could also have profound conservation management applications (McCallum 2013). In addition, the use of the ground-based techniques in concert with other estimates of absolute abundance not attempted here (e.g., DNA-based methods) may also provide information vital to the conservation and ongoing management of brush-tailed rock-wallabies.

5. Understorey and midstorey changes in grassy open – forest following removal of livestock grazing and frequent winter – spring burning: Implications for the survival of the brush-tailed rock-wallaby

5.1. Introduction

The effects of natural and anthropogenic disturbances on species composition and equilibrium conditions in temperate woodlands and grasslands have received much attention in community ecology (Gibson and Kirkpatrick 1989, McIntyre and Lavorel 1994, Tremont 1994, McIntyre et al. 1995, Pettit et al. 1995, Clarke 2003, Anderson et al. 2007, Prober et al. 2013). In the grassy woodlands of south-eastern Australia, grazing by domestic livestock has been a major factor influencing species composition and structure of the herbaceous layer (Lodge and Whalley 1989, Dorrough et al. 2004). Prior to European settlement, herbaceous biomass removal occurred primarily through wildfire, Aboriginal burning, and grazing by native marsupials (McIntyre et al. 1995). The population of these native herbivores may have been limited due to hunting by Aboriginal people, and predation by dingoes (Dorrough et al. 2004) and marsupial carnivores, as well as by intrinsic population regulation through delayed reproduction (Poole 1975). It is likely that the ground layer in grassy woodlands was dominated primarily by *Themeda australis*, *Sorghum leiocladum*, *Poa sieberiana*, *Cymbopogon refractus*, and *Austrostipa bigeniculata* over most of south-eastern Australia in the early 1800s (Moore 1953, Lodge and Whalley 1989). Since the 1830s, however, the introduction of domestic livestock and the sustained, and at times heavy, ungulate grazing, in conjunction with altered fire regimes, is generally considered to be the cause of the decline or local extinction of most native herbaceous plant species in temperate grasslands, grassy open forests and woodlands and the cause of major structural and compositional change in these formations throughout temperate Australia (Moore 1953, Moore 1970, McIntyre and Lavorel 1994, Tremont 1994, Gill and Bradstock 1995, Clarke 2003, Dorrough et al. 2004). In addition, changing fire regimes in the grassy woodlands of south-eastern Australia are also important in threatening plant populations and causing local extinctions (Gill and Bradstock 1995), with high frequency fire listed as a Key Threatening Process under the *Threatened Species Conservation* Act (1995).

Ungulate grazing influences patterns of plant species richness, evenness and coexistence via changes in species persistence, available biomass, seed production, competition and probability of dispersal and establishment (McIntyre and Lavorel 1994, Tremont 1994, Gough and Grace 1998, Olff and Ritchie 1998) to a greater degree than native fauna are likely to achieve (Fensham and Skull 1999). While ungulate grazing can increase the likelihood of local extinction through reduction in survival and seed production, particularly of tall palatable species (Hodgkinson 1992), grazing can also increase gap formation, favour establishment of grazing-tolerant species (O'Connor 1991) and increase plot scale species richness (Tremont 1994, Foster and Gross 1998), but has been shown to decrease richness at a landscape scale (Schultz et al. 2014). In addition, sustained heavy ungulate grazing can cause an increase in the encroachment of woody plants (Briggs et al. 2002), leading to reduced productivity of the herbaceous understorey (McPherson and Wright 1990, Lett and Knapp 2003, Price and Morgan 2008). In northern New South Wales, ungulate grazing can shift pastures dominated by *Themeda triandra, Sorghum leiocladum, Poa sieberiana* and *Cymbopogon refractus* to an alternative native state, dominated by less palatable or grazing-tolerant species such as *Aristida personata* or grazing-tolerant swards of *Sporobolus creber, Bothriochloa macra* and *Eragrostis leptostachya* (Lodge and Whalley 1989).

Frequent fire has moulded the evolution of the modern Australian flora over geological time (Bowman et al. 2003, Bradstock 2010). The intensity, frequency and season of fire affect the survival and reproduction of plants and changes in the fire regime can, like ungulate grazing, cause changes in dominance, population declines and extinctions (Gill and Bradstock 1995, Morrison and Renwick 2000, Watson 2001, Keith et al. 2002, Morrison 2002, Bradstock 2010). Throughout much of Australia, fire has historically been used as a tool to promote plant communities beneficial to primary producers and the pastoral industry (Hodgkinson et al. 1984, Lodge and Whalley 1989, Tasker and Bradstock 2006). In particular, in north-eastern NSW grassy woodlands and open forests, frequent low-intensity burning of the herbaceous understorey has been utilised to promote 'green pick' for livestock in the winter–spring feed drought, often on a 1–5 year interval (Tasker and Bradstock 2006).

In the last 30 years, the large areas of grassy woodland and open forest on the Northern Tablelands and in the adjacent gorge country in northern NSW that were once grazed commercially by cattle and sheep and managed with winter-spring burning (Lodge and Whalley 1989) have been incorporated into the national reserve network. Associated with this change in tenure has been a change in management. Protected area management in NSW requires outcomes that maintain or improve biodiversity (DoE 2014). The management practices most likely to produce this outcome are frequently debated. Ungulate grazing has been proposed as a management tool to reduce fine fuel loads and thus reduce the chances of unplanned fire (Williams et al. 2006). However, frequent ungulate grazing can promote exotic herbaceous species at the expense of native species (Hobbs 2001). In addition, fire can increase localised species density (i.e. species richness at plot scale) and thus has been promoted as a way to increase biodiversity (Fox and Fox 1986, Christensen and Abbott 1989). Still, species density is only one aspect of diversity and reliance on it as the sole criterion for decision making may lead to inappropriate management decisions and conservation recommendations (Hunter 2005, Schultz et al. 2014).

The primary aim of conservation management for any target species should be to improve overall persistence and maintain a sustainable carrying capacity of that species within the landscape (Norris 2004). The inclusion of Green Gully into the protected area network was driven by its wilderness and high conservation values, along with the prevalence of multiple populations of the brush-tailed rock-wallaby. Coincident with National Park management was the removal of ungulates from the Park and the cessation of annual, low-intensity burning (Piers Thomas, *pers. comm*) Here we present the results of a landscape-level study documenting rapid changes in the understorey and midstorey of the woodland and open forest at Green Gully over seven years. The aims of this study were 1) to assess the effect of the cessation of frequent winter–spring burning and removal of cattle grazing on floristic composition and understorey vegetation structure, and 2) consider the implications of the vegetative change on the food resources of the brush-tailed rock-wallaby, which is a major focus of the conservation management of the Green Gully section of the Park.

5.2. Methods

5.2.1. Floristic survey, ground cover and logs

Twenty-six 20 × 20 m permanent quadrats were established in Green Gully in June 2005 just after the removal of ungulate grazing and cessation of winter–spring burning (Morris 2005). Twelve quadrats were located on riparian flats along Green Gully Creek (elevation: 300m), and the remaining 14 were situated south of the creek at 100-m altitudinal intervals on paired north and south-facing slopes to a maximum elevation of 988 m. A full floristic survey of vascular plants was conducted in each quadrat. A count of all woody plants ≥1.5

m in height was made throughout the quadrat and identified by species. The number of individual woody plants ≤1.5 m in height was counted in a 5 × 5-m plot in the north-west corner of each quadrat and also identified to species. The point-intercept method was used to assess ground cover along each side of the 20 × 20-m quadrat at 0.5-m intervals along the boundary. Ground cover was recorded as herbaceous (identified to species and pooled into native and introduced categories), bare earth, dung, litter, rock, tree or lichen. The number and diameter of fallen logs with a diameter ≥20 mm were measured around the quadrat perimeter in four 10 m transects. Each vascular plant species was classified as perennial or annual; introduced or native, and forb, fern, vine, grass, tree or shrub. The same floristic and habitat measurements were repeated in all quadrats in September 2012. Most species were identified in the field. All unidentified species were bagged and refrigerated until identified in the laboratory. Nomenclature follows PlantNET (2012).

5.2.2. Univariate analysis

To ascertain the extent of change in individual floristic and structural variables between 2005 and 2012, differences in cover variables and functional group richness between years were analysed using paired t-tests, the null hypothesis being that there was no significant difference in cover or species density between sampling times. Individual species change between 2005 and 2012 was also analysed using paired t-tests.

Pattern diversity (species turnover) between 2005 and 2012 was estimated using a method described by Hunter (2005), in which Coleman curves (species accumulation curves) were calculated after 100 randomizations of both datasets using EstimateS (Colewell 2013). Species turnover was then estimated by calculating the slope of the Coleman curves generated from the production of log-log plots from each dataset (Hunter 2005).

5.2.3. Multivariate analysis

To determine whether linear or unimodal ordination was more appropriate to assess the incidence data collected, an initial detrended correspondence analysis (DCA) (using the program CANOCO 4.5) yielded a gradient length of much less than two standard deviations. A linear ordination technique was therefore deemed appropriate for the incidence data collected (all data collected was considered in the analysis) (ter Braak and Smilauer 2002). Principal Component Analysis (PCA; ter Braak and Smilauer 2002) was used to compare selected species' response to the removal of livestock grazing, and frequent fire between 2005 and 2012. Selected species were chosen based on their known response to grazing and conservation value within a pasture. Explanatory cover variables used in the analysis were: dung, trees, native and introduced herbaceous cover, bare earth, litter, lichen and rock. Additional explanatory variables plotted on the ordination were number of logs, summed area of logs, aspect, landscape position and elevation. Year of sampling was treated as a covariable.

5.3. Results

5.3.1. Variation in ground cover and species richness

The main ground cover components of the vegetation changed markedly between 2005 and 2012 (Table 5-1). Native ground cover increased significantly between the two sampling periods from 38% to 69%, while litter and bare earth declined from 46% to 26% and 11% to 1%, respectively. Changes in the minor ground cover components were not statistically significant.

The increase in herbaceous cover was mirrored in significant declines in species density of herbaceous life forms as well as overall numbers of native and perennial (mainly herbaceous) species per quadrat. Statistically significant declines between 2005 and 2012 were recorded in native and perennial species densities, as well as in the species densities of grasses, ferns and forbs (Table 5-1). Variables describing the woody floristics and structure were generally unchanged, except for the number of woody species <1.5 m in height, which increased significantly between 2005 and 2012.

Selected species response also varied significantly between 2005 and 2012 (Table 5-1). Statistically significant increases were recorded in the presence of *Themeda triandra,* and *Sorghum leiocladum* across the 26 sites. Statistically significant declines in the presence of *Bothrocloa macra* and *Sporobolus creber* were recorded between 2005 and 2012.

Table 5-1 Summary of results of analyses of mean (± 1.0 SE) cover and mean (± 1.0 SE) density per quadrat of a range of variables in 2005 and 2012 in Oxley Wild Rivers National Park.

	2005 average (mean ± 1SE)	2012 average (mean ± 1 SE)	*P*
Cover %			
Native herbaceous cover	38.3 ± 4.8	68.7 ± 3.5	<0.001
Introduced herbaceous cover	5.2 ± 3.5	6.1 ± 2.7	0.84
Bare earth	10.7 ± 2.3	1.1 ± 0.3	<0.001
Dung	0.2 ± 0.1	0.1 ± 0.03	0.19
Litter	46.3 ± 3.7	25.1 ± 3.4	<0.001
Rock	2.1 ± 0.5	0.8 ± 0.3	0.07
Tree	0 ± 0	0.3 ± 0.1	0.08
Lichen	0 ± 0	0.1 ± 0.1	0.32
Number of logs	2.1 ± 0.2	1.7 ± 0.2	0.30
Sum area of logs	2.7 ± 0.3	2.9 ± 0.06	0.68
Average number of species (per functional group) per quadrat/plot			
Trees (per quadrat)	3.5 ± 0.3	3.9 ± 0.3	0.41
Grasses (per quadrat)	9.0 ± 0.5	6.4 ± 0.3	<0.001

Vines (per quadrat)	0.2 ± 0.1	0.5 ± 0.2	0.26
Ferns (per quadrat)	1.5 ± 0.13	0.8 ± 0.1	0.004
Forbs (per quadrat)	22.4 ±1.1	16.4 ± 1.2	0.001
Shrubs (per quadrat)	4.6 ± 0.32	4.8 ± 0.4	0.66
Woody species <1.5m (per plot)	2.5 ± 0.71	10.7 ± 2.7	0.006
Woody species >1.5m (per quadrat)	13.6 ± 3.2	22.7 ± 4.6	0.11
Average number of species per quadrat			
Native	37.2 ± 1.7	29.7 ± 2.0	0.007
Introduced	4.3 ± 0.6	3.7 ± 0.5	0.48
Annual	3.1 ± 0.3	3 ± 0.2	0.79
Perennial	28.8 ± 1.2	20.6 ± 1.1	<0.001
Analysis of presence/absence data (average presence of species over 26 quadrats)			
Aristida jerichoensis	0.27 ± 0.08	0.15 ± 0.07	0.317
Eragrostis leptostachya	0.38 ± 0.09	0.34 ± 0.09	0.778
Bothriochloa macra	0.5 ± 0.10	0.15 ± 0.07	0.007
Sporobolus creber	0.46 ± 0.09	0.03 ± 0.03	<0.001
Aristida personata	0.50 ± 0.10	0.30 ± 0.09	0.163
Eragrostis elongata	0.26 ± 0.08	0.07 ± 0.05	0.069
Sorghum leiocladum	0.07 ± 0.05	0.30 ± 0.09	0.039
Aristida vagans	0.76 ± 0.08	0.42 ± 0.09	0.014
Cymbopogon refractus	0.73 ± 0.08	0.45 ± 0.07	0.519
Themeda triandra	0.54 ± 0.09	0.81 ± 0.07	0.039

Table 5-2 Slopes calculated from Coleman Curves (of Total, Native and Introduced species), based on 100 permutations from 2005 and 2012 in Oxley Wild Rivers National Park.

	2005	2012
Total Species	0.493	0.522
Native Species	0.479	0.509
Introduced Species	0.578	0.518

Changes in species turnover varied between 2005 and 2012 (Table 5-2). For the total number of species recorded between the two sampling events, the slope changed from 0.483 in 2005 to 0.552 in 2012, indicating a slight increase in species turnover (beta diversity) between the sites in 2012. Similarly, slopes of the accumulation curves also increased for total number of native species recorded between sites in 2012 (0.479 in 2005 and 0.509 in 2012), which is indicative of a greater turnover of these species. Conversely, the slopes of the accumulation curves for total introduced species declined from 0.578 in 2005 to 0.518 in 2012, indicating a reduction in beta diversity between the sampling periods (Table 5-2).

5.3.2. Response of selected species to reduced grazing and burning

Unconstrained ordination showed that the first two axes accounted for 15% of the variance in the data set (Fig. 5-1). The model demonstrated that relationships within the various cover class attributes existed. Native cover fell on the opposite side of the ordination space to the cover of litter and rock and also the number of logs, indicating that these functional groups increase at the expense of native cover. Grazing-sensitive species such as *Themeda triandra, Cymbopogon refractus* and *Sorghum leiocladum* aligned in ordination space with year, indicating an increase in these species between 2005 and 2012. Grazing-tolerant grasses such as *Eragrostis leptostachya, Sporobolus elongata* and *Bothriochloa macra* and unpalatableincreaser species such as *Aristida jerichoensis* fell on the opposite side of ordination space, indicating a decline in the abundance of these species between 2005 and 2012 (Figure 5-1).

Figure 5-1 Principal Components Analysis of selected species (squares: A. jer = *Aristida jerichoensis*, A. per = *Aristida personata*, A. vag = *Aristida vagans*, B. mac = *Bothriochloa macra*, C. ref = *Cymbopogon refractus*, E. elo = *Eragrostis elongata*, E. lep = *Eragrostis leptostachya*, S. cre = *Sporobolus creber*, S. lei = *Sorghum leiocladum*, T. tri = *Themeda triandra*), cover class environmental attributes (arrows: Int. cover = Introduced cover) and year of sampling (diamond = 2012).

5.4. Discussion

Considerable changes in plant composition occurred at Green Gully between 2005 and 2012 associated with the removal of livestock grazing and the cessation of frequent winter-spring burning. The herbaceous understorey shifted from a grazing-tolerant, native perennial grass assemblage dominated by species commonly encountered in commercially managed native pastures (Lodge and Whalley 1989) to a grazing-sensitive, high-conservation-value perennial grass sward typical of the pre-European situation (Whalley et al. 1978). In addition, a decline in the turnover of introduced species occurred between 2005 and 2012. Although overall native species density declined, the overall landscape (gamma diversity) remained the same due to an increase in turnover of native species and the cover of native species increased markedly and grazing-sensitive species such as *Themeda triandra* returned, resulting in significantly reduced bare ground and litter.

It has been suggested that an increase in the abundance and richness of exotic species in temperate Australian vegetation occurs as a result of grazing of domestic livestock (McIntyre and Lavorel 1994, Pettit et al. 1995, Schultz et al. 2011). In the case of Green Gully, and increase in richness of native species occurred followed stock exclusion. The number of midstorey shrubs <1.5 m in height increased significantly over the 7 years, a trend which has been documented repeatedly in dry sclerophyll vegetation ecosystems in Australia after the removal of livestock grazing (Clark 1988, Briggs et al. 2002, Tasker and Bradstock 2006). Removal of grazing and a reduction in fire frequency often cause increased thickening of midstorey woody vegetation. However, a significant increase in woody vegetation >1.5 m in height was not evident, suggesting that the increase in woody vegetation had not yet reached a height and density that impinged on the amount of available herbage. It is unclear; however, if this current ‘thickening’ is a transient episode and whether with longer periods of unburnt and ungrazed conditions, the system will eventually thin (Croft 2015). Repeated surveys in order to track these changes are therefore required.

Changes in the dominant ground layer life-forms were also evident at Green Gully between 2005 and 2012. Forbs accounted for the majority of species in each quadrat, and thus made the largest contribution to vegetation diversity. Similarly, in other temperate Australian grassy communities, forbs have been reported as an important component of overall diversity (McIntyre et al. 1993). Changes in the cover and frequency of the herbaceous understorey at Green Gully demonstrated directional change toward competitive dominance by fewer species after livestock grazing and frequent fire ceased. Gibson and Kirkpatrick (1989) and Lunt et al. (2007) suggested that long-term stock exclusion results in an increase in the biomass of grazing-sensitive grassy species, which can lead to a reduction in plant species density owing to competitive exclusion. The increase in the abundance of grazing-sensitive, perennial species such as *Themeda triandra,* and *Sorghum leiocladum* and the declines in grazing-tolerant species such as *Bothriochloa macra* and *Sporobolus creber* indicate that the herbaceous understorey is reacting favourably to the low disturbance conditions and shifting to a grass sward of high conservation value.

Species density is often relied upon as a single measure of the conservation value and the appropriateness of management practices (Meir et al. 2004). An attempt to maintain high species density may lead to homogenization and decreased diversity at landscape scale as there are fundamental interrelationships between the various components (alpha, beta, gamma and pattern diversity) of species richness (Hunter 2005, Fleishman et al. 2006, Schultz et al. 2014). At Green Gully, gamma diversity (total number of species recorded in quadrats) did not change significantly between 2005 and 2012. However, the removal of grazing led to an increase in turnover in both native and introduced species, lending support to recent findings in grazing research (Schultz et al. 2014). Therefore, when setting conservation goals and managing for conservation objectives, various measures of ecological integrity (e.g. species composition, species functions, and the ability to sustain ecological processes) as well as species richness at different scales need to be monitored in order to assume high levels of ecological function and biodiversity in the long term (Burgman et al. 2001, Davis and Slobodkin 2004). In the case of Green Gully, the transition from a grazing-tolerant grassland and understorey to a high-conservation-value, grazing sensitive, perennial assemblage highlights the importance of composition rather than high species density as an indicator of management efficacy. In addition, the return of understorey to a pre-European state is valuable, due to the extensive modification of grasslands and pastures throughout most of south-eastern Australia.

This study documented floristic and incipient structural changes in the vegetation of Green Gully, Oxley Wild Rivers National Park, over a 7-year period, with the removal of livestock grazing and cessation of frequent low-intensity burning. The impact of these changes on endangered populations of wildlife in Green Gully, such as brush-tailed rock-wallabies, remains to be seen. As a result, the finding that the herbaceous layer at Green Gully is shifting to a greater cover of pre-existing, high-conservation value, grazing-sensitive native pasture (with consequently less litter and bare earth) suggests that the conservation management of Green Gully since the cessation of commercial grazing and its incorporation in Oxley Wild Rivers National Park is appropriate. The finding that local native turnover has increased in compensation for an associated decrease in richness is an important relationship as it indicates that relying on species density alone can possibly lead to incorrect assumptions about decline. Continued implementation of conservation priorities (such as monitoring the various ecological communities and threatened populations) will assist not only in sustaining native biodiversity but the ecological functioning of the system.

5.4.1. Conservation management implications

Along with its wilderness value, the inclusion of Green Gully in Oxley Wild Rivers National Park was due in large part to the abundance of brush-tailed rock-wallabies (FNPW 2014) that occur as numerous colonies throughout the site (see map of colonies in Chapter 2)**.** Therefore, management of the reserve for the preservation of this species is of vital importance. When grazing, brush-tailed rock-wallabies preferentially select areas based upon the palatability of the forage available rather than the abundance of species that exist within the sward (Short 1989, Carter and Goldizen 2003). The changes observed throughout the sampling period indicate that removing the grazing of domestic livestock and periodic winter-spring burning enables the return of a high-conservation value herbaceous understorey.

This has implications for the survival of the metapoplation of brush-tailed rock-wallabies at Green Gully. For example, increased cover of available herbaceous plant species may allow for shorter foraging periods and smaller home ranges, which may arguably decrease predation risk, although this remains to be seen, and more research is required to quantify this. This approach is also beneficial for the survival of other fire-intolerant species; however any management actions need also take into account appropriate regimes to maintain vegetation that requires high-frequency fire.

Natural senescence of grassy understorey at Green Gully may reduce the attractiveness and nutritional value of the sward to brush-trailed rock-wallabies. The onset of senescence in forage is associated with a decrease in biomass production (Canunayon and Thorogood 2014). In addition, nutrients such as nitrogen and phosphorous mobilise and proteins degrade during senescence until the death of the leaf occurs (Gregersen et al. 2008). This has major implications if managers are seeking methods to maintain the food resources of these animals. The use of fire as a management option has been suggested to assist with the conservation of the brush-tailed rock-wallaby due to the potential to promote food resources within a colony’s home range (Tuft et al. 2012). However, the impacts on the ground-layer vegetation and the implications that frequent burning may have not been thoroughly addressed. Conservation managers are often confronted with a resource allocation problem and are constrained by budgets in selecting the most appropriate approaches for the management of certain species (Bottrill et al. 2009), and therefore targeted burning would have to be integrated within fine management guidelines for the wider landscape to conserve biodiversity more generally. It is essential that the interactions between grazing and fire, along with the resilience to recover, of the ground-layer are quantified if management is to be truly effective.

6. Fire and macropod grazing at Green Gully: Impacts on vegetation

6.1. Introduction

Fire is a key driver in many biomes, including boreal forests, savannas and grassy woodlands (Prober et al. 2007, Allen et al. 2014, Rodriguez and Kouki 2015) but patterns of response to fire can vary enormously between species and communities (Whelan et al. 2002). Within Australia's temperate grassy woodlands, fires can significantly alter forage quality and availability. These effects, in combination with grazing, can dramatically alter resources available to native herbivores (Tuft et al. 2012). Fire is also a popular management tool in south-eastern Australia's grassy woodlands, and, particularly for endangered ecosystems, the long term effects of contemporary vegetation management are of critical concern (Parr and Andersen 2006).

Acting as a non-selective tool for biomass removal (McIntyre et al. 2014), fire is usually followed by a pulse of resources that are exposed to heavy selection and sustained use by herbivores (Raynor et al. 2014). The preferential use of recently burnt vegetation have been documented in a wide variety of herbivores, for example, macropods (Meers and Adams 2003, Styger et al. 2011, Tuft et al. 2012), large African species (Klop et al. 2007, Eby et al. 2014), bison (Raynor et al. 2014, Ranglack and du Toit 2015) and white-tailed deer (Meek et al. 2008). The concentrated grazing pressure that follows a burn can maintain elevated forage quality and quantity (Raynor et al. 2014), due to the formation of patches from a combination of grazing, disturbance and deposition of urine and faeces (Morris et al. 1999, McIvor et al. 2005).

Grazing can also alter the spatial heterogeneity of vegetation, influencing ecosystem processes and biodiversity, however at a species level, response to grazing varies according to growth form and reproductive strategy (Pettit et al. 1995). In Australia's grassy woodlands, grazing can promote high diversity and richness (Tremont 1994). Otherwise increasing biomass of perennial grasses can reduce inter-tussock forbs in ungrazed swards (Tremont and McIntyre 1994). However, the effect of grazing on diversity is related to the overall productivity of the habitat (Kondoh 2001), the spatial distribution of grazing pressure (Sommer 1999) and grazer density (Abrams 2001).

The complex interactions occurring between the grazers and the effects of the fire make it difficult to distinguish between the independent effects of the two disturbances on the ground-layer vegetation (Tremont 1994). Clearly, both fire and grazing can affect ecosystem processes independently but their combined interaction is likely greater than their effects in isolation (Allred et al. 2011).Therefore, understanding the influence of fire and grazing on ecosystems is important for conservation (Kirkpatrick et al. 2011). Post-fire regeneration is usually more attractive to herbivores that the pre-fire vegetation (Bond and Keeley 2005), via the destruction of unpalatable biomass, a general tendency for grasses and herbs to be more prominent soon after fire (Sachro et al. 2005), and relatively greater nutritional quality of resprouting versus old grass shoots (Heckathorn and Delucia 1996). This resource pulse has been noted as driver for a positive correlation between recently burnt areas and increased macropod grazing pressure (Meers and Adams 2003, Styger et al. 2011).

In the gorge country of north east New South Wales, fire has long been used as a tool by livestock producers. Low-to-medium intensity burns were used for decades to produce grass deemed preferable for grazing by cattle, sheep and horses. These same areas remain vital refugia for the brush-tailed rock-wallaby, and it is conceivable that anthropogenic burning may have benefited rock wallabies by providing preferable food resources.

Today, much of this rugged gorge country is managed by the NSW National Parks & Wildlife Service (NPWS). Like public land managers Australia-wide, NPWS staff are faced with competing management imperatives, especially regarding fire. In Green Gully, these difficulties are manifest as pressure to increase burning by some groups and simultaneously reduce it by others, all the while attempting to maintain healthy populations of native species such as the brush-tailed rock-wallaby.

Based on local anecdotes and previous studies elsewhere in South Eastern Australia, I hypothesised that low-to-medium intensity burns would favour short-term grassy regrowth in Green Gully, potentially at the expense of certain grazing sensitive species. In turn, I expected that highly palatable and nutritious new grass would attract preferential grazing by local macropods, including brush-tailed rock wallabies. Although this might seem ideal for the wallabies, it clearly might also add to impacts of fire on local vegetation.

In order to test these hypotheses, and inform local managers about the consequences of this type of burning, I conducted a controlled experiment comparing burnt and unburnt plots. Some plots were fenced to exclude herbivores and others open to them, and these were positioned at varying distances from known brush-tailed rock-wallaby refuges. I aimed to determine consequences of low-intensity burning and grazing – for local ground-layer vegetation (i.e ground cover, species density and evenness) and its role as food resources for brush-tailed rock-wallabies.

6.2. Methods

6.2.1. Site selection

Five brush-tailed rock wallaby colonies within Green Gully were selected as reference locations for this experiment. Although all were representative in terms of species composition and in structure of common types of understorey found beneath the grassy woodlands at Green Gully, they necessarily differed in aspect, elevation and minimum-known-alive (MKA) of brush-tailed rock-wallabies.

6.2.2. Experimental design

I used a nested, repeated-measures design, with pairs of 10 × 10 m vegetation plots at two distances, near (~10 m) and far (~100 m) from five brush-tailed rock wallaby colonies. Within pairs, plots were randomly assigned fire treatments (burnt or unburnt). Access issues prevented the implementation of a ‘far’ treatment at one colony (Fig. 6-1) thus a total of nine pairs of plots were established.

Within each burnt/unburnt plot, a 5 × 5 m subplot was enclosed within a 1.5 m tall fence at a randomly assigned corner of the plot. Two permanent 2 × 2 m quadrats were located within each subplot (fenced/unfenced) and the position marked with wooden or steel pegs driven into the ground so the tops of the pegs remained visible for the duration of the study. Fences were constructed with ring lock wire, with a 30 cm high ring of chicken wire fixed to the top of the fence, along with a ring partially buried around the base.

A HC600 Reconyx camera trap was placed in each of two randomly assigned fenced subplots to measure the effectiveness of the fences at excluding brush-tailed rock-wallabies and other herbivores. Settings used were the same utilised for plot monitoring in Chapter 7.

No formal testing of fence effects was conducted as Tuft et al. (2012) had previously demonstrated a nil response to the presence of similarly constructed exclusion fences. Brush-tailed rock-wallaby and sympatric herbivore monitoring was conducted throughout the duration of the study, in order to ascertain grazing pressure on the treatment plots (Chapter 7).

Under my instruction, NPWS staff burned the treatment (burnt) plots in September 2013, following the construction of fences for grazing exclusion. Burns were of low to moderate intensity, sufficient to remove the majority of the standing vegetation (Fig. 6-2).

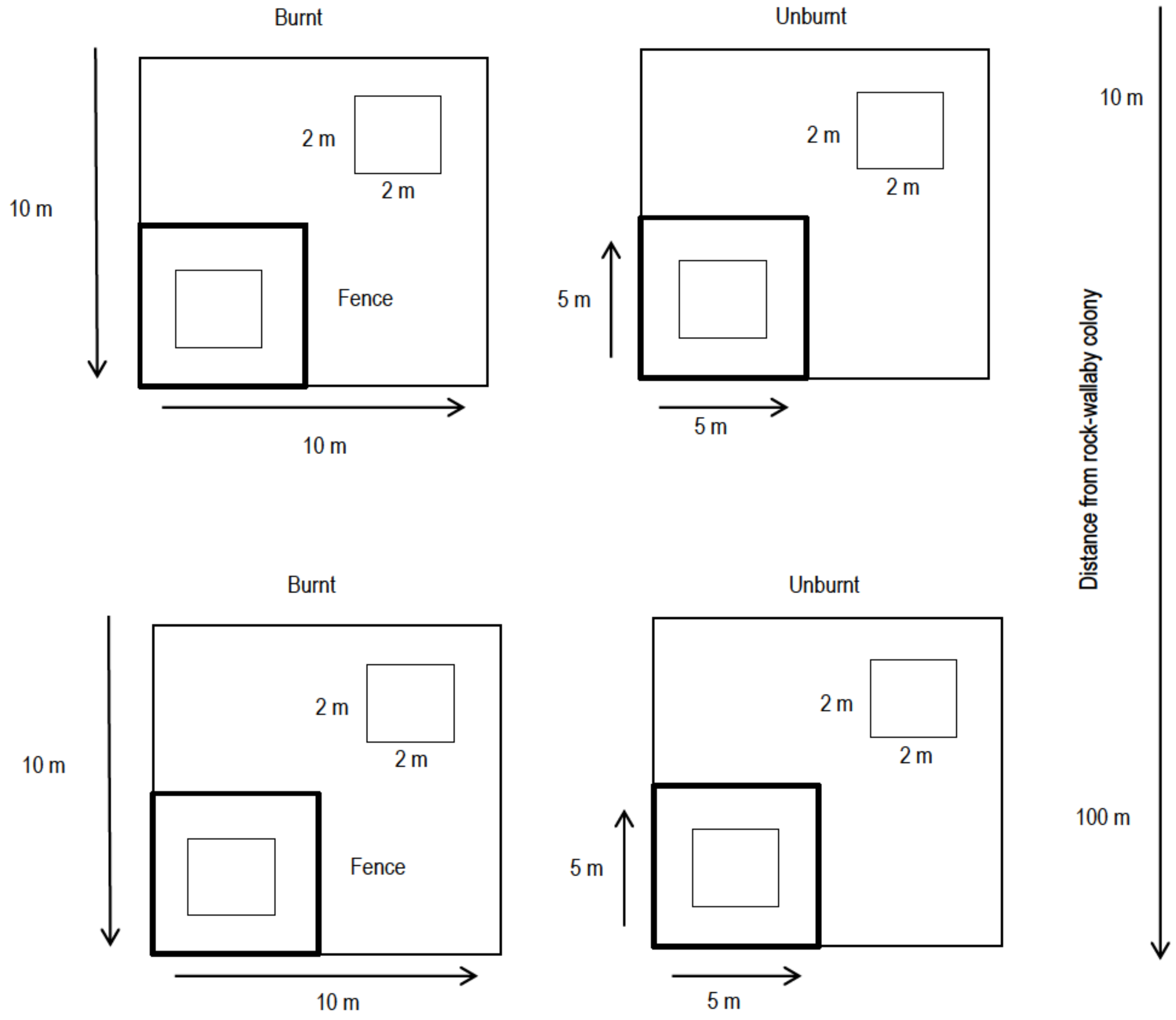


Figure 6-1 Experimental design with fire and grazing treatments represented in two pairs

Figure 6-2 Example of the controlled burn within one of the "Burnt" treatment plots a) during and b) after, at Horse Gate colony, 19th September 2013

6.2.3. Vegetation surveys

Initial vegetation survey data were collected two weeks prior to burning, and then at 3, 6, 9 and 13 months after burning. Fenced and unfenced areas of each plot contained a 2 × 2 m quadrat (n = 36) used for floristic composition assessment. Each 2 × 2 m quadrat was further sub-divided into 16 smaller 0.5 × 0.5 m quadrats. The presence/absence of vascular plant species was assessed in each of the 16 sub-quadrats and combined to derive a score out of 16 per 2 × 2m quadrat. This score reflected the density of species within each plot, rather than total cover or biomass. Where possible, plants were identified to at least genus level and usually to species level. Nomenclature follows PlantNET (2014).

In each 2 × 2m quadrat, the cover/abundance afforded by all species of vascular plant was visually estimated and allocated to one of six classes in a modified Braun-Blanquet scale cover-abundance scale: +, <5% cover and <10 individuals, 1, 5% cover with 10 or more individuals; 2, 5-25% cover; 3, 25-50% cover; 4, 50-75% cover and 5, 75-100% cover (Braun-Blanquet 1932). Cover, as defined for this purpose, was the fraction of the total quadrat area that is obscured by a particular species when viewed from directly above. The conversion of the Braun-Blanquet scale to a midpoint cover range enabled the estimation of total groundcover within each quadrat (Wikum and Shanholtzer 1978).

6.2.4. Species data analysis

6.2.4.1. Univariate analysis

The effects of burning, grazing and time since fire on ground-cover, species evenness and change in species density was analysed using a multivariate linear mixed model (MLMM) approach. For each trait, the model was fitted using asreml (Butler, 2009) under R (R Core Team, 2015), with tests for fixed effects based on Wald F statistics for small samples (Kenward and Roger, 1997).

For the purpose of the experiment the whole plots for the trail are the Colony × Distance combinations, the sub-plots are the Colony × Distance × Trt combinations, and the sampling units are the Colony × Distance × Trt × Fenced combinations. Each sampling unit (plot) was measured over time and these correspond to the repeated measures, which will be correlated within a plot.

The vectors of measurements for each plot (measurements over time), conditionally on the given mean for the plot, were assumed to be from distribution having a common variance – covariance structure.

When describing the effects, the following notation is used:

Colony Corresponds to the five colonies included in the trial;

Distance Near or far from the colony of wallabies (Dist1 and Dist2 resp.)

Trt Burnt or Unburnt respectively

Fenced Factor indicating if an areas was fenced

FTime Factor indicating sampling period (i.e. times 1, 2, …, (4 or 5)).

To model the plot means at each FTime, the initial model fitted was:

Trait = baseline + Colony + Distance + Trt + Distance:Trt + Fenced + FTime
+ FTime:Distance + FTime:Trt + FTime:Distance:Trt + FTime:Fenced
+ **Colony:Distance** + **Colony:Trt** + **Colony:Distance:Trt**
+ **Colony:Distance:Fenced** + **Colony:Trt:Fenced**
+ **FTime:Colony** + **FTime:Colony:Distance** + **FTime:Colony:Trt**
+ **FTime:Colony:Distance:Trt**
+ **error**

The terms in bold italic are fitted as random effects, with the error terms modelled are described above. The initial model was subsequently simplified by removing non-marginal fixed effect terms not significant the P = 0.05 level.

The evenness index E_{var} (Smith and Wilson 1996) was used to calculate species evenness and is based on the variance in abundance over the species. This variance is taken over log abundances, to examine proportional differences. The variance is then converted by 1-2/π arctan() to a 0-1 range, with 0 representing the minimum evenness, and 1 the maximum (Smith and Wilson 1996).

$$E_{var} = 1 - 2/\pi \arctan\left\{ \sum_{s=1}^{S} \left(\ln(x_s) - \sum_{t=1}^{S} \ln(x_t)/S \right)^2 \Big/ S \right\}$$

Differences in species composition between burnt and unburnt plots prior to the implementation of the treatments, necessitated that difference between pre-fire and corresponding post-fire species density was calculated.

Pairwise comparisons using Least Significant Difference (LSD) rankings were used to compare significant interactions and treatments. Predicted means without a letter in common differed by more than twice the standard error of their difference, and hence were significantly different at the *P* = 0.05 level.

6.2.4.2. Multivariate analysis

To assist in interpreting the interactions between grazing, fire and distance from the rock-wallaby colony, all herbaceous species including forbs, browse and grasses were assigned into functional groups based on height and lateral spread, seed and fruit size, specific leaf area (SLA), storage organs and clonality. Principal Component Analysis (PCA; ter Braak and Smilauer 2002) using CANOCO 5 was used to compare the response of functional groups to macropod grazing, fire and distance from the rock-wallaby colony. Explanatory variables plotted on the ordination included in this analysis were Distance, Burnt/Unburnt, and Grazed/Ungrazed.

Visual classification of compositional change sequence over time was also analysed in CANOCO 5. Average species composition was estimated for all treatment plots during each sampling period. PCA was utilised to examine the temporal sequence of the compositional change in the treatment plots at Green Gully. The ordination was fitted using polylines connecting points that represented the same treatment plot.

6.3. Results

6.3.1. Fire, macropod grazing and groundcover, species density and species evenness

The fences were deemed to be effective at excluding herbivores. No images of macropods, including brush-tailed rock-wallabies, were recorded via camera traps within the fenced plots, nor were there other signs (e.g. scats or grazing damage) that incursions had occurred. Excluding unidentified graminoid species, 89 native and introduced species were sampled across all sites. A full list of species recorded throughout the duration of the study is presented in Appendix 2.

Standing ground-cover at Green Gully was significantly influenced by Time and Treatment (Burnt/Unburnt) ($P < 0.01$) (Table 6-1). LSD indicated that the Burnt and Unburnt plots did not have significantly different amounts of ground-cover prior to the implementation of experimental burns (Table 6-2) (Fig 6-3). However, standing ground-cover was significantly less in the burnt plots than in unburnt plots in the 13 months' post fire (Fig 6-3). Colony and time (P = 0.01 and P < 0.01 respectively) had a significant main effect on the average percentage standing ground-cover within the sampling period (Table 6-1).

Table 6-1 Results of multivariate linear mixed modelling for total percentage ground-cover at Green Gully. F-values and significance values for interactions are shown**.** Significant (*P* < 0.05) values are indicated in bold

Ground-cover	***F***	***P***
FTime:Distance:Trt	0.38	0.82
Distance:Trt	0.30	0.60
FTime:Fenced	1.84	0.14
Fenced	2.47	0.15
FTime:Distance	1.95	0.12
Distance	0.14	0.73
Colony	9.50	**0.01**
Trt	0.85	0.37
FTime	23.61	**<0.01**
FTime:Trt	26.43	**<0.01**

Table 6-2 Results of LSD comparing ground-cover in burnt and unburnt plots at Green Gully. Predicted means without a letter in common differ by more than twice the standard error of their difference and hence are significantly different at the *P* = 0.05 level.

FTime	Trt	Predicted Ground-cover	Std Error	LSD Rank
Time 1	Burnt	82.0	4.5	de
Time 1	Unburnt	82.4	4.5	de
Time 2	Burnt	33.1	4.5	a
Time 2	Unburnt	77.9	4.5	d
Time 3	Burnt	51.6	4.5	b
Time 3	Unburnt	82.4	4.5	e
Time 4	Burnt	50.7	4.5	b
Time 4	Unburnt	83.7	4.5	e
Time 5	Burnt	58.8	4.5	c
Time 5	Unburnt	83.3	4.5	de

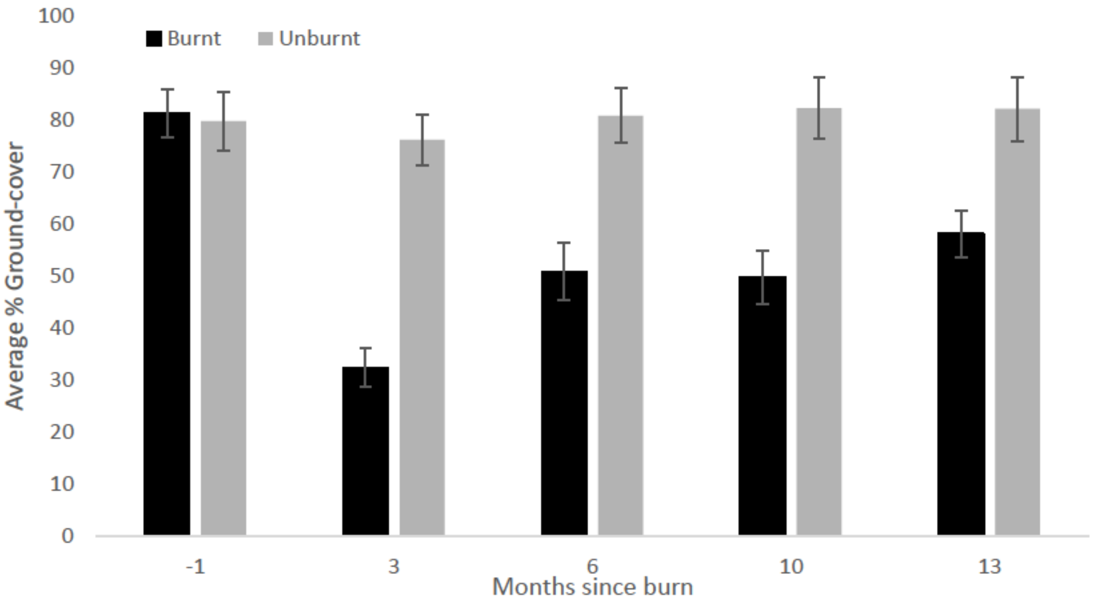


Figure 6-3 Groundcover % (±1 SE) in the burnt and unburnt plots before and after experimental burns at Green Gully, Oxley Wild Rivers National Park

Changes in species density was significantly influenced by Distance ×Trt (*P* = 0.03) (Table 6-3).

Table 6-3 Results of multivariate linear mixed modelling for the change in species density following fire at Green Gully. F-values and significance values for each interaction are shown. Significant ($P < 0.05$) values are indicated in bold.

Change in Species Density	***F***	***P***
FTime:Distance:Trt	1.19	0.33
FTime:Trt	0.73	0.54
Colony	1.38	0.30
FTime:Distance	2.23	0.17
FTime:Fenced	2.25	0.11
Fenced	0.25	0.63
Distance	0.50	0.53
Trt	1.42	0.28
Distance:Trt	1.73	**0.03**
FTime	14.84	**<0.01**

The above results indicate a marginally significant Distance × Trt interaction ($P = 0.03$) (Table 6-3). However, if the means for each Distance × Trt combination, averaged over the other factors, are pairwise compared, no pair of means differed by more than twice the standard error of their difference (Table 6-5). The significant Distance × Trt effect is a result of the larger mean for close unburnt plots compared with the other three combinations (Fig 6-4) (Table 6-5). A significant peak in richness across all plots was observed 6 months after the experimental burns were implemented (Table 6-4) (Fig 6-4).

Table 6-4 Results of LSD comparing the change in richness over time burnt and unburnt plots at Green Gully. Predicted means without a letter in common differ by more than twice the standard error of their difference and hence are significantly different at the $P = 0.05$ level.

FTime	Predicted change in richness	Std Error	LSD Rank
Time 1	-1.7	1.3	a
Time 2	4.3	1.3	c
Time 3	2.1	1.3	b
Time 4	2.0	1.3	b

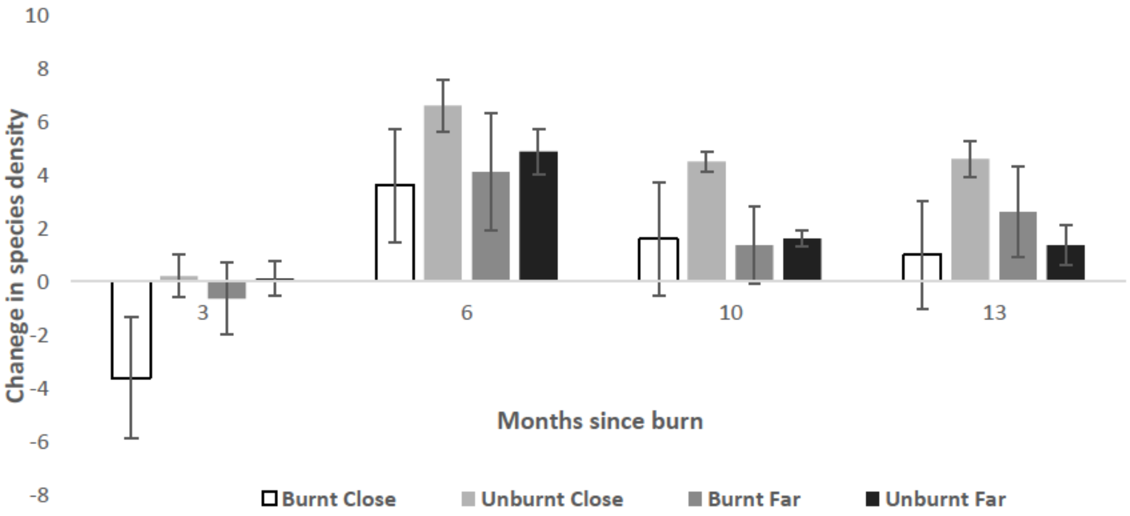


Figure 6-4 Species density change (± 1 SE) in the burnt and unburnt plots following experimental burns at Green Gully, Oxley Wild Rivers National Park

Table 6-5 Results f LSD comparing the change in richness over time burnt and unburnt plots at Green Gully**.** Predicted means without a letter in common differ by more than twice the standard error of their difference and hence are significantly different at the $P = 0.05$ level.

Distance	**Trt**	**Change in richness**	**Std Error**	**LSD Rank**
Close	Burnt	0.6	1.8	a
Close	Unburnt	3.9	1.8	a
Far	Burnt	0.6	1.9	a
Far	Unburnt	1.6	1.9	a

Species evenness at Green Gully was significantly influenced by burning ($P = 0.04$) (Table 6-6). Burnt plots remained significantly more even in comparison to the unburnt plots, regardless of distance (Table 6-7) (Fig 6-5).

Table 6-6 Results of multivariate linear mixed modelling on species evenness following fire at Green Gully. F-values and significance values for each interaction are shown. Significant ($P < 0.05$) values are indicated in bold.

Evenness	***F***	***P***
FTime:Fenced	0.12	0.97
Colony	0.30	0.85
FTime:Distance:Trt	1.03	0.40
FTime:Trt	0.70	0.60
FTime	0.88	0.49
Fenced	1.02	0.34
Distance:Trt	3.18	0.11
Distance	3.18	0.54
Trt	6.15	**0.04**

Table 6-7 Results of LSD comparing species evenness in burnt and unburnt plots at Green Gully. Predicted means without a letter in common differ by more than twice the standard error of their difference and hence are significantly different at the *P* = 0.05 level.

Trt	Evenness	Std Error	LSD Rank
Burnt	0.60	0.014	b
Unburnt	0.55	0.014	a

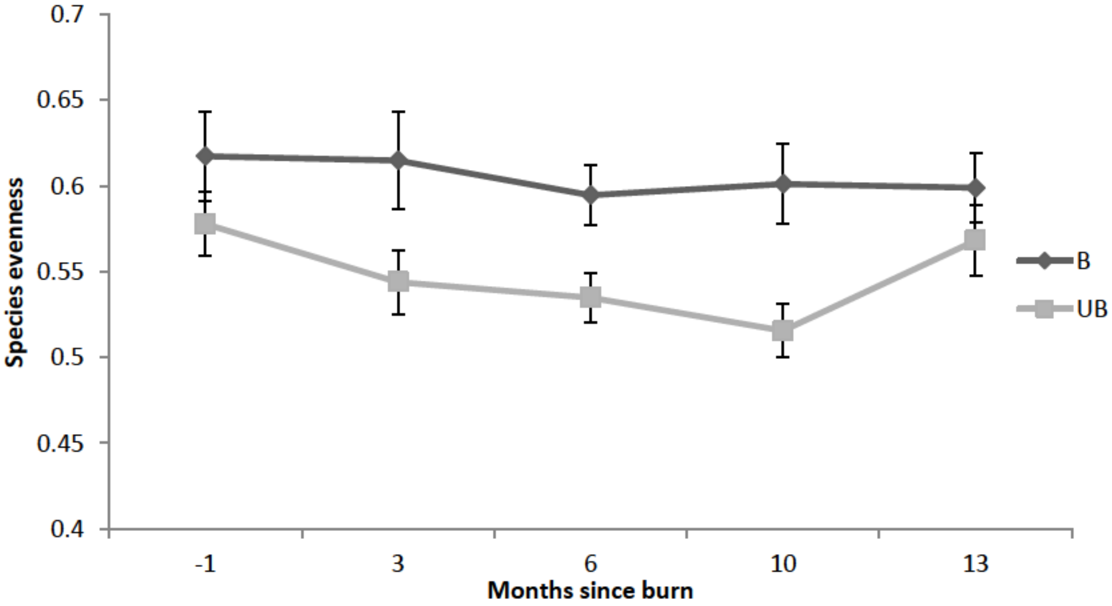


Figure 6-5 Changes in species evenness (±1 SE) following experimental burns at Green Gully, Oxley Wild Rivers National Park

Figure 6-6 Six months post-fire (March 2014) at Fig Tree Colony, Green Gully. a) Burnt*Ungrazed*Far b) Burnt*Grazed*Far c) Unburnt*Ungrazed*Far d) Unburnt*Grazed*Far e) Burnt*Ungrazed*Close f) Burnt*Grazed*Close g) Unburnt*Ungrazed*Close h) Unburnt*Grazed*Close

6.3.2. Fire, macropod grazing and compositional change

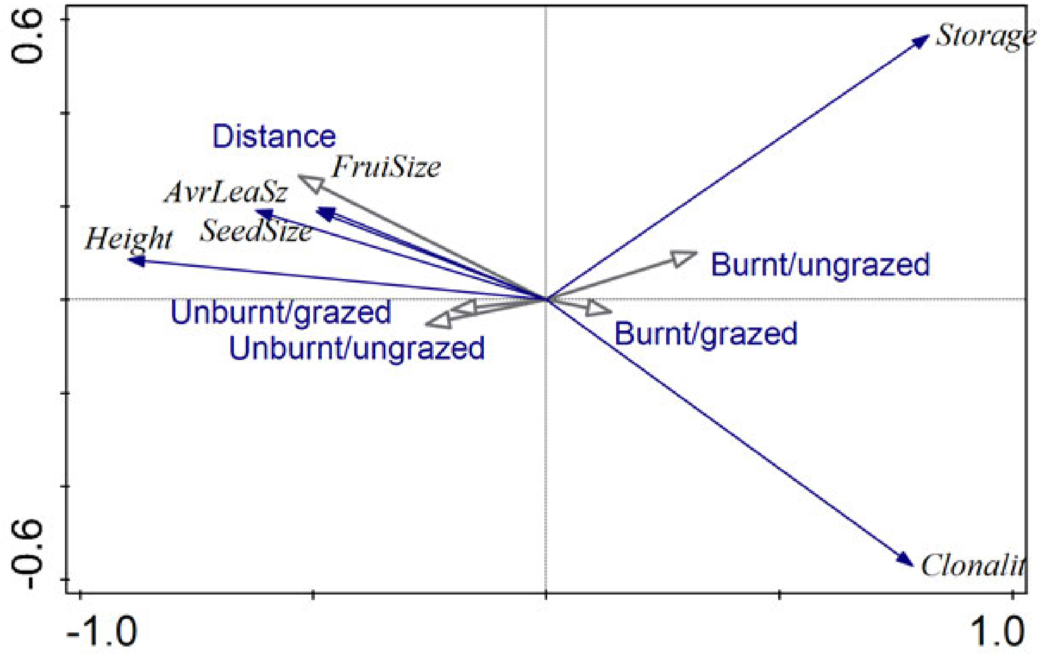


Figure 6-7 Principal Components Analysis of functional traits against management treatments at Green Gully, Oxley Wild Rivers National Park. These axes account for 31.1% of the variance. Grey lines represent explanatory variables, (grazing and burning treatments). Blue lines are the response variables, including height, average leaf, seed and fruit size, clonality and the presence of storage organs.

The first two axis accounted for 31.1% of the total explained variance and 22.2% of the adjusted variance. A positive deviation of tall plants with respect to the species pool was observed in relation to the burning/grazing treatments, and was strongly associated with increasing distance from rock wallaby colonies. Increasing seed size, fruit size and leaf size were positively correlated with increasing distance from the colonies, and lay on the opposite side of the ordination space to burning and grazing, suggesting a negative relationship between these species and treatments (Fig 6-7). Both clonality and presence of storage organs increased in association with burning and macropod grazing. The length of the functional trait arrows suggests a strong directional force, and the proximity of the blue response variable (Storage organs and clonality) with the grey explanatory variables (burning and grazing) indicates a strong correlation (Fig 6-7).

Composition within all plots changed along the same gradient within the 13 months of post burning (Fig 6-8). The greatest peak in compositional change for all plots occurred during the autumn sampling (Time 3) (Fig 6-8). Differences existed in species composition between the burnt and unburnt plots prior to implementing treatments. The amplitude of change in autumn (sample period 3) was as great as the entire change across the 13 months (Fig 6-8).

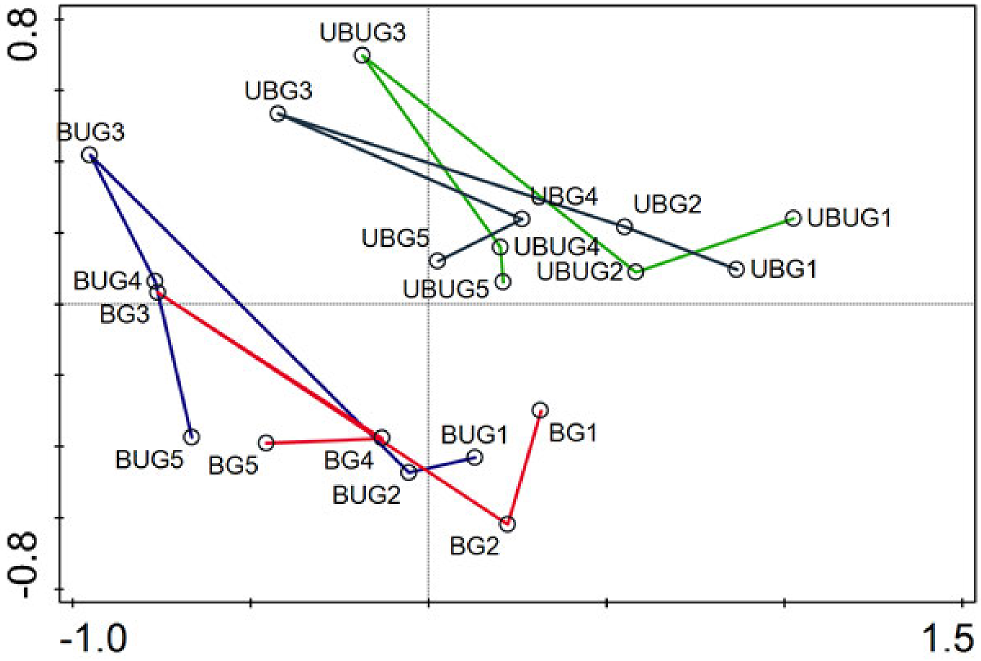


Figure 6-8 Species compositional change over time of burnt and unburnt plots at Green Gully, Oxley Wild Rivers National Park. Time 1 is the initial sample, time 5 is 13 months post burn. B = Burnt, G= Grazed, UB = Unburnt, UG = Ungrazed

6.4. Discussion

6.4.1. Effects of fire and macropod grazing on species density and evenness, and groundcover

Together, macropod grazing and a single low-intensity burn resulted in vegetation in plots that were both burnt and grazed was still significantly different in terms of functional groups and groundcover 13 months’ post-burn. Unsurprisingly, burning had a significant immediate impact on the vegetation surrounding the colonies. According to Archibald et al. (2005), fire acts as an ‘ecological magnet’ for many grazer species, and following the prescribed burns, grazing in conjunction with burning had a greater impact than burning alone. Indeed, the effect of the fire and grazing on standing groundcover was evident 13-month post-burn. Herbaceous cover can act as a surrogate for biomass or productivity (Bowles and Jones 2013), and the changes in the burnt and grazed plots indicates that the observed intensities of burning and grazing could influence the grassy woodland composition at Green Gully.

In contrast to the findings of other south east Australian studies’, exclusion of grazing alone was not associated with a decline in species density (Morgan and Lunt 1999, Schultz et al. 2011). This could be explained by relatively low grazing pressure exerted on unburnt sites; I observed that rock-wallabies generally faced little competition from sympatric macropods (Chapter 7). Further, it is possible that the 13-month time frame of this study was too short to detect changes in species density with grazing exclusion alone. After an initial decline in species density was measured at three months’ post burn, plots other than those both burnt and grazed increased in species density, suggesting that the treatments of burning and grazing negatively affected the establishment of additional species. A number of authors have suggested that the presence of bare ground within the ungrazed plots provides space for seedling germination (Lunt 1990, Noy-Meir 1995), allowing richness to increase in the burnt ungrazed plots. However, the preferential selection of the ‘green pick’ in burnt grazed plots may have limited an increase in richness in these areas, regardless of the burnt patches providing favourable areas for establishment. The lack of a difference in species density within 13 months of burning was unexpected, however the trend has also been documented in semi-arid eastern Australia (Lewis et al. 2010).

When selecting sites for the establishment of plots, difficulties were encountered locating homogeneous areas for replication of treatments. However, analysis did not suggest that this factor caused a significant difference in the results achieved. Brush-tailed rock-wallabies are central-place foragers, that appear to exploit available resources closest to their refuge first before depleting patches further away, following optimal foraging rules and the associated travel and predation risks (Schoener 1979, Stephens and Krebs 1986). Like Argentine ants (Holway and Case 2000), beavers (Raffel et al. 2009) and the European rabbit (Bakker et al. 2005), central-place foragers can have profound effects on the habitat and ground-layer vegetation surrounding refugia (Bakker et al. 2005).

Brush-tailed rock-wallabies foraging behaviour could also account for the differences in species density observed between the near and far plots prior to the implementation of the burning. Species density within all close plots was generally higher before burning, and the formation of these patches might be attributed to historical macropod grazing prior to the experiment. Patches can form due to a combination of grazing, disturbances such as trampling, and the deposition of urine and faeces by animals (Morris et al. 1999, McIvor et al. 2005), and these areas might have been relatively species rich and more heterogeneous (through nutrient enrichment and runoff from the colonies, overgrazing and patches of regular trampling) compared to those located further away.

The reduction in evenness observed in the unburnt plots in comparison to the burnt plots hints to the severity of the patch selection around the colonies. This effect was most pronounced within the unburnt areas, as evenness decreased due to the thickening of vegetation that wasn't being grazed. A reduction in evenness can be attributed to an increase in dominance of competitive species, reducing their abundances and allowing other species to increase in abundance (Tozer and Bradstock 2003). Evenness declined in the burnt patches, however not to as great an extent as in the unburnt patches, which could be a function of greater grazing pressure on these areas. Overall, the changes to species evenness were small, possibly due to the fact that the over-dominance of individual species (such as thick tussock swards of *Themeda* spp., *Cymbopogogon* spp. or *Aristida* spp.) is not a key defining feature of the majority of these sites, and due to this, even in the absence of grazing and fire the competitive strength of the dominant grassy species are reduced. These grassy systems could be naturally limited by other means such as tree cover, shallow infertile soils on steep slopes, or exposure; although more research is required to validate this.

6.4.2. Effects of fire and macropod grazing on functional groups

While species density did not appreciably change at the conclusion of the monitoring period, changes in composition were highlighted with the promotion of different species in the grazed and burnt sites than those in the ungrazed and unburnt sites. With respect to functional traits, the combined effects of burning and grazing had a significant impact on the species composition of the plots, with clonal and stoloniferous species correlated with the highly disturbed sites. The use of functional traits as an indicator of the effects of fire and especially grazing has been cited as a particularly powerful method to model the response of ecological systems to changes in disturbance regimes (Lavorel et al. 1999).

McIvor et al. (2005) reported a significant relationship between heavy defoliation by grazing with an increase in stoloniferous species, and this trend was also evident in the increase in stoloniferous species being positively associated with burning and grazing at Green Gully. A trend towards clonal growth has also been documented in grazing and burning experiments, with Kahmen and Poschlod (2008) demonstrating that clonal growth was highly correlated with these disturbances in grasslands.

Timing of the treatment may have favoured species with particular nutrient storage traits. Late winter/early spring burns may have favoured the establishment of species with capacity to withdraw and store their nutrients prior to burning, e.g. via bulbs, corms and tubers, as the stored nutrients survive burning and support subsequent establishment (Kahmen and Poschlod 2008). The same mechanism, and possible interaction with selective grazing of non-stoloniferous species, could explain the promotion of species with storage organs in other studies on grassland management (Köhler 2001), even leading to a greater representation in comparison to sexually reproducing plants.

Plant height, seed size and leaf area were all negatively associated with burning and macropod grazing, with the unburnt plots tending to have species which were taller, broader and produced larger seeds and fruits. In addition, distance from the colony was also a major contributing factor to the observed changes in expressed functional traits. Many studies have documented that a major response to grazing and burning is the promotion of rosette and ground-layer species, and smaller leaf size (Moog et al. 2005), due to the vertical defoliation (Dupré and Diekmann 2001, McIvor et al. 2005, Kahmen and Poschlod 2008), so it was unsurprising that a similar result emerged from this study. The tendency for plants to be relatively shorter in the burnt areas could also be a photo-morphogenic response to high light intensity experienced during germination in freshly burnt areas. This has been found to result in shorter internodes and tiller angles closer to the horizontal (Noy-Meir 1995).

6.4.3. Effects of seasonality and rainfall on species composition

Seasonality, incorporating both rainfall and the time of year when it occurred, acted as a major driver of change. Compositional change of treatment and nil-treatment plots followed the same response trajectory, suggesting season had a greater influence on compositional change observed within the plots than the interactive effects of fire and grazing. The peak in compositional change and species density during autumn 2014 followed heavy late summer rains. In addition, a lag in the recovery of the ground-layer vegetation in the three months following the burns was associated with scarce spring rainfall. According to Prober et al. (2013), slow recovery of species density following fire can be directly attributed to low rainfall and additional disturbance, in this case manifest as intense grazing by macropods. Variability in inter-seasonal rainfall was an overwhelming determinant of local scale native density and vegetative cover in this study, similar to studies of grasslands in Africa (Buitenwerf et al. 2011, Prober et al. 2013) and Australia (Schultz et al. 2011).

Hobbs et al. (2007) noted that the dominating effect of rainfall on ground-layer vegetation are temporary in its persistence, and possibly due to the flush of species with below-ground organs that remain dormant during dry years. Therefore, the acknowledgement of the influence of climate variability on compositional change in vegetation swards is vital as this variation is strongly linked to fluctuations in populations (Jongejans et al. 2010, Schultz et al. 2014).

6.4.4. Management recommendations

The resilience of grasslands' is strongly dependent on the capacity of resident species to recover from disturbance as well as the ability of alternative species to restore ecosystem functions (Prober et al. 2007). Much of the vegetation around Green Gully's brush-tailed rock -wallaby colonies fit this model, i.e. disturbed grasslands whose recovery is affected by disturbance (fire) and then by the influence of other species (wallabies). In this study, low intensity fire and macropod grazing had no substantial, enduring effect on vegetation species density. However, after 13 months of heavy macropod grazing in the burnt patches, groundcover within the burnt and grazed plots remained significantly lower than in the ungrazed and unburnt plots. Further, there was evidence of a drastic compositional change, within the burnt and grazed plots close to rock wallaby colonies, to a sward dominated by basal, rosette forming and small SLA plants.

Several factors are important when planning future management of vegetation and rock wallabies, within Green Gully. It seems likely that my burnt patches (at 10m x 10m) were too small to be considered a viable management option. Indeed, if managers limited future burns to this size they would likely overly concentrate macropod grazing (Meers and Adams 2003). Grasslands experience grazing pressure after a burn due to the reduced availability of high quality forage (Kirkpatrick et al. 2011),and therefore smaller patches will have relatively greater grazing pressure. Although this means it should be easier for animals to keep vegetation in a short, developing state of high nutritional value, similar to grazing lawns (Allred et al. 2011) with associated benefits of localised 'fuel-management' (Leonard et al. 2010) the changes in composition and groundcover should not be ignored.

It is important to note that rock-wallabies appeared to preferentially feed on nutritious areas closest to the colonies (within 50 m), and this must be taken into account when planning prescribed burns. Establishing a heterogeneous local environment by maintaining patches in various states of succession is important to provide food-resources for rock-wallabies to remain (Tuft et al. 2012). It is not possible nor feasible to recommend a burn of a certain extent, as the appropriate size will vary, depending on the colonies' habitat size, condition of the understorey and the surrounding landscape features. 'Complete' burns around rock-wallaby colonies should be avoided, as based on the results collected here, available food resources could be limited for as long as three months. Likewise, extensive burns may force wallabies to travel further from shelter for food and water, subjecting them to water conservation issues and an increased risk of predation.

The timing of any planned burns is also a critical consideration. It is not recommended that burns be conducted during spring or autumn at Green Gully, as brush-tailed rock-wallabies demonstrate a peak in feeding activity at all plots during these seasons (Chapter 7). Since summer burns are unlikely to be manageable, winter would be the preferred option. In addition, if drought conditions are predicted, burning should not be undertaken, as it will be important to maintain existing vegetation and retain palatable browse plants until conditions improve. Further, the impacts of seasonality (prevailing weather conditions) should be carefully considered, as the rainfall that occurs within the post-burn recovery period will determine the overall impacts of the burn on the ground-layer vegetation (Lewis et al. 2010).

Lastly, the observed recovery of the standing vegetation within all the burnt plots in the three months' post-burn suggests that plans for supplementary feeding of rock-wallabies should be considered in instances where burns are followed by drought conditions. Food supplementation has been documented as being beneficial (increases in body mass) for numerous species and many populations around the world, and rodents (Banks and Dickman 2000), bandicoots (Broughton and Dickman 1991), bats (Law 1995) and eastern pygmypossums (Tulloch and Dickman 2007) provide some well-studied examples. Indeed, food supplementation may be a vital component of management in the event that not only that seasonal conditions are poor post-fire but also under circumstances where planned burns become too extensive, too intense, or both.

7. Behavioural response of brush-tailed rock-wallabies to prescribed management burns

7.1. Introduction

Anthropogenic burning is an important land management tool world-wide but its impacts on fauna are sometimes poorly understood. Fire, as am ecological disturbance, can cause changes in vegetation, thereby affecting interactions within and between species (Arthur et al. 2012). Fires have the capacity to significantly alter forage quality and availability, and therefore the intensity and spatial distribution of grazing (Meers and Adams 2003, Archibald and Bond 2004, Klop et al. 2007). Consequently, understanding the response of herbivores to fires is essential for their conservation and management (Klop et al. 2007).

Fire scars in grasslands are an ‘ecological magnet’ for many species (Archibald et al. 2005), resulting in heavy selection and sustained use of regrowth in post-burned areas by macropods and other herbivores (Southwell and Jarman 1987, Meers and Adams 2003, Klop et al. 2007, Styger et al. 2011, Tuft et al. 2012, Eby et al. 2014, Raynor et al. 2014). This preference for burned areas has been linked to the higher nutritional quality of the freshly burnt understorey (Sensenig et al. 2010), as essential macronutrients such as phosphorous and nitrogen are greater in the regrowth following fire (Van de Vijver et al. 1999). Higher protein levels in the vegetation following burning (Van de Vijver et al. 1999) has been linked to preferential feeding and for weight gain (Woolfolk et al. 1975).

The possibility that some herbivores are attracted to burnt areas due to an increased level of protection from predation has also been suggested by some authors (Burkepile et al. 2013), as burnt vegetation potentially offers herbivores more opportunity to detect predators (Eby et al. 2014). The risk of predation often results in prey species selecting foraging areas that are perceived to represent a lower predation risk (Creel and Christianson 2008), and for some macropods, increasing distance from a refuge or appropriate cover is often associated with increased predation risk (While and McArthur 2005).

Brush-tailed rock-wallabies are often referred to as refuge dependent, and rely on the resources immediately surrounding their colonies in rocky refuges in order to satisfy their necessary nutritional requirements (Tuft et al. 2011). Bumble bees (Munidasa and Toquenaga 2010), chipmunks (McAleer and Giraldeau 2006), beavers (Raffel et al. 2009), ants (Campos et al. 2014) and human hunter – gatherers (Codding et al. 2011) provide well-studied examples of central place foraging. In addition, continual exploitation of the resources closest to the refuge can result in the modification of the ground-layer vegetation to a more species-rich, diverse and heterogeneous sward (Gálvez-Bravo et al. 2011).

In an attempt to determine the spatial and temporal responses of rock-wallabies to low-intensity burning, I implemented a manipulative experiment in Oxley Wild Rivers National Park. In this chapter, I present the findings from 13 months of continuous monitoring of burnt and unburnt patches adjacent to brush-tailed rock-wallaby colonies, and discuss the potential implications of burning habitat for rock-wallabies along with sympatric native and introduced herbivores. Specifically, I aimed to 1) determine if rock-wallabies preferentially selected burnt patches and, in doing so, examine the degree to which these patches were selected in comparison to unburnt patches; 2) document potential competitive interactions with sympatric macropods, and 3) examine the potential of prescribed fire as a tool for managing the food resources of this endangered rock-wallaby.

7.2. Methods

7.2.1. Colony selection

The behavioural response of rock-wallabies to low-intensity burns was examined by choosing five colonies at Green Gully, which were subject to the fire and macropod grazing manipulative experiment reported in Chapter 6. Population estimates (MKA) were derived for all treatment colonies during work outlined in Chapters 3 and 4.

7.2.2. Camera placement and period of observations

Reconyx HC600 Hyperfire camera traps were installed at the 5 colonies (n_1 = 4, n_2 = 4, n_3 = 4, n_4 = 4, n_5 = 2). Cameras were placed to monitor herbivore activity on the grazed burnt and unburnt plots as outlined in Chapter 6. Two additional cameras were placed at one colony, to monitor the effectiveness of the fences at excluding macropods. Cameras were programmed to record ten images per activation, with no delay, when triggered by heat-in-motion. Cameras were mounted 0.5 m above ground level on steel posts and directed at the vegetation plots. Cameras were offset approximately 1.5 m from the edge of the 10 × 10 m plots in order to detect animals utilising the burnt and unburnt areas (Fig 7-1).

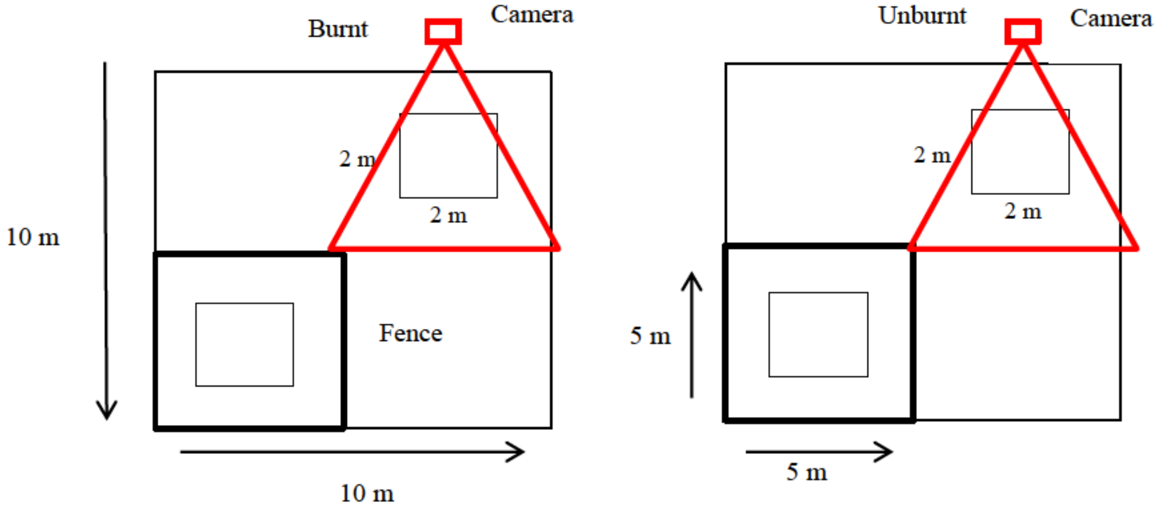


Figure 7-1 Example of camera trap layout – red square indicates location of the camera and red triangle is the field of view.

Camera data were downloaded and batteries changed every two weeks during the summer months and every six weeks during the winter months. Cameras were operational between 7 September 2013 and 26 October 2014.

7.2.3. Processing camera trap data

Each image was reviewed using ExifPro 2.1 software. Metadata tags for species, number of individuals and behaviour were assigned to each sequence of ten images, in addition to the time and date metadata tags automatically added to each image by the cameras. Customised software written by Dr Gerhard Körtner was used to collate metadata tags and export summaries of data for each camera at each colony. Sequences of events were tagged as the first behaviour observed by an individual (Fig 7-2). Behaviours of individuals were classed as 'feed,' 'flight,' 'groom,' 'fight,' 'reproduction,' and 'alert.' Sequences of images were subsequently grouped into 'events.' For this study, a sequence was defined as the images (n

= 10) resulting from a single trigger. For this purpose, an event was defined as consecutive sequences less than five minutes apart.

Figure 7-2 Examples of behaviours exhibited by brush-tailed rock-wallabies: a) groom, b) alert, c) feed, d) flight.

7.2.4. Data analysis

Within each event, time spent feeding was estimated using the number of sequences tagged as ‘feed’ behaviour. Utilising the time stamps for the beginning and end of each sequence, the total amount of time spent feeding (in minutes and seconds) was calculated for each event was summed per treatment and per colony. The average number of rock-wallabies (NRW) per event was also calculated to compare grazing pressure between burnt and unburnt plots, along with distance from the colony. Average time (AvTime) spent feeding was determined by averaging the feeding time of each event across all plots.

time spent grazing. Ordinary least squares regression was used to consider the relationship between colony size and the average number of animals observed per event.

For each trait (AvTime, NRW), a split-plot analysis was performed, with Colony × Distance combinations taken as the whole-plots. Factors included in each analysis were Colony, Distance, Trt (Burnt/Unburnt) and interactions between Colony × Trt and Distance × Trt. Terms not significant at the 0.1 level were dropped from the model. $P \leq 0.1$ was the level chosen for significance due to the small sample size and subsequent small statistical power.

The behaviours from each sequence of images were pooled into three categories, feeding, locomotion (moving through the plots) and interaction (including groom, reproductive and fight behaviours). Chi– squared analyses were conducted to test if differences in rock-wallaby behaviour were associated with 1) burning and 2) distance from colonies.

Plot usage over time was compared by summing the total number of events occurring in each month post – burn and calculating this as the proportion of total events occurring throughout the 13 months of observations. Average rainfall data from the nearest available weather station was collected to ascertain if rock-wallaby activity on the plots was a function of rainfall experienced at the study site.

7.2.5. Activity patterns and overlap

The daily activity patterns of rock-wallabies along with the overlap in activity with sympatric macropods were calculated for burnt and unburnt plots. All broad-scale overlap analysis was conducted using the package “overlap” (Meredith and Ridout 2013) in the statistics package R. For each overlap comparison, the coefficient of overlap was bootstrapped 1000 times to generate 95% confidence limits. To satisfy the sample size requirements of the analysis package, macropod events were pooled due to the paucity of samples. The analysis was repeated using data from winter (1 April – 30 September) and summer (1 October – 31 March) to account for pronounced changes in photoperiod between seasons.

7.3. Results

Camera traps were deployed at five colonies for a total of 8280 trap nights (414 days per camera). Each camera was active for approximately 90% of the total survey period. I recorded 3030 independent events rock-wallaby events and 259 independent events involving sympatric macropods. Known predators of rock-wallabies (e.g. wild dogs, cats, foxes and spotted-tailed quolls) and introduced pests (e.g. feral horses, goats and pigs) were also recorded throughout the study (Fig 7-3), however at such a low rate of occurrence that they were excluded from analysis.

Figure 7-3 Examples of known rock-wallaby predators and pest species recorded on the heat-in-motion cameras at Green Gully.

7.3.1. Behaviour and plot usage

Brush-tailed rock-wallabies generally utilised burnt plots close to the colonies to a greater degree than the burnt plots located 100 m from colonies (Table 7-1). A preference for close plots was also observed when comparing the use of unburnt plots (Table 7-1). More feeding occurred to a greater extent in burnt plots than unburnt plots when distance from colony was not considered (Table 7-1). No feeding events were recorded at the unburnt far plot at Fig Tree colony (Table 7-1).

Table 7-1 Total time spent (hh:mm:ss) by brush-tailed rock-wallabies feeding on experimental plots at Green Gully (Yarrowitch Lower did not have an assigned 'far' treatment)

	Burnt Close	**Burnt Far**	**Unburnt Close**	**Unburnt Far**
Horse Gate	10:38:08	5:36:08	1:19:30	0:25:40
Fig Tree	10:48:02	3:00:16	0:24:52	0
Old Camera	4:50:11	16:21:24	4:10:38	1:23:48
Yarrowitch Pinch	6:23:50	3:01:00	1:59:48	1:25:48
Yarrowitch Lower	0:26:36		0:03:24	

The average time spent feeding per event was similar in all plots, regardless of treatment (Table 7-2). On average, brush-tailed rock-wallabies spent 1-9 minutes per event actively foraging (Table 7-2). However, after removing all non-significant terms (P >0.01), there was a significant difference in the average time spent feeding between colonies (P = 0.08)

Table 7-2 Average time spent actively foraging by brush-tailed rock-wallabies at Green Gully (mm:ss±1SE), on different types of plot

	Burnt Close	**Burnt Far**	**Unburnt Close**	**Unburnt Far**
Yarrowitch Pinch	03:31±00:24	03:29±00:39	04:26±01:02	08:35±06:13
Old Camera	02:52±00:32	04:28±00:39	02:57±00:25	02:37±00:30
Fig Tree	03:09±00:20	03:24±00:33	02:04±01:04	0
Horse Gate	02:59±00:12	02:38±00:18	01:24±00:12	02:24±00:38
Yarrowitch Lower	01:20±00:27		01:08±00:34	

When considering the average number of rock-wallabies per feed event, treatment significantly influenced the number of individuals, with more rock-wallabies feeding on the burnt than the unburnt sites (*P* = 0.07) (Table 7-3). Neither distance (*P* >0.1), nor colony (*P* >0.1) influenced the average number of rock-wallabies per event (Table 7-3).

Table 7-3 Average number of brush-tailed rock-wallabies recorded per event on experimental plots at Green Gully

	Burnt Close	**Burnt Far**	**Unburnt Close**	**Unburnt Far**
Yarrowitch Pinch	1.45±0.06	1.23±0.05	1.33±0.11	1.3±0.21
Old Camera	1.62±0.06	1.25±0.03	1.32±0.05	1.25±0.08
Fig Tree	1.64±0.03	1.11±0.05	1.08±0.08	0
Horse Gate	1.65±0.02	1.09±0.03	1.19±0.05	1.16±0.12
Yarrowitch Lower	1.15±0.08		1.00±00.00	

Number of individuals observed within plots was not significantly correlated with colony MKA. The least squares regression yielded a non-significant (*P* = 0.735), and a poor R^2 value of 0.022 (Fig 7-4).

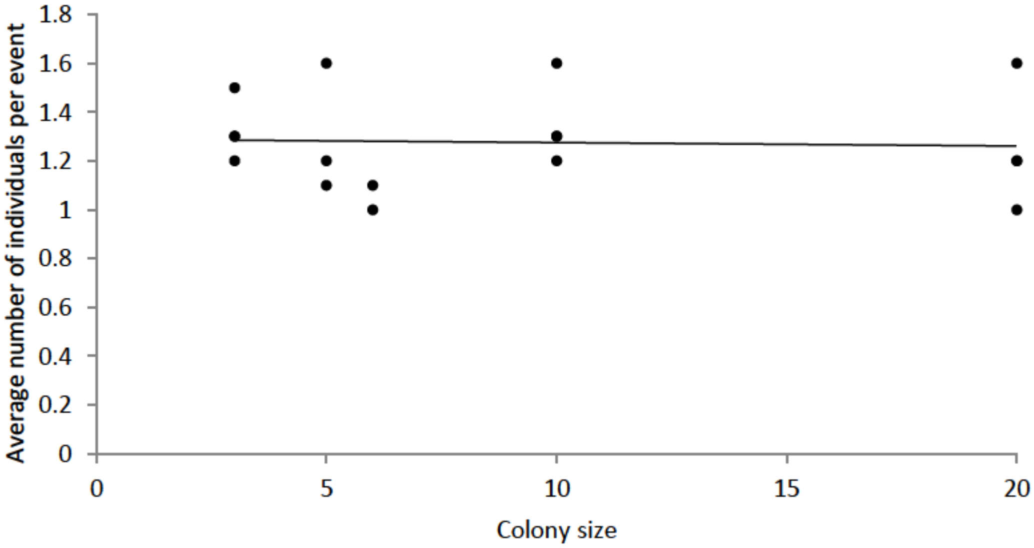


Figure 7-4 Least squares regression of the average number of individuals per event on colony size (R^2 = 0.022)

Results of the Chi-square analysis suggested that foraging behaviours were significantly greater ($P = \leq 0.01$) in burnt than unburnt plots and also between close and far burnt plots ($P = \leq 0.01$) (Fig 7-5). Locomotive behaviour was observed more often than expected in the unburnt versus burnt patches ($P = \leq 0.01$), and interactive behaviours were also significantly greater in burnt plots ($P = \leq 0.01$). These behaviours were also significantly impacted by distance ($P = \leq 0.01$ and $P = \leq 0.01$, respectively; Fig 7-5).

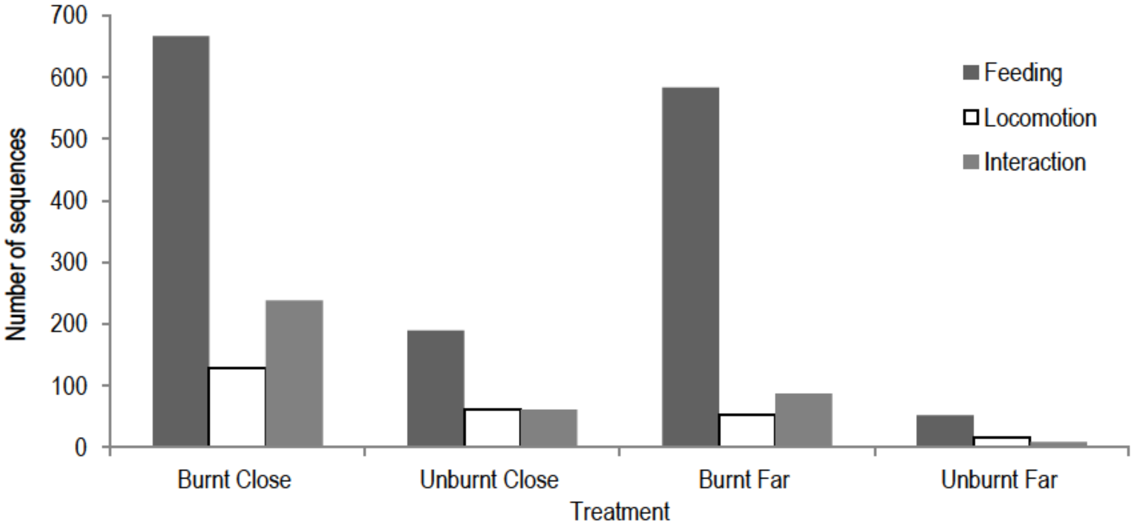


Figure 7-5 Behaviours exhibited by brush-tailed rock-wallabies on treatment plots at Green Gully, represented as average number of sequences (±1SE)

Regression analysis indicated a significant relationship between monthly rainfall (mm) and brush-tailed rock-wallaby activity on the unburnt sites (*P*= <0.01) (Table 7-2). Activity on the burnt sites was significantly correlated with high rainfall when it occurred in the previous month (*P* = <0.01) (Table 7-5). Limited activity occurred in burnt and unburnt plots in September – October immediately following fire (Figs 7-6, 7-7); however, activity in November was pronounced more so on burnt plots (Fig 7-6), than unburnt plots (Fig 7-7). Two additional peaks in brush-tailed rock-wallaby activity occurred in burnt plots over the 13-month study period, in March and September (Fig 7-6). The autumn peak in activity was also pronounced in unburnt plots, but the September peak in activity in burnt plots was not mirrored in unburnt plots, and occurred in August (Fig 7-7).

Table 7-5 Regression analysis between monthly rainfall (mm) and brush-tailed rock-wallaby activity at Green Gully.

Treatment	Relationship between rainfall and activity	Relationship between rainfall and activity with 1-month lag
Burnt	P > 0.05, R^2 0.116	P – <0.01, R^2 0.221
Unburnt	P <0.01, R^2 0.1791	P – >0.1, R^2 0.049

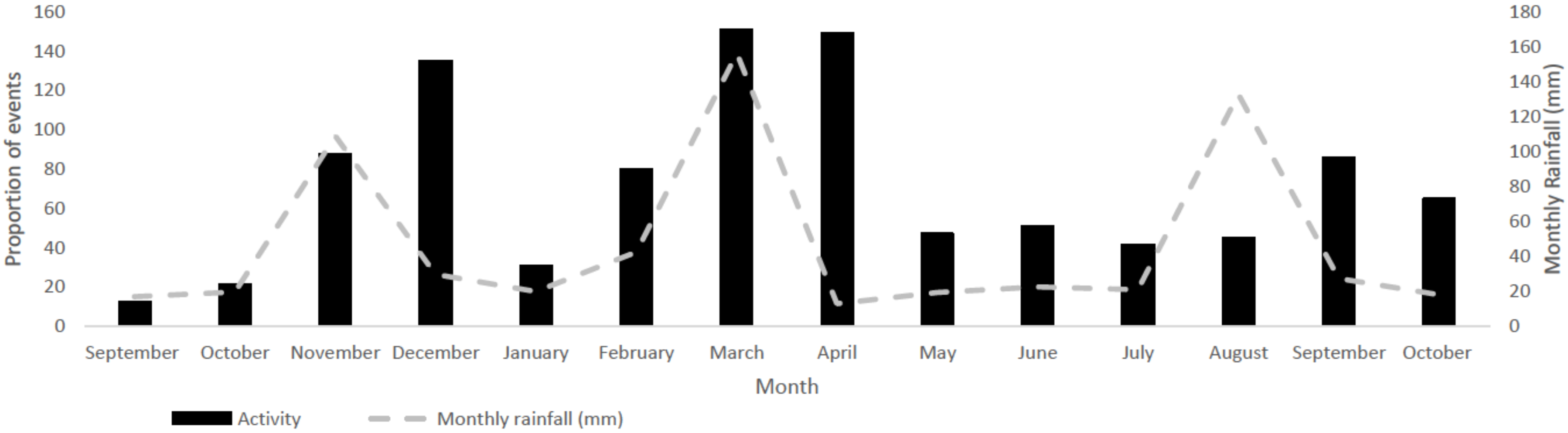


Figure 7-6 Brush-tailed rock-wallaby activity in burnt plots at each colony over the 13 months of monitoring, represented as the combined proportion of events for each colony, including average rainfall (mm).

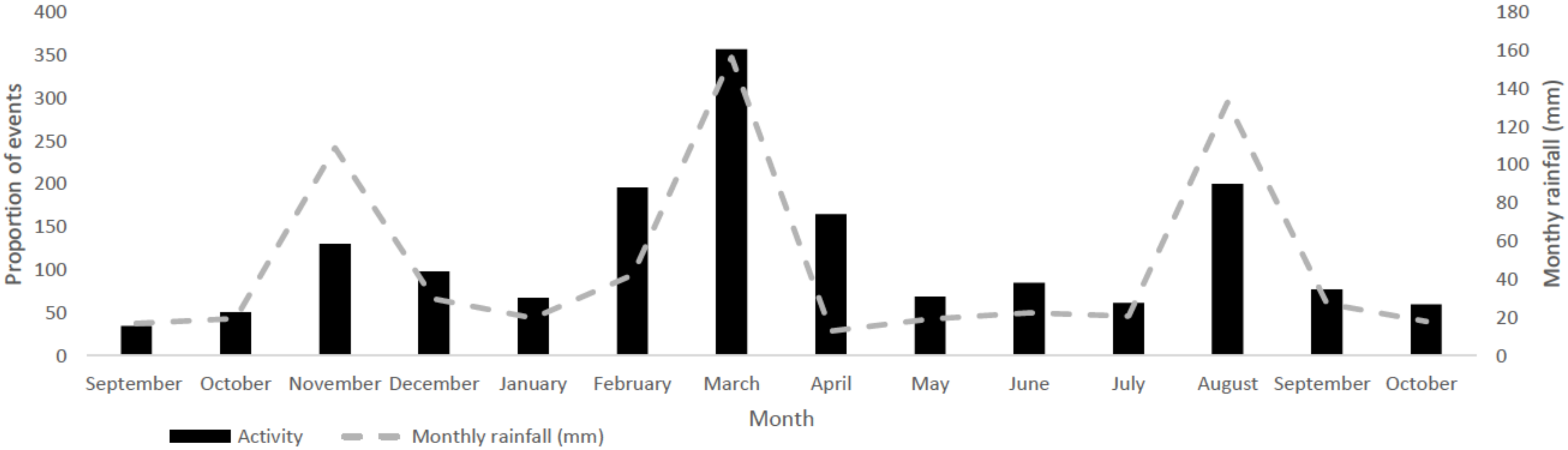


Figure 7-7 Brush-tailed rock-wallaby activity in unburnt c plots at each colony over the 13 months of monitoring, represented as the combined proportion of events for each colony, including average rainfall (mm)

7.3.2. Daily activity patterns

Brush-tailed rock-wallaby activity on plots was consistently nocturnal, with a tendency to become active on the plots just after sunset and ceasing just before sunrise (Fig 7-8). Activity density functions revealed a bimodal pattern with one activity peak just after sunset and the other more noticeable peak occurring approximately two hours before sunrise (Fig 7-8).

Temporal overlap of brush-tailed rock-wallabies utilising burnt and unburnt plots was high (87.2%; 95% CI 82.0 – 90.5) (Fig 7-9).

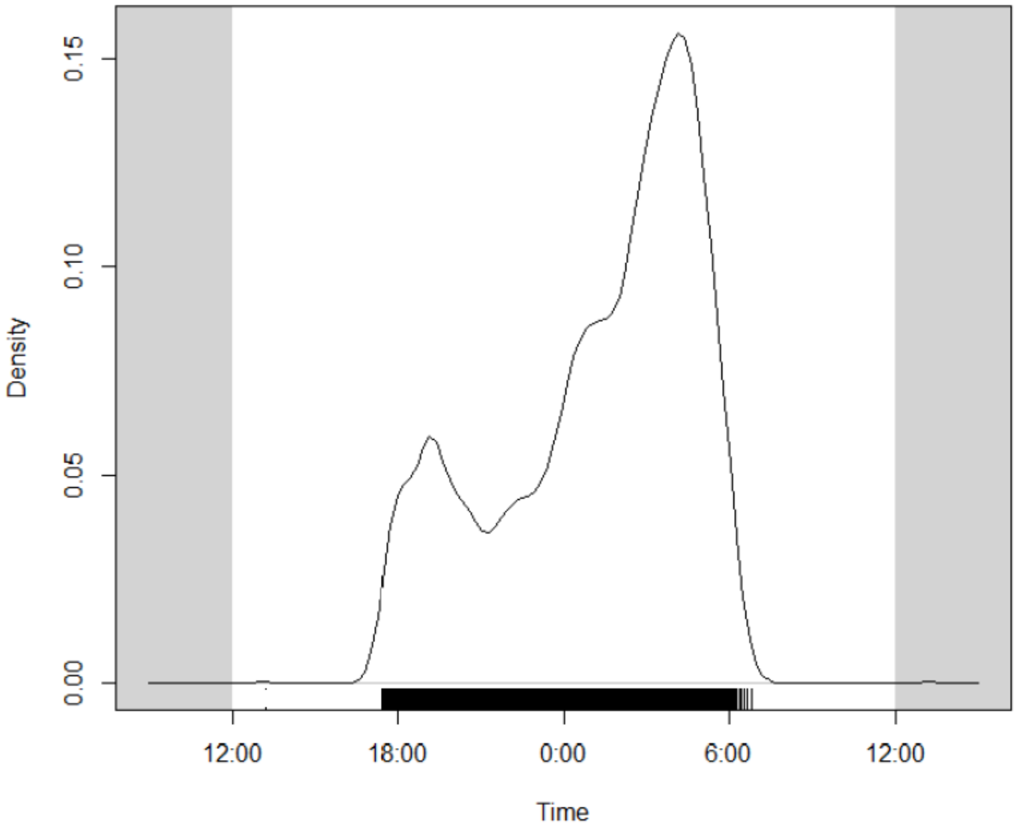


Figure 7-8 Activity profile of brush-tailed rock-wallabies at Green Gully. The black bar represents a record of all independent rock-wallaby events used to generate the activity profile

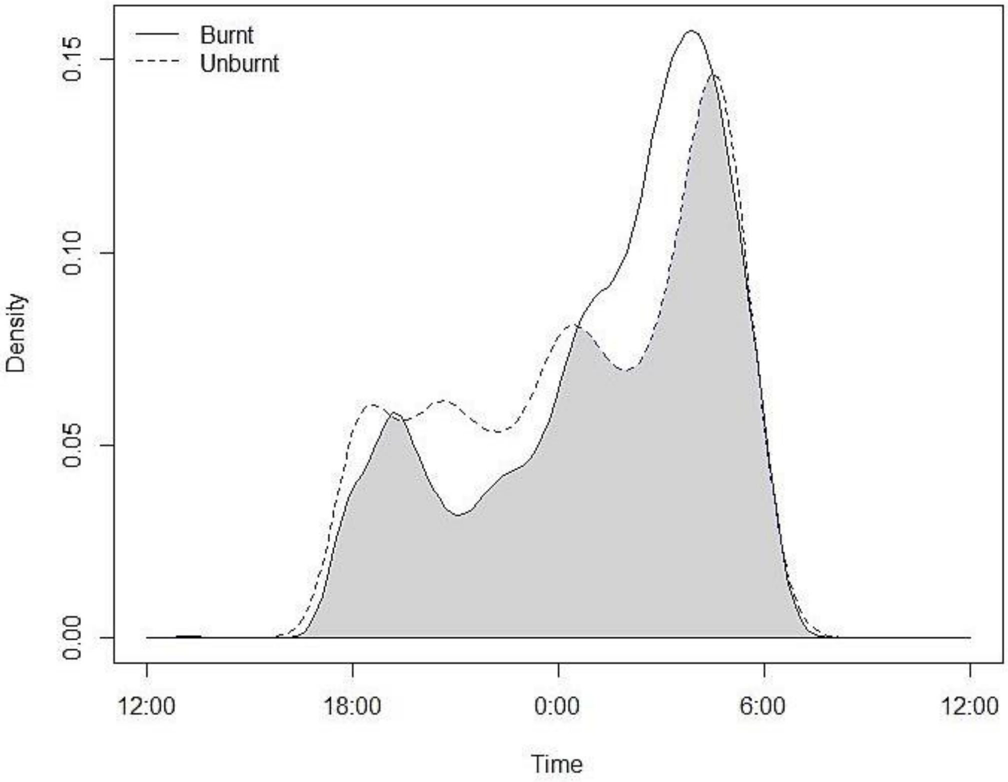


Figure 7-9 Activity overlap between brush-tailed rock-wallabies feeding on burnt (solid line) and unburnt (dashed line) plots

7.3.2.1. Seasonal variations in activity

Brush-tailed rock-wallaby activity patterns were broadly similar between summer and winter, although winter activity began an hour earlier and ended an hour later (Fig 7-10). A temporal overlap of 72.5% (95% CI: 69.9 – 75.5) was estimated. A higher peak in activity before sunrise was observed in summer (Fig 7-10).

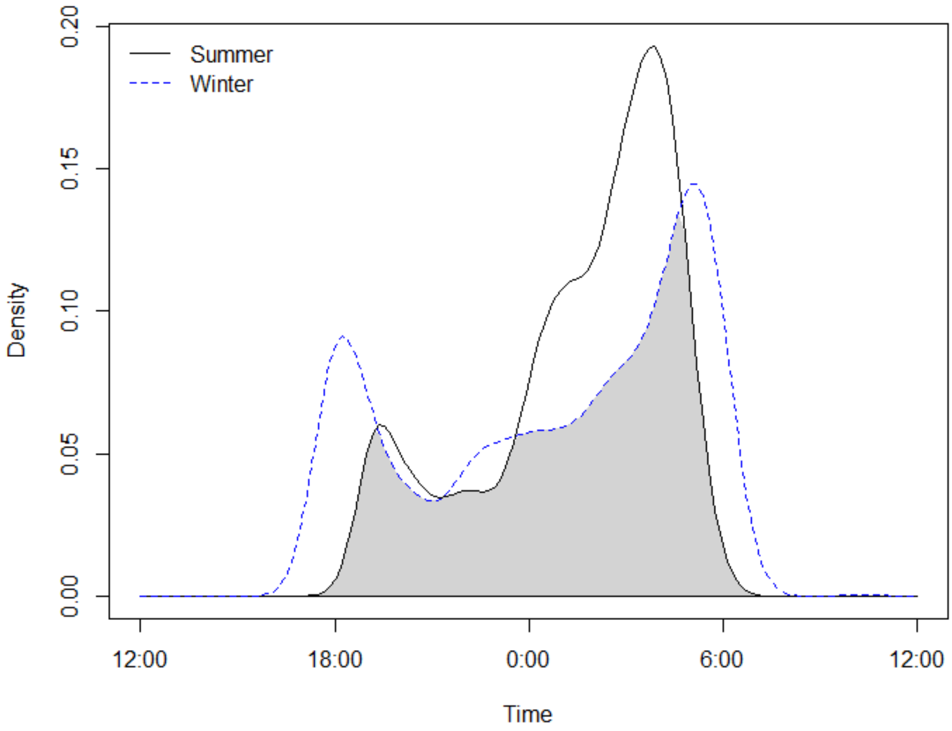


Figure 7-10 Daily activity patterns of brush-tailed rock-wallabies during summer (solid line) and winter (dashed line). Sunrises and sets in summer at 0600 and 1800 respectively. In winter, sunrise and sunset are at 0650 and 1720 respectively.

7.3.2.2. Interactions with sympatric macropods

Other macropods recorded on cameras included red-necked wallabies (*M. rufogriseus*), swamp wallabies (*Wallabia bicolor*), eastern-grey kangaroos (*M. giganteus*) and wallaroos (*M. robustus*). Overlap of other macropods in temporal activity with brush-tailed rock-wallabies was low, at 44.0% (95% CI: 41.6 – 51.7%) (Fig 7-11). Whereas the brush-tailed rock-wallabies exhibited nocturnal behaviour, the macropod assemblage at Green Gully was largely diurnal. Other macropods had an activity peak from sunrise to approximately 0900 h, declining during day before a second peak around sunset (Fig 7-11). Sympatric macropod activity on monitoring plots was minimal at midnight (Fig 7-11).

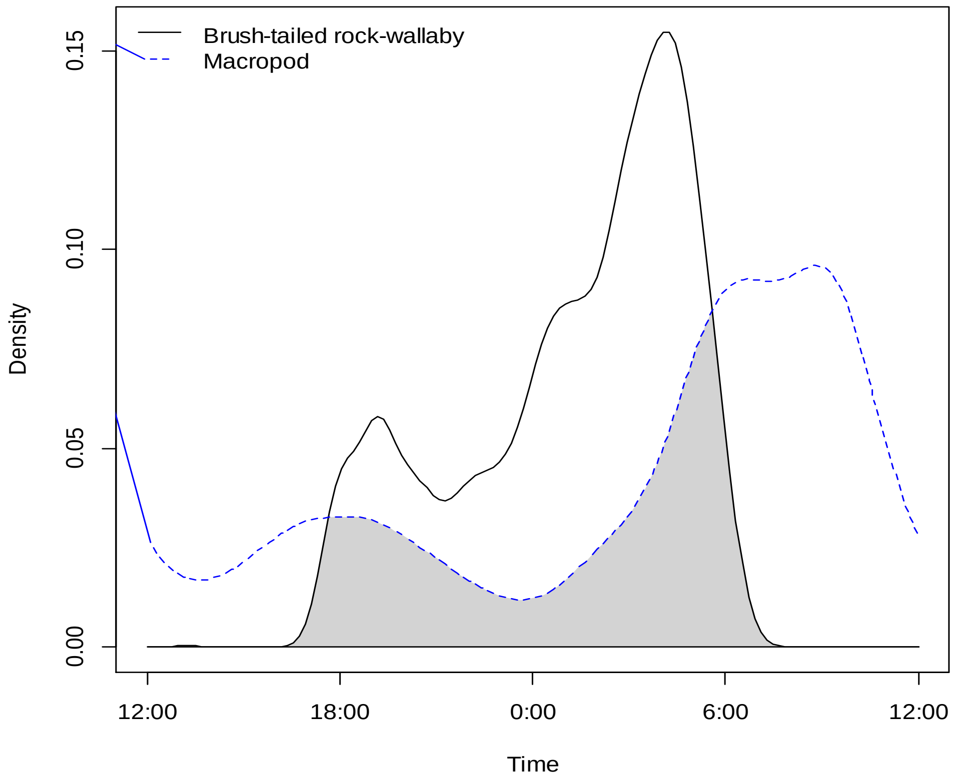


Figure 7-11 Temporal overlap in activity of brush-tailed rock-wallabies and other macropods (red-necked wallabies, eastern-grey kangaroos, swamp wallabies and wallaroos) at Green Gully.

7.4. Discussion

7.4.1. Selection of microhabitat by foraging brush-tailed rock-wallabies

I was able to quantify the spatial and temporal influence of fire on rock-wallaby foraging patterns, and demonstrate that brush-tailed rock-wallabies exhibit a strong preference for recently burnt patches close to their refuges. Individuals actively fed on the burnt patches of vegetation at Green Gully to a greater extent than unburnt patches for the entire 13 months of monitoring. In addition, the average number of individuals utilising the burnt patches closer to the colony was the same as the number of animals travelling further to forage. The broad-scale observational and experimental work in this study reveals that patch burning has a strong influence on animal behaviour and that the interaction between fire and grazing itself is strong.

The preference for rock-wallabies to remain close to their refuge (in this case, within 100m) can be explained by optimal foraging rules. Senft et al. (1987) presented a hierarchical foraging model where a grazer will select a patch of vegetation first for its richness, then its productivity and thirdly the palatability. However, grazers are faced with a trade-off between the potential energy gained from a foraging event against the perceived risk of predation (Lima and Dill 1990). The limitations imposed on central-place foragers results in the depletion of resources from patches closest to their refuge before the exploitation of areas at a greater distance away (Schoener 1979). However, for rock-wallabies, the distance from a colony or refuge can result in the plural issue of predation and water conservation, and this is possibly the reason that the number of rock-wallabies feeding on a patch was influenced by treatment. In some instances, an individual may accept more risk in order to forage on ‘good’ patches, although the quality of these areas needs to be significantly greater to make the risk worthwhile. Not only do green shoots have higher nutrient contents, they also generally have a higher water content per unit of biomass (Jarman and Evans 2010) which can potentially assist with the water conservation issue mentioned previously. The trade-off between risk and reward has been documented in a number of species, including gerbilline rodents, where foraging only occurred in risky patches if the quality of the forage available was four to eight times more valuable than the patches that afforded more cover (Kotler et al. 1993). Red-necked wallabies (*Macropus rufogriseus*) were also observed to actively forage on patches that offered no on-site shelter, but only when the habitat offered a high cover of edible vegetation (While and McArthur 2005).

Brush-tailed rock-wallabies also preferentially selected burnt patches over unburnt vegetation approximately 100 m from the colony. Results from a concurrent telemetry project (Ballard et al, unpublished data) suggest that local individuals will travel greater distances from the colony than this to feed. Research has shown that the home-range size of male brush-tailed rock-wallabies is greater than those of females (Laws and Goldizen 2003). Establishing if males represent a large proportion of the individuals travelling further from the refuge to forage provides potential for future research as an attempt to distinguish between characteristics of individuals and sexes was not undertaken here.

Aggregation, or the perceived ‘safety in numbers,’ is a common behavioural response of foraging macropods exposed to the threat of predation, and has been documented in Tasmanian pademelons (*Thylogale billardierii*) (Blumstein and Daniel 2003b), red kangaroos *(Macropus rufus*) (Blumstein and Daniel 2003a), eastern grey kangaroos (*Macropus giganteus*) (Carter et al. 2009, Favreau et al. 2010, Dannock et al. 2013), yellow-footed rock-wallabies (*Petrogale xanthopus*) (Blumstein et al. 2001a) and quokkas (*Setonix brachyurus*) (Blumstein et al. 2001b). However, regardless of colony size, the number of brush-tailed rock-wallabies recorded foraging together was low, with on average, 1–2 individuals feeding within the same event. This is unsurprising, as previous work has demonstrated that brush-tailed rock-wallabies are not as social as larger macropods, and are often observed alone for more than 60% of the night (Laws and Goldizen 2003).

Interestingly, brush-tailed rock-wallabies were present on burnt plots in the three months after the burns when patches were virtually devoid of vegetation. At the time of burning, and within the three months’ post burning, the study area experienced prolonged drought, which limited the amount of forage available to rock-wallabies. The regrowth that occurs after burning is known to contain higher nutrient levels than unburnt vegetation (Van de Vijver et al. 1999), and sustained grazing has been shown to retain vegetation in a nutrient-rich state (Van der Wal et al. 2000), facilitating the formation and preservation of marsupial grazing lawns (Roberts 2009). The continued selective use of these areas may therefore be a function of the rock-wallabies selecting for the most nutrient-rich patches in their home ranges. A consequence of selective grazing on recently burnt areas, however, is that it can limit the potential for vegetation to recover by depleting a plants' energy reserves, resulting in a significant shift in how regenerating plant species' allocated their energy resources (Chapter 6).

The influence of rainfall on the use of treatment plots was also examined. A strong seasonal relationship in foraging effort was evident in all treatment plots, but to a lesser degree in unburnt far patches. Peaks in use of unburnt areas occurred immediately following rainfall events in early summer and autumn, with the green flush of vegetation (for results, see Chapter 6) making unburnt patches more favourable than they were prior to the rain. Other authors have noted that rainfall can alter the preferential use of burnt patches by macropods; Caughley et al. (1985) monitored the distribution of red and grey kangaroos before and after burning in western New South Wales, finding that selective use of burnt areas did not occur, possibly as result of higher than average winter rainfall. The lack of foraging response after the August rainfall in 2014 was likely a result of the vegetation not being able to take advantage of the rainfall due to temperature being limiting. An intrinsic link exists between rainfall and when plants are able to respond in terms of temperature, and the peak in activity in September-October could be a result of water remaining in the soil until temperatures became sufficiently warm to allow vegetative growth. The peaks in rock-wallaby activity were lower in September and October 2014 indicating that rainfall coincident with the thermal period for warm season growth can potentially allow for increased foraging at more extensive areas around the colonies.

7.4.2. Activity and behaviour of brush-tailed rock-wallabies

Brush-tailed rock-wallabies at Green Gully were nocturnal, with a bimodal pattern of activity at night. The cyclic pattern showed a small peak in activity in the hours just after sunset, and was similar to the activity pattern reported by Carter and Goldizen (2003). However, Carter and Goldizen (2003) did not observe the higher, more pronounced second peak in activity in this study, which was identified in the few hours before sunrise. The second peak in activity may be linked to the circadian variation in leaf water content. Jarman and Evans (2010) demonstrated that the leaf water content of a number of grass species (including *Themeda triandra*, a dominant grass species at Green Gully) was highest just before dawn, dropping significantly throughout the day, before rising again throughout the night. The possibility that diel cycle in water content of the dominant grasses at Green Gully drives the circadian variation in brush-tailed rock-wallaby activity is suggested by the observed peak in activity just before sunrise and the corresponding high water content of grass leaf at this time (Jarman and Evans 2010). This preference for feeding on grasses when their water content is at its peak could be a function of predator avoidance. If an individual brush-tailed rock-wallaby’s needs are met by its intake of grass at certain times of the day, the costs and risks of travelling to free standing water (sometimes kilometres away) would be significantly reduced. In addition, a favourable colony microclimate and nocturnal activity pattern may explain why brush-tailed rock-wallabies do not need to travel to open water sources. This cost-saving behaviour has also been observed in eastern-grey kangaroos, where individuals’ dietary requirements were met by the intake of grass at non-uniform times throughout the day (Jarman and Evans 2010). The first observed peak may be a direct result of the nutritional need for rock-wallabies to forage immediately after having spent the day fasting and tied to their rocky shelter where food resources (save for figs in some instances) are limited (personal observation).

A higher observed peak in pre-dawn foraging of brush-tailed rock-wallabies was observed in summer in comparison to winter. The diel cycle of the water content in plants may provide an explanation for this evident pattern. Jarman and Evans (2010) found that grasses preferentially selected by macropods in the Northern Tablelands had a higher moisture content in summer before dawn than in winter (where the relative water content of the grasses remained similar) and it is possible the rock-wallabies were following this pattern to take advantage of this. More research is needed however to validate this theory.

The onset of nocturnal foraging differed slightly between summer and winter. Nouvellet et al. (2012) stated that some animal activities are likely to be strongly influenced by the time of sunset and sunrise, and studies should take into account the variation in day length when considering the behavioural responses of target species. For brush-tailed rock-wallabies, activity during the winter months began approximately one hour earlier and ended one hour later than the same activities observed for these animals during the summer months. These seasonal variations in activity were likely a result of the shifting photoperiod throughout the study. This included decidedly shorter day lengths during winter which were associated with extended nocturnal foraging.

Boyd et al. (2014) suggested that relatively few studies have examined the time allocation that central place foragers give to certain behaviours, such as foraging, even though this information is vital when considering foraging strategies in terms of energy and time budgets. The behaviours exhibited by brush-tailed rock-wallabies varied significantly when comparing burnt and unburnt patches of vegetation. Contrary to the predictions of other authors in relation to brush-tailed rock-wallabies (Carter and Goldizen 2003), interactive behaviours (such as vigilance) increased in burnt patches with significantly less vegetative cover. This finding coincides with other studies on macropod vigilance levels, where red kangaroos remain more vigilant in grassy open habitats in comparison to scrubby vegetation types (Blumstein and Daniel 2003a). Increased vigilance in these patches may be a function of predator detection, due to lower amounts of concealing vegetation (Moe and Wegge 1997), or as a result of an increased degree of social interaction with other individuals using these areas. Locomotive behaviour (moving through a patch without pausing to feed) was significantly greater in the burnt close plots, and it is likely this occurred due to these plots' proximity to the colonies, with the heat-in-motion cameras capturing rock-wallabies as they travel to different areas to forage. Brush-tailed rock-wallabies have previously been shown to preferentially select areas with short green grass and high levels of forbs (Carter and Goldizen 2003, Tuft et al. 2012), in agreement with the impacts on vegetation as outlined in Chapter 6.

7.4.3. Competition with sympatric macropods

Surprisingly, competition with sympatric macropods at Green Gully did not appear to be a limiting factor for brush-tailed rock-wallabies accessing resources surrounding their colonies, specifically, such as the green pick that followed fire. Other authors suggest that competition with macropods restricts the foraging potential of rock-wallabies (Tuft et al. 2011), but this was not clear in this study, due to the very limited number of recordings of sympatric macropods on the camera traps. In addition, the recorded activity of the other macropods was not biased towards burnt patches, and most individuals were travelling through patches rather than pausing to actively feed. In African ungulates, the selection of post-fire regrowth may be governed by bodyweight, with Wilsey (1996) hypothesising that larger herbivores should continually ingest low-quality feed due the ability to digest low-quality resources through longer gut retention times, and smaller herbivores should preferentially select post-fire regrowth due to them requiring higher quality feed. Similar to the limited spatial overlap, there was relatively little temporal overlap in activity between rock-wallabies and of other macropod species.

The macropod assemblage at Green Gully, not including rock-wallabies, hold the competitive advantage in that they are not resource-limited by central place foraging behaviours. Therefore, the presence of small patches of freshly burnt vegetation in the landscape may not represent as great a ‘magnet’ for these species. However, if an extensive burn occurs around a colony and drought conditions prevail, competition with sympatric macropods may become more intense, due to a 50–60% dietary overlap between the brush-tailed rock-wallabies and the sympatric macropod species (Tuft 2005). Due to this, there is a need to actively monitor the densities of the other species of macropods over time, in order to establish if changes in their population levels will impact on the availability of resources for the brush-tailed rock-wallabies. This potential difference between future real-world scenarios and this controlled experiment should be taken into account by managers when considering the use of fire to promote food resources for brush-tailed rock-wallabies.

7.4.4. Conservation and management implications

The interactions between the spatial variation in feeding behaviour and fire-induced nutritional enhancement of forage suggests that understanding brush-tailed rock-wallaby foraging activity is dependent on examining foraging at the patch scale. The finding that rock-wallabies preferentially select post-burn regrowth, and do so for up to 13 months, suggests that patch burning around colonies could be used to positively influence the availability of food resources for rock-wallabies. However, it was also apparent that heavy grazing pressure imposed on small burnt areas close to colonies altered the composition of the understorey (such as the change in the dominant functional groups) (Chapter 6), and so these longer-term changes should be considered alongside the short-term benefits to rock-wallabies. Underwood and Christensen (1981) reported that the utilisation of forested areas in Western Australia by western brush wallabies and western grey kangaroos were three times higher in the first year following a burn when compared with utilisation five years later. Thus, further research examining the long-term use of burnt patches and the potential for the grassy understorey to recover will be beneficial, as grazing pressure may reduce over time.

At Green Gully, competition with sympatric macropods was not considered to limit resources for rock-wallabies, which is contrary to other studies comparing the competitive interactions between these species (Tuft et al. 2011). However, monitoring sympatric macropods' abundance would assist in identifying the potential for negative competitive interactions.

Studies on the nutritional content of the grass species in African grasslands have linked frequent fires (every 0–5 years) with decreasing nutritional content (Anderson et al. 2007), which, if similar changes occur in Australian woodlands, would have severe implications for the maintenance of forage quality for rock-wallabies. Plant species that can cope with frequent disturbance often allocate resources to less palatable and more resistant biomass. Along with recommendations made for managing the grassy understorey in Chapter 6, the results in this Chapter suggests that fire should be managed to create a mosaic of asynchronously burnt patches, large enough to potentially reduce grazing pressure on other areas and allow vegetation to recover. The size of an appropriate patch size will depend on a number of factors, including the colony size, quality of the understorey and the surrounding landscape features. In addition, the burning of patches during drought conditions or when drought is predicted may have serious implications for the spatial movements and welfare of rock-wallabies. It would appear that during a good season, the activity range of individuals is relatively limited, and if my hypothesis that rock-wallabies focus their foraging activity on the water content of grasses proves true, a wide spread burn during a dry period would force surviving rock-wallabies to travel far from their refuge in order to satisfy their dietary and water intake requirements, likely exposing them to greater predation risk. Examining the changes in water content of preferentially selected species, along with the use of remote cameras to track the presence of predators within the system (to test an alternate hypothesis, predator avoidance) could be used in order to test this hypothesis. Similar expansions of activity have been observed internationally, for example in agoutis (*Dasyprocta punctata*). Emsens et al. (2013) showed individuals of that species will increase their home range size in response to food scarcity at the cost of their safety. This likelihood reinforces the need for managers to take into account prevailing and predicted climate patterns when planning patch burns near brush-tailed rock-wallaby habitat, regardless of whether the fire is primarily intended to provide benefits to the wallabies or not.

8. Synthesis and conclusions

8.1. Introduction

The impetus for undertaking this research was an apparent decade long decline in brush-tailed rock-wallaby numbers at Green Gully, within Oxley Wild Rivers National Park. Within the context of the apparent management problem, this study addressed the key knowledge gaps in brush-tailed rock-wallaby conservation and management in north-east NSW. I primarily did this by testing options for population monitoring, and by examining the potential to use fire as a management tool around known rock-wallaby habitat.

This final chapter of my thesis summarises and explores the implications of the research findings presented in this thesis by highlighting the main findings of the research, concluding with a list of management recommendations.

In particular, this research aimed to:

1. To establish appropriate estimation protocols for estimating population size and related trends of brush-tailed rock-wallabies

2. To examine the response of the grassy understorey to changes in tenure and associated management approaches

3. Identify the responses of refuge dependent herbivores to vegetation manipulation

8.2. Main findings, contribution to scientific theory and practice and future research

8.2.1. Monitoring the population size of rock-wallabies

I examined the behavioural responses of rock-wallabies to aerial survey in response to concerns by local managers that increasing aversion to helicopter-based survey was negatively biasing population estimates, in turn creating the observed decline in population size (Fig 1-1). Although a number of authors have reported that the behavioural response of macropods to aerial survey significantly impacts on the accuracy of aerial survey (Short and Bayliss 1985, Southwell 1994), in the current study, the ecological parameters I measured suggests that the response of brush-tailed rock wallabies to helicopters was not a significant source of bias for aerial survey of rock-wallabies in Green Gully. Importantly, aerial survey appeared to have no obvious effects on the behaviour or welfare of the animals I observed. Indeed, the lack of response may be an important limitation for the technique, as animals that remain camouflaged against a similarly coloured background are less likely to be observed (Clancy 1999) than those who flush in response to the helicopter’s approach. I did not examine differences in behavioural responses between sexes or age classes; these issues may be the focus of future research should managers disregard my suggestion and persist with aerial survey as a monitoring technique. Essentially, my comparison of census techniques showed that the NPWS aerial survey underestimated population size but did not do so consistently. Consequently, it could not be used as an index of abundance. However, there is some potential for aerial survey to be utilised as a rapid assessment of brush-tailed rock-wallaby status within the region, and, in this regards, be a useful tool to establish conservation priorities as well as efficient management programs.

SLR individual recognition, and photographic mark – recapture returned better population estimates of brush-tailed rock-wallaby colonies. As a bonus, these ground-based techniques also provide demographic data which could be used by managers to undertake more sensitive assessments of population trends, including population viability. However, comparative evaluations of different areas of preferred brush-tailed rock-wallaby habitat, in order to determine the most applicable population estimation method.

It is possible that limited behavioural response to helicopter and high vegetative cover might be reasons for the relatively low estimates derived via aerial survey but further research would be required to quantify this relationship. All three ground-based techniques for population estimation showed considerable promise for future monitoring programs. Further, the ability to re-use equipment suggests it will be possible to make considerable savings in future monitoring programs. Development of automated recognition software, such as that in development for dingoes and quolls (Falzon et al, unpublished data) would make techniques reliant upon camera-traps even more cost effective in the future.

The use of these methods may not be limited to brush-tailed rock-wallabies. On an international scale, these methods may be suitable for species that are restricted to refuges, display preferential behaviours (e.g. basking) and are individually distinct. Therefore, more research is required to test the applicability of these methods over a range of species.

8.2.2. Managing brush-tailed rock-wallaby food resources

By re-surveying 26 sites across the former grazing property of Green Gully, I was able to consider how the exclusion of livestock grazing and the absence of frequent fires, influenced local vegetation. These management changes appeared to have significantly influenced the composition of the grassy understorey between 2005 and 2012, and native species turnover increased, in concert with a decline in introduced species turnover.

Principle components analysis demonstrated in increase in grazing sensitive, high-conservation-value species such as *Themeda triandra* and *Sorghum leiocladum* and an associated decline in grazing tolerant species such as *Sporobolus elongata* and *Eragrotis leptostachya*. The number of midstorey shrubs <1.5 m in height increased significantly over the seven years, a trend which has been documented repeatedly in dry sclerophyll vegetation ecosystems in Australia after the removal of livestock grazing (Clark 1988, Briggs et al. 2002, Tasker and Bradstock 2006). It was unclear; however, if the current 'thickening' is a transient episode and whether with longer periods of unburnt and ungrazed conditions, the system will eventually thin (Croft 2015). Repeated surveys will be needed to track these changes into the future. Such research could also improve our ability to develop management strategies that successfully maintain heterogeneity, and extend the applicability and theoretical advances made from the observation of broad-scale diversity patterns.

The landscape methods used in this Chapter, such as species accumulation curves, helped to identify the landscape-scale patterns evident in the region. The knowledge of such patterns may enable ecologists to more readily identify the processes that are responsible for them (Gering et al. 2003). From a NPWS management perspective, the major alteration of the grassy understorey to a high-conservation value sward in a relatively short time frame highlights the importance of continually including parcels of land into the protected area network, for the continued conservation and management of the flora and fauna.

The perceived benefits from the changes in the grassy understorey following livestock removal may only be applicable when the natural senescence of the grassy sward does not impact on overall nutritional value. The likelihood that these circumstances in combination are rare in natural systems provided the impetus for work carried out in subsequent chapters. Anecdotal evidence suggested that small scale, low intensity fires might address issues of sward senescence thereby benefiting rock wallabies, and therefore I implemented a controlled experiment to examine the combined effects of a single low intensity burn and sustained macropod grazing on the grassy understorey at Green Gully. Low intensity fire and macropod grazing had no substantial, enduring effect on vegetation species density but after 13 months of grazing in the burnt patches, groundcover within the burnt and grazed plots was significantly lower than in the ungrazed and unburnt plots. Further, PCA analysis indicated there was evidence of a drastic compositional change, within the burnt and grazed plots close to rock wallaby colonies, to a sward dominated by basal, rosette forming and small leaf area plants. In addition, the compositional differences were greater between seasons than between treatments, leading me to conclude that seasonal variability (particularly growth in autumn) had a greater influence on compositional change observed within the plots than the interactive effects of fire and grazing. Further monitoring of the effects of inter-seasonal variability is needed, as any experimental design must take into account rainfall variability and the effects that it has on ground layer vegetation, regardless of the treatments imposed.

I showed that individual rock-wallabies actively fed on the burnt patches of vegetation at Green Gully to a greater extent than unburnt patches for the entire 13 months of monitoring. I determined that brush-tailed rock-wallabies exhibit a bimodal nocturnal activity pattern that is possibly due to water conservation. Competition with sympatric macropods was not identified as a significant factor when considering rock-wallabies’ access to resources around the colonies, however further research investigating this relationship is needed, especially if patch burns are of a larger scale. Camera trapping did indicate the use of the isolated burnt patches far from the colonies. Past studies on home ranges indicates that males have larger home ranges (Laws and Goldizen 2003), further research could be directed at examining if a bias exists for the sex of individuals travelling further from the colony to forage.

Additional research is also required to determine if a perceived risk of predation is linked to the observed brush-tailed rock-wallabies' foraging habits. Fleming (1997) suggests that the regulation of macropods by wild dogs is occurring within the Northern Tablelands. Since the main focus of this thesis was not to look at potential predation on rock-wallabies, further research over a longer time-frame is required to confirm the applicability of this theory to rock-wallaby/wild dog (or other predators, such as foxes or cats) interactions.

8.3. Implications and recommendations

The findings of this thesis have several implications not only for the management of the brush-tailed rock-wallaby, but for the management of naturally forested ecosystems throughout North-eastern Australia. This study demonstrated significant seasonal variation in the vegetation survey data. Assuming the patterns observed are consistent and repeatable over time, data such as these can be used to estimate the proportion of diversity not captured by a single survey. For example, Schultz et al. (2014) estimated that in the spring in a year of average to above rainfall on the North-West Slopes 40 – 70% of the species present on the site would be revealed. Conversely, if patterns are inconsistent or unpredictable, reliable data for total species density will only be achieved by repeat surveys in different seasons.

A changing climate will have significant implications for the management of brush-tailed rock-wallabies at not only Green Gully, but throughout their current range. With predictions of hotter summers and an increased risk of bush fire, the appropriate management of food-resources for these animals is of paramount importance. Warmer average temperatures will also have implications if my hypothesis that the activity patterns of rock-wallabies follow a water conservation strategy proves to be true. A higher dew point may limit the amount of time where wallabies are able to meet their water requirements from the grass, and the implications that warmer temperatures have on the overall water content of grasses needs to be examined. In addition, being unable to meet the water requirements from their forage results in individuals needing to travel, thus potentially increasing their predation risk. This hypothesis has implications not only for rock-wallabies but for all central place foragers, where resources are restricted and free standing water may not be available.

The importance of disturbance interactions for structuring grasslands has been widely recognised, and res-establishing fire-grazing interactions is often identified as a priority for maintaining biodiversity within these systems. Many authors have suggested that fire and herbivory can interact strongly in space and time to shape the structure of vegetation communities (Royo et al. 2010, Koerner and Collins 2014). While my experiments were small scale, the contrasting responses of the ground-layer vegetation to herbivory and fire indicate that a mixed management strategy promoting a heterogeneous understorey should be implemented. This is applicable to land managers seeking to not only assist with the persistence of the brush-tailed rock-wallaby throughout its range, but also by those aiming to achieve a heterogeneous landscape as a desired goal of land management. Although both fire and herbivory are often actively managed in forested systems (Gordon et al. 2004), these processes are usually considered independently (Royo and Carson 2006). However, the interactive effects of fire and herbivory observed in this study indicate that integrating herbivore management with fire management practices is likely to be important for achieving vegetation heterogeneity in forests. My study has highlighted the value of experimental studies that quantify disturbance responses collectively, and emphasises the importance of considering multiple flora and fauna species in the management of disturbance regimes.

A major challenge for the conservation of threatened species’ is incomplete knowledge of their distribution and ecology. My findings are likely to apply to other populations of brush-tailed rock-wallabies, and further research is required to ascertain the applicability of the population estimation techniques in other metapopulations. My goal was to provide knowledge regarding brush-tailed rock-wallabies, primarily to inform management at a local scale, and to inform the development of guidelines for broader conservation of the species. My findings have some clear guidelines for the not only the management of rock-wallabies at Green Gully, but across this species' current range, and they can be summarised as follows:

- Aerial survey, as implemented to date, should not continue to be used for population monitoring, however has potential as a broad-scale measure of presence/absence
- A rigorous, camera-trap based mark – recapture monitoring program should be implemented in order to effectively monitor a representative subset of the broader population within Oxley Wild Rivers National Park
- Small, low-intensity patch burns, preferably undertaken in winter, could be used to stimulate nutritious food resources for rock-wallabies, however 'complete' burns around rock-wallaby colonies should be avoided. Further, it would be advisable to:
    - Establish a heterogeneous local environment by maintaining patches in various states of succession (no less than 5 year intervals between burns).
    - If drought conditions are predicted, burning should not be undertaken, as it will be important to maintain existing vegetation and retain palatable browse plants until conditions improve.
    - Plans for supplementary feeding of rock-wallabies should be developed, to follow burns that are followed by drought conditions.

8.4. Conclusions

For the management of threatened species, like brush-tailed rock-wallabies, rigorous estimates of population size and change that can be defended in scientific and public fora are imperative if management is to be effective. To ensure the persistence of a species within the landscape, a better understanding of that species ecology is vital. This research provided new insight into the ecology and conservation of brush-tailed rock-wallabies at Green Gully, Oxley Wild Rivers National Park. In particular, it is evident how we can improve monitoring of the species and responsibly promote vital, native food resources within the landscape. By incorporating these lessons into management and recovery plans, conservation managers can improve long term conservation and management of this iconic, endangered macropod.

Literature Cited

Abrams, P. A. 2001. The effect of density-independent mortality on the coexistence of exploitative competitors for renewing resources. The American Naturalist **158**:459-470.

Act, E. 1999. Environment Protection and Biodiversity Conservation Act 1999 (Cth).

Act, T. 1995. Threatened Species Conservation Act 1995. NSW.

Allen, D. E., P. M. Bloesch, R. A. Cowley, T. G. Orton, J. E. Payne, and R. C. Dalal. 2014. Impacts of fire on soil organic carbon stocks in a grazed semi-arid tropical Australian savanna: accounting for landscape variability. Rangeland Journal **36**:359-369.

Allred, B. W., S. D. Fuhlendorf, D. M. Engle, and R. D. Elmore. 2011. Ungulate preference for burned patches reveals strength of fire–grazing interaction. Ecology and Evolution **1**:132-144.

Alonso, R. S., B. T. McClintock, L. M. Lyren, E. E. Boydston, and K. R. Crooks. 2015. Mark-recapture and mark-resight methods for estimating abundance with remote cameras: a carnivore case study. PloS one **10**:e0123032.

Andersen, A. N. 1991. Responses of ground-foraging ant communities to three experimental fire regimes in a savanna forest of tropical Australia. Biotropica:575-585.

Andersen, A. N., T. Hertog, and J. C. Woinarski. 2006. Long-term fire exclusion and ant community structure in an Australian tropical savanna: congruence with vegetation succession. Journal of Biogeography **33**:823-832.

Anderson, T. M., M. E. Ritchie, E. Mayemba, S. Eby, J. B. Grace, and S. J. McNaughton. 2007. Forage nutritive quality in the Serengeti ecosystem: the roles of fire and herbivory. The American Naturalist **170**:343-357.

Archibald, S., and W. Bond. 2004. Grazer movements: spatial and temporal responses to burning in a tall-grass African savanna. International Journal of Wildland Fire **13**:377-385.

Archibald, S., W. Bond, W. Stock, and D. Fairbanks. 2005. Shaping the landscape: fire-grazer interactions in an African savanna. Ecological Applications **15**:96-109.

Arthur, A. D., P. C. Catling, and A. Reid. 2012. Relative influence of habitat structure, species interactions and rainfall on the post-fire population dynamics of ground-dwelling vertebrates. Austral Ecology **37**:958-970.

Bakker, E. S., R. C. Reiffers, H. Olff, and J. M. Gleichman. 2005. Experimental Manipulation of Predation Risk and Food Quality: Effect on Grazing Behaviour in a Central-Place Foraging Herbivore. Oecologia **146**:157-167.

Banks, P. B., and C. R. Dickman. 2000. Effects of winter food supplementation on reproduction, body mass, and numbers of small mammals in montane Australia. Canadian Journal of Zoology **78**:1775-1783.

Bayliss, P. 1986. Factors Affecting Aerial Surveys of Marine Fauna, and Their Relationship to a Census of Dugongs in the Coastal Waters of the Northern-Territory. Wildlife research **13**:27-37.

Bayliss, P., and J. Giles. 1985. Factors affecting the visibility of kangaroos counted during aerial surveys. The Journal of Wildlife Management:686-692.

Bayne, P., B. Harden, K. Pines, and U. Taylor. 2000. Controlling feral goats by shooting from a helicopter with and without the assistance of ground-based spotters. Wildlife research **27**:517-523.

Bengsen, A., J. Butler, and P. Masters. 2011. Estimating and indexing feral cat population abundances using camera traps. Wildlife research **38**:732-739.

Bleich, V. C., R. T. Bowyer, A. M. Pauli, R. L. Venroy, and R. W. Anthes. 1992. Responses of mountain sheep to helicopter surveys: V. C. Bleich, R. T. Bowyer, A. M. Pauli, R. L. Vernoy & R. W. Anthes, California Fish & Game, 76(4), 1990, pp 197–204. Biological conservation **59**:276.

Blumstein, D., and J. Daniel. 2003a. Red kangaroos (Macropus rufus) receive an antipredator benefit from aggregation. acta ethologica **5**:95-99.

Blumstein, D. T., and J. C. Daniel. 2003b. Foraging behavior of three Tasmanian macropodid marsupials in response to present and historical predation threat. Ecography **26**:585-594.

Blumstein, D. T., J. C. Daniel, and C. S. Evans. 2001a. Yellow-Footed Rock-Wallaby Group Size Effects Reflect A Trade-Off. Ethology **107**:655-664.

Blumstein, D. T., J. C. Daniel, and I. G. McLean. 2001b. Group size effects in quokkas. Australian Journal of Zoology **49**:641-649.

Blumstein, D. T., J. C. Daniel, M. R. Schnell, J. G. Ardron, and C. S. Evans. 2002. Antipredator behaviour of red-necked pademelons: a factor contributing to species survival? Animal Conservation **5**:325-331.

Bodie, W. L., E. O. Garton, E. R. Taylor, and M. McCoy. 1995. A sightability model for bighorn sheep in canyon habitats. The Journal of Wildlife Management:832-840.

BOM. 2015. Climate Summary Statistics: Walcha Post Office. Bureau of Meteorology

Bond, W. J., and J. E. Keeley. 2005. Fire as a global ‘herbivore’: the ecology and evolution of flammable ecosystems. Trends in ecology & evolution **20**:387-394.

Bottrill, M. C., L. N. Joseph, J. Carwardine, M. Bode, C. Cook, E. T. Game, H. Grantham, S. Kark, S. Linke, and E. McDonald-Madden. 2009. Finite conservation funds mean triage is unavoidable. Update **24**.

Bouskila, A., and D. T. Blumstein. 1992. Rules of thumb for predation hazard assessment: predictions from a dynamic model. American Naturalist:161-176.

Bowles, M. L., and M. D. Jones. 2013. Repeated burning of eastern tallgrass prairie increases richness and diversity, stabilizing late successional vegetation. Ecological Applications **23**:464-478.

Bowman, D. M., I. Abbott, and N. Burrows. 2003. Australian landscape burning: a continental and evolutionary perspective. Pages 107-118 *in* Fire in ecosystems of south-west Western Australia:

impacts and management. Symposium proceedings (Volume I), Perth, Australia, 16-18 April 2002. Backhuys Publishers.

Boyd, C., A. E. Punt, H. Weimerskirch, and S. Bertrand. 2014. Movement models provide insights into variation in the foraging effort of central place foragers. Ecological Modelling **286**:13-25.

Bradstock, R., M. Tozer, and D. Keith. 1997. Effects of high frequency fire on floristic composition and abundance in a fire-prone heathland near Sydney. Australian Journal of Botany **45**:641-655.

Bradstock, R. A. 2010. A biogeographic model of fire regimes in Australia: current and future implications. Global Ecology and Biogeography **19**:145-158.

Braun-Blanquet, J. 1932. Plant sociology. The study of plant communities. Plant sociology. The study of plant communities. First ed.

Briggs, J. M., A. K. Knapp, and B. L. Brock. 2002. Expansion of woody plants in tallgrass prairie: a fifteen-year study of fire and fire-grazing interactions. The American Midland Naturalist **147**:287-294.

Broughton, S., and C. Dickman. 1991. The effect of supplementary food on home range of the southern brown bandicoot, Isoodon obesulus (Marsupialia: Peramelidae). Australian Journal of Ecology **16**:71-78.

Bruce, V., P. R. Green, and M. A. Georgeson. 2003. Visual perception: Physiology, psychology, & ecology. Psychology Press.

Buitenwerf, R., A. M. Swemmer, and M. J. Peel. 2011. Long-term dynamics of herbaceous vegetation structure and composition in two African savanna reserves. Journal of Applied Ecology **48**:238-246.

Burgman, M. A., H. P. Possingham, A. J. J. Lynch, D. A. Keith, M. A. McCarthy, S. D. Hopper, W. L. Drury, J. A. Passioura, and R. J. Devries. 2001. A method for setting the size of plant conservation target areas. Conservation biology **15**:603-616.

Burkepile, D. E., C. E. Burns, C. J. Tambling, E. Amendola, G. M. Buis, N. Govender, V. Nelson, D. I. Thompson, A. D. Zinn, and M. D. Smith. 2013. Habitat selection by large herbivores in a southern African savanna: the relative roles of bottom-up and top-down forces. Ecosphere **4**:art139.

Cairns, S. 2003. A report to the New South Wales National Parks & Wildlife Service on the consultancy:‘Kangaroo monitoring–New England Tablelands Helicopter Survey’. Unpublished report to New South Wales National Parks and Wildlife Service, Dubbo, NSW.

Cairns, S. C., G. W. Lollback, and N. Payne. 2008. Design of aerial surveys for population estimation and the management of macropods in the Northern Tablelands of New South Wales, Australia. Wildlife research **35**:331-339.

Calef, G. W., E. A. DeBock, and G. M. Lortie. 1976. The reaction of barren-ground caribou to aircraft. Arctic:201-212.

Campos, D., F. Bartumeus, V. Méndez, and X. Espadaler. 2014. Reorientation patterns in central-place foraging: internal clocks and klinokinesis. Journal of the Royal Society, Interface / the Royal Society **11**:20130859.

Canunayon, M., and D. Thorogood. 2014. QTL Identification for Leaf Senescence in Perennial Ryegrass (Lolium perenne L.). Pages 267-271 Quantitative Traits Breeding for Multifunctional Grasslands and Turf. Springer.

Carter, A. J., O. Pays, and A. W. Goldizen. 2009. Individual variation in the relationship between vigilance and group size in eastern grey kangaroos. Behavioral Ecology and Sociobiology **64**:237-245.

Carter, K., and A. W. Goldizen. 2003. Habitat choice and vigilance behaviour of brush-tailed rock-wallabies (Petrogale penicillata) within their nocturnal foraging ranges. Wildlife research **30**:355-364.

Caughley, G., B. Brown, and J. Noble. 1985. Movement of Kangaroos After a Fire in Mallee Woodland. Wildlife research **12**:349-353.

Caughley, G., and J. Goddard. 1975. Abundance and distribution of elephants in the Luangwa Valley, Zambia. African Journal of Ecology **13**:39-48.

Choquenot, D. 1995a. Assessing visibility bias associated with helicopter counts of feral pigs in Australia's semi-arid rangelands. Wildlife research **22**:569-577.

Choquenot, D. 1995b. Species-and habitat-related visibility bias in helicopter counts of kangaroos. Wildlife Society Bulletin:175-179.

Christensen, P., and I. Abbott. 1989. Impact of fire in the eucalypt forest ecosystem of southern Western Australia: a critical review. Australian Forestry **52**:103-121.

Clancy, T. 1999. Choice of survey platforms and technique for broad-scale monitoring of kangaroo populations. Australian Zoologist **31**:267-274.

Clancy, T., A. Pople, and L. Gibson. 1997. Comparison of helicopter line transects with walked line transects for estimating densities of kangaroos. Wildlife research **24**:397-409.

Clark, S. 1988. Effects of hazard-reduction burning on populations of understorey plant species on Hawkesbury sandstone. Australian Journal of Ecology **13**:473-484.

Clarke, P. J. 2003. Composition of grazed and cleared temperate grassy woodlands in eastern Australia: patterns in space and inferences in time. Journal of Vegetation Science **14**:5-14.

Codding, B. F., D. W. Bird, and R. B. Bird. 2011. Provisioning offspring and others: risk-energy trade-offs and gender differences in hunter-gatherer foraging strategies. Proceedings: Biological Sciences **278**:2502-2509.

Colewell, R. K. 2013. EstimateS: statistical estimation of species richness and shared species from samples EstimateS 9.1.0 User’s Guide.

Cote, S. D. 1996. Mountain goat responses to helicopter disturbance. Wildlife Society Bulletin:681-685.

Creel, S., and D. Christianson. 2008. Relationships between direct predation and risk effects. Trends in ecology & evolution **23**:194-201.

Croft, P. 2015. Habitat features of open forests and woodlands in relation to disturbance by fire. University of New England, Armidale

Cutler, T. L., and D. E. Swann. 1999. Using remote photography in wildlife ecology: a review. Wildlife Society Bulletin:571-581.

Czech, B. 1991. Elk behavior in response to human disturbance at Mount St. Helens National Volcanic Monument. Applied Animal Behaviour Science **29**:269-277.

Dannock, R. J., S. P. Blomberg, and A. W. Goldizen. 2013. Individual variation in vigilance in female eastern grey kangaroos. Australian Journal of Zoology **61**:312-319.

Davis, M. A., and L. B. Slobodkin. 2004. The science and values of restoration ecology. Restoration Ecology **12**:1-3.

De Bondi, N., J. G. White, M. Stevens, and R. Cooke. 2010. A comparison of the effectiveness of camera trapping and live trapping for sampling terrestrial small-mammal communities. Wildlife research **37**:456-465.

DEC. 2005. Oxley Wild Rivers National Park, Plan of Management *in* D. o. E. a. Conservation, editor.

Delaney, R., P. Temple-Smith, and N. McLean. 2005. Translocation strategy for the brush-tailed rock-wallaby in Victoria. EnviroSense Consulting and Department of Sustainability and Environment, Melbourne.

DoE. 2014. National Reserve System protected area requirements.*in* D. o. t. Environment, editor.

Dorrough, J., J. Ash, and S. McIntyre. 2004. Plant responses to livestock grazing frequency in an Australian temperate grassland. Ecography **27**:798-810.

Dupré, C., and M. Diekmann. 2001. Differences in species richness and life-history traits between grazed and abandoned grasslands in southern Sweden. Ecography **24**:275-286.

Eby, S. L., T. M. Anderson, E. P. Mayemba, and M. E. Ritchie. 2014. The effect of fire on habitat selection of mammalian herbivores: the role of body size and vegetation characteristics. Journal of Animal Ecology **83**:1196-1205.

Eldridge, M., and R. Close. 1995. Brush-tailed rock-wallaby Petrogale penicillata. The mammals of Australia:383-385.

Eldridge, M. D. B., T. L. Browning, and R. L. Close. 2001. Provenance of a New Zealand brush-tailed rock-wallaby (Petrogale penicillata) population determined by mitochondrial DNA sequence analysis. Molecular Ecology **10**:2561-2567.

Emsens, W. J., L. Suselbeek, B. T. Hirsch, R. Kays, A. J. S. Winkelhagen, and P. A. Jansen. 2013. Effects of Food Availability on Space and Refuge Use by a Neotropical Scatterhoarding Rodent. Biotropica **45**:88-93.

Falzon, G. 2014. Computer-assisted identification of Australian rodents in camera trap imagery. In 'Camera Trapping in Wildlife Research and Management'.(Eds PD Meek, AG Ballard, PB Banks, AW Claridge, PJS Fleming, JG Sanderson and DE Swann.) pp. 299–306. CSIRO Publishing: Melbourne.

Fashing, P. J., and M. Cords. 2000. Diurnal primate densities and biomass in the Kakamega Forest: an evaluation of census methods and a comparison with other forests. American Journal of Primatology **50**:139-152.

Favreau, F.-R., A. W. Goldizen, and O. Pays. 2010. Interactions among social monitoring, anti-predator vigilance and group size in eastern grey kangaroos. Proceedings of the Royal Society of London B: Biological Sciences:rspb20092337.

Fensham, R., and S. Skull. 1999. Before Cattle: A Comparative Floristic Study of Eucalyptus Savanna Grazed by Macropods and Cattle in North Queensland, Australia1. Biotropica **31**:37-47.

Fewster, R. M., C. Southwell, D. L. Borchers, S. T. Buckland, and A. R. Pople. 2008. The influence of animal mobility on the assumption of uniform distances in aerial line-transect surveys. Wildlife research **35**:275-288.

Fleishman, E., R. F. Noss, and B. R. Noon. 2006. Utility and limitations of species richness metrics for conservation planning. Ecological Indicators **6**:543-553.

Fleming, P. 1997. Aspects of the Management of Wild Dogs ('Canis familiaris') in North-eastern New South Wales.

Fleming, P. J., and J. P. Tracey. 2008. Some human, aircraft and animal factors affecting aerial surveys: how to enumerate animals from the air. Wildlife research **35**:258-267.

Foster, B. L., and K. L. Gross. 1998. Species richness in a successional grassland: effects of nitrogen enrichment and plant litter. Ecology **79**:2593-2602.

Fox, M. D., and B. J. Fox. 1986. The susceptibility of natural communities to invasion. Pages 57-66 *in* R. H. B. Groves, J.J, editor. Ecology of biological invasions: An Australian perspective. Australian Academy of Science

Gaidet-Drapier, N., H. Fritz, M. Bourgarel, P.-C. Renaud, P. Poilecot, P. Chardonnet, C. Coid, D. Poulet, and S. Le Bel. 2006. Cost and efficiency of large mammal census techniques: comparison of methods for a participatory approach in a communal area, Zimbabwe. Biodiversity & Conservation **15**:735-754.

Gálvez-Bravo, L., A. López-Pintor, S. Rebollo, and A. Gómez-Sal. 2011. European rabbit (Oryctolagus cuniculus) engineering effects promote plant heterogeneity in Mediterranean dehesa pastures. Journal of Arid Environments **75**:779-786.

Garden, J. G., C. A. McAlpine, H. P. Possingham, and D. N. Jones. 2007. Using multiple survey methods to detect terrestrial reptiles and mammals: what are the most successful and cost-efficient combinations? Wildlife research **34**:218-227.

Gering, J. C., T. O. Crist, and J. A. Veech. 2003. Additive partitioning of species diversity across multiple spatial scales: implications for regional conservation of biodiversity. Conservation biology **17**:488-499.

Gibson, N., and J. Kirkpatrick. 1989. Effects of the cessation of grazing on the grasslands and grassy woodlands of the Central Plateau, Tasmania. Australian Journal of Botany **37**:55-63.

Gilkinson, A. K., H. C. Pearson, F. Weltz, and R. W. Davis. 2007. Photo-identification of sea otters using nose scars. The Journal of Wildlife Management **71**:2045-2051.

Gill, A., and R. Bradstock. 1995. Extinction of biota by fires. Conserving biodiversity: threats and solutions:309-322.

Gordon, I. J., A. J. Hester, and M. Festa-Bianchet. 2004. The management of wild large herbivores to meet economic, conservation and environmental objectives. Journal of Applied Ecology **41**:1021-1031.

Goswami, V. R., M. Madhusudan, and K. U. Karanth. 2007. Application of photographic capture–recapture modelling to estimate demographic parameters for male Asian elephants. Animal Conservation **10**:391-399.

Gough, L., and J. B. Grace. 1998. Herbivore effects on plant species density at varying productivity levels. Ecology **79**:1586-1594.

Gowen, C., and K. Vernes. 2014. Population estimates of an endangered rock-wallaby (Petrogale penicillata) using time-lapse photography from camera traps. Camera Trapping: Wildlife Management and Research:61.

Green, R. 1985. Estimating the abundance of breeding Snipe. Bird Study **32**:141-149.

Gregersen, P. L., P. B. Holm, and K. Krupinska. 2008. Leaf senescence and nutrient remobilisation in barley and wheat. Plant Biology **10**:37-49.

Griffin, A. S., A. S. Griffin, and C. S. Evans. 2003. Social learning of antipredator behaviour in a marsupial. Animal Behaviour **66**:485-492.

Grigg, G., A. Pople, and L. Beard. 1997. Application of an ultralight aircraft to aerial surveys of kangaroos on grazing properties. Wildlife research **24**:359-372.

Guthlin, D., I. Storch, and H. Kuchenhoff. 2014. Is it Possible to Individually Identify Red Foxes from Photographs? Wildlife Society Bulletin **38**:205-210.

Harrington, F. H., and A. M. Veitch. 1991. Short-term impacts of low-level jet fighter training on caribou in Labrador. Arctic:318-327.

Heckathorn, S., and E. Delucia. 1996. Restranslocation of Shoot Nitrogen to Rhizomes and Roots in Prairie Grasses May Limit Loss of N to Grazing and Fire during Drought. Functional Ecology:396-400.

Heilbrun, R. D., N. J. Silvy, M. J. Peterson, and M. E. Tewes. 2006. Estimating bobcat abundance using automatically triggered cameras. Wildlife Society Bulletin **34**:69-73.

Hill, G. J., A. Barnes, and G. Wilson. 1985. Time of day and aerial counts of grey kangaroos. The Journal of Wildlife Management:843-849.

Hobbs, R. J. 2001. Synergisms among habitat fragmentation, livestock grazing, and biotic invasions in southwestern Australia. Conservation biology **15**:1522-1528.

Hobbs, R. J., S. Yates, and H. A. Mooney. 2007. Long-term data reveal complex dynamics in grassland in relation to climate and disturbance. Ecological monographs **77**:545-568.

Hodgkinson, K. 1992. Elements of grazing strategies for perennial grass management in rangelands. The Linnean Society of London, London.

Hodgkinson, K., G. Harrington, G. Griffin, J. Noble, and M. Young. 1984. Management of vegetation with fire. Management of Australia's rangelands/GN Harrington, AD Wilson, MD Young [editors].

Hohnen, R., J. Ashby, K. Tuft, and H. McGregor. 2013. Individual identification of northern quolls (Dasyurus hallucatus) using remote cameras. Australian Mammalogy **35**:131-135.

Holway, D. A., and T. J. Case. 2000. Mechanisms of dispersed central-place foraging in polydomous colonies of the Argentine ant. Animal Behaviour **59**:433-441.

Hone, J. 1990. Predator prey theory and feral pig control, with emphasis on evaluation of shooting from a helicopter. Wildlife research **17**:123-130.

Hunter, J. 2005. Geographic variation in plant species richness patterns within temperate eucalypt woodlands of eastern Australia. Ecography **28**:505-514.

Isaac, J. L., L. E. Valentine, and B. A. Goodman. 2008. Demographic responses of an arboreal marsupial, the common brushtail possum (Trichosurus vulpecula), to a prescribed fire. Population ecology **50**:101-109.

Jachmann, H. 2002. Comparison of aerial counts with ground counts for large African herbivores. Journal of Applied Ecology **39**:841-852.

Jackson, R. M., J. D. Roe, R. Wangchuk, and D. O. Hunter. 2006. Estimating Snow Leopard Population Abundance Using Photography and Capture-Recapture Techniques. Wildlife Society Bulletin **34**:772-781.

Jarman, P., M. Jones, C. Johnson, C. Southwell, R. Stuartdick, K. Higginbottom, and J. Clarke. 1989. Macropod studies at Wallaby Creek. 8. Individual recognition of kangaroos and wallabies. Wildlife research **16**:179-185.

Jarman, P. J., and S. M. Capararo. 1997. Use of rock-wallaby faecal pellets for detecting and monitoring populations and examining habitat use. Australian Mammalogy **19**:257-264.

Jarman, P. J., and M. C. Evans. 2010. Circadian variation in resource quality: leaf water content and its relevance to eastern grey kangaroo Macropus giganteus and common wombat Vombatus ursinus. Austral Ecology **35**:176-188.

Joblin, K. 1984. Behaviour and Ecology of the Brushtailed Rockwallaby: Petrogale Penicillata, in the New England Region. University of New England.

Johnson, C. N. 1989. Grouping and the Structure of Association in the Red-Necked Wallaby. Journal of Mammalogy **70**:18-26.

Jones, M. E., D. Paetkau, E. Geffen, and C. Moritz. 2004. Genetic diversity and population structure of Tasmanian devils, the largest marsupial carnivore. Molecular Ecology **13**:2197-2209.

Jongejans, E., H. De Kroon, S. Tuljapurkar, and K. Shea. 2010. Plant populations track rather than buffer climate fluctuations. Ecology Letters **13**:736-743.

Kahmen, S., and P. Poschlod. 2008. Effects of grassland management on plant functional trait composition. Agriculture, Ecosystems & Environment **128**:137-145.

Karanth, K. U., and J. D. Nichols. 1998. Estimation of tiger densities in India using photographic captures and recaptures. Ecology **79**:2852-2862.

Keith, D., J. Williams, J. Woinarski, R. Bradstock, and A. Gill. 2002. Fire management and biodiversity conservation: key approaches and principles. Flammable Australia: the fire regimes and biodiversity of a continent:401-425.

Keith, D. A. 2004. Ocean shores to desert dunes: the native vegetation of New South Wales and the ACT. Department of Environment and Conservation (NSW).

Keith, D. A., and R. A. Bradstock. 1994. Fire and competition in Australian heath: a conceptual model and field investigations. Journal of Vegetation Science **5**:347-354.

Kingsford, R., and J. Porter. 2009. Monitoring waterbird populations with aerial surveys–what have we learnt? Wildlife research **36**:29-40.

Kirkpatrick, J., J. Marsden-Smedley, and S. Leonard. 2011. Influence of grazing and vegetation type on post-fire flammability. Journal of Applied Ecology **48**:642-649.

Klop, E., J. van Goethem, and H. H. de Iongh. 2007. Resource selection by grazing herbivores on post-fire regrowth in a West African woodland savanna. Wildlife research **34**:77-83.

Koerner, S. E., and S. L. Collins. 2014. Interactive effects of grazing, drought, and fire on grassland plant communities in North America and South Africa. Ecology **95**:98-109.

Köhler, B. 2001. Mechanisms and extent of vegetation changes in differently managed limestone grasslands. Diss., Naturwissenschaften ETH Zürich, Nr. 14227, 2001.

Kondoh, M. 2001. Unifying the relationships of species richness to productivity and disturbance. Proceedings of the Royal Society of London B: Biological Sciences **268**:269-271.

Kotler, B. P., J. S. Brown, and W. A. Mitchell. 1993. Environmental Factors Affecting Patch use in two Species of Gerbilline Rodents. Journal of Mammalogy **74**:614-620.

Krausman, P. R., B. D. Leopold, and D. L. Scarbrough. 1986. Desert mule deer response to aircraft. Wildlife Society Bulletin:68-70.

Lavorel, S., C. Rochette, and J.-D. Lebreton. 1999. Functional Groups for Response to Disturbance in Mediterranean Old Fields. Oikos **84**:480-498.

Law, B. S. 1995. The Effect of Energy Supplementation on the Local Abundance of the Common Blossom Bat, Syconycteris australis, in South-Eastern Australia. Oikos **72**:42-50.

Laws, R. J., and A. W. Goldizen. 2003. Nocturnal home ranges and social interactions of the brushtailed rock-wallaby Petrogale penicillata at Hurdle Creek, Queensland. Australian Mammalogy **25**:169-176.

Lazell, J. D., T. W. Sutterfield, and W. D. Giezentanner. 1984. The population of rock wallabies (genus Petrogale) on Oahu, Hawaii. Biological conservation **30**:99-108.

Le Mar, K., C. Southwell, and C. McArthur. 2001. Evaluation of line-transect sampling to estimate nocturnal densities of macropods in open and closed habitats. Wildlife research **28**:9-16.

Lenarz, M. 1974. The reaction of Dall sheep to an FH-1100 helicopter. Arctic Gas Biological.

Leonard, S., J. Kirkpatrick, and J. Marsden-Smedley. 2010. Variation between grassland structural types in the effects of vertebrate grazing on fire potential. Journal of Applied Ecology **47**:876-883.

Lett, M. S., and A. K. Knapp. 2003. Consequences of shrub expansion in mesic grassland: resource alterations and graminoid responses. Journal of Vegetation Science **14**:487-496.

Lewis, T. O. M., N. Reid, P. J. Clarke, and R. D. B. Whalley. 2010. Resilience of a high-conservation-value, semi-arid grassland on fertile clay soils to burning, mowing and ploughing. Austral Ecology **35**:464-481.

Lim, T., and J. Giles. 1987. Studies on the yellow-footed rock-wallaby, Petrogale xanthopus Gray (Marsupialia: Macropodidae). 3. Distribution and management in western New South Wales. Wildlife research **14**:147-161.

Lima, S. L., and L. M. Dill. 1990. Behavioral decisions made under the risk of predation: a review and prospectus. Canadian Journal of Zoology **68**:619-640.

Linklater, W. L., and E. Z. Cameron. 2002. Escape behaviour of feral horses during a helicopter count. Wildlife research **29**:221-224.

Lodge, G., and R. Whalley. 1989. Native and natural pastures on the Northern Slopes and Tablelands of New South Wales: a review and annotated bibliography. Technical Bulletin-New South Wales Agriculture and Fisheries (Australia).

Lunt, I. D. 1990. Impact of an autumn fire on a long-grazed Themeda triandra (Kangaroo Grass) grassland: implications for management of invaded, remnant vegetations. Victorian Naturalist **107**:45-51.

Lunt, I. D., A. Jansen, D. L. Binns, and S. A. Kenny. 2007. Long-term effects of exclusion of grazing stock on degraded herbaceous plant communities in a riparian Eucalyptus camaldulensis forest in south-eastern Australia. Austral Ecology **32**:937-949.

Lyra-Jorge, M. C., G. Ciocheti, and V. R. Pivello. 2008a. Carnivore mammals in a fragmented landscape in northeast of São Paulo State, Brazil. Biodiversity and Conservation **17**:1573-1580.

Lyra-Jorge, M. C., G. Ciocheti, V. R. Pivello, and S. T. Meirelles. 2008b. Comparing methods for sampling large-and medium-sized mammals: camera traps and track plots. European Journal of Wildlife Research **54**:739-744.

MacArthur, R. A., V. Geist, and R. H. Johnston. 1982. Cardiac and behavioral responses of mountain sheep to human disturbance. The Journal of Wildlife Management:351-358.

Masters, P. 1993. The effects of fire-driven succession and rainfall on small mammals in spinifex grassland at Uluru National Park, Northern Territory. Wildlife research **20**:803-813.

McAleer, K., and L.-A. Giraldeau. 2006. Testing central place foraging in eastern chipmunks, Tamias striatus, by altering loading functions. Animal Behaviour **71**:1447-1453.

McCallum, J. 2013. Changing use of camera traps in mammalian field research: habitats, taxa and study types. Mammal Review **43**:196-206.

McIntyre, S., R. B. Cunningham, C. F. Donnelly, and A. D. Manning. 2014. Restoration of eucalypt grassy woodland: effects of experimental interventions on ground-layer vegetation. Australian Journal of Botany **62**:570-579.

McIntyre, S., Z. Huang, and A. Smith. 1993. Patterns of Abundance in Grassy Vegetation of the New-England Tablelands; Identifying Regional Rarity in a Threatened Vegetation Type. Australian Journal of Botany **41**:49-64.

McIntyre, S., and S. Lavorel. 1994. How environmental and disturbance factors influence species composition in temperate Australian grasslands. Journal of Vegetation Science **5**:373-384.

McIntyre, S., S. Lavorel, and R. Tremont. 1995. Plant life-history attributes: their relationship to disturbance response in herbaceous vegetation. Pages 31-44 Journal of Ecology.

McIvor, J. G., S. McIntyre, I. Saeli, and J. Hodgkinson. 2005. Patch dynamics in grazed subtropical native pastures in south-east Queensland. Austral Ecology **30**:445-464.

McPherson, G. R., and H. A. Wright. 1990. Effects of cattle grazing and Juniperus pinchotii canopy cover on herb cover and production in western Texas. American Midland Naturalist:144-151.

Meek, M. G., S. M. Cooper, M. K. Owens, R. M. Cooper, and A. L. Wappel. 2008. White-tailed deer distribution in response to patch burning on rangeland. Journal of Arid Environments **72**:2026-2033.

Meek, P. D., G. Ballard, and P. Fleming. 2012. An introduction to camera trapping for wildlife surveys in Australia. Canberra: Invasive Animals CRC.

Meek, P. D., G. A. Ballard, K. Vernes, and P. J. S. Fleming. 2015. The history of wildlife camera trapping as a survey tool in Australia. Australian Mammalogy **37**:1-12.

Meers, B. T., and R. Adams. 2003. The impact of grazing by Eastern Grey Kangaroos (Macropus giganteus) on vegetation recovery after fire at Reef Hills Regional Park, Victoria. Ecological Management & Restoration **4**:126-132.

Meir, E., S. Andelman, and H. P. Possingham. 2004. Does conservation planning matter in a dynamic and uncertain world? Ecology Letters **7**:615-622.

Melville, G. J., J. P. Tracey, P. J. S. Fleming, and B. S. Lukins. 2008. Aerial surveys of multiple species: critical assumptions and sources of bias in distance and mark–recapture estimators. Wildlife research **35**:310-348.

Menkhorst, P., and P. Jarman. 2004. National recovery plan for the brush-tailed rock wallaby Petrogale penicillata (Gray 1825).

Meredith, M., and M. Ridout. 2013. overlap: Estimates of Coefficient of Overlapping for Animal Activity Patterns. R package version 0.2. 0. URL: http://CRAN. R-project. org/package= overlap.

Moe, S. R., and P. Wegge. 1997. The effects of cutting and burning on grass quality and axis deer (Axis axis) use of grassland in lowland Nepal. Journal of Tropical Ecology **13**:279-292.

Moog, D., S. Kahmen, and P. Poschlod. 2005. Application of CSR-and LHS-strategies for the distinction of differently managed grasslands. Basic and Applied Ecology **6**:133-143.

Moore, C. W. E. 1953. The vegetation of the south-eastern Riverina, New South Wales. I. The climax communities. Australian Journal of Botany **1**:485-547.

Moore, R. 1970. South-eastern temperate woodlands and grasslands. 1 edition. Australian National University Press, Canberra.

Morgan, J. W., and I. D. Lunt. 1999. Effects of time-since-fire on the tussock dynamics of a dominant grass (Themeda triandra) in a temperate Australian grassland. Biological conservation **88**:379-386.

Morris, C., J. Derry, and M. Hardy. 1999. Effect of cattle and sheep grazing on the structure of Highland Sourveld swards in South Africa. Tropical Grasslands **33**:111-121.

Morris, L. 2005. Vegetation and Floristic Composition of the Green Gully addition to Oxley Wild Rivers National Park University of New England Armidale

Morrison, D. A. 2002. Effects of fire intensity on plant species composition of sandstone communities in the Sydney region. Austral Ecology **27**:433-441.

Morrison, D. A., and J. A. Renwick. 2000. Effects of variation in fire intensity on regeneration of co-occurring species of small trees in the Sydney region. Australian Journal of Botany **48**:71-79.

Muir, K. 2012. NSW wilderness update. Nature New South Wales **56**.

Munidasa, D. T., and Y. Toquenaga. 2010. Do pollen diets vary among adjacent bumble bee colonies? Ecological Research **25**:639-646.

Murray, J. V., S. Low Choy, C. A. McAlpine, H. P. Possingham, and A. W. Goldizen. 2008. The importance of ecological scale for wildlife conservation in naturally fragmented environments: A case study of the brush-tailed rock-wallaby (Petrogale penicillata). Biological conservation **141**:7-22.

Norris, K. 2004. Managing threatened species: the ecological toolbox, evolutionary theory and declining-population paradigm. Journal of Applied Ecology **41**:413-426.

Nouvellet, P., G. Rasmussen, D. Macdonald, and F. Courchamp. 2012. Noisy clocks and silent sunrises: measurement methods of daily activity pattern. Journal of Zoology **286**:179-184.

Noy-Meir, I. 1995. Interactive effects of fire and grazing on structure and diversity of Mediterranean grasslands. Journal of Vegetation Science **6**:701-710.

NPWS. 2002. Warrumbungle Brush-tailed Rock-wallaby Endangered Population Recovery Plan. NSW National Parks and Wildlife Service Hurstville, NSW

Olff, H., and M. E. Ritchie. 1998. Effects of herbivores on grassland plant diversity. Trends in ecology & evolution **13**:261-265.

Oliveira-Santos, L. G. R., C. A. Zucco, P. C. Antunes, and P. G. Crawshaw. 2010. Is it possible to individually identify mammals with no natural markings using camera-traps? A controlled case-study with lowland tapirs. Mammalian Biology-Zeitschrift für Säugetierkunde **75**:375-378.

Parker, G., S. Sundaresan, G. Chege, and T. O’Brien. 2011. Using sample aerial surveys to estimate the abundance of the endangered Grevy’s zebra in northern Kenya. African Journal of Ecology **49**:56-61.

Parr, C. L., and A. N. Andersen. 2006. Patch mosaic burning for biodiversity conservation: a critique of the pyrodiversity paradigm. Conservation biology **20**:1610-1619.

Pettit, N. E., R. H. Froend, and P. G. Ladd. 1995. Grazing in remnant woodland vegetation: changes in species composition and life form groups. Journal of Vegetation Science **6**:121-130.

Piggott, M. P., S. C. Banks, N. Stone, C. Banffy, and A. C. Taylor. 2006a. Estimating population size of endangered brush-tailed rock-wallaby (Petrogale penicillata) colonies using faecal DNA. Molecular Ecology **15**:81-91.

Piggott, M. P., S. C. Banks, and A. C. Taylor. 2006b. Population structure of brush-tailed rock-wallaby (Petrogale penicillata) colonies inferred from analysis of faecal DNA. Molecular Ecology **15**:93-105.

Pollock, K. H., and W. L. Kendall. 1987. Visibility bias in aerial surveys: a review of estimation procedures. The Journal of Wildlife Management:502-510.

Pollock, K. H., H. D. Marsh, I. R. Lawler, and M. W. Alldredge. 2006. Estimating animal abundance in heterogeneous environments: an application to aerial surveys for dugongs. Journal of Wildlife Management **70**:255-262.

Poole, W. 1975. Reproduction in the Two Species of Grey Kangaroos, Macropus Giganteus Shaw and M. Fuliginosus (Desmarest). Ii. Gestation, Parturition and Pouch Life. Australian Journal of Zoology **23**:333-353.

Porter, J., R. Kingsford, and S. Hunter. 2006. Aerial Surveys of Wetland Birds in Eastern Australia. Department of Environment and Conservation NSW, Hurstville.

Preen, A., and H. Marsh. 1995. Response of dugongs to large-scale loss of seagrass from Hervey Bay, Queensland Australia. Wildlife research **22**:507-519.

Price, J. N., and J. W. Morgan. 2008. Woody plant encroachment reduces species richness of herb-rich woodlands in southern Australia. Austral Ecology **33**:278-289.

Prober, S., and K. Thiele. 2005. Effects of the 2003 Bogong Complex fires on brush-tailed rock-wallaby habitat and Acacia doratoxylon scrubs in Little River Gorge, East Gippsland. Ecological Interactions report, Victoria.

Prober, S. M., K. R. Thiele, and I. D. Lunt. 2007. Fire frequency regulates tussock grass composition, structure and resilience in endangered temperate woodlands. Austral Ecology **32**:808-824.

Prober, S. M., K. R. Thiele, and J. Speijers. 2013. Management legacies shape decadal-scale responses of plant diversity to experimental disturbance regimes in fragmented grassy woodlands. Journal of Applied Ecology **50**:376-386.

Raffel, T. R., N. Smith, C. Cortright, and A. J. Gatz. 2009. Central place foraging by beavers (Castor canadensis) in a complex lake habitat. The American Midland Naturalist **162**:62-73.

Ranglack, D. H., and J. T. du Toit. 2015. Wild bison as ecological indicators of the effectiveness of management practices to increase forage quality on open rangeland. Ecological Indicators **56**:145-151.

Ransom, J. I. 2012. Detection Probability in Aerial Surveys of Feral Horses. The Journal of Wildlife Management **76**:299-307.

Raynor, E. J., A. Joern, and J. M. Briggs. 2014. Bison foraging responds to fire frequency in nutritionally heterogeneous grassland. Ecology.

Roberts, C. M. 2009. Marsupial grazing lawns in Tasmania: maintenance, biota and the effects of climate change. University of Tasmania.

Rodriguez, A., and J. Kouki. 2015. Emulating natural disturbance in forest management enhances pollination services for dominant Vaccinium shrubs in boreal pine-dominated forests. Forest ecology and management **350**:1-12.

Rovero, F., F. Zimmermann, D. Berzi, and P. Meek. 2013. "Which camera trap type and how many do I need?" A review of camera features and study designs for a range of wildlife research applications. Hystrix-Italian Journal of Mammalogy **24**:148-156.

Royo, A. A., and W. P. Carson. 2006. On the formation of dense understory layers in forests worldwide: consequences and implications for forest dynamics, biodiversity, and succession. Canadian Journal of Forest Research **36**:1345-1362.

Royo, A. A., R. Collins, M. B. Adams, C. Kirschbaum, and W. P. Carson. 2010. Pervasive interactions between ungulate browsers and disturbance regimes promote temperate forest herbaceous diversity. Ecology **91**:93-105.

Russell-Smith, J., P. G. Ryan, and D. C. Cheal. 2002. Fire regimes and the conservation of sandstone heath in monsoonal northern Australia: frequency, interval, patchiness. Biological Conservation **104**:91-106.

Sachro, L., W. Strong, and C. Gates. 2005. Prescribed burning effects on summer elk forage availability in the subalpine zone, Banff National Park, Canada. Journal of Environmental Management **77**:183-193.

Schnabel, Z. E. 1938. The estimation of total fish population of a lake. American Mathematical Monthly:348-352.

Schoener, T. W. 1979. Generality of the size-distance relation in models of optimal feeding. American Naturalist:902-914.

Schultz, N. L., J. W. Morgan, and I. D. Lunt. 2011. Effects of grazing exclusion on plant species richness and phytomass accumulation vary across a regional productivity gradient. Journal of Vegetation Science **22**:130-142.

Schultz, N. L., N. Reid, G. Lodge, and J. T. Hunter. 2014. Seasonal and interannual variation in vegetation composition: Implications for survey design and data interpretation. Austral Ecology **39**:755-766.

Schumacher, F., and R. Eschmeyer. 1943. The estimation of fish populations in lakes and ponds. Journal of the Tennessee Academy of Science **18**.

Schwarz, C. J., and G. A. Seber. 1999. Estimating animal abundance: review III. Statistical Science:427-456.

Senft, R., M. Coughenour, D. Bailey, L. Rittenhouse, O. Sala, and D. Swift. 1987. Large herbivore foraging and ecological hierarchies. BioScience:789-799.

Sensenig, R. L., M. W. Demment, and E. A. Laca. 2010. Allometric scaling predicts preferences for burned patches in a guild of East African grazers. Ecology **91**:2898-2907.

Short, J. 1982. Habitat Requirements of the Brush-Tailed Rock-Wallaby, Petrogale penicillata, in New South Wales. Wildlife research **9**:239-246.

Short, J. 1989. The diet of the brush-tailed rock-wallaby in New-South-Wales. Wildlife research **16**:11-18.

Short, J., and P. Bayliss. 1985. Bias in aerial survey estimates of kangaroo density. Journal of Applied Ecology:415-422.

Short, J., and J. Hone. 1988. Calibrating aerial surveys of kangaroos by comparison with drive counts. Wildlife research **15**:277-284.

Short, J., and G. Milkovits. 1990. Distribution and status of the brush-tailed rock-wallaby in south-eastern Australia. Wildlife research **17**:169-179.

Silveira, L., A. T. Jacomo, and J. A. F. Diniz-Filho. 2003. Camera trap, line transect census and track surveys: a comparative evaluation. Biological conservation **114**:351-355.

Smith, B., and J. B. Wilson. 1996. A consumer's guide to evenness indices. Oikos:70-82.

Sommer, U. 1999. The impact of herbivore type and grazing pressure on benthic microalgal diversity. Ecology Letters **2**:65-69.

Southwell, C. 1994. Evaluation of walked line transect counts for estimating macropod density. The Journal of Wildlife Management:348-356.

Southwell, C. 2005. Response behaviour of seals and penguins to helicopter surveys over the pack ice off East Antarctica. Antarctic Science **17**:328-334.

Southwell, C., D. Borchers, C. G. M. Paxton, L. Burt, and W. de la Mare. 2007. Estimation of Detection Probability in Aerial Surveys of Antarctic Pack-Ice Seals. Journal of Agricultural, Biological, and Environmental Statistics **12**:41-54.

Southwell, C., M. Fletcher, and T. Scotney. 1986. Improving the aerial survey method for monitoring kangaroo populations in Queensland: results of preliminary investigations. Unpublished report prepared for the Queensland National Parks and Wildlife Service, Brisbane.

Southwell, C., and P. Jarman. 1987. Macropod studies at Wallaby Creek. 3. The effect of fire on pasture utilization by macropodids and cattle. Wildlife research **14**:117-124.

Southwell, C., K. E. Weaver, S. C. Cairns, A. Pople, A. N. Gordon, N. W. Sheppard, and R. Broers. 1995. Abundance of macropods in north-eastern New South Wales, and the logistics of broad-scale ground surveys. Wildlife research (East Melbourne).

Stankowich, T., and D. T. Blumstein. 2005. Fear in animals: a meta-analysis and review of risk assessment. Proceedings of the Royal Society of London B: Biological Sciences **272**:2627-2634.

Stephens, D. W., and J. R. Krebs. 1986. Foraging theory. Princeton University Press.

Stoll, R. J., M. W. McClain, J. C. Clem, and T. Plageman. 1991. Accuracy of helicopter counts of white-tailed deer in western Ohio farmland. Wildlife Society Bulletin:309-314.

Styger, J. K., J. B. Kirkpatrick, J. O. N. Marsden-Smedley, and S. W. J. Leonard. 2011. Fire incidence, but not fire size, affects macropod densities. Austral Ecology **36**:679-686.

Taggart, D. M., P. & Lunney, D. 2008. *Petrogale penicillata,* The IUCN Red List of Threatened Species. Version 2015.

Tasker, E. M., and R. A. Bradstock. 2006. Influence of cattle grazing practices on forest understorey structure in north-eastern New South Wales. Austral Ecology **31**:490-502.

Telfer, W. R., A. D. Griffiths, and D. M. J. S. Bowman. 2006. Scats can reveal the presence and habitat use of cryptic rock-dwelling macropods. Australian Journal of Zoology **54**:325-334.

Thomas, J. 2003. Fire management in Oxley Wild Rivers National Park: A review and synthesis of available data. NSW National Parks and Wildllife Service.

Tozer, M., and R. Bradstock. 2003. Fire-mediated effects of overstorey on plant species diversity and abundance in an eastern Australian heath. Plant Ecology **164**:213-223.

Tracey, J. P., and P. J. Fleming. 2007. Behavioural responses of feral goats (Capra hircus) to helicopters. Applied Animal Behaviour Science **108**:114-128.

Tracey, J. P., P. J. Fleming, and G. J. Melville. 2005. Does variable probability of detection compromise the use of indices in aerial surveys of medium-sized mammals? Wildlife research **32**:245-252.

Tracey, J. P., P. J. Fleming, and G. J. Melville. 2008. Accuracy of some aerial survey estimators: contrasts with known numbers. Wildlife research **35**:377-384.

Trainor, C., and J. Woinarski. 1994. Responses of lizards to three experiments fires in the savanna forests of Kakadu National Park. Wildlife research **21**:131-147.

Tremont, R. 1994. Life-history attributes of plants in grazed and ungrazed grasslands on the Northern Tablelands of New South Wales. Australian Journal of Botany **42**:511-530.

Tremont, R., and S. McIntyre. 1994. Natural grassy vegetation and native forbs in temperate Australia: structure, dynamics and life-histories. Australian Journal of Botany **42**:641-658.

Trolle, M., and M. Kéry. 2003. Estimation of ocelot density in the Pantanal using capture-recapture analysis of camera-trapping data. Journal of Mammalogy **84**:607-614.

Tuft, K. 2005. Brush-tailed rock-wallabies (Petrogale penicillata) in the Warrumbungle National Park: potential for dietary competition with sympatric macropods. B. Sc. Honours) Thesis, University of Sydney.

Tuft, K., M. Crowther, K. Connell, S. Müller, and C. McArthur. 2011. Predation risk and competitive interactions affect foraging of an endangered refuge-dependent herbivore. Animal Conservation **14**:447-457.

Tuft, K. D., M. S. Crowther, and C. McArthur. 2012. Fire and grazing influence food resources of an endangered rock-wallaby. Wildlife research **39**:436-445.

Tulloch, A., and C. Dickman. 2007. Effects of food and fire on the demography of a nectar-feeding marsupial: a field experiment. Journal of Zoology **273**:382-388.

Underwood, R. J., and P. Christensen. 1981. Forest fire management in Western Australia. Forests Department of Western Australia.

Van de Vijver, C., P. Poot, and H. Prins. 1999. Causes of increased nutrient concentrations in post-fire regrowth in an East African savanna. Plant and Soil **214**:173-185.

Van der Wal, R., H. van Wijnen, S. van Wieren, O. Beucher, and D. Bos. 2000. On facilitation between herbivores: how brent geese profit from brown hares. Ecology **81**:969-980.

Vermeulen, C., P. Lejeune, J. Lisein, P. Sawadogo, and P. Bouché. 2013. Unmanned aerial survey of elephants. PloS one **8**:1-7.

Vernes, K. 1999. Pellet counts to estimate density of a rainforest kangaroo. Wildlife Society Bulletin:991-996.

Vernes, K., M. Freeman, and B. Nesbitt. 2009. Estimating the density of free-ranging wild horses in rugged gorges using a photographic mark-recapture technique. Wildlife research **36**:361-367.

Vernes, K., S. Green, and P. Thomas. 2011. Estimating brush-tailed rock-wallaby population size using individual animal recognition. Australian Mammalogy **33**:228-234.

Walther, F. R. 1969. Flight behaviour and avoidance of predators in Thomson's gazelle (Gazella thomsoni Guenther 1884). Behaviour **34**:184-220.

Wang, S. W., and D. W. Macdonald. 2009. The use of camera traps for estimating tiger and leopard populations in the high altitude mountains of Bhutan. Biological conservation **142**:606-613.

Watson, P. 2001. The role and use of fire for biodiversity conservation in south-east Queensland: Fire management guidelines derived from ecological research. SEQ Fire & Biodiversity Consortium.

Weisenberger, M. E., P. R. Krausman, M. C. Wallace, D. W. D. Young, and O. E. Maughan. 1996. Effects of Simulated Jet Aircraft Noise on Heart Rate and Behavior of Desert Ungulates. The Journal of Wildlife Management **60**:52-61.

Whalley, R., G. Robinson, and J. Taylor. 1978. General effects of management and grazing by domestic livestock on the rangelands of the Northern Tablelands of New South Wales. The Rangeland Journal **1**:174-190.

Whelan, R., L. Rodgerson, C. R. Dickman, and E. F. Sutherland. 2002. Critical life cycles of plants and animals: developing a process-based understanding of population changes in fire-prone landscapes. Flammable Australia: the fire regimes and biodiversity of a continent:94-124.

While, G. M., and C. McArthur. 2005. Foraging in a risky environment: a comparison of Bennett's wallabies Macropus rufogriseus rufogriseus (Marsupialia: Macropodidae) and red-bellied pademelons Thylogale billiardierii (Marsupialia: Macropodidae) in open habitats. Austral Ecology **30**:756-764.

Wikum, D., and G. F. Shanholtzer. 1978. Application of the Braun-Blanquet cover-abundance scale for vegetation analysis in land development studies. Environmental Management **2**:323-329.

Williams, R. J., C. H. WAHREN, R. A. Bradstock, and W. J. Müller. 2006. Does alpine grazing reduce blazing? A landscape test of a widely-held hypothesis. Austral Ecology **31**:925-936.

Wilsey, B. 1996. Variation in use of green flushes following burns among African ungulate species: the importance of body size. African Journal of Ecology **34**:32-38.

Woinarski, J. 1990. Effects of fire on the bird communities of tropical woodlands and open forests in northern Australia. Australian Journal of Ecology **15**:1-22.

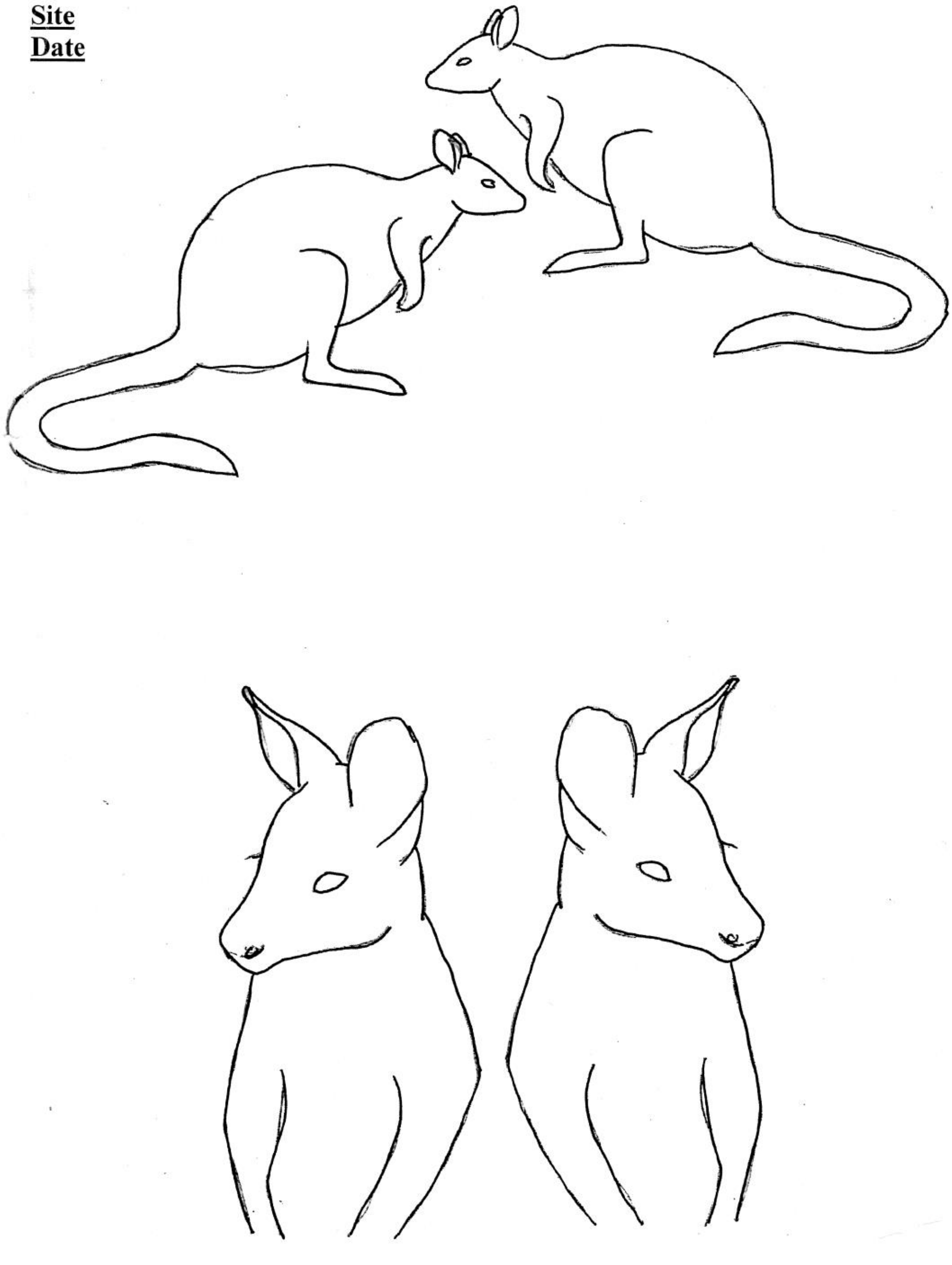
Woinarski, J., C. Brock, A. Fisher, D. Milne, and B. Oliver. 1999. Response of birds and reptiles to fire regimes on pastoral land in the Victoria River District, Northern Territory. The Rangeland Journal **21**:24-38.

Woolfolk, J. S., E. F. Smith, R. R. Schalles, B. E. Brent, L. H. Harbers, and C. E. Owensby. 1975. Effects of nitrogen fertilization and late-spring burning of bluestem range on diet and performance of steers. Journal of Range Management:190-193.

York, A. 2000. Long-term effects of frequent low-intensity burning on ant communities in coastal blackbutt forests of southeastern Australia. Austral Ecology **25**:83-98.

APPENDIX 1

Example of brush-tailed rock-wallaby template utilised during SLR individual recognition



APPENDIX 2

Species list for all taxa represented in metadataset (T represents time sampling period)

				Sample Frequency					Ave % Cover				
Species	Annual/ Perennial	Life Form	Native/ Introduced	T1	T2	T3	T4	T5	T1	T2	T3	T4	T5
Abutilon oxycarpum	Perennial	Shrub	Native	1	0	8	3	1	0.1	0.0	0.1	0.1	2.5
Ajuga australis	Perennial	Herb	Native	0	0	1	1	0	0.0	0.0	0.1	0.1	0.0
Allocasuarina torulosa	Perennial	Tree	Native	1	1	1	1	1	2.50	2.50	2.5	2.5	2.5
Aristida jerichoensis	Perennial	Grass	Native	33	29	33	34	34	26.8	12.6	15.01	16.5	14.8
Arthropodium milleflorum	Perennial	Lilly	Native	1	1	3	1	0	0.1	2.5	1.7	2.5	0.0
Asperula conferta	Perennial	Herb	Native	4	5	7	7	8	0.7	0.8	0.4	0.1	0.1
Austrodanthonia racemosa	Perennial	Grass	Native	1	0	1	1	1	0.1	0.0	0.1	0.0	0.0
Austrostipa scabra	Perennial	Grass	Native	1	0	0	0	0	0.1	0.00	0.0	0.0	0.0
Bidens pilosa	Perennial	Browse	Introduced	4	1	2	0	0	0.1	2.50	1.3	0.0	0.0
Boerhavia dominii	Perennial	Herb	Native	0	3	6	6	1	0.0	0.10	0.5	0.1	2.5
Bothriochloa macra	Perennial	Grass	Native	6	1	4	4	2	0.9	0.10	0.7	0.1	1.3
Brachypodium sp.	Perennial	Grass	Native	0	0	1	1	1	0.0	0.0	0.1	0.0	0.0
Brachyscome nova-anglica	Perennial	Herb	Native	1	0	1	0	0	0.0	0.0	0.1	0.0	0.0
Bromus catharticus	Annual	Grass	Introduced	2	0	0	0	0	1.3	0.0	0.0	0.0	0.0
Brunonia australis	Perennial	Herb	Native	0	0	0	0	0	0.7	0.1	0.0	0.0	0.0
Brunoniella australis	Perennial	Herb	Native	7	11	16	13	17	0.1	0.6	0.7	0.5	1.0
Bursaria spinosa	Perennial	Shrub	Native	1	0	1	1	0	0.1	0.1	0.1	0.1	0.0
Calandrinia sp.	Perennial	Herb	Native	0	0	1	0	0	0.0	0.0	0.1	0.0	0.0
Calotis cuneifolia	Perennial	Herb	Native	0	0	1	0	1	0.0	0.0	0.1	0.0	2.5
Carex inversa	Perennial	Herb	Native	2	0	0	0	0	0.1	0.0	0.0	0.0	0.0
Cenchrus caliculatus	Perennial	Grass	Native	1	0	0	1	0	0.1	0.0	0.0	0.1	0.0

Chamaesyce drummondii	Perennial	Browse	Native	0	3	14	5	9	0.0	0.5	0.3	0.5	0.3
Cheilanthes sp.	Perennial	Fern	Native	9	4	12	15	9	0.1	0.7	0.5	0.7	0.7
Chenopodium	Perennial	Browse	Native	0	1	3	1	0	0.0	0.0	1.3	0.1	0.0
Chrysocephalum apiculatum	Perennial	Herb	Native	2	0	2	1	1	0.1	0.0	0.1	0.1	0.1
Cissus hypoglauca	Perennial	Vine	Native	1	0	0	0	0	0.0	0.0	0.0	0.0	0.0
Commelina cyanea	Perennial	Herb	Native	0	1	6	6	3	0.0	0.1	0.6	0.5	0.1
Conyza bonariensis	Annual	Herb	Introduced	3	0	0	0	0	0.1	0.0	0.0	0.0	0.0
Cymbopogon refractus	Perennial	Herb	Native	21	18	19	22	19	10.	12.0	14.1	14.9	15.7
Cynodon dactylon	Perennial	Herb	Native	8	5	21	18	25	0.1	0.3	0.4	0.7	1.0
Cyperus gracilis	Perennial	Herb	Native	35	35	36	36	36	0.0	0.0	0.0	0.0	0.0
Desmodium varians	Perennial	Browse	Native	8	2	4	9	7	1.7	2.2	2.3	2.1	2.2
Dianella revoluta	Perennial	Lilly	Native	5	0	8	7	3	0.1	0.1	0.1	1.0	1.8
Dichanthium sericeum	Perennial	Grass	Native	5	1	3	0	4	0.5	0.0	6.3	2.9	14.1
Dichelachne micrantha	Perennial	Grass	Native	17	19	23	23	22	0.1	0.1	1.3	1.3	0.7
Dichondra repens	Perennial	Herb	Native	1	0	0	0	0	1.2	1.1	1.0	1.0	1.8
Digitaria brownii	Perennial	Grass	Native	0	0	1	1	1	0.1	0.0	0.0	0.0	0.0
Echinopogon caespitosus	Perennial	Grass	Native	7	4	5	7	4	0.0	0.0	0.1	0.0	0.0
Eragrostis leptostachya	Perennial	Grass	Native	1	0	0	0	0	7.9	0.1	1.0	2.5	1.7
Eremophila debilis	Perennial	Shrub	Native	0	0	0	0	0	0.0	0.0	0.0	0.0	0.0
Eucalyptus caliginosa	Perennial	Tree	Native	8	5	5	10	0	0.0	0.0	0.0	0.0	0.0
Eucalyptus cameronii	Perennial	Tree	Native	0	0	0	0	0	0.1	2.5	2.5	2.5	2.5
Eucalyptus campanulata	Perennial	Tree	Native	7	5	5	1	8	0.0	0.0	0.0	0.0	0.0
Eucalyptus melliodora	Perennial	Tree	Native	0	0	0	2	5	0.1	2.5	2.5	2.5	0.0
Eucalyptus tereticornis	Perennial	Tree	Native	0	0	2	0	0	0.0	2.5	2.5	2.5	2.5
Evolvulus sp.	Perennial	Herb	Native	3	1	3	4	2	0.0	0.0	0.1	0.0	0.1
Geranium solanderi	Perennial	Herb	Native	4	18	22	2	16	0.1	0.1	0.1	0.1	0.1
Glossocardia bidens	Perennial	Browse	Native	6	0	6	0	0	0.1	0.3	0.3	0.1	0.8
Glycine clandestina	Perennial	Vine	Native	1	0	2	1	1	0.1	0.1	0.1	0.0	0.0
Gomphocarpus fruticosus	Perennial	Shrub	Introduced	1	5	1	2	4	0.1	0.1	0.1	0.1	2.5

Hibbertia obtusifolia	Perennial	Shrub	Native	1	1	3	2	0	0.1	0.1	0.1	0.1	0.1
Hypericum japonicum	Perennial	Herb	Native	1	4	2	1	2	0.1	0.1	0.1	0.1	0.0
Jasminum lineare	Perennial	Shrub	Native	0	1	9	1	2	0.1	0.1	0.1	0.1	0.1
Lagenifera stipitata	Perennial	Herb	Native	1	0	5	3	2	0.0	1.3	0.6	0.1	2.5
Lespedeza juncea	Perennial	Browse	Native	1	0	0	0	0	0.0	0.0	0.5	0.1	2.5
Leucopogon lanceolatus	Perennial	Shrub	Native	0	0	0	0	0	0.1	0.0	0.0	0.0	0.0
Lomandra multiflora	Perennial	Herb	Native	13	13	18	20	13	2.1	2.3	1.8	2.1	2.5
Mentha diemenica	Perennial	Herb	Native	1	3	1	1	1	2.5	0.1	2.5	2.5	2.5
Microlaena stipoides	Perennial	Grass	Native	1	3	0	0	0	0.1	0.1	0.0	0.0	0.0
Opercularia aspera	Perennial	Browse	Native	8	1	1	1	0	0.4	0.1	0.0	0.1	0.0
Orchid sp.	Perennial	Orchid	Native	1	0	5	2	0	0.1	0.0	0.1	0.1	0.0
Oxalis perennans	Perennial	Herb	Native	1	3	9	3	2	0.0	0.1	0.1	0.1	0.1
Panicum simile	Perennial	Herb	Native	2	0	1	1	0	0.1	0.0	0.1	0.0	0.0
Paspalidium gracile	Perennial	Grass	Native	1	0	5	1	1	0.1	0.0	0.1	0.1	0.1
Phyllanthus sp.	Perennial	Shrub	Native	0	0	3	1	1	0.0	0.0	0.1	0.0	0.0
Pimelea neo-anglica	Perennial	Shrub	Native	1	0	0	0	0	0.1	0.0	0.0	0.0	0.0
Poa sieberiana	Perennial	Grass	Native	5	3	3	10	13	1.3	0.1	0.1	1.0	0.4
Podolepis neglecta	Perennial	Herb	Native	0	0	1	0	4	0.0	0.0	0.1	0.0	1.0
Pomax umbellata	Perennial	Browse	Native	1	5	6	2	1	2.5	0.9	0.5	1.3	2.5
Pratia purpurascens	Perennial	Herb	Native	4	6	4	9	7	0.9	0.5	0.1	0.1	0.4
Ranunculus lappaceus	Perennial	Herb	Native	1	1	2	3	3	0.1	0.1	0.1	0.1	0.1
Rostellularia adscendens	Perennial	Browse	Native	4	0	10	0	4	0.1	0.1	0.5	0.0	0.1
Scleranthus biflorus	Perennial	Herb	Native	1	0	0	0	0	0.1	0.0	0.0	0.0	0.0
Scleria mackaviensis	Perennial	Herb	Native	19	24	29	31	29	1.4	0.7	0.6	1.1	1.1
Senecio pinnatifolius	Perennial	Herb	Native	6	0	0	0	0	0.1	0.0	0.0	0.0	0.0
Sorghum leiocladum	Perennial	Grass	Native	3	3	9	7	8	11.	25.0	28.0	25.0	29.3
Sporobolus creber	Perennial	Grass	Native	6	0	12	6	5	0.5	0.9	2.5	1.7	2.5
Stackhousia viminea	Perennial	Herb	Native	2	13	22	15	19	0.0	0.1	0.2	0.7	1.8
Stellaria flaccida	Perennial	Herb	Native	0	0	0	0	2	0.0	0.0	0.0	0.0	0.1

Themeda triandra	Perennial	Grass	Native	26	22	27	28	28	53.1	39.4	36.3	35.5	37.6
Vernonia cinerea	Perennial	Herb	Native	5	15	23	13	23	0.0	0.2	0.3	0.2	0.3
Veronica calycina	Perennial	Herb	Native	0	0	1	4	0	0.0	0.0	0.0	0.0	0.1
Vittadinia cuneata	Perennial	Browse	Native	0	0	2	0	0	0.0	0.1	0.1	0.	0.0
Wahlenbergia stricta	Perennial	Herb	Native	8	4	11	12	14	0.4	0.4	0.9	0.9	1.1
Xerochrysum bracteatum	Perennial	Browse	Native	0	0	0	2	0	0.0	0.0	0.0	0.0	0.0
Zornia sp.	Perennial	Herb	Native	0	0	2	1	0	0.0	0.0	1.3	2.5	0.0