Understanding the causes of human–wombat conflict and exploring non-lethal damage mitigation strategies for the southern hairy-nosed wombat (*Lasiorhinus latifrons*)

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Declaration

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I acknowledge the support I have received for my research through the provision of an Australian Government Research Training Program Scholarship.

Signature: _ _

Date: __18/04/19____

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Abstract

Human–wildlife conflict is a widespread and growing threat to conservation worldwide. It encompasses a wide range of problems and negatively impacts a large diversity of species, with far-reaching environmental, social, economic, political, health, and safety outcomes. Effective conflict management requires an integrative approach, encompassing the ecological and human aspects of the problem. For many species like the southern hairy-nosed wombat (*Lasiorhinus latifrons*), there is insufficient data to make informed and effective conflict management decisions. Conflicts between *L. latifrons* and the agricultural sector have been ongoing for decades, because of damage caused by their burrowing behaviour. Culling is used to manage damages, but it fails to provide long-term relief from conflict and raises ethical and conservation concerns. Landholders have expressed support for the development of non-lethal damage mitigation strategies in the past, but little research has been done to quantify the effectiveness of such measures. This thesis investigates landholder perceptions of human–wombat conflict and assesses the effectiveness of potential non-lethal damage mitigation techniques, to improve *L. latifrons* management.

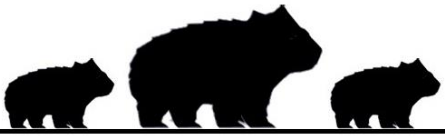
The perceptions of landholders living throughout the range of *L. latifrons* were surveyed using a mail questionnaire. Survey questions aimed to gather a variety of information on landholder opinions of *L. latifrons*, the causes and costs of damage and the species' management. Surveys distributed by mail in the Murraylands received a 3.2% (*n* = 122), response rate, while those distributed at wombat workshops in the Far West received an estimated 41.0% (*n* = 33) response rate. Of the respondents with *L. latifrons* on their properties, 81.2% reported damage, mainly as a result of burrowing. Despite this, there is strong support for *L. latifrons* conservation, though support decreased among respondents with *L. latifrons* present and those who were financially dependent on their properties. To improve management, many respondents suggested the development of alternative non-lethal management options, such as translocation.

Translocation provides the opportunity to reduce conflicts while restoring declining populations of *L. latifrons*. Animals were translocated from conflict zones to a private grazing property where the species had declined following a drought. GPS and VHF monitoring revealed translocated *L. latifrons* displayed high site fidelity, though they initially ranged further than their resident counterparts. However, animals were difficult to capture and translocation failed to reduce conflicts, as neighbouring *L. latifrons* quickly recolonised vacated burrows. These findings indicate translocation is not suitable for reducing human–wombat conflict, but it may prove useful for restoring declining populations of *L. latifrons*.

Deterrents may be a more cost-effective and efficient means of reducing human-wombat conflict. The effectiveness of four treatments for deterring free-living *L. latifrons* from their burrows was assessed: dingo (*Canis lupus dingo*) urine; dingo faeces; blood and bone (Brunnings Pty Ltd); and compact discs (CDs). Remote cameras monitored *L. latifrons* behaviour before and after the treatments were applied. The number of visits to burrows decreased significantly following the application of CDs. No other treatments deterred *L. latifrons*. This research suggests that visual cues may be more effective than olfactory ones in deterring *L. latifrons*. However, responses to threats can be context dependent. Further research into the use of deterrents in different contexts is needed to gain a better understanding of how *L. latifrons* detect and respond to threats.

Chapter 1. General Introduction

A southern hairy-nosed wombat basking in the afternoon sun



1.1 Human–wildlife conflict

Human–wildlife conflicts (HWCs) have occurred for many millennia, but are becoming an issue of increasing concern in the 21st century (Conover 2002; Woodroffe *et al.* 2005b; Madden and McQuinn 2014). Growing human civilisations are transforming the world's landscapes, dramatically altering ecosystem services and modifying habitats (Houghton *et al.* 1999; Bennett *et al.* 2001; Gleick 2003; Foley *et al.* 2005). Large scale land clearance is driving habitat loss, increasingly restricting the world's biodiversity to small, fragmented patches within human-dominated landscapes (McCloskey and Spalding 1989; Hansen *et al.* 2013). Over half of the earth's land mass has been converted to human-dominated use (Watson *et al.* 2016), and an estimated 83% is affected by human activities (Sanderson *et al.* 2002). The expansion of human activities into the remotest places in the world has resulted in a greater overlap in the resource requirements of humans and wildlife. This leads to greater competition between humans and wildlife over declining space and resources, resulting in increased incidence of HWC.

Human–wildlife conflicts occur when the actions of either humans or wildlife have a detrimental effect on the other. Conflicts can result in a wide range of outcomes, such as the loss of life, injury to humans and wildlife, competition for resources, property damage, the spread of infectious diseases, and habitat loss and fragmentation (Woodroffe *et al.* 2005a; Sillero-Zubiri *et al.* 2007; Dunham *et al.* 2010; Wilson *et al.* 2011). Conflicts negatively affect millions of people and threaten a large diversity of wildlife worldwide, with far-reaching environmental, social, economic, health, and safety impacts. Areas with a high degree of human–wildlife overlap tend to be associated with declines in wildlife populations, thus conflicts present a genuine threat to species conservation (Woodroffe *et al.* 2005b). Considering the current human population growth rate (United Nations Department of Economics and Social Affairs Population Division (UNDESAPD) 2017), increasing demand for resources, and access to land, it is clear that HWC will not be overcome in the near future. Conflict resolution will become increasingly important to foster environmental and production sustainability, and maximise wildlife and human wellbeing.

1.2 Conflicts with agriculture

Agriculture is one of the most extensive terrestrial land uses on the planet, occupying ~ 40% of ice-free land (Ramankutty and Foley 1999; Foley *et al.* 2005). Few human land uses have as great an impact on wildlife resources or as much contact with wildlife as agriculture. It is one of the most prevalent threats facing the conservation of wildlife, with more than 60% of threatened or near threatened species on the International Union for Conservation of Nature (IUCN) Red List affected by agriculture (Maxwell *et al.* 2016). Conversion of land for agricultural purposes is one of the chief drivers of habitat loss and degradation (Ramankutty and Foley 1999; Watson *et al.* 2016). Many ecosystems and biomes have fallen to food production (Hoekstra *et al.* 2005). The loss of previously wild habitats has restricted biodiversity to small fragmented patches within agricultural landscapes, increasing the potential for conflicts between humans and wildlife, as they clash over space and resources. Conflicts involve a diverse range of species and encompass a wide range of problems that negatively affect millions of people and threaten a considerable diversity of species worldwide, with varying degrees of severity. Carnivores depredate livestock and in extreme cases can cause loss of human lives (Stahl *et al.* 2002; Treves and Karanth 2003; Inskip and Zimmermann 2009). Herbivores consume crops, compete with livestock for food, and damage property (Ramp and Roger 2008; Delahay *et al.* 2009; Hockings 2009; Hedges and Giunaryadi 2010; Nugent *et al.* 2011). Increased contact between livestock, humans, and wildlife can lead to the transmission of infectious disease (Vercauteren *et al.* 2008; Wilson *et al.* 2011). Humans fragment and alter habitats, and decrease the productivity of natural ecosystems through overgrazing and overcropping.

Wildlife damage can have severe economic impacts, affecting the viability of farms and increasing food costs for consumers (Conover 1997; Naughton-Treves and Treves 2005). In Australia for instance, crop grazing, competition with livestock for food resources, and fence damage by kangaroos (*Macropus* spp.) are estimated to cost the agricultural industry AU$44 m per annum (McLeod and Norris 2004). Dingoes and dingo-dog hybrids (*Canis lupus dingo*) cost the Australian economy ~AU$48.3 m each year, although much of this cost is for controlling dingoes rather than in livestock losses (Allen and West 2013). On a smaller landholder scale in the United States (US), wildlife damage to crops resulted in a 72% loss in value per acre over a 6-month period (Drake and Grande 2002). The burden of damage management typically falls on private landowners, adding further to the cost of conflicts. Wildlife can pose health risks to humans, livestock, and pets when they act as hosts of infectious disease (Daszak *et al.* 2000). The spread of bovine tuberculosis to cattle has been linked to badgers (*Meles meles*) (Wilson *et al.* 2011) and white-tailed deer (*Odocoileus virginianus*) (Vercauteren *et al.* 2008). In extreme circumstances conflicts can result in injuries or loss of human life, creating fear among people (Woodroffe *et al.* 2005a). In Mozambique alone, wildlife was responsible for killing 265 people in a 27-month period, with crocodiles accounting for 66% of deaths (Dunham *et al.* 2010). Conflicts can also have hidden psychological and social impacts, resulting from injury or fatality, loss of livelihoods, and food security, including missed school or work, loss of sleep, fear, and restriction of travel (Barua *et al.* 2013).

Humans typically respond to conflicts with the retaliatory killing of wildlife, via trapping, poisoning, and shooting. These methods often carry substantial animal welfare costs and raise conservation concerns (Treves *et al.* 2016). Worldwide, 47% of cheetah (*Acinonyx jubatus*), 46% of Eurasian lynx (*Lynx lynx*), and 50% of tiger (*Panthera leo*) deaths are attributed to retaliatory killing (Inskip and Zimmermann 2009). Governments have sponsored nationwide eradication schemes for species perceived as highly problematic, such as prairie dogs (*Cynomys* spp.) in the US (Reading *et al.* 2005). Large-scale lethal control programs have eliminated wolf, bear, and lynx populations throughout Western Europe (Linnell *et al.* 1996), and are thought to have contributed to the extinction of the thylacine (*Thylacinus cynocephalus*) in Australia (Guiler 1985). Lethal control can have devastating impacts on wildlife, contributing to species declines and range contractions (Abbott 1933; Fuller *et al.* 1992; Woodroffe *et al.* 2005b). Following widespread persecution, African wild dogs (*Lycaon pictus*) remain in only 14 of the 39 countries they once occupied and number fewer than 5,000 individuals (Ginsberg *et al.* 1997). In extreme circumstances, lethal control has been blamed for species extinctions, including the Carolina parakeet (*Conuropsis carolinensis*) (Powell 2006) and the Falklands wolf (*Dusicyon australis*) (Sillero-Zubiri et al. 2004).

If the global trend in human population growth continues, it is expected that 9.8 billion people will inhabit the earth by 2050 (UNDESAPD 2017). To meet the rise in global food demand, agricultural production is expected to increase by 60–100%, requiring an additional 34,000,000 km^2 of farmland, an area larger than Africa (Tilman *et al.* 2011; Alexandratos and Bruinsma 2012). Conflict between agriculturalists and wildlife will likely increase in the near future. The resolution of conflicts on agricultural land will become increasingly important to ensure species conservation, food security and production sustainability.

1.3 Conflict management

The management of HWC is complex. It encompasses a broad range of environmental, health, economic, political, social, and animal welfare impacts that are unique to each conflict situation. Because of the diversity of causes and circumstances in which conflicts occur, no single management procedure will effectively reduce HWC. The most effective approach is likely to be an integrative one, informed not only by ecological science but also the human aspects of conflict situations. Dubois *et al.* (2017) recommends seven stepwise principals to ensure wildlife is managed in the most ethical, socially acceptable, effective, and efficient way possible. These include modifying human behaviours where possible, justifying the need for control, ensuring the objective is achievable, causing the least harm to wildlife, considering community values and scientific evidence, using systematic long term planning, and warranting decisions. For many conflict situations, the necessary information to make informed and effective management decisions are lacking. Deficiencies in knowledge include (1) the perceptions of stakeholder groups, (2) actual versus perceived economic losses, (3) information on the timing and distribution of conflicts, and (4) quantitative data on the effectiveness of mitigation strategies in the field. Better monitoring should be a priority to improve the available data to underpin all stages of the management decision process.

1.3.1 Human perceptions

Although wildlife damage is commonly cited as the primary cause of HWC, perceptions of conflict and responses to them are not always in direct proportion to actual damages (Siex and Struhsaker 1999; 2000; Madden 2004; Dickman 2010). People’s attitudes and responses towards wildlife can be influenced by cultures, societal values, economics, and personal experience (Zinn *et al.* 1998; Madden 2004; St John *et al.* 2013; Rust *et al.* 2016). Biological science alone cannot provide a complete understanding of HWC, as the underlying causes are often human driven (Conover 2002; Zimmermann *et al.* 2005). Human attitudes and values towards wildlife vary both among and within different sectors of society (Elmore *et al.* 2007; Baker *et al.* 2016). The success of programs designed to resolve HWC will rest in part on the ability of wildlife managers to understand and incorporate differing stakeholder values, attitudes, and beliefs in the policy-making process. Failure to do so can cause animosity between stakeholder groups and result in a lack of co-operation, hindering management efforts (Kellert 1981; Zinn *et al.* 1998). This has been demonstrated by the sabotage of efforts to create community-based wildlife management areas in Tanzania by locals who were not involved in the planning process and felt their needs were superseded for conservation outcomes (Igoe and Croucher 2007; Sachedina 2008).

Agriculturalists' views towards wildlife differ substantially from other stakeholder groups, which is understandable given the impact wildlife can have on agricultural production, economic viability, and livelihoods (Kellert 1981). Understanding and incorporating agriculturalists' attitudes towards wildlife into management decisions will be critical to the success or failure of wildlife management on farmland (Mascia *et al.* 2003). However, agricultural producers are increasingly being asked to incorporate public values into wildlife management while absorbing the costs of damages (McIvor and Conover 1994b), even though the public rarely understands how HWC are created or should be managed (Messmer 2000). When wildlife damage is not adequately addressed and the needs of wildlife are given priority over agriculturalists, conflicts often escalate and can lead to disagreement among stakeholders on how to manage wildlife (Madden and McQuinn 2014). Focusing research efforts on agriculturalists perceptions of damage can provide a general index of where to direct management efforts (Wywialowski and Beach 1991). Increased public awareness of the origin, ecology, and impacts of HWC may help to lessen conflicts among stakeholders.

1.3.2 Lethal control

Historically, HWC has been managed using lethal controls such as trapping, shooting, or poisoning with little regard for their impact on animal welfare of wildlife populations (Treves and Naughton-Treves 2005). Culling is often implemented in the absence of crucial ecological information including the population size and dynamics of the target species, and their flow-on effects are rarely evaluated properly (Treves *et al.* 2016). Culling conducted at unsustainable levels can result in range contractions, population isolation, and has contributed to species extinctions (Fuller *et al.* 1992; Powell 2006). Large-scale eradication campaigns have contributed to a 98% reduction in the range of black-tailed prairie dogs (*Cynomys ludovicianus*) (Miller *et al.* 1994) and are thought to be the primary cause of the extinction of the Guadalupe caracara (*Caracara lutosa*) (Abbott 1933). Removing insufficient number of animals, however, can fail to reduce conflicts to acceptable levels (Hone 2007).

Culling may encourage population growth and increased immigration in some species, perpetuating perceived conflict (King and Powell 2011; Treves *et al.* 2016; Newsome *et al.* 2017). The culling of coyotes (*Canis latrans*), for instance, disrupts pack structure and territorial behaviour, resulting in higher population growth rates than in non-culled populations (Knowlton *et al.* 1999). Similarly, culling dingoes changes the age and experience of social groups, disrupts pack structure, and increases the likelihood of livestock predation (Allen 2014). Furthermore, lethal control can have unpredictable impacts on ecosystem dynamics (Greenwood *et al.* 1995; Wallach *et al.* 2010). Hebblewhite *et al.* (2005) found the removal of wolves (*Canis lupus*) resulted in an increase in greater elk (*Cervus canadensis*) numbers, leading to increased browsing pressure on aspen and willow plants. The reduced production of aspen and willow in turn had a negative effect on riparian songbird diversity and abundance (Hebblewhite *et al.* 2005).

The use of culling as a management tool has become a highly contentious issues over recent decades (Littin *et al.* 2014). For people living in direct conflict with wildlife, culling provides the opportunity to take direct action, satisfying their need for revenge and thus reducing the likelihood of illegal culling (Horton and Craven 1997; Zinn and Andelt 1999). Culling may also benefit local communities by generating income from hunting revenue or meat production, thereby building local support for conservation efforts (Treves and Naughton-Treves 2005). However, increased awareness the limitations of lethal control have contributed to growing public concern for animal welfare and justification for the use of culling (Treves *et al.* 2016; Woodroffe *et al.* 2005b). Although substantial effort has been invested in improving the humaneness of lethal controls (Warburton and Norton 2009) wildlife managers are facing increased pressure to implement non-lethal management techniques. In many circumstances, effective and economical non-lethal techniques are not avaliable and culling remains the best management option. The use of lethal controls to reduce HWC requires careful evaluation of ecological knowledge, animal welfare, social acceptability, and ongoing monitoring to ensure conflicts are reduced without threatening species conservation.

1.3.3 **Non-lethal control**

Changing public attitudes, protective legislation, animal welfare concerns, and increasing awareness of the potential limitations of lethal wildlife control have seen a shift towards the use of non-lethal damage mitigation alternatives (Conover and Conover 2001; Shivik *et al.* 2003; Massei *et al.* 2010). Such techniques include changing human behaviours (Dubois *et al.* 2017), the translocation of problem animals (Imam *et al.* 2002; Butler *et al.* 2005b), use of deterrents (Cox *et al.* 2015), physical barriers (Okello and D'Amour 2008; Honda *et al.* 2011), fertility control (Massei and Cowan 2014), and habitat manipulation (Sadlier and Montgomery 2004). However, many of these techniques have not been adequately researched in natural environments, are impractical on a large scale, and fail to provide long-term relief from conflict (Koehler *et al.* 1990; Mason 1998; Barlow *et al.* 2010). Although they are perceived as humane, they can have negative ecological or animal welfare impacts. The use of exclusion fencing, for example, can prohibit the movement of non-target animals and disrupt migratory movements or access to resources (Hoare 1992). Evidence-based approaches that are tailored to specific ecological, social, cultural, and economic realities are needed, as one solution is unlikely to be appropriate for all circumstances (Dubois *et al.* 2017). Although much research is being done to find effective non-lethal control methods, as yet they have limited utility. There may always be some situations in which lethal control is the best intervention (Warburton and Norton 2009).

1.3.3.1. Translocation

Translocation has become one of the most widely used tools for managing HWC, particularly when dealing with protected or endangered species (International Union for Conservation of Nature Species Survival Commission (IUCN/SSC) 2013; Miller *et al.* 2014; Germano *et al.* 2015). Viewed as humane and species specific by many, it involves the deliberate movement of wild individuals or populations from one part of their range to another (Craven *et al.* 1998; Massei *et al.* 2010; IUCN/SSC 2013). Many thousands of animals are translocated annually for the purposes of conflict reduction, but, its effectiveness in resolving HWC in a humane manner is disputed (Craven *et al.* 1998; Massei *et al.* 2010). Driven by the desire to resolve conflict, rather than establish populations, mitigation translocations are often carried out in a manner that is not consistent with IUCN translocation guidelines (Germano *et al.* 2015). Important biological and ecological factors such as capture stress, disease transmission, habitat suitability, sociality, genetics, and competition with conspecifics are frequently neglected and contribute to low survival. Furthermore, the fate of translocated animals is rarely monitored over the long-term to determine if individuals establish or integrate into populations. Individuals that survive often display extensive movement and non-establishment, and suffer from malnutrition, dehydration, and immunodeficiency (Massei *et al.* 2010; Germano *et al.* 2015; Sullivan *et al.* 2015). This was documented in translocated urban raccoons, which displayed disorientation and extensive post-release movement, and suffered 50% mortality within 3 months of release as a result of starvation, road mortalities, shooting, and poisoning (Rosatte and MacInnes 1989).

Although the main aim of mitigation translocations is to reduce conflicts, its ability to do so is disputed. Many studies fail to report whether and for how long conflicts are reduced; those that do indicate that long-term conflict resolution is rare because of population growth, immigration of neighbouring animals, or the homing behaviour of translocated individuals (Phillips *et al.* 1991; Clarke and Schedvin 1997; Van Vuren *et al.* 1997; Bradley *et al.* 2005; Massei *et al.* 2010; Germano *et al.* 2015). For example, translocated wolves (*Canis lupus*) often failed to establish territories in their release location because of competition from rival wolf packs, and returned home if released within 64 km of their capture site (Fritts *et al.* 1984). Translocations also run the risk of transferring the conflict to the release site as the problem behaviour exhibited by animals is not resolved by relocating them (Linnell *et al.* 1997; Bradley *et al.* 2005). The translocation of stock-raiding lions (*Panthera leo*) often transferred the problem to the release site, with animals continuing to kill stock in their new location (Stander 1990). The use of translocation as a conflict resolution tool needs to be properly evaluated on a case-by-case basis to determine if it is a suitable intervention to reach management objectives for both humans and wildlife.

1.3.3.2. Deterrents

A wide variety of deterrents, including taste-aversion deterrents (Murray *et al.* 2006; Baker *et al.* 2008), frightening devices (Bomford and OBrien 1990; Gilsdorf *et al.* 2002), physical barriers (Honda *et al.* 2009) and painful stimuli (Andelt *et al.* 1999) are increasingly being marketed as non-lethal means of reducing HWC (Breitenmoser et al. 2005). Although there have been numerous studies on the use of deterrents, many devices have not been quantitatively field tested and their perceived effectiveness lies in testimonial and anecdotal reports (Koehler *et al.* 1990). Controlled laboratory studies often do not translate to field situations where numerous uncontrollable environmental factors can influence the manner in which animals detect and respond to threats (Koehler *et al.* 1990; Mason 1998; Barlow *et al.* 2010). Even where devices have been field tested they often produce conflicting results because of differences in survey designs or species-specific and context-dependent (e.g. differing habitat structure, individual fitness, or predator densities) responses (Lima and Dill 1990; Mason and Clark 1992; Apfelbach *et al.* 2005; Schakner and Blumstein 2013). Reflective tape, for instance, produced avoidance in blackbirds (*Agelaius phoeniceus*) in some studies (Bruggers *et al.* 1986; Dolbeer *et al.* 1986) but failed to deter them from damaging crops in others (Conover and Dolbeer 1989). Despite these differences, many methods are considered successful in reducing damages caused by wildlife (Schakner and Blumstein 2013).

Devices that successfully deter animals are often limited in their effectiveness because of habituation, with animals becoming less wary of devices that are used continuously, in predictable patterns, or when unsupported by negative consequences (Breitenmoser *et al.* 2005). Altering the positions of devices, using them sparingly, or in combination with other deterrents, may prolong their effectiveness (Koehler *et al.* 1990; Belant *et al.* 1996; Shivik *et al.* 2003; Stone *et al.* 2017). Time to habituation was extended when using motion-activated propane exploders to deter white-tailed deer from agricultural crops, compared with those exploding at predicted intervals; however, the former were only effective for a few weeks (Belant *et al.* 1996). Deterrents relying on innate responses or animal learning to become effective may have longer lasting effects (Muller-Schwarz 1974; Boag and Mlotkiewicz 1994; Gilsdorf *et al.* 2002). Woodchucks (*Marmota monax),* for instance, did not habituate to bobcat urine following repeated exposure over consecutive years (Swihart 1991). There is a clear need to tailor the use of deterrent strategies to the target species and the environment in which they occur (Nolte *et al.* 1993).

1.4 Human–wombat conflict

Following European settlement in Australia, all three Vombatid species, the common (*Vombatus ursinus*), northern hairy-nosed (*Lasiorhinus krefftii*), and southern hairy-nosed (*Lasiorhinus latifrons*) wombats have experienced conflicts with humans. Large-scale land clearance for agricultural and urban development, and the introduction of European rabbits (*Oryctolagus cuniculus*) are thought to have played a significant role in the range contractions of all three species (Figure 1.1) (Aitken 1971; Gordon *et al.* 1985; Buchan and Goldney 1998; Cooke 1998; Temby 1998; McIlroy 2008). Rabbits compete with wombats for food and shelter in their warrens. In a bid to control rabbits, wombat warrens were often gassed and destroyed (Aitken 1971; Mallett and Cooke 1986; Temby 1998; Swinbourne *et al.* 2017). The decline of the northern hairy-nosed wombat has also been attributed to competition with livestock (Crossman *et al.* 1994); it has suffered the greatest range contraction of all three species. By 1982, the northern hairy-nosed wombat was restricted to a single population of approximately 30 animals (Gordon *et al.* 1985) and listed as endangered under the *Nature Conservation Act 1992* (QLD). The common wombat was afforded no such protection, listed as vermin in Victoria by 1906, under provisions of the *Vermin Destruction Act 1890 (Vic)*, because of the damage they caused to netting fences designed to exclude rabbits from agricultural land. The government went so far as to place a bounty on the common wombat's head in 1925, which was abolished in 1971, followed by removal of the vermin label in 1984 (Temby 1998).

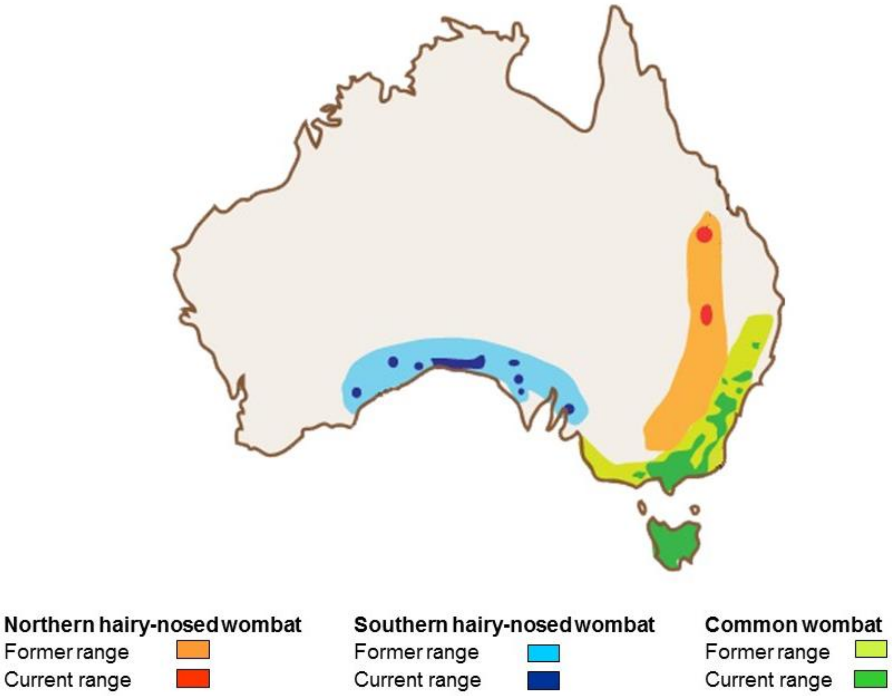


Figure 1.1. The estimated pre-European and current distributions of the three extant species of wombat in Australia (Department of Environment and Heritage Protection 2018).

1.4.1 **Southern hairy-nosed wombats**

1.4.1.1. Distribution and conservation status

Prior to European settlement, the distribution of *L. latifrons* was thought to be continuous, extending from the Murraylands in the east through to Western Australia (St John and

Saunders 1989) (Figure 1.1). However, a lack of historical records makes this difficult to confirm (Aitken 1971). Recent research has suggested that at the time of European settlement, *L. latifrons*' range was split into two main populations on either side of the Spencer Gulf (Swinbourne *et al.* 2017). Following European settlement, the range of *L. latifrons'* contracted considerably, because of the introduction of rabbits and the conversion of land for human settlement and agricultural development (Aitken 1971; Swinbourne *et al.* 2017). Currently, much of the *L. latifrons* distribution falls across agricultural land, within five genetically distinct mainland populations, on the Nullarbor Plains, Gawler Ranges, Eyre Peninsula, Yorke Peninsula, and the Murraylands (St John and Saunders 1989; Alpers *et al.* 1998) (Figure 1.2). There is also an introduced population on Wedge Island in the Spencer Gulf (St John and Saunders 1989). Due to *L. latifrons'* cryptic and burrowing nature, obtaining accurate abundance estimates is difficult. Population sizes are thought to range from 100,000 animals on the Nullarbor, to a few hundred on the Yorke Peninsula (St John and Saunders 1989; Sparrow 2009). As a whole, the species is classified as near threatened (Woinarski and Burbidge 2016); only the Nullarbor population is viewed as secure, while the highly fragmented Yorke Peninsula population is considered endangered (St John and Saunders 1989; Walker 2004; Sparrow 2009).

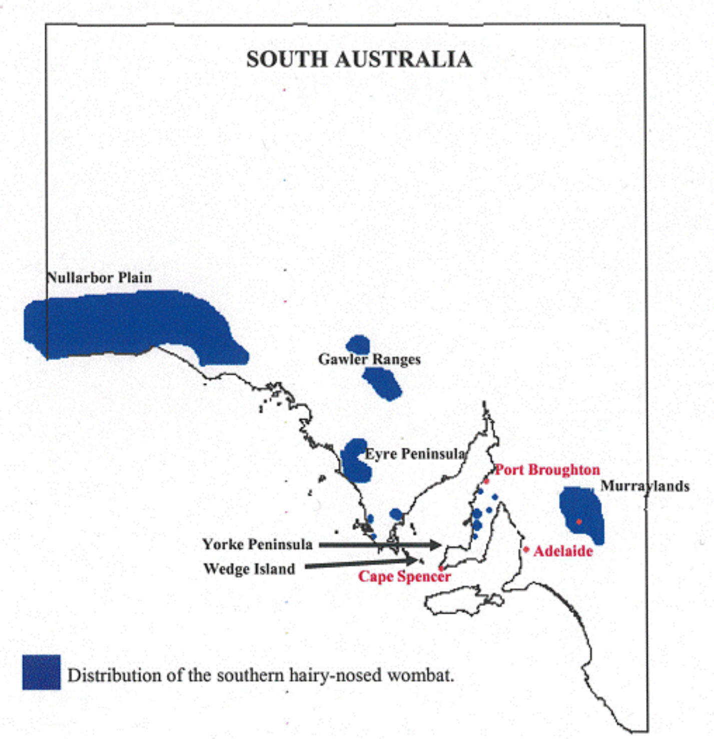


Figure 1.2. The distribution of *L. latifrons* in southern Australia (Furbank 2010).

1.4.1.2. Causes of conflict

Conflicts between agriculturalists and *L. latifrons* have been ongoing since the commencement of farming throughout the species range. In recent years, conflicts between rural landholders and *L. latifrons* have escalated, with reports that *L. latifrons* populations have expanded into new, previously uninhabited regions (Taggart *et al.* 2008). Conflicts arise as a result of the grazing and burrowing habits of this species. As one of the largest burrowing herbivores in the world, *L. latifrons* create warren complexes that vary in size from single entrance burrows to large warren complexes with up to 80 burrows and spanning up to 3,000 m^2 (Loffler and Margules 1980; Shimmin *et al.* 2002; Triggs 2009). Warrens undermine infrastructure such as roads, dams, fences, irrigation banks, windmills, and gravesites (St John and Saunders 1989; Stott 1998). Burrows can cause injury to humans, damage to vehicles, and heavy machinery when tunnels collapse under their weight (St John and Saunders 1989; Stott 1998). *Lasiorhinus latifrons* provide grazing competition for stock and consume crops, with large grazing halos evident around their warrens (Loffler and Margules 1980; St John and Saunders 1989; Stott 1998). Much of the damage that *L. latifrons* cause or are perceived to cause is yet to be quantified.

Agricultural development and human activities have negatively affected *L. latifrons.* Competition from livestock, overgrazing, and the introduction of rabbits have contributed to the decline of *L. latifrons* (Wells 1995; Swinbourne *et al.* 2017). In some regions, a loss of native grass species has led to a dietary shift towards introduced weed species that are high in toxins, severely affecting *L. latifrons* health (Woolford *et al.* 2014). Retaliatory culling to control *L. latifrons* damage is also thought to have played a role in their decline, but, its impacts are poorly understood (Tartowski and Stelmann 1998; Taggart *et al.* 2008).

1.4.1.3. Conflict management

The management of *L. latifrons* is a highly contentious issue. Problems arise when trying to strike a balance between the competing interests of farmers, conservationists, the general public, and the needs of the species. Although *L. latifrons* is protected under state and federal legislation; destruction permits can be issued by the Department of Environment and Water in circumstances where *L. latifrons* cause damage or threaten human safety (*section 53.1c of the National Parks and Wildlife Act 1972 (SA)*). The impacts of culling on *L. latifrons* populations are poorly understood, and raise ethical and conservation concerns for the species (Tartowski and Stelmann 1998; Taggart *et al.* 2008). Due to a lack of scientific data, current permit allocations are not based on evidence or knowledge of the species, making the permit system ineffective for wombat conservation and management (Taggart *et al.* 2008). Many landholders are dissatisfied with the permit system, as they feel they are rarely allowed to destroy a realistic number of animals to mitigate damage, with recolonisation of burrows being an ongoing problem (St John and Saunders 1989; Stott 1998; Taggart *et al.* 2008). Subsequently, many landholders resort to culling outside of the permit system, resulting in inadequate data on the numbers of *L. latifrons* culled and the intensity of conflicts (Taggart *et al.* 2008).

Conservation and animal welfare groups concerned about the impacts of culling on the long-term survival of *L. latifrons* populations advocate for banning lethal controls and implementing non-lethal conflict mitigation measures. To date, there has been little quantitative research on the effectiveness of non-lethal management options for *L. latifrons*. Along the Nullarbor Plain, the electrification of the dog fence, a 5,600 km fence designed to exclude dingoes from agricultural properties in south east Australia has reduced damage caused by *L. latifrons* digging behaviour (St John and Saunders 1989). However, electric fencing is not cost effective on a small-landholder scale. Effective, socially acceptable, and economically viable management options are not yet available for this species. Quantified research on the extent and impact of conflicts and the effectiveness of non-lethal conflict mitigation measures is required to guide decision makers and develop a successful management strategy that balances the needs of *L. latifrons* and landholders.

1.5 Project aims

The current management of conflicts between agriculturalists and *L. latifrons* is limited in its effectiveness and causes concern for the long-term survival of the species. Past surveys of landholder opinions of *L. latifrons* have identified support for the development of non-lethal conflict mitigation strategies (St John and Saunders 1989). Despite this, there has been little quantitative research on the effectiveness of non-lethal damage mitigation measures for this species. As opinions can change over time, this thesis seeks to reinvestigate landholder perceptions of *L. latifrons* and their management and assess the effectiveness of non-lethal conflict mitigation measures, to improve species management and foster an environment of co-existence. Specifically this thesis aims to:

1. Quantify landholder perceptions of *L. latifrons,* the damage they cause, and their management throughout the species’ range, with a focus on the impact of the

presence of *L. latifrons* on properties, experience with damage, and financial dependence on properties on people's attitudes.

2. Assess the effectiveness of translocation as a tool for resolving human-wombat conflict, by examining the effect of translocation on *L. latifrons* survival, establishment, and conflict reduction at the source site.
3. Examine the effectiveness of deterrents in reducing *L. latifrons* use of their burrows, with the aim of preventing the recolonisation of vacated burrows.

These aims are addressed across four chapters presented as stand-alone manuscripts, which are written in a style for publication following the submission of this thesis. Therefore, some repetition in the introduction and methods of these chapters is unavoidable. The analysis presented in Chapter 2 addresses landholder attitudes towards *L. latifrons* and their management. Based on the findings from Chapter 2, translocation was trialled as a non-lethal conflict mitigation measure for *L. latifrons,* as outlined in Chapter 3. Chapter 4 and Chapter 5 assess the effectiveness of deterrents in reducing *L. latifrons* use of their burrows. Chapter 6 summarises and synthesises the findings of Chapter 2–5, provides recommendations for the management of *L. latifrons*, and identifies areas of future research pertinent to improving the management of this species, and resolving human–wombat conflict.

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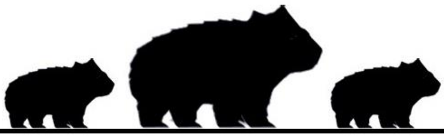
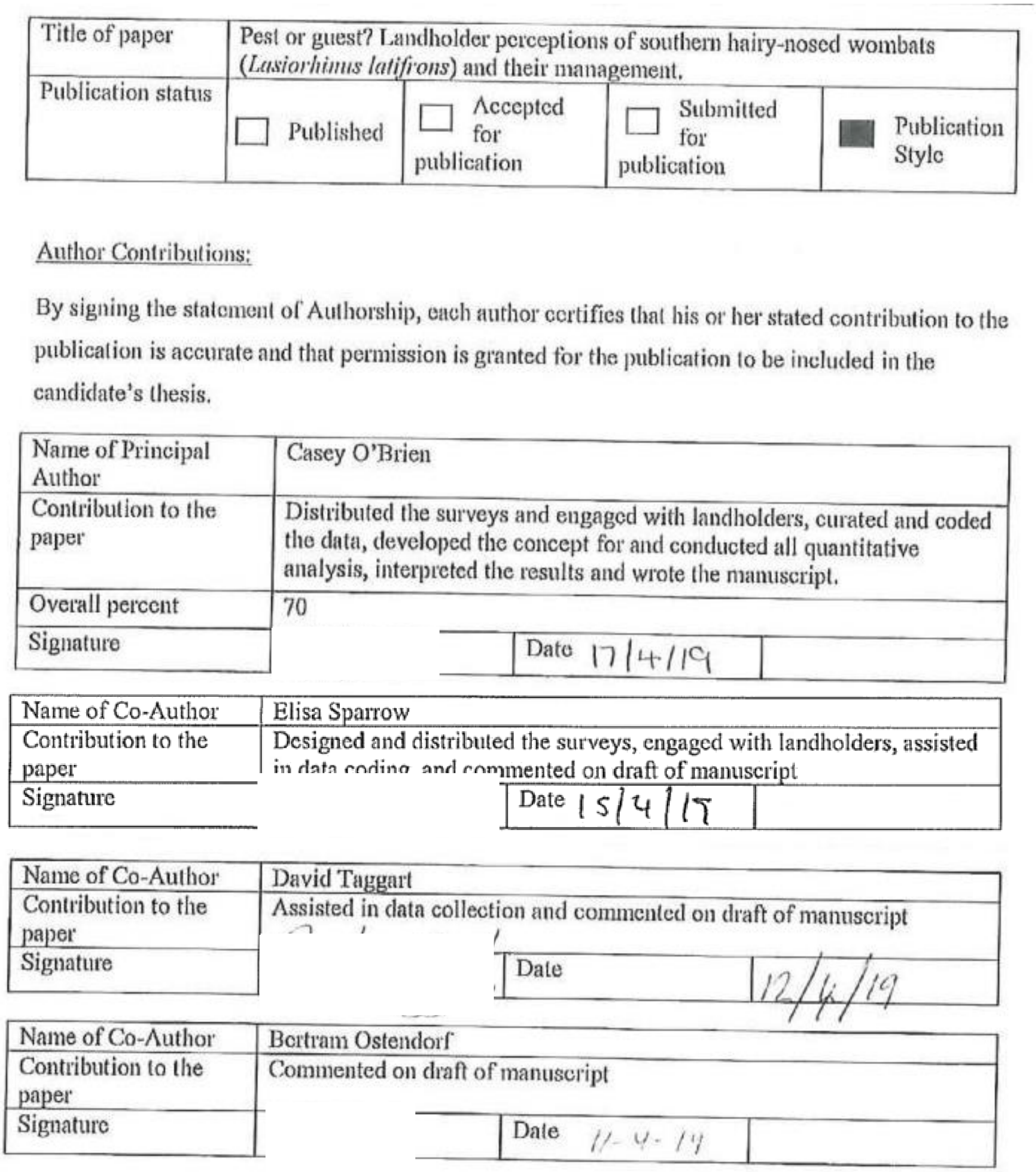
Chapter 2. Pest or guest? Landholder perceptions of southern hairy-nosed wombats (*Lasiorhinus latifrons*) in South Australia.

Consulting with landholders about the problems southern hairy-nosed wombats cause for agriculture



2.1 Statement of Authorship



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Abstract

Human–wildlife conflict is typically thought of as a damage management problem, with research focusing on managing problem wildlife with technical solutions. However, the underlying causes of conflicts are often human-driven. To make informed and effective management decisions, the human dimensions of conflicts need to be taken into consideration. This study examines landholder perceptions of the southern hairy-nosed wombat (*Lasiorhinus latifrons*), a species that has conflicted with agricultural interests for decades, using qualitative surveys distributed throughout the species' range. Of the respondents with *L. latifrons* on their properties, 81.2% reported damage, largely caused by the species' burrowing behaviour. Despite this, the majority of respondents supported *L. latifrons* conservation. However, support was lower among respondents with *L. latifrons* present and those who were financially dependent on their properties. Respondents who had experienced damage and were financially dependent on their properties were more likely to use culling to reduce damages. The largest proportion of respondents suggested the development of alternative non-lethal management options to improve *L. latifrons* management. These results highlight the need for improved management strategies that reduce damages and enhance co-existence between *L. latifrons* and landholders.

Keywords: co-exist, human–wildlife conflict, *Lasiorhinus latifrons*, survey, wildlife management, wombat.

2.2 Introduction

The management of human–wildlife conflict (HWC) typically focuses on controlling wildlife in a bid to reduce the damages caused to people's livelihoods. A wide range of strategies have been used to alleviate conflicts, but even where these have successfully reduced damage, long-term conflict resolution is rare (Marker 2002; Baruch-Mordo *et al.* 2009; Dickman 2010). Farmers, for instance, continued to remove cheetahs (*Acinonyx jubatus*) from farmland in Namibia despite the implementation of non-lethal mitigation measures that reduced predation on livestock (Marker 2002). This suggests that landholder perceptions of HWC and responses to them are not always in direct proportion to the cost of damages (Siex and Struhsaker 1999; 2000; Madden 2004; Dickman 2010). Research needs to look beyond damage control measures and consider the human aspects of conflicts.

The capacity of landholders to cope with losses has a strong influence on their perceptions of wildlife. Tolerance for wildlife typically decreases as reliance on agricultural income increases, even for wildlife that does not cause damage (Gillingham and Lee 1999; Van Tassell *et al.* 2000; Naughton-Treves and Treves 2005). Catastrophic or costly damage events play a greater role in shaping people's perceptions than more frequent small-scale losses (Naughton-Treves and Treves 2005). Bears, for example, have been viewed as bloodthirsty killers by many Japanese people since a rare fatal attack killing seven people in 1915, and their sightings often spark fear and retaliatory killing (Knight 2000). Cultural and societal values or other people's experiences can elevate fear of damage and drive negative responses towards wildlife (Zinn *et al.* 1998; Sillero-Zubiri and Laurenson 2001; Prokop *et al.* 2009; St John *et al.* 2013). The movie Jaws escalated fears of sharks in millions of people across the globe, with some refusing to enter the water 20 years later, despite the minimal risk of an attack (Harrison 1999). It is evident from these examples that focusing research solely on managing problem wildlife will limit the ability to resolve conflicts. The successful mitigation of HWC requires a broader approach, integrating the social dimensions of conflicts to understand the dynamics of the situation.

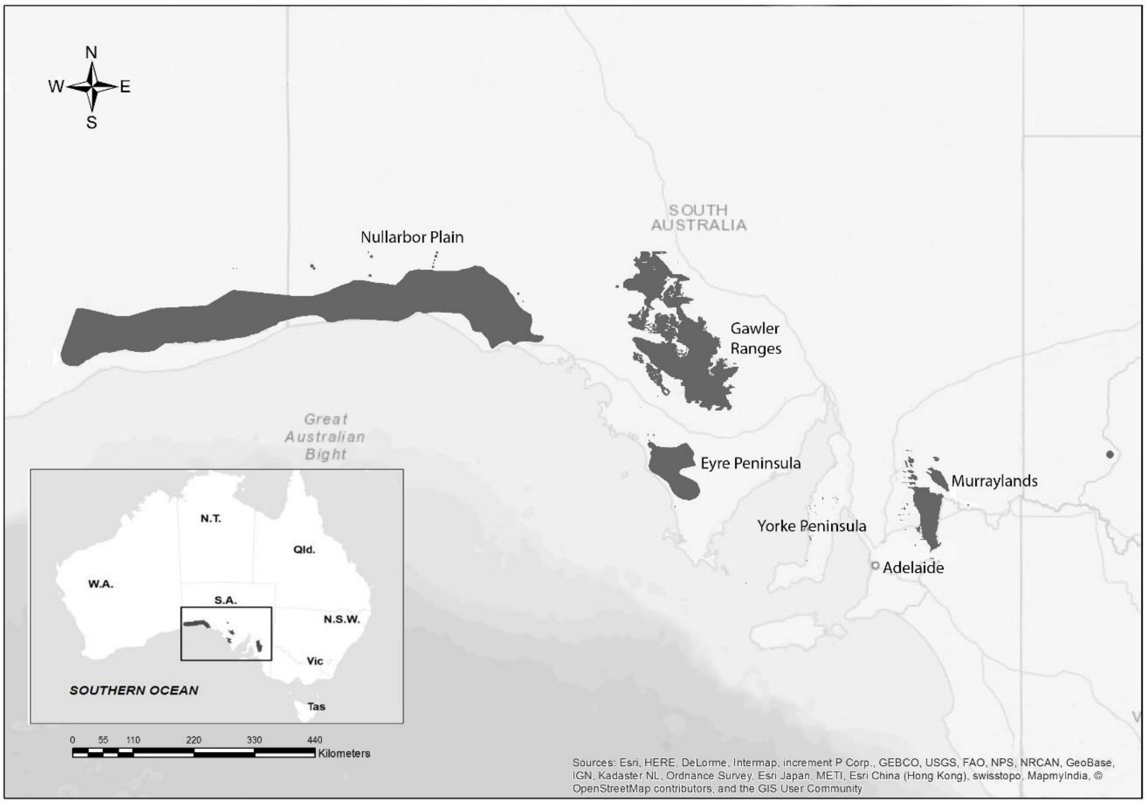
More informed management decisions can be made when the human dimensions of conflict are taken into consideration (Baruch-Mordo *et al.* 2009; Dickman 2010). In the absence of such information, assumptions can be made about people's perceptions and behaviours that are mismatched with actual perceptions (Knuth *et al.* 1992; Dickman 2010). This can lead to controversial management decisions, and create animosity among stakeholder groups, hindering management efforts. People who oppose or feel management decisions have been imposed on them are less likely to co-operate (Kellert 1981; Zinn *et al.* 1998). Efforts to create community-based wildlife management areas in Tanzania have been sabotaged by locals who were not involved in the planning process and felt their needs were superseded for conservation (Igoe and Croucher 2007; Sachedina 2008). Involving people in the decision-making process ensures their values are represented, making them more likely to co-operate (Fielding *et al.* 2005; Moon and Cocklin 2011).

The southern hairy-nosed wombat (*Lasiorhinus latifrons*) has a history of conflict with agriculturalists, the causes of which vary widely (Stott 1998; Temby 1998). *Lasiorhinus latifrons* consume crops and compete with livestock for food. The large warren complexes they create can damage vehicles, heavy machinery, and farm infrastructure such as roads and fences, and cause human injury (St John 1998; Stott 1998). The resulting damage can lead to substantial financial loss (St John and Saunders 1989). Management of *L. latifrons* has focused on alleviating damages through the implementation of a culling permit system monitored by the South Australian Department of Environment and Water (DEW) (*National Parks and Wildlife Act 1972* (SA)). Frustrations with the permit system and its failure to resolve conflicts often lead to indiscriminate culling (Taggart *et al.* 2008). To successfully manage *L. latifrons* across the agricultural land on which they are largely distributed (St John and Saunders 1989), understanding and incorporating landholder views into management decisions will be essential. Landholder opinions of *L. latifrons* were surveyed in 1985 (St John and Saunders 1989); however, people’s perceptions can change over time. Consequently, this study aimed to quantify the perceptions of landholders living throughout the range of *L. latifrons*. Surveys focused on examining the influence of (i) the presence/absence of *L. latifrons* on properties, (ii) experience of *L. latifrons* damage, and (iii) financial dependence on properties, on landholder perceptions of the species and its management. Information gained from the surveys will assist in identifying and understanding the problems associated with living with *L. latifrons* and provide recommendations for the future management of the species on private land.

2.3 Methods

2.3.1 Study area

This study focused on the five mainland regions within South Australia where *L. latifrons* occurs (Murraylands, Yorke Peninsula, Eyre Peninsula, Gawler Ranges, and the Nullarbor Plain; Figure 2.1). These regions are comprised of different environmental and geographic features and contain varying densities of *L. latifrons* (St John and Saunders 1989). All five regions are predominantly agricultural, containing a mixture of cropping and grazing land. The Murraylands region contains hobby farms and lifestyle properties because of its close proximity to Adelaide and the Murray River. In all five regions, *L. latifrons* have been reported to cause damage, the extent of which varies between regions (St John 1998).



Sources: Esri, HERE, DeLorme, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), swisstopo, MapmyIndia, © OpenStreetMap contributors, and the GIS User Community

Figure 2.1. The distribution of *L. latifrons* across the five distinct mainland populations on the Nullarbor Plain, Gawler Ranges, Eyre Peninsula, Yorke Peninsula and the Murraylands (Swinbourne *et al.* 2018).

2.3.2 Survey design

The survey (Appendix A) was designed by Zoos South Australia (SA), University of Adelaide, and University of South Australia scientists in consultation with DEW and the South Australian Farmers Federation (SAFF), in accordance with University of Adelaide Human Ethics (permit #H-025-2011). The survey focused on gathering information on a broad range of topics, to collect data relevant to numerous stakeholders. The structure of questions varied from binary yes/no statements to open-ended questions. Questions covered respondent demographics (age, gender, and region of residence), perceptions of *L. latifrons*, the damage they cause, and the species' management. Surveys were enclosed in unaddressed envelopes, accompanied by a reply paid envelope for return postage. A cover letter was included, detailing the purpose of the study, confidentiality information, instructions on how to complete the survey and how the information would be used. Due to the sensitive nature of some questions, the survey was anonymous to encourage full disclosure in responses.

2.3.3 Survey distribution

Surveys were distributed to regional landholders within the five mainland regions where *L. latifrons* occur between March and May 2011. Community wombat workshops were held within major regional centres on the Nullarbor at Nundroo and Penong; the Eyre Peninsula at Elliston and Cummins; the Gawler Ranges at Wudinna; and the Yorke Peninsula at Maitland, where approximately 80 surveys were distributed. The workshops aimed to engage with communities and foster relationships with landholders. They were run by Zoos SA staff, University of Adelaide scientists, and local Natural Resource Management (NRM) officers. The surveys and workshops were advertised on regional and statewide radio stations, in regional and metropolitan newspapers, on the Zoos SA and SAFF websites, and were promoted by local NRM staff. Presentations were given by Zoos SA and University of Adelaide researchers, addressing current knowledge on *L. latifrons* biology, ecology, and the purpose of the surveys. Attendees were encouraged to ask questions and share their opinions throughout the workshops. In addition, local NRM staff distributed an unknown number of surveys to interested residents who were unable to attend the workshops. Therefore, the exact number of surveys distributed across the Yorke Peninsula, Eyre Peninsula, Gawler Ranges, and the Nullarbor is unknown.

Within the Murraylands, 3,840 surveys were mailed to residents with post office boxes from the major regional centres of Mt Pleasant, Cambrai, Sedan, Swan Reach, Blanchetown, Morgan, Mannum, Angaston, Kapunda, Truro, Eudunda, Manoora, Saddleworth, Riverton, and Burra. Surveys were distributed in this manner to target regional landholders. The surveys were advertised on regional and statewide radio stations and in regional and metropolitan newspapers. Workshops were not conducted in the Murraylands as they had been carried out the previous year to address landholder concerns about living with *L. latifrons*.

2.3.4 Survey data screening

The numbers of survey responses received from the Nullarbor Plain (n = 6), Eyre Peninsula (n = 16), Gawler Ranges (n = 11) and Yorke Peninsula (n = 4) were too small to analyse individually. Respondents from the Nullarbor Plain, Eyre Peninsula, and Gawler Ranges regions had similar age and gender demographics (Appendix B) and were pooled to create the Far West region for the purpose of analysis. The Yorke Peninsula was excluded from analysis because of the small number of respondents, likely resulting from low numbers of *L. latifrons* in this region (Sparrow 2009). Because of the different survey distribution methods used in the Murraylands and its differing demographics from the Far West, it was treated as a separate region. Ages were reported in 10-year intervals. Because of the underrepresentation of some age classes, they were amalgamated into categories of ≤ 44, 45–64 and ≥ 65 years of age, to create an even spread of data for analyses.

2.3.5 Statistical analyses

Missing and non-applicable responses were dropped from the analyses, resulting in unequal sample sizes for each question. Responses were considered non-applicable if the question had not been answered, or consisted of ‘don’t know’, ‘no opinion’ or joke responses. Age and gender did not influence responses to any binary yes/no question in either the Murraylands or Far West (chi-squared: $P \geq 0.05$, Appendix C), and thus were disregarded in subsequent analyses. Differences in responses to binary yes/no questions were compared between (i) presence/absence of *L. latifrons* on properties, (ii) experience with damage, and (iii) financial dependence on properties within each region. Comparisons were made using chi-squared statistics and where significance was found, phi coefficients were used to examine the strength of relationships. Where sample sizes were small (one or more cells had expected counts ≤ 5), Fisher’s exact tests were applied. The Murraylands and Far West were not compared statistically as differences in the survey distribution methods and response rates between regions likely resulted in varying response biases and thus may not provide an accurate representation of disparities in opinions between regions.

Content analyses were conducted on open-ended questions, with responses sorted into categories based on themes found within the data (Hsieh and Shannon 2005). If a respondent's opinions did not fall within a pre-defined category, they were placed in the 'other' category. To account for the multiple opinions expressed by some respondents, the multiple dichotomy coding method was used (De Vaus 2002). Two independent raters coded the data to account for human error in interpreting responses and Cohen's kappa tests were applied to assess inter-rater agreement (Viera and Garrett 2005). Inter-rater agreement was sufficient for all questions (Cohen's Kappa ≥ 0.7, Appendix D). Descriptive statistics and cross-tabulations were used to assess trends in the data. All analyses were conducted in SPSS (version 20.0).

2.4 Results

2.4.1 Response rate and demographics

Within the Murraylands, 122 of the 3,840 surveys distributed were returned in usable form, resulting in a 3.2% response rate. Because of the distribution methods used in the Far West, the number of surveys delivered was unknown. Based on workshop attendances it was estimated that ~80 surveys were distributed. Of those, 33 were returned in usable form, giving an estimated response rate of 41%. Due to the anonymity of the surveys, non-response bias could not be quantified. The response ratio of males to females was ~1:1 and the mean age of respondents was 45-54 years of age in both regions. The majority (78.8%) of Far West and 39.5% of Murraylands respondents had *L. latifrons* on their properties. Three quarters (76.0%) of the Far West and 39.0% of Murraylands respondents were financially dependent on their properties. Significantly more respondents with *L. latifrons* on their properties relied on them as a main source of income in the Murraylands ($\chi^2_1 = 8.24$, $P = 0.00$, $\varphi = -0.33$) and the Far West (Fisher's Exact: $P = 0.00$). Due to a lack of data on the demographics of the regions sampled, it is unknown whether the respondents sampled were representative of their regions.

2.4.2 Perceptions of damage

The majority of respondents with *L. latifrons* on their property, had experienced damage (77.3%, $n = 46$ Murraylands; 84.6%, $n = 26$ Far West). The most prevalent form of damage in both regions was caused by *L. latifrons* digging behaviour (Figure 2.2). There was no relationship between reports of damage and financial dependence on properties within either region (Fisher's Exact, Murraylands: $P = 0.08$; Far West: $P = 0.09$). Monetary and time loss was calculated for respondents with *L. latifrons* on their properties who reported a value for damages. Estimated monetary losses due to *L. latifrons* damage ranged from $0–100,000 per annum, with the largest proportion of respondents within both regions reporting losses of ≥$10,000 per annum (Table 2.1). Time spent repairing *L. latifrons* damage ranged from 0–250 hours a year, with the largest proportion of respondents in both regions spending 10–99 hours repairing damages annually (Figure 2.2). When these two factors are considered together, 60% of Murraylands and 68.2% of Far West respondents incurred losses in time or money due to *L. latifrons* damage.

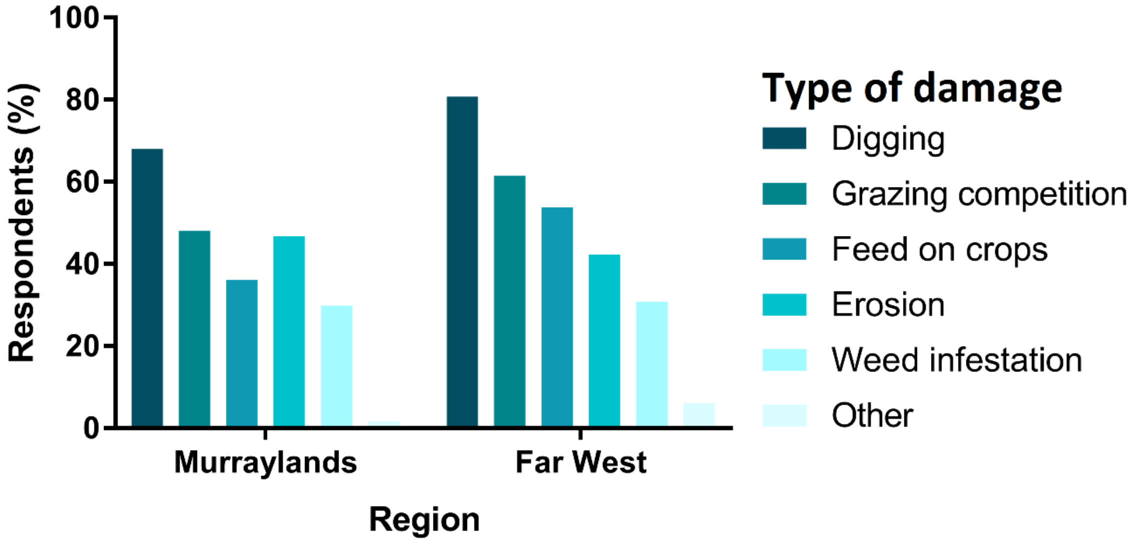


Figure 2.2. The percentage of survey respondents with *L. latifrons* on their properties that reported types of damage incurred by the species in the Murraylands (*n* = 46) and Far West (*n* = 26) regions of South Australia.

Table 2.1. The percentage of survey respondents with *L. latifrons* on their properties that reported time (hr) and monetary losses, incurred by the species in the Murraylands and Far West regions of South Australia.

Monetary loss ($)	**≥ 10,000**	**1,000 – 9,999**	**≤ 999**	**0**
Murraylands (*n* = 28)	39.2	39.2	14.2	7.1
Far West (*n* = 10)	50.0	50.0	-	-
Time lost (hr)	**≥ 100**	**10-99**	**≤ 9**	**0**
Murraylands (*n* = 26)	19.2	38.5	19.2	23
Far West (*n* = 12)	8.3	83.4	-	8.3

2.4.3 Conservation and co-existence

There was strong support for *L. latifrons* conservation in both regions, with 90.5% of Murraylands and 81.25% of Far West respondents finding it important (Table 2.2). Support for conservation was more likely among Murraylands respondents who did not have *L. latifrons* on their properties ($\chi^2_1 = 5.55$, $P = 0.03$, $\varphi = -0.22$). Far West respondents' support for conservation was not influenced by the presence/absence of *L. latifrons* on their property (Fisher's exact, $P = 0.29$; Table 2.2). Murraylands respondents who were financially dependent on their properties were less likely to support *L. latifrons* conservation (Fisher's exact, $P = 0.00$; Table 2.2). Far West respondents' support for conservation did not differ with financial dependence on their property (Fisher's exact, $P = 0.29$; Table 2.2). No significant difference was found between respondents' support for *L. latifrons* conservation and experience of damage in either region (Fisher's exact, Murraylands: $P = 0.17$; Far West: $P = 0.54$; Table 2.2).

The largest proportion (37.0% Murraylands; 30.4% Far West) of respondents who supported *L. latifrons* conservation did so because they are an iconic Australian species (Table 2.3). This was closely followed by the view that *L. latifrons* are part of the environment (25.8% Murraylands; 17.4% Far West; Table 2.3). Of the Far West respondents that felt conservation of *L. latifrons* is important, 17.4% also recognised the need for populations to be controlled (Table 2.3). The majority of Far West (66.7%) and a third of Murraylands respondents that did not support *L. Latifrons* conservation felt the species was unlikely to become extinct (Table 2.3). A third of Far West respondents felt the conservation of *L. latifrons* should be dependent on land use, for example, 'not appropriate in cropland' (Table 2.3). The majority (66.6%) of responses from Murraylands respondents fell into the 'other' category, with reasons ranging from safety and damage concerns to *L. latifrons* numbers being too high. The remainder of Murraylands respondents thought conservation was unnecessary, as extinction would never happen.

Table 2.2**.** The percentage of survey respondents within the Murraylands and Far West regions of South Australia that felt conservation of *L. latifrons* was important or unimportant, based on the presence/absence of *L. latifrons*, financial dependence on properties, and experience of *L. latifrons* damage.

	Murraylands			**Far West**		
	Important	Unimportant	*n*	Important	Unimportant	*n*
***L. latifrons* presence:**						
Present	82.6	17.4	46	76.0	14.0	25
Absent	95.7	4.3	70	100.0	0.0	7
Financial dependence on property:						
Dependent	71.8	28.2	30	71.4	28.6	21
Independent	97.6	2.4	41	100.0	0.0	7
***L. latifrons* damage:**						
Experienced	72.2	27.8	36	71.4	28.6	21
Not experienced	100.0	0.0	8	100.0	0.0	4

Table 2.3. Survey respondents' reasons for thinking conservation of *L. latifrons* was or was not important in the Murraylands (*n* = 125) and Far West (*n* = 29) regions of South Australia, where *n* is the total number of responses.

	Murraylands			**Far West**		
Responses:	Important	Unimportant	*n*	Important	Unimportant	*n*
Iconic Australian species	37.0	-	43	30.4	-	7
Part of the environment	25.8	-	30	17.4	-	4
Prevent biodiversity loss	13.8	-	16	8.7	-	2
Responsibility to protect	2.6	-	3	4.4	-	1
Declining species	5.2	-	6	-	-	-
Require control	3.5	-	4	17.4	-	4
Dependent on land use	3.5	-	4	8.7	33.3	2
Extinction unlikely	-	33.4	3	-	66.7	4
Other	8.6	66.6	16	13.0	-	3

The majority of respondents from both regions (Murraylands 87.1%, $n = 122$; Far West 80%, $n = 33$) believe co-existence between *L. latifrons* and primary producers is possible. Within the Murraylands, the belief that co-existence is possible was less likely among respondents with *L. latifrons* on their property ($\chi^2_1 = 4.15$, $P = 0.04$, $\varphi = -0.19$; Table 2.4), and respondents who were financially dependent on their property (Fisher's exact, $P = 0.05$; Table 2.4). No relationship was found between Murraylands respondents' opinions of co-existence and experience of *L. latifrons* damage (Fisher's exact, $P = 0.17$). The sample size for the Far West region was not large enough for statistical analysis of this question, though trends were similar to those in the Murraylands (Table 2.4).

When asked why they thought co-existence was possible, the largest proportion (18.7%) of Murraylands respondents felt land could be shared, while 13.7% of Far West respondents thought *L. latifrons* are a part of the environment (Table 2.5). This view was closely followed by the opinion that co-existence was possible if *L. latifrons* were managed (17.7% Murraylands; 13.7% Far West; Table 2.5). Of those who thought co-existence was not possible, half of Murraylands and three quarters (71.4%) of Far West respondents felt *L. latifrons* cause too much damage (Table 2.5). When asked where they thought *L. latifrons* should live, 29.4% of Murraylands respondents said within their natural range, 16.8% said where they choose, and 16.1% thought they should live in conservation parks (Table 2.6). The largest proportion (25.7%) of Far West respondents thought *L. latifrons* should be restricted to non-arable land, while 20% thought they should live within their natural range (Table 2.6).

Table 2.4. The percentage of survey respondents in the Murraylands and Far West regions of South Australia that felt co-existence between *L. latifrons* and primary producers was possible or impossible.

	Murraylands			**Far West**		
	Possible	Impossible	*n*	Possible	Impossible	*n*
***L. latifrons* presence:**						
Present	79.0	21.0	43	75.0	25.0	24
Absent	92.4	7.6	66	100.0	0.0	6
Financial dependence on property:						
Dependent	71.4	28.6	29	70.0	30.0	24
Independent	92.1	7.9	38	100.0	0	7
***L. latifrons* damage:**						
Experienced	71.8	28.2	32	70.0	30.0	24
Not experienced	100.0	0.0	9	100.0	0.0	4

Table 2.5. Respondents’ reasons for thinking co-existence was or was not possible within the Murraylands (*n* = 100) and Far West (*n* = 29) regions of South Australia, where *n* is the total number of responses.

	Murraylands			**Far West**		
Responses:	Possible	Impossible	*n*	Possible	Impossible	*n*
Can share land	18.7	-	18	4.5	-	1
If *L. latifrons* controlled	17.7	-	17	13.7	-	3
Change human attitudes	16.7	12.5	17	4.5	-	1
Other	15.6	37.5	18	40.9	14.3	10
If landholders supported	9.4	-	9	4.5	-	1
If *L. latifrons* are protected	9.4	-	9	4.5	-	1
Part of the environment	8.3	-	8	13.7	-	3
Area dependent	2.1	-	2	13.7	14.3	4
Cause too much damage	2.1	50.0	2	-	71.4	5

Table 2.6. Respondent’s opinions on where *L. latifrons* should live, within the Murraylands (*n* = 143) and Far West (*n* = 35) regions of South Australia, where *n* is the total number of responses.

	Murraylands		**Far West**	
Location:	*n*	%	*n*	%
Where they choose	24	16.8	4	11.4
Natural range	42	29.4	7	20.0
Sanctuary/zoo	5	3.5	1	2.9
Non-arable land	14	9.7	9	25.7
Bush/scrub	13	9.1	4	11.4
Conservation park	23	16.1	4	11.4
Other	22	15.4	6	17.2

2.4.4 Management of *Lasiorhinus latifrons*

Culling of *L. latifrons*, primarily by shooting was employed by 65.4% (*n* = 26) of Far West and 33.3% (*n* = 45) of Murraylands respondents with animals on their properties, in a bid to reduce damages. Within both regions, respondents who were financially dependent on their properties were more likely to cull *L. latifrons* (Murraylands: $\chi^2_1 = 13.01$, $P = 0.00$, $\varphi = 0.54$; Far West: Fisher’s exact $P = 0.01$; Table 2.7). Far West respondents who had experienced *L. latifrons* damage were more likely to use culling as a control measure ($\chi^2_1 = 8.93$, $P = 0.01$, $\varphi = 0.59$). The use of culling was not influenced by Murraylands respondents’ experience with *L. latifrons* damage ($\chi^2_1 = 2.17$, $P = 0.23$, $\varphi = 0.36$; Table 2.7). Half of Far West (*n* = 16) and 60% (*n* = 15) of Murraylands respondents who had used culling as a control measure found it to be successful at reducing damage. Few respondents (Murraylands: *n* = 7, Far West: *n* = 5) reported on the use of non-lethal conflict mitigation measures, and of those that did only a third had employed them. Methods used included the blocking of burrow entrances, burrow destruction, electric fencing, and the application of creosote, blood and bone based fertilisers, and D-Ter® around burrows. Among the methods used only two respondents reported success in reducing damages by blocking burrow entrances.

When asked for their suggestions on what could be researched to improve the control of *L. latifrons*, 53.3% of Murraylands and 25.0% of Far West respondents suggested the development of effective non-lethal control measures (Table 2.8). Of the Murraylands respondents who suggested non-lethal control, 47.0% of Murraylands and 9.0% of Far West respondents had *L. latifrons* on their properties. Suggestions for non-lethal control measures included the relocation of *L. latifrons,* use of physical barriers, and birth control. When asked how *L. latifrons* management could be improved, 37.6% of Far West respondents suggested alternative management options (Table 2.9), including non-lethal control, compensation schemes, and individual property plans. A third of Murraylands and a quarter of Far West respondents thought more research and education was required (Table 2.9).

Table 2.7. The percentage of respondents with *L. latifrons* on their properties that used culling as a management technique in the Murraylands and Far West regions of South Australia.

	Murraylands		**Far West**	
	n	%	*n*	%
Financial dependence on property:				
Dependent	25	56.0	22	77.2
Independent	20	5.0	4	0.0
***L. latifrons* damage:**				
Experienced	35	40.0	22	77.2
Not experienced	8	12.5	4	0.0

Table 2.8. Respondents' suggestions on what could be researched to improve the control of *L. latifrons* within the Murraylands (n = 30) and Far West (n = 8) regions of South Australia, where n is the total number of responses.

	Murraylands		**Far West**	
Suggestions:	*n*	%	*n*	%
Non-lethal practices	16	53.3	2	25.0
Adjust current practices	2	6.7	-	-
Ban culling	4	13.3	-	-
Destroy warrens	2	6.7	2	25.0
Increase protection	4	13.3	2	25.0
Other	2	6.7	2	25.0

Table 2.9. Respondents' suggestions for improving *L. latifrons* management within the Murraylands (n = 45) and Far West (n = 16) of South Australia, where n is the total number of responses.

	Murraylands		**Far West**	
Suggestions:	*n*	%	*n*	%
Research and education	15	33.3	4	25.0
Increase protection	6	13.3	-	-
Decrease protection	5	11.2	3	18.8
Work together	4	8.9	1	6.2
Alternative management	6	13.3	6	37.6
Landholder attitudes	2	4.4	1	6.2
Other	7	15.6	1	6.2

2.5 Discussion

Few human land uses have as great an impact on wildlife resources as agriculture (Maxwell *et al.* 2016; Watson *et al.* 2016). The support of agriculturalists is therefore essential for maintaining wildlife on private land. Understanding agriculturalists' perceptions of wildlife and their management will be critical for effectively managing HWC. This study examined landholder perceptions of *L. latifrons* and their management throughout the species range, to gain a better understanding of the factors driving human – wombat conflict. In particular, the study focused on how landholder perceptions were influenced by (i) the presence of *L. latifrons* on their property, (ii) experiences with *L. latifrons* damage, and (iii) financial dependence on their property. It was evident from these findings that *L. latifrons* were perceived as a considerable problem by landholders with the species on their properties. Overall attitudes towards *L. latifrons* were largely positive, but those directly affected by the species were more likely to have negative views towards it.

2.5.1 Response rate and demographics

The survey response rate for the Murraylands region was low, at 3.2%. Low response rates are common in postal surveys (White *et al.* 2005) and may reflect a lack of interest in the survey topic (Martin 1994). Because of the blanket postal distribution methods used in the Murraylands, it is likely that a large volume of individuals to whom the topic was irrelevant received surveys (e.g. town residents or landholders living outside the range of *L. latifrons*). In contrast, the high estimated response rate of 41% obtained for the Far West may have resulted from the distribution of surveys at workshops to landholders interested in *L. latifrons* management. Response rates in the Murraylands may have been improved by distributing the surveys to a pre-selected group of respondents representative of the sample population, personalising the survey, using pre-contact forms, or providing monetary incentives (Edwards *et al.* 2002; Dillman *et al.* 2014). Follow-up contact with non-respondents has also been shown to improve response rates (Dillman *et al.* 2014), but was not possible in this study due to the anonymity of the survey.

Low survey response rates often result in a significant non-response bias (Groves 2006). Due to the anonymity of the surveys, and a lack of data on the demographics of the regions sampled, non-response bias could not be assessed. The inability to compare the respondents' details with those of the sample population limits the wider application of these results (Williams 2003). Thus, the findings presented relate only to the opinions of respondents, and not the whole population. Regardless, these results may be especially relevant to wildlife managers, as those most likely to respond to the survey are people to whom the survey topic is relevant, and are most likely to take action or influence decisions (Groves *et al.* 1992; Messmer *et al.* 1999).

The demographics sampled varied between regions, with most Far West respondents having *L. latifrons* on their properties and relying on them as a main source of income, while this was true for less than half of Murraylands respondents. Respondents from both regions that had *L. latifrons* on their properties relied on them significantly more as a main source of income. This was expected, as over 90% of *L. latifrons* distribution falls across agricultural land (St John 1998). Differences in the demographics sampled between regions could be attributed to the different distribution methods and may account for differences in responses between regions. The postal survey distribution in the Murraylands likely resulted in non-target individuals receiving the surveys. Furthermore, the Murraylands contains more regional towns with residential properties and hobby farms and lifestyle properties (non-income earning rural properties) than the Far West because of its proximity to the Murray River and Adelaide, which may contribute to the different demographics sampled. The low sample of respondents from the Far West region who did not have *L. latifrons* and were not financially dependent on their properties limited the ability for statistical comparisons with those who did. However, given this region is dominated by agricultural production it is likely that the sample was representative of the population.

2.5.2 Perceptions of damage

It is clear from these results that respondents with *L. latifrons* on their properties perceived them to cause considerable damage. The most frequently reported form of damage in both regions was caused by *L. latifrons* digging behaviour. This is consistent with past reports of *L. latifrons* damage on the Nullarbor Plain (Stott 1998). Reports of grazing competition, crop consumption, erosion, and weed infestation varied between regions, most likely due to differing perceptions, land uses, and environmental factors. The largest proportion of landholders in both regions estimated *L. latifrons* damage to cost ≥ $10,000 per annum, with some reporting damage of as much as $100,000 per annum. This has increased from 1985 survey reports of $2,000–5,000 damages per annum, which equates to $5,061–12,654 at 2011 rates (St John and Saunders 1989; Reserve Bank of Australia 2017). This suggests *L. latifrons* damage or perceptions thereof may have intensified over time. However, people’s perceptions of damage are often disproportionate to actual damages (Wigley and Garner 1986; Knight 2000; Madden 2004; Dickman 2010). This was observed in relation to prairie dogs (*Cynomys ludovicianus*), which are perceived to reduce the carrying capacity of agricultural land, even though studies show they improve grassland and forage quality (Reading *et al.* 2005). Similarly, greater sandhill cranes (*Grus canadensis tabida*) were perceived to cause significant damage to barley fields, but actual damage was ≤ 3% of total crop yields (McIvor and Conover 1994a).

Landholders’ perceptions and tolerance of damage are often shaped by their financial dependence on their crop or stock (Van Tassell *et al.* 2000; Naughton-Treves and Treves 2005). Interestingly, no relationship was found between respondents financial dependence on properties and reports of *L. latifrons* damage, in either region. Similarly, Holmern and Roskaft (2014) found people’s financial dependence on poultry, did not influence their perceptions of carnivore depredation. In contrast, Lacey *et al.* (1993) found that as landowners dependence on agricultural income increased, so too did reports of crop damage by ungulates. Quantifying the extent of *L. latifrons* damage to agricultural properties, and ground truthing survey participant's estimates of damage is recommended for future research. Regardless of whether the damage is real or perceived landholder perceptions are still greatly important, as their attitudes influence the success of wildlife management programs.

2.5.3 Conservation and co-existence

Respondents' attitudes towards conservation and co-existence with *L. latifrons* were largely positive. Within the Murraylands, support for conservation decreased significantly among respondents who were financially dependent upon or had *L. latifrons* on their properties. Although similar trends were observed in the Far West, they were not significant. This is most likely due to the small sample of respondents who were financially independent of their properties and did not have *L. latifrons* present, limiting the power of comparisons between groups. This is consistent with other studies that found people were more likely to have negative views towards a conflict species by which they were directly affected (Zinn and Andelt 1999; Elmore *et al.* 2007; Schumann *et al.* 2012). Decreased tolerance of *L. latifrons* from respondents who are financially dependent on their property is understandable, as the impact of resource damage would be intensified by a lack of other assets (Dickman 2010). People who are not financially affected and do not have to deal with the problems associated with a species are more likely to have positive attitudes towards them (Lamb and Cline 2003). Support for conservation and co-existence was not influenced by respondents' experience with *L. latifrons* damage in either region. This is inconsistent with findings from other studies where landholders were more likely to have negative attitudes towards species that had caused damage (Gillingham and Lee 1999; Schumann *et al.* 2012).

Experience of damage may not influence support for conservation in this study because of respondents' awareness of the broader value of wildlife within the landscape. Support for *L. latifrons* conservation was largely driven by the view that they are an iconic Australian species and play an important ecological role. Similarly, Schumann *et al.* (2012) found farmers who understood that carnivores have an ecological role were more likely to have positive attitudes towards them and perceive lower levels of conflict. Respondents who did not support *L. latifrons* conservation thought the species would never become extinct. These views may be a result of localised experience with *L. latifrons*, rather than a broad understanding of the species' conservation status. Thoughts on whether co-existence was possible varied between regions, with the largest proportion of Murraylands respondents believing *L. latifrons* and landholders could share land, while the bulk of Far West respondents felt co-existence was not possible on arable land. The variation in respondents' opinions both between and within regions highlights the need for area-specific management approaches to meet the needs of differing land users. Focusing damage reduction efforts on those who are most vulnerable to the costs of damage and improving knowledge about the ecological role and conservation status of *L. latifrons* may lead to increased support for conservation and co-existence with the species.

2.5.4 **Management of *Lasiorhinus latifrons***

Culling was found to be the most commonly used control measure for the abatement of *L. latifrons* damage, which is unsurprising given the lack of alternative options. The use of culling was significantly higher among respondents who were financially dependent on their property or those who had experienced *L. latifrons* damage. Similarly, other studies have found that the acceptance of lethal control techniques was higher in people who had personally experienced wildlife related problems (Decker and Gavin 1987; McIvor and Conover 1994b; Zinn and Andelt 1999; Schumann *et al.* 2012). Despite landholder preference for the use of culling to control *L. latifrons*, only half of respondents found this method to be successful at reducing damage. The inability to accurately estimate *L. latifrons* numbers makes it difficult for state agencies to set balanced culling quotas and often results in ineffective conflict resolution, which leads to landholder frustration and indiscriminate culling (Taggart *et al.* 2008). A greater understanding of landholder perceptions of the permit system, the impacts of culling on *L. latifrons* population viability, and more accurate means of estimating *L. latifrons* abundance is needed to improve the culling permit system and reduce conflicts without threatening species conservation.

There was strong support for the development of non-lethal control techniques, despite few landholders using them or finding them successful. Electric fencing, which is costly and time consuming to maintain on a small-landholder scale was reported as unsuccessful at resolving conflicts, despite being found to reduce *L. latifrons* damage to the dog proof fence by 80% (St John and Saunders 1989). Landholder perceptions of success may not be influenced solely by damage reduction, but also by the time and costs associated with implementing and maintaining control measures. The lethal control of ungulates was favoured by landholders over alternative measures such as electric fencing or deterring devices that are costly to maintain (Horton and Craven 1997). The limited use of non-lethal control methods observed may be due to the absence of cost-effective strategies, rather than a lack of preference for these methods. This suggests non-lethal control measures are more likely to be accepted if they are cost effective, efficient and reduce damages. Such non-lethal techniques are currently lacking for *L. latifrons*, highlighting the need for further research.

Suggestions for improving the management of *L. latifrons* varied between the two regions examined, indicating the need for site-specific management plans. One suggestion that ranked as a high priority in both regions was additional research and education. This is promising, as it indicates landholders are open to new ideas and are willing to work with researchers and wildlife managers to improve *L. latifrons* management and enhance co-existence. Further research will undoubtedly increase our knowledge of the species and lead to more informed management decisions being made. Increased education of all stakeholder groups can have an important role to play in improving tolerance and co-operation, as people who are more knowledgeable about a problem are more accepting of it (Caro *et al.* 2003; Treves and Karanth 2003; Prokop *et al.* 2009).

2.5.5 Management recommendations

These findings highlight that the damage *L. latifrons* cause is of great concern to landholders. However, perceived and actual damages often differ, and the timing, intensity, and costs of damages need to be quantified to better direct appropriate management actions to where they are needed. Regardless of whether damage is real or perceived, landholder opinions need to be taken seriously by wildlife managers, as their attitudes will influence the success of wildlife management on private land. The differences in landholder perceptions within and between regions, highlight the need for locally adapted management plans that take into consideration differing land use, economic situations, frequency and intensity of damage, as well as *L. latifrons* densities. Prioritising efforts to reduce damages in high conflict zones and for landholders who are most affected by conflicts may enhance co-existence with *L. latifrons*. The use of culling will most likely continue as a preferred method of damage reduction until effective alternative damage abatement methods are identified. At present, the impact of culling on the behavioural ecology, breeding, recruitment, dispersal of *L. latifrons*, and its effectiveness at damage reduction is poorly understood. Further research is required to improve the management and implementation of culling permits. Furthermore, quantitative research into potential non-lethal damage mitigation strategies is recommended, to identify alternative management options to culling. Finally, the involvement of stakeholders in the future management and research of *L. latifrons* will be vital to developing and implementing successful strategies that balance the conservation needs of the species with the needs of the agricultural sector.

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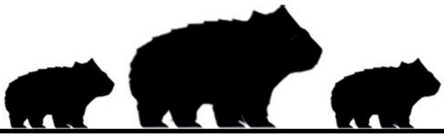
Chapter 3. Resolving human-wombat conflict: Is translocation the answer?

A translocated wombat with a GPS collar captured on remote camera



3.1 Statement of Authorship

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Contribution to the paper	Designed the experiment, conducted all field work and data collection, conducted all analysis interpreted the results and wrote the manuscript		
Overall percent	80		
Signature		Date	17/4/19

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Contribution to the paper	Assisted in the experimental design and data collection, and commented on draft of manuscript		
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Contribution to the paper	Assisted in the experimental design and data collection, and commented on draft of manuscript		
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3.2 Abstract

Translocation is a widely used non-lethal management tool for removing ‘problem individuals’ from sites of conflict, but its effectiveness is rarely properly evaluated. Although viewed as humane, translocations typically result in low survival, extensive movement, and non-establishment of animals, and often fail to resolve conflicts. Translocation has been suggested as a non-lethal management tool for the southern hairy-nosed wombat (*Lasiorhinus latifrons*), which is often in conflict with agricultural interests. This study assessed the viability of translocation for this species, by examining the (1) ease of capture of problem *L. latifrons*, (2) survival and establishment of translocated animals, and (3) reduction of conflict at the release site. *Lasiorhinus latifrons* were translocated from agricultural properties where conflicts arose and released on a private grazing property in the Murraylands of South Australia. The behaviour of translocated *L. latifrons* was monitored using GPS and VHF technology and compared with that of resident animals. The targeted removal of problem individuals proved time-consuming, with low capture success. No post-release mortality was observed, though many *L. latifrons* went missing soon after release, largely due to collar failures. The translocated *L. latifrons* that were tracked for ≥ 3 months displayed high site fidelity, though they initially ranged further than residents, before settling into similar behaviour patterns. Neighbouring *L. latifrons* quickly recolonised vacated burrows at the source sites; thus, conflicts were not resolved. These findings indicate translocation is not a suitable human–wombat conflict mitigation tool but may prove useful for restoring declining populations.

Keywords: co-exist, human–wildlife conflict, *Lasiorhinus latifrons*, translocation, wildlife management.

3.3 Introduction

As the world's landscapes become increasingly anthropogenic, human–wildlife conflict (HWC) is expected to become more common. Historically, conflicts have been managed using lethal controls, but growing public concern about their impact on wildlife has led to mounting pressure to implement non-lethal human–wildlife conflict mitigation measures (Shivik 2004; Massei *et al.* 2010; Germano *et al.* 2015). Viewed as humane, species specific and effective, translocation is the most widely used non-lethal conflict mitigation technique (Massei *et al.* 2010; Miller *et al.* 2014). Many thousands of problem animals across a wide range of taxa are translocated annually to ease problems ranging from crop damage (Craven *et al.* 1998), predation on stock (Waite and Phillips 1994), spread of disease (Beringer *et al.* 2002), property damage (Mosillo *et al.* 1999), and threats to human life (Butler *et al.* 2005a). However, mitigation translocations may not be as humane as they are perceived to be and there is growing concern about their appropriate use (Baker and Macdonald 2012; Germano *et al.* 2015).

Driven by the desire to resolve conflict, rather than establish populations, mitigation translocations are often poorly planned and implemented. Important biological and ecological factors such as capture stress, disease transmission, habitat suitability, genetics, sociality, and competition with conspecifics are neglected. This commonly results in low survival, extensive movement, and non-establishment of translocated animals (Germano *et al.* 2015; Sullivan *et al.* 2015). For example, the survival of gray squirrels (*Sciurus carolinensis*) translocated from urban to forested areas was low, with 97% either dying or disappearing soon after release (Adams *et al.* 2004). Similarly, wolves (*Canis lupus*) translocated into areas with competing resident packs failed to establish territories (Fritts *et al.* 1984). Thus, there is a clear need for better planning and implementation of mitigation translocations to ensure greater survival and persistence of animals.

Furthermore, the ability of translocations to reduce conflicts remains questionable. Many studies fail to report on conflict resolution, or only produce short term results, because of population growth, immigration of neighbouring animals, or homing behaviour of translocated individuals (Clarke and Schedvin 1997; Van Vuren *et al.* 1997; Bradley *et al.* 2005; Massei *et al.* 2010; Germano *et al.* 2015). The territories of translocated golden eagles, for instance, were re-occupied by immigrating neighbours within 1–8 days of their removal (Phillips *et al.* 1991), and half of all translocated black bears (*Ursus americanus*) returned home to resume their problem behaviour (Linnell *et al.* 1997). In contrast, translocating aggressive Australian magpies (*Gymnorhina tibicen*) reduced conflicts by 98% (Jones and Nealson 2003). Conflicts can also be transferred to the release site (Linnell *et al.* 1997; Bradley *et al.* 2005), as seen in urban white-tailed deer (*Odocoileus virginianus*), which damaged gardens and were involved in vehicle collisions at their release site (Cromwell *et al.* 1999). To determine if translocations successfully reduce conflict, post-release monitoring at the source and release sites is essential.

Translocation has been suggested as a conflict management tool for the southern hairy-nosed wombat (*Lasiorhinus latifrons*) (Chapter 2), which is considered an agricultural pest throughout much of its range. Conflicts largely arise because *L. latifrons* burrows undermine infrastructure such as dams and roads, damage vehicles and heavy machinery, and cause human injuries (St John and Saunders 1989; Stott 1998). *Lasiorhinus latifrons* also provide grazing competition for stock and consume crops (St John and Saunders 1989; Stott 1998). The resulting damage can cause large financial losses, decreased production, and a loss of time in reparations (St John 1998). To reduce damages, *L. latifrons* are culled under a permit system monitored by the Department of Environment and (*National Parks and Wildlife Act 1972* (SA)). Because of a lack of data on *L. latifrons* abundance and the effects of culling on populations, it is difficult to set balanced culling quotas often rendering culling ineffective for conflict resolution (Taggart *et al.* 2008). Wildlife managers are under increasing pressure from conservationists and the public to implement non-lethal conflict mitigation strategies.

The translocation of *L. latifrons* provides a unique opportunity to relocate problem animals from conflict zones to areas where populations are in decline, potentially reducing conflicts, and aiding in conservation. Currently, the impact of translocation on *L. latifrons* is poorly understood. Previous reintroductions of *L. latifrons* have been met with mixed success; some populations failed to establish (Copley 1995), while others such as those on Wedge Island are still present (Thornett *et al.* 2017). Due to a lack of information on the implementation and monitoring of these reintroductions, the reasons for their success or failure are unknown. This study aimed to assess the viability of translocation as a conflict management tool for *L. latifrons* based on three criteria: (1) capture success, (2) the survival and establishment of translocated animals, and (3) the reduction of conflicts at the source site.

3.4 Methods

3.4.1 Study sites

This study was conducted in the Murraylands of South Australia, a semi-arid region with low and unpredictable rainfall (annual mean 300 mm), and periodic but prolonged drought (Finlayson *et al.* 2005). *Lasiorhinus latifrons* were translocated from two sites near Eudunda (34.10˚S, 139.05˚E) and Morgan (34.02˚S, 139.40˚E), where they were in conflict with agricultural interests, and consequently being culled under destruction permits (*National Parks and Wildlife Act 1972)*. The release site was located on a private grazing property near Swan Reach (34.34˚S, 139.36˚E), approximately 60 km from the source colonies. The release site had an abundance of unoccupied burrows and competition from conspecifics was expected to be low, as the population was recovering from decline following a major drought in 2002 (D. Taggart pers. comm.). Semi-arid vegetation is common to all three sites, with remnant patches of mallee eucalypts throughout, while the shrub and herb layers varied across sites. The site at Eudunda is mainly used for cropping, and dominated by wheat and introduced forbs (*Asteraceae* spp.). In Morgan, land use was predominantly pastoral, and the site was dominated by native grasses (*Stipa* spp.) and Saltbush (*Atriplex* spp.). The shrub layer in Swan Reach consisted of saltbush (*Atriplex* spp.), and blue bush (*Maireana* spp.), with an underlayer of native grasses (*Stipa* spp.), introduced thread iris (*Moraea setifolia)*, and ward's weed (*Carrichtera annua*).

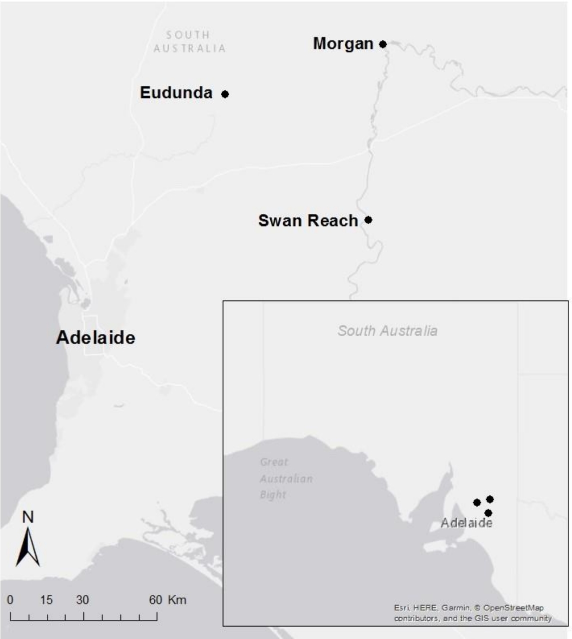

Figure 3.1. The locations of the two source sites, Eudunda and Morgan, and the release site, Swan Reach, in the Murraylands of South Australia.

3.4.2 Capture and processing

Lasiorhinus latifrons were translocated from agricultural properties near Eudunda and Morgan and released on a grazing property near Swan Reach between November 2010 and June 2013. To reduce site disturbances that may cause *L. latifrons* to remain underground and thus reduce capture success, warrens in conflict zones were fenced 2 weeks prior to the commencement of translocation. Fences consisted of 120-mm hex mesh with 30-mm hex holes supported by 120-mm star pickets, inserted to a depth of ~ 30–40 cm. To prevent *L. latifrons* digging under the fence, 40 cm of mesh was folded inwards, laid across the soil and secured with rocks to create a foot netting (Figure 3.2). Weldmesh cage traps (37 × 37 × 107 cm) were positioned around the perimeter of the fence line and wired open to allow *L. latifrons* to enter and exit freely. The traps faced both inwards and outwards to capture *L. latifrons* entering and exiting the warren. Upon commencement of translocation, the traps were set and checked every morning at sunrise. Captured *L. latifrons* were transferred to a hessian sack and transported by vehicle to a field station for processing. To obtain an index of trapping success the number of adult/sub-adult *L. latifrons* captured was divided by the number of trap-nights (No. of traps set per night × No. of nights traps set).

Lasiorhinus latifrons resident to the release site in Swan Reach were captured for comparison, using long-handled hoop nets while spotlighting at night (Taggart *et al.* 2003). Captured *L. latifrons* were transferred to a hessian sack and transported to a field station. The capture location of each *L. latifrons* was recorded using a Garmin® etrex 10 GPS (± 5 m accuracy), to ensure it was returned to the same location the following day. All captured *L. latifrons* were anaesthetised with an intramuscular injection of Zoletil (3 mg/kg, Vibrac Australia Pty Ltd) and given a health assessment. As part of this procedure, *L. latifrons* were weighed and given a body condition score of 1–5, with one being very poor and five being excellent condition (Table 3.1). Skin biopsies were collected and assessed for signs of sarcoptic mange (*Sarcoptes scabiei*) under a light microscope. The age classes (adult, sub-adult, juvenile) of male *L. latifrons* were determined by head length and width, measured to the nearest millimetre using callipers (Taggart *et al.* 2007). Female age class and reproductive status were determined by pouch condition (depth, moisture content, cleanliness, colour, and teat length) (Tyrell 2001).

Figure 3.2. A) and B) A fenced warren at the source site in Morgan from which *L. latifrons* were captured; C) A *L. latifrons* caught in a weldmesh trap.

Table 3.1. The ratings used to assess the body condition of *L. latifrons* (Sparrow 2009)

Rating	Status	Description
1	Very poor	Obviously protruding vertebral processes (by touch), pelvic and shoulder girdles, and very high external parasite load
2	Poor	Protruding vertebral processes, pelvic and shoulder girdles, dull fur, and an elevated external parasite load
3	Average	Average cover across vertebral processes, pelvic and shoulder girdles, shiny fur, and moderate external parasite load
4	Good	Good cover across vertebral processes (not protruding), pelvic and shoulder girdles, shiny fur, and low external parasite load
5	Excellent	Excellent cover across vertebral processes (difficult to detect), pelvic and shoulder girdles, shiny fur, and little or no external parasite load

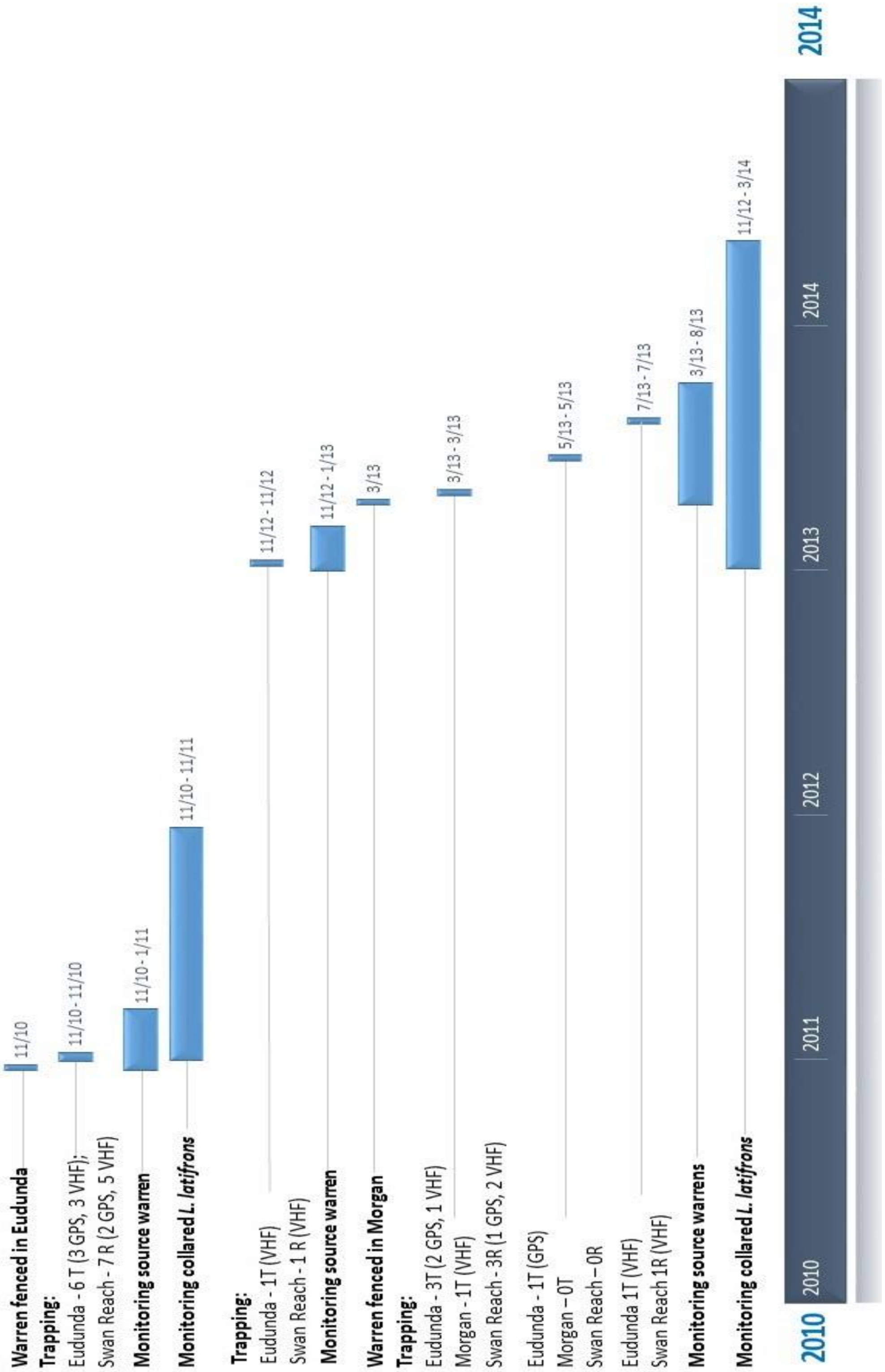

Figure 3.3. A timeline of events showing the dates (month/year) warrens were fenced at the source sites, source warrens were monitored with cameras, translocated (T) *L. latifrons* were trapped/captured from the two source sites (Swan Reach and Eudunda), residents (R) were captured at the release site (Swan Reach) and collared *L. latifrons* were monitored at the release site.

Fourteen translocated and 13 resident mange-free adult/sub-adult *L. latifrons* with a body condition ≥ 3 were selected for the trials; the remainder were returned to their capture site. Selected *L. latifrons* were microchipped (Trovan Ltd) and fitted with either a Titley electronics™ double-stage radio transmitter (VHF) or GPS collar. All collars were made of soft nylon webbing containing a biodegradable element designed to disintegrate within 18–24 months. Collars weighed 180–420 g, less than 2% of an average animal's body weight (~25 kg). During 2010, VHF collars were fitted to three translocated and five resident *L. latifrons*. In addition, three translocated and two resident *L. latifrons* were fitted with Sirtrack® data-logging GPS collars containing a VHF component to allow manual tracking with a yagi antenna. The Sirtrack® collars operated on a cycle of half-hourly fixes between 4 pm and 6 am with fixes only every 3 hours between 6 am and 4 pm to save battery power during the day when *L. latifrons* were underground. In 2012, one translocated and one resident *L. latifrons* were fitted with VHF collars. During 2013, three translocated and one resident *L. latifrons* were fitted with Ecotone® E.P 3.5 Harrier GPS data-logging collars, operating on the same cycle as the Sirtrack® collars. The Ecotone® collars communicated with an EP BS-12T LR long-range base station using UHF signals, which allowed manual tracking with a directional antenna and the transfer of data from the collar to the base station when *L. latifrons* were within a 1 km range. The remaining three translocated and three resident *L. latifrons* were fitted with VHF collars that contained temperature data loggers (Maxim Integrated iButtons™) programmed to record the temperature (˚C) every hour. The iButtons™ were attached to the collars using Araldite® epoxy adhesive and heat-shrink tubing. Comparison of this data with ambient and burrow temperatures (˚C) recorded every hour using iButtons™ enabled the time *L. latifrons* spent above ground to be calculated (Finlayson *et al.* 2003).

3.4.3 Release and monitoring

Following processing and recovery from anaesthesia translocated *L. latifrons* were hard released into vacant burrows within large warrens containing a combination of empty and occupied burrows. Soft release methods were not used as *L. latifrons* are territorial, and confining animals into warrens with resident conspecifics could result in aggressive behaviour. Release locations were selected based upon food availability and the presence of vacant burrows, identified by the presence of plant matter, and a lack of fresh scats, and digging around the burrow entrance. Resident *L. latifrons* were released at their capture locations and monitored to ensure safe passage into a burrow. During the first week post-release, nightly locations of all *L. latifrons* were sought from monitoring positions across a 17 km^2 area. Thereafter, the locations of all *L. latifrons* were sought once a week, for up to a year or until the animals were recaptured and collars removed. Upon detection, the date, time, and location of collared *L. latifrons* was recorded to an accuracy of ± 5 m, using a GPS. In addition to manual tracking, the Ecotone® base station was also deployed within the 17 km^2 study area between monitoring trips to optimise detection of GPS collared *L. latifrons*. Week-long field trips were conducted every 3 months to monitor and recapture all collared *L. latifrons* for health checks using long-handled hoop nets, and weld mesh cage traps placed in burrow entrances. During monitoring trips, the daily locations of *L. latifrons* were sought, to determine minimum overnight travel distances (distance between successive daily locations). Recaptured *L. latifrons* were given a health check and released at their site of capture within 24 hours. Animals caught more than 5 months post translocation had their collars removed before being released. This research was undertaken with a University of Adelaide animal ethics permit (s2011-197C), and a scientific research permit (A25828-1) and animal ethics approval (A25828-1) from the Department of Environment and Water in South Australia.

3.4.4 **Conflict reduction**

The source sites were monitored with motion sensor cameras (Scoutguard KG680V and Reconyx HC600, Faunatech Ausbat, Australia), to determine if the removal of individuals reduced *L. latifrons* activity. Four cameras were fixed to fence posts within the perimeter of each warren 2 weeks prior to translocation and remained in place for 8 weeks post-translocation to monitor *L. latifrons* activity inside the fenced area. Cameras were checked and photos downloaded on a fortnightly basis. The number of individuals inhabiting the warrens was determined each week by identifying individuals from the photographs, based on scars, ear markings, and fur patterns (Figure 3.4). Although individuals are unable to be identified in all photos, all *L. latifrons* using a warren can be identified after 1 week of monitoring (Koenig 2012). The number of *L. latifrons* using the warrens before and after the translocation events was compared to determine the time taken for neighbouring animals to recolonise vacant burrows.

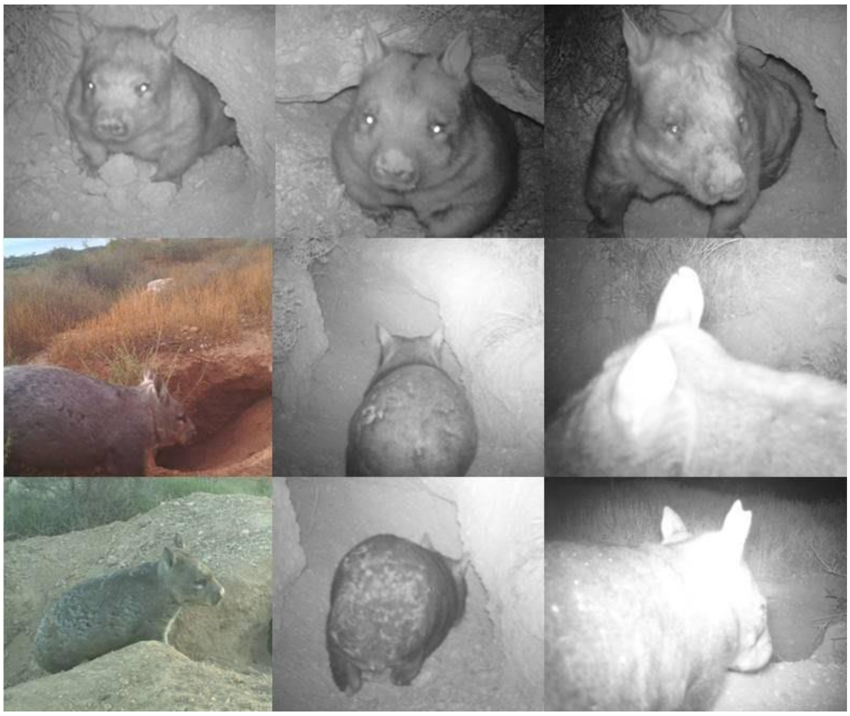

Figure 3.4. A Sample of *L. latifrons* images captured with the motion sensor cameras, each showing a different individual, identified based on fur patterns, scars, and ear markings.

3.4.5 **Data analyses**

On average, collared *L. latifrons* were tracked for 90.8 ± 92.7 days, as a large number of animals went missing shortly after release. Therefore, only data from *L. latifrons* with ≥ 15 locations recorded in the first 3 months post-translocation were included in analyses. No significant differences have been found in the ranging behaviour of male and female *L. latifrons* (Finlayson *et al.* 2005), so data were pooled across genders to increase sample size and strengthen the analysis. All data were mapped using ArcMap10 and analysed in R-*3.3.2* (R Core Team 2014). The furthest recorded distance travelled from release locations, calculated as the straight-line distance between VHF points, was log transformed to meet the assumptions of normality, and distances travelled by resident and translocated *L. latifrons* were compared using ANOVA. The number of burrows and warrens that individuals were observed using was determined from visual observations and VHF data. Variation in burrow and warren use was compared between resident and translocated *L. latifrons* using ANOVA. *Lasiorhinus latifrons* that established home ranges that included their release warren and travelled ≤ 1 km from their release location were considered to display site fidelity. The measure for site fidelity was set at 1 km as it was the maximum distance travelled by a resident *L. latifrons* in this study.

Nightly distance travelled, time above ground, and home range estimates were calculated using GPS data only, as insufficient location data were gathered from VHF collared

L. latifrons. No iButtons™ were recovered for analysis; they appeared to have become dislodged from the collars through wear and tear. The Sirtrack GPS collars recorded a Horizontal Dilution of Precision (HDOP), which assessed the accuracy of locations. Fixes with an associated HDOP value ≥ 10 were discarded from the analysis, as they are associated with large location errors (D'Eon and Delparte 2005). The distance travelled and the time between consecutive GPS locations were generated using the movement pathmetrics tool in the Geospatial Modelling Environment (GME) ArcMap 10.0 extension software (Beyer 2012). Nightly distances travelled were calculated as the sum of straight-line distances between consecutive GPS locations. Time above ground was calculated nightly, by summing the time between consecutive half-hourly GPS locations. When fixes were greater than half an hour apart, it was assumed the animals had entered a burrow. Due to the nocturnal activity of *L. latifrons*, nights constituted a 24-hour period beginning and ending at 6 am.

Monthly home ranges of translocated and resident GPS collared *L. latifrons* were generated in GME for the first 3 months post-release. Despite their inefficiency in calculating home ranges, 100% minimum convex polygons (MCPs) were generated as they indicate more unusual movements (Nilsen *et al.* 2008), which were of interest for the translocated *L. latifrons*. Five percent incremental area analysis was conducted on monthly MCPs, and asymptotes were examined to determine the mean number of locations required to develop a stable home range (Kenward and Hodder 1996). Monthly home ranges were also calculated using kernel density estimates (KDE), as they are considered the most robust and unbiased home range indicators (Borger *et al.* 2006; Nilsen *et al.* 2008). Gaussian fixed kernels with a smooth cross-validation algorithm were generated in GME using the R package ‘*ks*’ (Duong 2013), as they produced the most appropriate kernels for closely spaced GPS locations (Hemson *et al.* 2005). Percentage isopleths were calculated for each kernel using GME. Ninety-five percent isopleths were discontinuous and are recommended to be avoided because of bias in the outlying points (Borger *et al.* 2006). Thus, 90% isopleths were selected as indicators for home range, and calculated for each kernel. Fifty percent isopleths were used to examine core areas, as incremental utilisation plots revealed discontinuity in the line at 50% of the fixes (Kenward *et al.* 2001). Home range estimates were also calculated for the translocated *L. latifrons* in the first fortnight to assess its ranging behaviour prior to the collapse of its release warren in the third week post-release following an unseasonably large rainfall event (108 mm in 2 days). Due to variation in the number of fixes collected by each collar, Pearson's product–moment correlation was conducted to determine the relationship between the number of fixes recorded per month and home range size. Home range and core area sizes were calculated for each individual and compared between translocated and resident *L. latifrons*. Small sample size limited the ability to undertake statistical analysis; therefore, results are presented as means ± standard deviations and displayed graphically.

3.5 Results

3.5.1 Trapping success

Trapping at the source sites occurred over 252 trap–nights, during which 15 adult/sub-adult *L. latifrons* were caught, giving an overall trapping rate of 5.95% (Table 3.2). The four juveniles and one female with a young at foot were returned to their capture site. The remaining 14 adult/sub-adult *L. latifrons* were transported to Swan Reach for translocation. *Lasiorhinus latifrons* were most trappable in November (10.53%) and least trappable in May (1.19%; Table 3.2). Trapping success was lowest in Morgan (2.00%) and highest in Eudunda (6.95%). Thirteen adult/sub-adult *L. latifrons* resident to the release site at Swan Reach were captured for comparison.

3.5.2 Survival and health

Of the 32 *L. latifrons* captured, two died from heat stress during recovery from anaesthesia in 2012, giving an overall pre-release mortality rate of 6.25%. No mortalities were recorded following release, though a large number of *L. latifrons* (nine translocated, nine residents) went missing shortly thereafter (Table 3.3). The three translocated and one resident *L. latifrons* fitted with Ecotone collars could not be tracked due to repeated malfunctions of the base station. Extensive search efforts resulted in the relocation or recapture of five translocated and five resident *L. latifrons* at or near their release locations, 50% of which had malfunctioning collars (Table 3.3). The reasons for the disappearance of the remaining four translocated and four resident *L. latifrons* remains unknown. Based on the number of *L. latifrons* known to be alive 6 months post-release (translocated = 7, resident = 7), the minimum survival rate was 53.84% for translocated and 58.33% for resident animals.

Throughout the study, four translocated and four resident *L. latifrons* were recaptured, within 5–24 months post-release. As *L. latifrons* were captured at different times, post-release body condition scores were not compared between translocated and resident animals. All *L. latifrons* were in a condition that was to be expected for the time of year at which they were caught (D. Taggart, pers. comm.). One translocated female recaptured 7 months post-release had a pouch young. There were no signs of lesions or discomfort from wearing the collars, though there was some minor hair loss beneath the collar band. Three intact collars of two translocated and one resident *L. latifrons* were retrieved from burrows. There was no evidence of mortality or human interference on the recovered collars, or at the site of recovery.

Table 3.2. The trapping success of *L. latifrons* at the source sites, Eudunda and Morgan. Capture success is calculated for adult (A) and sub-adult (SA) *L. latifrons* only. Juveniles (J) were not included in calculations of trapping success as they were not translocated**.**

Location	**Date**	**Nights trapped**	**No. of traps**	**Trap – nights**	***L. latifrons* caught**	**Capture success (%)**
Eudunda	Nov 2010	14	5	70	4A, 2SA	8.57
Eudunda	Nov 2012	4	6	24	3A, 1J	12.50
Eudunda	Mar 2013	7	6	42	3A, 1J	7.14
Eudunda	May 2013	7	6	42	1A	2.38
Eudunda	Jul 2013	4	6	24	1A	4.16
Morgan	Mar 2013	5	5	25	1A,1J	4.00
Morgan	May 2013	5	5	25	1J	0.00
Total		**46**	**39**	**252**	**13A, 2SA, 4J**	**5.95**

Table 3.3. The success of collar deployments for translocated and resident *L. latifrons*.

	Translocated			Resident		
Parameter:	**VHF**	**GPS**	**Total**	**VHF**	**GPS**	**Total**
Collared *L. latifrons*	7	6	13	9	3	12
Recaptured *L. latifrons*	4	-	4	2	1	3
Missing *L. latifrons*	4	5	9	7	2	9
- Recaptured	2	-	2	2	-	2
- Resighted	2	1	3	2	1	3
- Collars recovered	-	2	2	-	2	2
Collar failures	2	4	6	2	1	3
- Recaptured	1	-	1	2	-	2
- Resighted	1	1	2	-	-	1
- Collars recovered	-	1	1	-	-	-
- Missing Ecotone collars	-	2	2	-	1	1
Sufficient data obtained	4	1	5	5	3	8
- Displayed site fidelity	3	-	3	5	3	-
- Home range established	-	1	1	-	3	3

Table 3.4. The date of release, tracking period and fate (M = missing, R = recaptured) of each collared *L. latifrons* during the translocation trials. Sufficient data were obtained for analysis from individuals highlighted in bold. Individuals with ibuttons attached to their VHF collars are denoted by a ^.

Individual	Collar type	Release date	No. of fixes	Tracking period (days)	Fate	Comments
Resident:						
1.5	GPS	Nov-10	**452**	**87**	M	Collar recovered Nov 2012
1.8	GPS	Nov-10	**7,321**	**150**	R	Caught April 2012
2.1	VHF	Nov-10	**18**	**133**	R	Collar failed Caught March 2012
1.4	VHF	Nov-10	5	13	M	Last signal Nov 2010
1.6	VHF	Nov-10	**15**	**286**	M	Last signal Sept 2011
1.7	VHF	Nov-10	4	13	M	Last signal Nov 2010
1.9	VHF	Nov-10	5	48	R	Collar failed Caught April 2011
Waffles	VHF	Nov-12	**18**	**162**	M	Last signal Feb 2013
Maple	GPS	Mar-13	**233**	**1**	M	Base station failed Resighted Oct 2013 Collar recovered Sept 2015
Pepper	VHF^	Mar-13	**37**	**116**	R	Caught Oct 2014
Ron	VHF^	Mar-13	**25**	**220**	M	Last signal Oct 2013 Resighted Oct 2014
Sorbet	VHF^	Jul-13	10	74	M	Last signal July 2013
Translocated:						
2.3	GPS	Nov-10	**554**	**77**	M	Collar recovered July 2012
2.2	GPS	Nov-10	3	25	M	Collar failed Collar recovered Feb 2011
1.3	GPS	Nov-10	3	3	M	Last Signal Oct 2010
1.2	VHF	Nov-10	**17**	**289**	R	Collar failed Caught Nov 2013
1.1	VHF	Nov-10	2	2	M	Last Signal Sept 2011
2.4	VHF	Nov-10	**17**	**136**	R	Caught March 2012
Bacon	VHF	Nov-12	2	14	M	Collar failed Resighted Aug 2013
Womble	VHF^	Mar-13	**37**	**221**	R	Caught Oct 2013
Chomp	GPS	Mar-13	1	1	M	Base station failed No recorded locations
Squeak	GPS	Mar-13	1	1	M	Base station failed No recorded locations
Omelette	VHF^	Mar-13	3	7	M	Last Signal July 2013
Milo	GPS	May-13	2	1	M	Base station failed Resighted Nov 2013
Sparkle	VHF^	Jul-13	**35**	**171**	R	Caught Dec 2013

3.5.3 Ranging behaviour

Sufficient VHF data were obtained from six resident and four translocated *L. latifrons*, including VHF location data for one GPS collared resident. The furthest distance travelled from release locations was greater for translocated (0.55 km ± 0.54) than resident *L. latifrons* (0.27 km ± 0.29), though this was not significant ($F_{1,8} = 0.465$, $P = 0.51$). Warren and burrow use did not differ significantly between resident and translocated *L. latifrons* ($F_{1,8} = 2.53$, $P = 0.15$; $F_{1,8} = 0.11$, $P = 0.75$ respectively, Figure 3.5). All resident ($n = 6$) and 60% of translocated ($n = 5$) *L. latifrons* displayed release site fidelity. The two translocated *L. latifrons* that did not display release site fidelity established home ranges that did not contain their release warrens, and one travelled 1.3 km from its release site following the collapse of its release warren. No other *L. latifrons* travelled ≥ 1 km from its release site or was known to have experienced a warren collapse throughout the duration of the study.

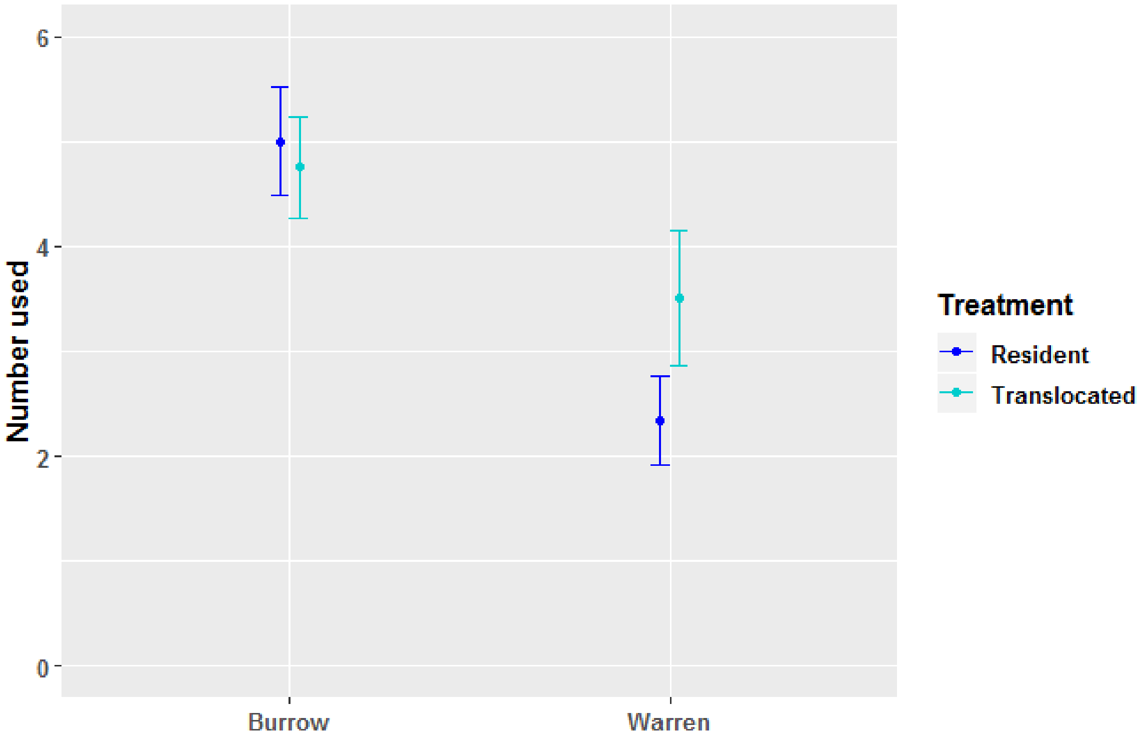

Figure 3.5. The mean ± SD of the number of burrows and warrens used by resident ($n = 6$) and translocated ($n = 4$) *L. latifrons* in the first 3 months post-release.

Five GPS collars were recovered, one of which failed to record any data. The remaining four collars collected data from translocated female 2.3, and three residents; male 1.8, and females 1.5, and Maple, for periods of 2–18 months. The greatest single nightly foray of 3.36 km was conducted by the translocated *L. latifrons* following the collapse of its release warren; 3 weeks post-release. Prior to its warren collapse, the greatest nightly foray by the translocated *L. latifrons*' was 2.20 km. In comparison, conducted by a resident Maple was 1.07 km during the first week post-release. The mean nightly distance travelled by the translocated *L. latifrons* (0.67 ± 0.72 km) was greater than that of the residents (0.39 ± 0.22 km). Over time, the nightly distance travelled by the translocated *L. latifrons* decreased to resemble the residents' patterns (Figure 3.7), which all followed a similar trend (Figure 3.6).

The translocated *L. latifrons* spent a mean of 2.56 ± 1.58 hr above ground per night, less than its resident counterparts (3.66 ± 1.16 hr; Figure 3.8). However, individual variation in the amount of time spent above ground was observed (Figure 3.9). Resident 1.5 behaved similarly to the translocated *L. latifrons*, spending a mean of 2.38 ± 0.98 hr above ground per night, while Maple spent on average 3.19 ± 1.14 hr above ground per night. Resident 1.8 spent more time above ground than all *L. latifrons* (4.95 ± 1.45 hr), and the greatest amount of time (6.50 hr) above ground in a single night. In the third week post-release, a mass rainfall event resulted in a decrease in the amount of time spent above ground by resident 1.8, but did not appear to affect the other *L. latifrons* (Figure 3.9).

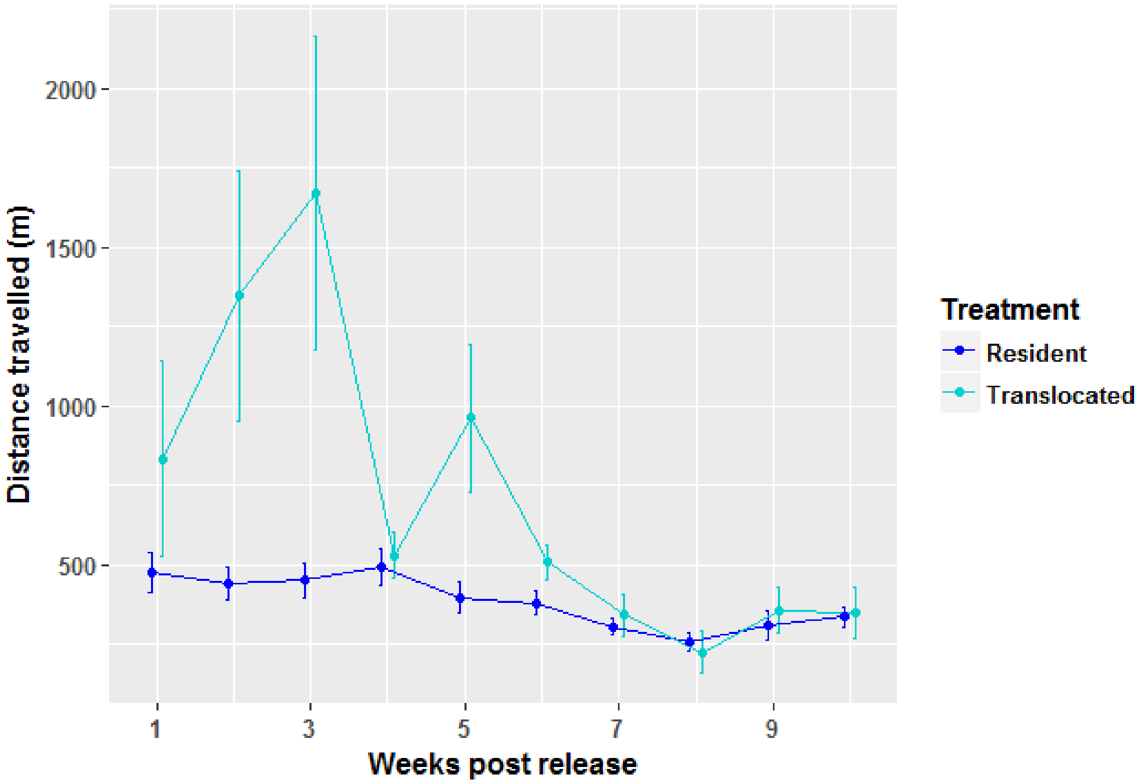

Figure 3.6. The mean nightly distance travelled (m) by the translocated (*n* = 1), and resident (*n* = 3) GPS collared *L. latifrons* over time (weeks).

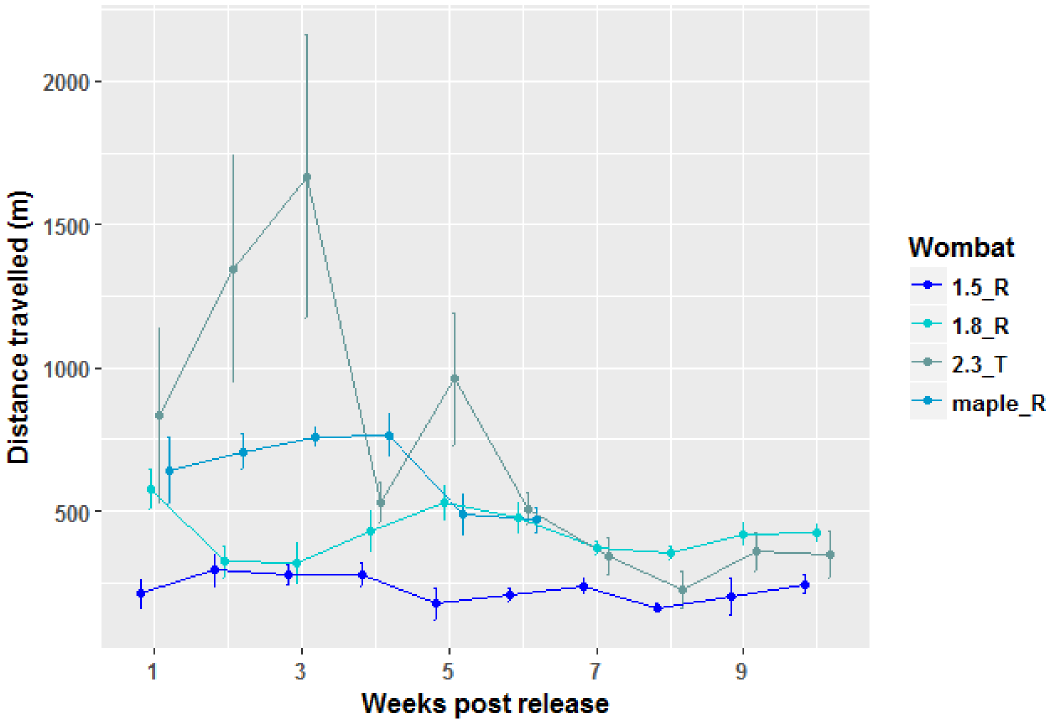

Figure 3.7. The mean distance (m) travelled by individual GPS collared *L. latifrons* per week, where 1.8 is the male and 1.5 the female residents (R) released in 2010, Maple is the resident female released in March of 2013 and 2.3 is the female translocated (T) in 2010.

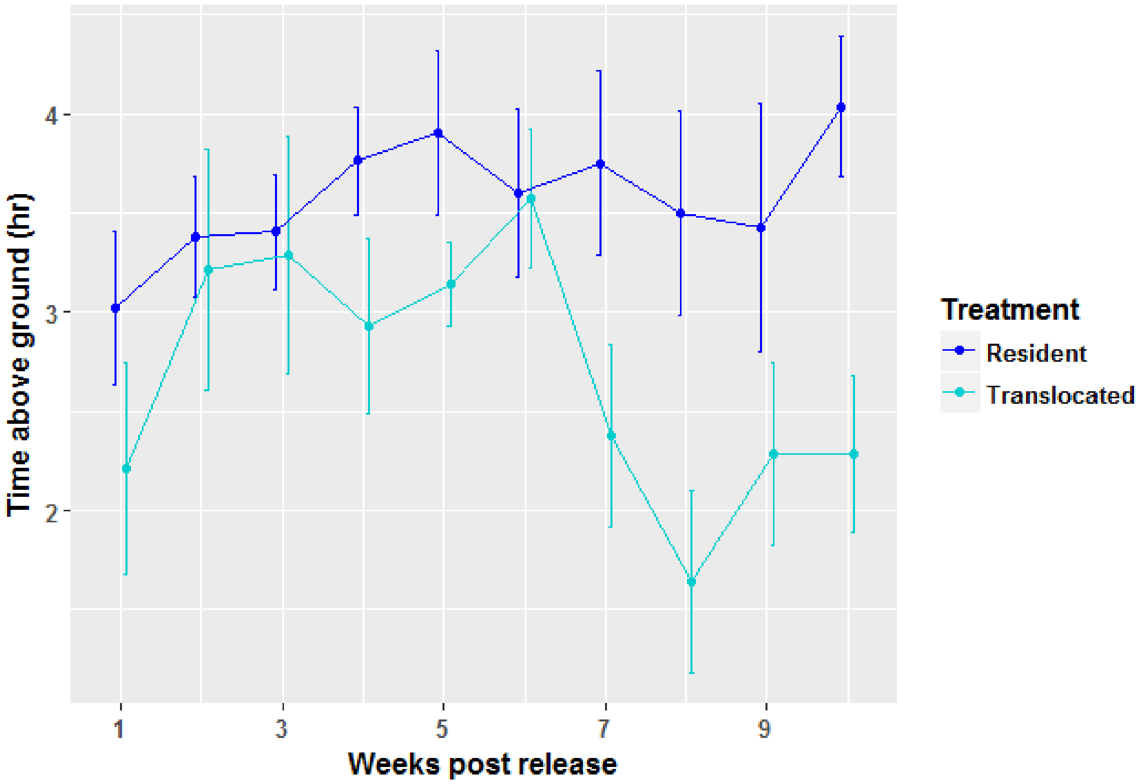

Figure 3.8. The mean time spent above ground (hr) by the resident (*n* = 3) and translocated (*n* = 1) GPS collared *L. latifrons* over time (weeks).

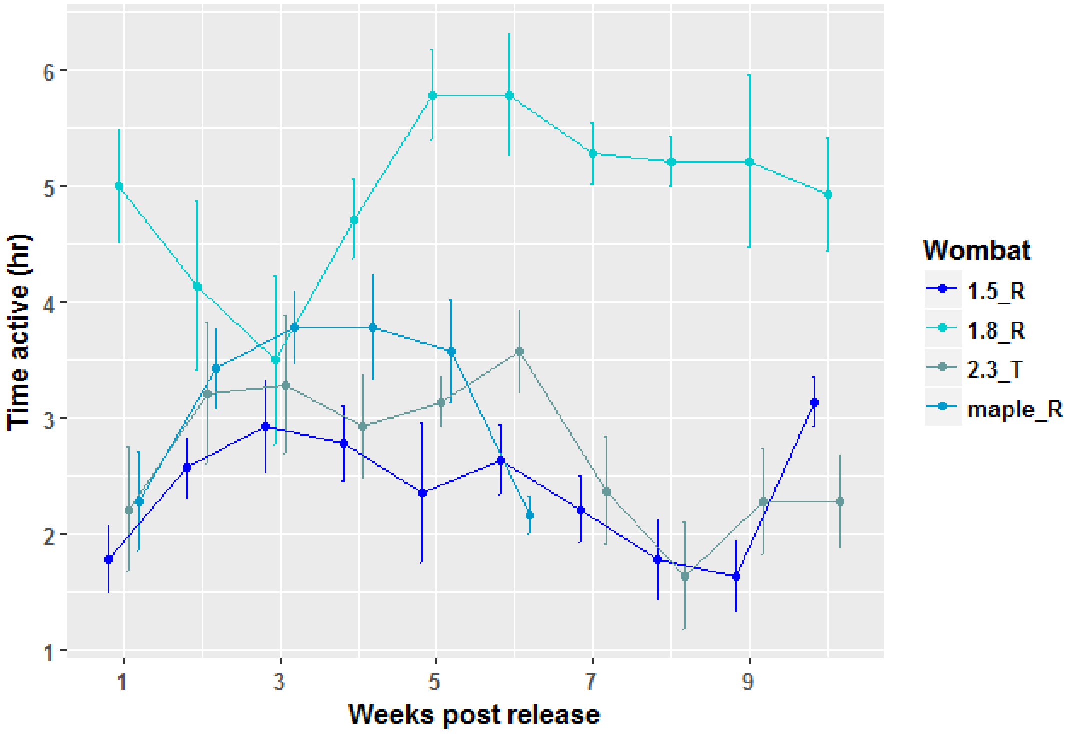

Figure 3.9. The mean time spent above ground (hr) by each individual GPS collared *L. latifrons* per week, where 1.8 is the male and 1.5 the female residents (R) released in 2010, Maple is the resident female released in March of 2013 and 2.3 is the female translocated (T) in 2010.

Incremental area analyses revealed the home ranges of each of the residents stabilised at 50 locations, while the home range of the translocated *L. latifrons* stabilised at 180 locations in the first month, and 40 locations thereafter. Sufficient locations were recorded to estimate reliable monthly home ranges for the translocated *L. latifrons* and all three residents. There was no correlation between the number of fixes obtained per month and the overall home range size (MCP: r = 0.26, *n* = 11, *P* = 0.94). The translocated *L. latifrons* exhibited extensive exploratory behaviour, as evident from its mean MCP home range of 92.02 ± 118.01 ha, much greater than its mean 90% KDE of 46.81 ± 62.86 ha. In comparison, the mean MCP for the resident *L. latifrons* was 6.15 ± 4.16 ha, and 90% KDE was 4.11 ± 3.27 ha. The translocated *L. latifrons* had the largest 90% KDE home range area observed in the first month following release (119.34 ha). In comparison, Maple had the largest home range (11.56 ha) of the residents, followed by resident 1.8 (5.05 ha), both of which occurred in the first month post-release. The mean core area (50% KDE) of the translocated *L. latifrons* (10.16 ± 11.41 ha) was much greater than the residents' mean core area of 1.39 ± 1.33 ha. The translocated *L. latifrons* had the greatest core area of 23.32 ha, observed in the first month post-release. In comparison, Maple had the largest core area of a resident at 4.52 ha, and resident 1.8 the second largest at 1.43 ha, both of which were recorded in the first month post-release. The MCP, 90% KDE and core area of all four *L. latifrons* decreased overtime; none more so than the translocated *L. latifrons*, whose home range became similar in size to the residents by the third month post-release (Figure 3.10). It is important to note that the release warren for the translocated *L. latifrons* collapsed in the third week post-release, as a result of an unseasonably large rainfall event (108 mm in 2 days). The extent to which this affected the ranging behaviour of the translocated *L. latifrons* is unknown. Prior to the rainfall event, it's home range (MCP = 110 ha, 90% KDE = 50 ha, 50% KDE = 7 ha) was smaller than the monthly estimate, though it is important to note that an asymptote had not been reached at this point.

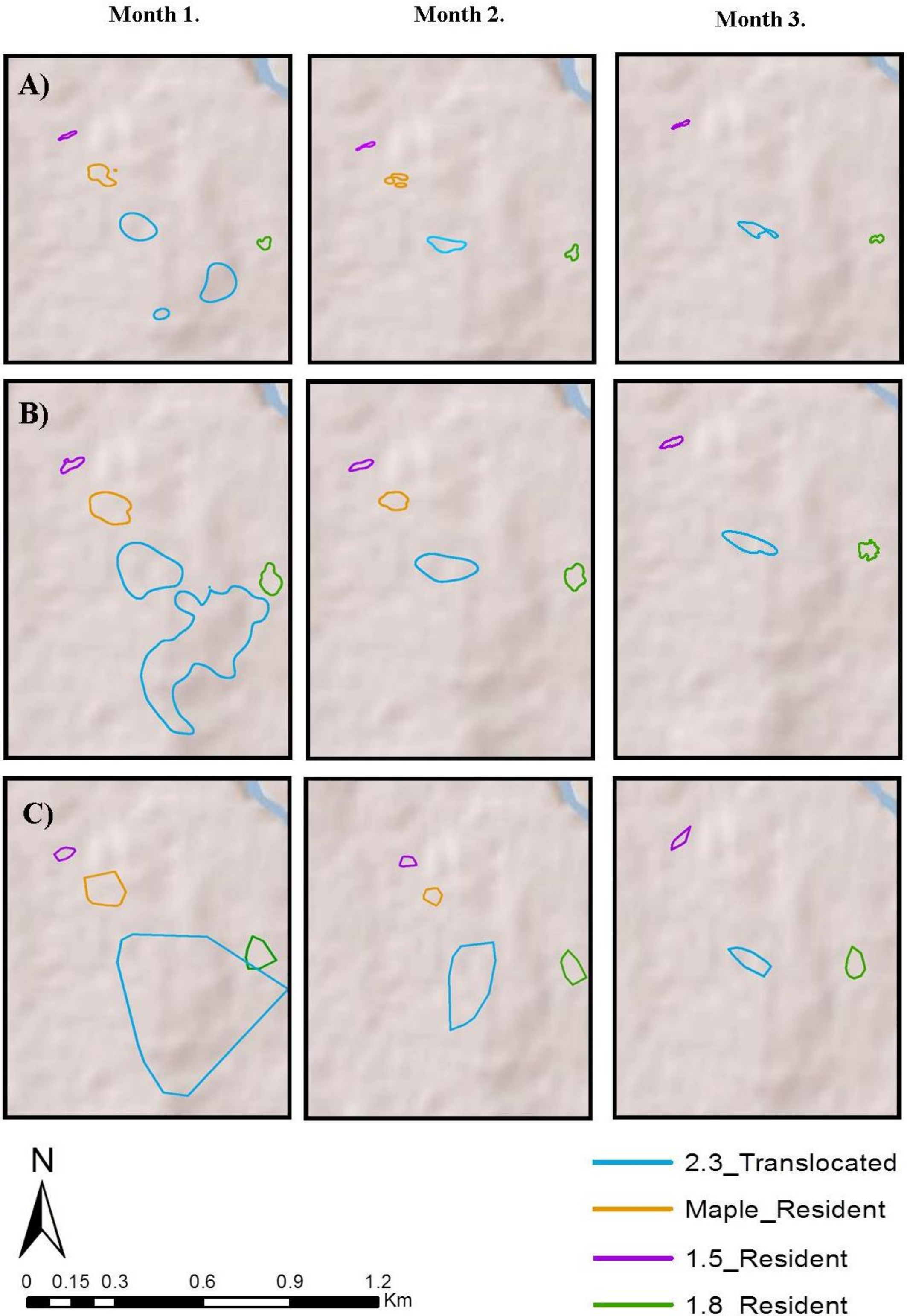

Figure 3.10. The monthly home range estimates of resident and translocated *L. latifrons* at Swan Reach, South Australia, for A) 50% KDE, B) 90% KDE and C) 100% MCP.

3.5.4 Conflict reduction

Camera malfunctions at Eudunda in 2010 and Morgan in 2013 prevented accurate calculations of the number of *L. latifrons* using the warrens prior to the translocation events and these data were thus excluded from the analysis. Throughout the remainder of the four translocation events in Eudunda, 23,861 photos of *L. latifrons* were recorded. Fewer photos of *L. latifrons* were recorded during translocation events (87.93 ± 76.02) than before (127.64 ± 104.81) or after (184.29 ± 133.91). Identification of individuals from the photos revealed that the mean ±SD number of *L. latifrons* using the warrens before (6.67 ± 1.52), during (5.61 ± 0.89), and after (6.67 ± 1.15) translocation showed little variation. In one instance the number of *L. latifrons* using the warren following a translocation event ($n = 6$) was greater than the pre-translocation number ($n = 5$) despite an *L. latifrons* being removed.

3.6 Discussion

Translocation is widely used as a non-lethal HWC mitigation measure, but its effectiveness is disputed (Massei *et al.* 2010; Germano *et al.* 2015). Mitigation translocations are often poorly planned and fail to take into account the biological needs of the species. As such, survival of translocated animals is often low, due to stress, disorientation, release into unsuitable habitat, and competition from conspecifics. In addition, mitigation translocations may either fail to resolve conflicts or transfer them to release sites. Translocation of *L. latifrons* has been suggested as a means of reducing conflicts with the agricultural sector; thus this study aimed to assess its viability. *Lasiorhinus latifrons* were translocated from areas where they were causing conflict to a property on which *L. latifrons* numbers were low. While *L. latifrons* responded well to translocation, with high survival and site fidelity, they were difficult to capture, making translocation impractical on a large scale. In addition, neighbouring *L. latifrons* quickly recolonised vacated burrows at the source site. Thus, conflicts were not resolved.

3.6.1 Trapping success

Lasiorhinus latifrons activity decreased during translocation events despite no difference in the number of individuals observed. Decreased activity was likely a result of the increased disturbance of warrens during translocation, which causes *L. latifrons* to remain underground (D. Taggart pers. comm). Digging under fences to evade traps was also frequently observed during translocation events (C. O'Brien pers. obs.). This resulted in a low overall trapping success of 5.95%. Common (V*ombatus ursinus*) and northern hairy-nosed (*L. krefftii*) wombats are also known to avoid capture by remaining underground for extended periods of time, with trapping success of *L. krefftii* as low as 1–2% (McIlroy 1977; Alan Horsup, pers. Comm. 2018). Similarly, trapping California ground squirrels using live traps placed in burrow entrances resulted in 4% capture success (Van Vuren *et al.* 1997). In contrast, the capture of Gunnison's prairie dogs (*Cynomys gunnisoni*) for translocation yielded a 38.5% trapping rate (Nelson and Theimer 2012). Trapping success was higher at Eudunda than Morgan. This may have been due to differences in the abundance of *L. latifrons* at each site, though this could not be ascertained due to camera malfunctions at Morgan. It is also possible that different soil types affected trap rates, with the harder calcareous–clay soil at Eudunda making it more difficult to dig under the fencing and evade traps than the soft clay soil at Morgan. Interestingly, trapping success was highest in November when *L. latifrons* are less active, and lowest in May, when they are more active (Finlayson *et al.* 2010). This could be attributed to energy conservation, with declining body condition in November, caused by diminishing summer food resources, resulting in less energy to expend evading traps. In contrast, improved body condition in May–July, as a result of abundant winter food resources may enable *L. latifrons* to expend more energy digging and evading traps. The time-intensive nature of trapping *L. latifrons*, particularly during heightened periods of activity when they are more likely to be involved in conflicts, suggests the use of translocation on a large scale or over the long term would be infeasible.

3.6.2 **Survival**

Two *L. latifrons* died during recovery from anaesthesia, resulting in a pre-release mortality rate of 6.25%, which is unusually high for this species (D. Taggart pers. comm.). Capture-related mortality during translocations differs considerably among species, the capture and handling methods used, and the experience of the handler (Salas *et al.* 2004; Arnemo *et al.* 2006; Lemckert *et al.* 2006; Massei *et al.* 2010). On this occasion, the deaths occurred on a day when temperatures rose above 30^{o}C. Despite following standard practices of placing animals in a cool well ventilated area with continuous monitoring, they succumbed to heat stress. Similar heat-related deaths have occurred in microbats despite the use of standard handling protocols (Lemckert *et al.* 2006). Capture protocols for *L. latifrons* were changed following this event with no trapping taking place when temperatures rose over 30^{o}C, to prevent future mortalities.

Contrary to many translocation studies (reviewed in Griffith *et al.* 1989; Fischer and Lindenmayer 2000; Germano *et al.* 2015), post-release mortality was not observed in this trial. However, 69% of translocated and 75% of resident *L. latifrons* went missing shortly after release. Translocated animals commonly disappear as a result of mortality, dispersal, or collar loss and failure (Mosillo *et al.* 1999; Adams *et al.* 2004; Bradley *et al.* 2005; Dickens *et al.* 2009; Gammons *et al.* 2009; Matthews *et al.* 2013). Following extensive search efforts, 55% of the missing *L. latifrons* were relocated or recaptured, and the majority had failed collars. Overall collar failures accounted for 50% of all missing *L. latifrons,* ruling out mortality or dispersal as the main causes of disappearances. Collar failure could have occurred as a result of antenna and wiring damage from wear and tear, as evidenced on some of the recaptured *L. latifrons*. Damaged collars with poor quality signals would be difficult to locate, particularly when signals were obscured underground. Water ingression is another possible cause of collar failures for the *L. latifrons* released in 2010, they experienced a record rainfall event (108 mm in 2 days) shortly after their release. The collars of some of the missing *L. latifrons* were recovered intact with no signs of mortality or human interference. It is likely that these collars slipped off as the *L. latifrons* lost weight over summer, a common phenomenon for this species in this environment (D. Taggart pers. comm.). Collars lost underground, particularly in limestone burrows, can be undetectable, and may have contributed to the number of missing *L. latifrons*.

Despite extensive efforts, collared *L. latifrons* were difficult to recapture, as they only spend ~26% of their time above ground (Finlayson *et al.* 2010). The failure to recapture *L. latifrons* within the first few months post-release meant the impact of translocation on their health could not be assessed. Those that were recaptured ≥ 5 months post-release were all in a condition which was to be expected for the time of year at which they were caught (D. Taggart pers. comm.). One translocated female recaptured 7 months post-release had a pouch young, providing preliminary evidence to suggest translocation does not have negative long-term effects on health and reproduction. Similarly, reintroduced northern hairy-nosed wombats reproduced within a year of being relocated (Department of Environment and Heritage Protection 2017). Because of the small sample size and a lack of long-term data, these results should be interpreted with caution.

3.6.3 Ranging behaviour

Translocated *L. latifrons* travelled further from their release locations than did residents, though the difference was not significant. Despite this, translocated *L. latifrons* displayed high release site-fidelity, establishing home ranges that included their release warrens and occupying a similar number of warrens and burrows to residents. Similarly, translocated gopher tortoises (*Gopherus polyphemus*) displayed high release-site fidelity, despite travelling further than residents, which was attributed to the use of soft release techniques and habitat suitability (Bauder *et al.* 2014). In contrast, many animals disperse considerable distances soon after translocation, as a result of homing behaviour, competition, disorientation, or exploration (Bright and Morris 1994; Sjoasen 1997; Letty *et al.* 2007; Dickens *et al.* 2009). Translocated suburban raccoons (*Procyon lotor*), for example, displayed low release-site fidelity in rural forests, with 82% travelling outside of the release area to establish dens (Mosillo *et al.* 1999). It is possible that some of the missing *L. latifrons* dispersed outside the release area and the results of this study could be due to small sample size. This is considered unlikely as *L. latifrons* has a small home range (4 ha) (Finlayson *et al.* 2005), competition from conspecifics was low, and ample vacant burrows were available; all factors that have contributed to high release site fidelity in other studies (Griffith *et al.* 1989; Truett *et al.* 2001; Letty *et al.* 2007). European otters (*Lutra lutra*) for instance, displayed higher release-site fidelity in areas with no conspecifics than those with competing conspecifics (Sjoasen 1997), and European rabbits (*Oryctolagus cuniculus)* displayed higher release-site fidelity when translocated into areas with a greater number of available burrows (Drees *et al.* 2009). Furthermore, *L. latifrons* are highly dependent on burrows to provide diurnal shelter and assist in energy conservation (Shimmin *et al.* 2002), which would discourage long-distance movements away from the release site.

The translocated *L. latifrons* initially travelled greater nightly distances and utilised a larger home range than the residents before settling into similar behaviour patterns 3 months post-release. This has also been observed in translocated swift foxes (*Vulpes velox*), which initially ranged further than their resident counterparts, before settling into similar behavioural patterns ≥ 50 days post-release (Moehrenschlager and Macdonald 2003). The greater ranging behaviour displayed by translocated animals could be due to exploration or disorientation as a result of their unfamiliarity with the release site (Pinter-Wollman *et al.* 2009). Given wombats are known to follow distinct tracks and rely heavily on olfactory cues to mark their territories (Triggs 2009), disorientation caused by a lack of familial cues is likely. The greater ranging behaviour of the translocated *L. latifrons* was also influenced by the collapse of its release warren following a record rainfall event in the third week post-release, forcing it to relocate to another warren. This could have prolonged the time taken for the translocated *L. latifrons* to settle into its new environment.

Despite travelling greater nightly distances than residents, the translocated *L. latifrons* spent less time above ground. This suggests the translocated *L. latifrons* spent more time engaged in locomotion than relaxing or foraging, when above ground, most likely because of the stress of being placed in an unfamiliar environment. The energy expended travelling long distances may have required the translocated *L. latifrons* to spend more time below ground to rest. It is also possible that the translocated *L. latifrons* spent more time below ground exploring, modifying unfamiliar warren systems, or defending burrows to establish a territory.

3.6.4 **Conflict reduction**

Human-wombat conflicts were not reduced at source sites, as neighbouring *L. latifrons* quickly recolonised the warrens from which translocated animals were removed. Landholders using culling as a control measure to reduce conflicts with *L. latifrons* have reported similar results, with neighbouring animals recolonising vacated warrens (St John and Saunders 1989). These findings are consistent with other studies, where conspecifics recolonised the vacated territories of removed animals through immigration or population growth (Bradley *et al.* 2005; Cooley *et al.* 2009). The territories of relocated golden eagles (*Aquila chrysaetos*), for example, were re-occupied by neighbouring pairs within 1–8 days of their removal (Phillips *et al.* 1991). Translocating a greater number of *L. latifrons* may result in a more sustained reduction in human–wombat conflict, but this may be unattainable given the time-consuming nature of capturing them.

3.6.5 Management implications

Although preliminary, the findings of this study indicate that translocation is not a viable conflict management tool for *L. latifrons*. Given the short time taken for neighbouring *L. latifrons* to recolonise vacated burrows, translocation failed to resolve conflicts at the source site. Translocating a larger number of *L. latifrons* may have a more sustained effect, but given the time-consuming nature of capturing *L. latifrons*, this would most likely be infeasible. Furthermore, the ongoing translocation of large numbers of *L. latifrons* would over time lead to shortages of suitable conflict free release sites within the species natural range. To reduce conflict and build a foundation for co-existence between *L. latifrons* and the agricultural sector, efforts may be better focused on developing preventative measures at the site of conflict, such as habitat manipulation and deterrent use (Blackwell *et al.* 2002a; Conover 2002; Chelliah *et al.* 2010; Kaplan *et al.* 2011).

Despite the problems associated with the use of translocation as a conflict mitigation measure, it shows promise as a conservation tool, for supplementing declining populations of *L. latifrons*. Though they initially ranged further than residents, the survival and release-site fidelity of translocated *L. latifrons* was high. However, these results should be interpreted with caution, because of the small sample size and lack of long-term data. To understand the implications of translocation on *L. latifrons*, a multi-year study with a larger sample size is recommended.

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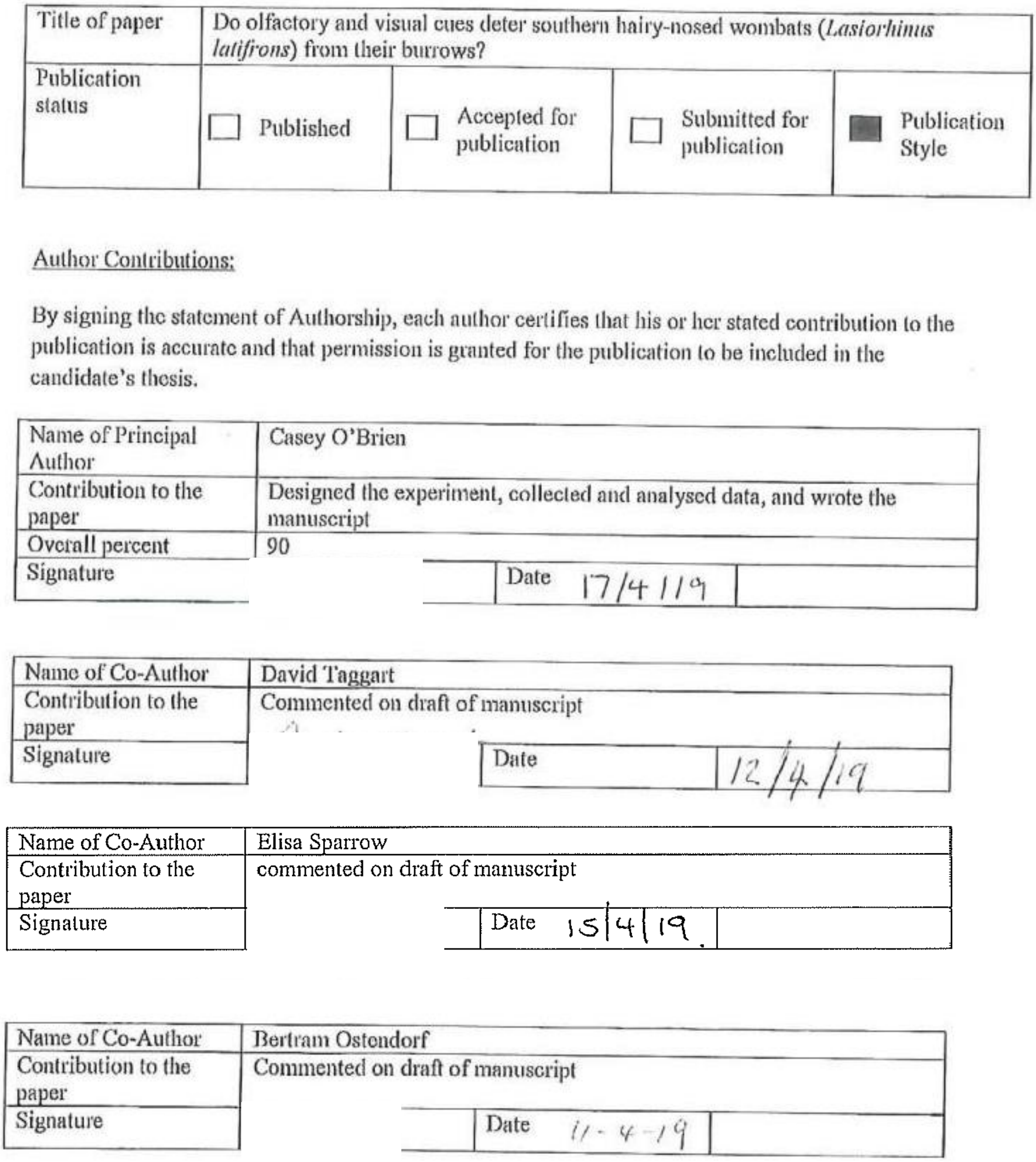
Chapter 4. Do olfactory and visual cues deter southern hairy-nosed wombats (*Lasiorhinus latifrons*) from their burrows?

A southern hairy-nosed wombat approaching a CD

Statement of authorship



Title of paper	Do olfactory and visual cues deter southern hairy-nosed wombats (*Lasiorhinus latifrons*) from their burrows?			
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Name of Principal Author	Casey O'Brien		
Contribution to the paper	Designed the experiment, collected and analysed data, and wrote the manuscript		
Overall percent	90		
Signature		Date	17/4/19

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4.1 Abstract

As agricultural production expands to meet the resource demands of growing human populations thereby shrinking wild habitats, conflicts with wildlife over limited natural resources are intensifying. Human–wildlife conflicts threaten a huge diversity of species and affect the livelihoods of millions of people. Thus there is an increasing need to develop management strategies that meet the needs of both humans and wildlife. The use of deterrents could provide a potentially cost effective and efficient means of reducing wildlife damage to agriculture. For many conflict species like the southern hairy-nosed wombat (*Lasiorhinus latifrons*), there is a lack of scientific data on the effectiveness of potential deterrents in natural environments. Anecdotal accounts suggest that blood and bone (BB), a plant fertilizer, and compact discs (CDs) may deter *L. latifrons.* This study aimed to assess their effectiveness in deterring free-living *L. latifrons* from their burrows. Remote cameras were used to monitor *L. latifrons* activity and behaviour before and after BB and CDs were applied to burrows. Mixed-effects models revealed a significant decrease in the number of visits to the burrows following the application of CDs. Blood and bone did not deter *L. latifrons* from their burrows, but increased exploratory, and decreased travelling behaviour were observed. This suggests visual cues may be more effective in repelling *L. latifrons* from their burrows, as they are more easily detected from further away, whereas olfactory cues require closer investigation.

Keywords: deterrent, *Lasiorhinus latifrons*, human–wildlife conflict, repellent, wildlife management.

4.2 Introduction

As the world's landscapes become increasingly human dominated, and competition for limited space and natural resources intensifies, human–wildlife conflicts (HWCs) are escalating (Madden 2004; Woodroffe *et al.* 2005c). Conflicts commonly arise when wildlife compete with humans for agricultural resources and affect people's livelihoods (Mason and Clark 1992; Conover *et al.* 1995; Barua *et al.* 2013). Humans typically retaliate with lethal control, often resulting in species range contractions and population isolation (Fuller *et al.* 1992; Woodroffe *et al.* 2005b; Powell 2006). Human persecution is thought to be one of the leading causes of the extirpation of African wild dogs (*Lycaon pictus*) from 93% of their former range (Ginsberg *et al.* 1997; Wolf and Ripple 2017). Similarly, the range of black-tailed prairie dogs (*Cynomys ludovicianus*) has contracted by 99%, largely due to extensive eradication campaigns (Dolan 1999). There is an increasing need to develop conflict mitigation strategies that ensure production sustainability and conserve species. Deterrents designed to capitalise on an animal's mechanisms of threat detection, with aversive stimuli could provide a means of reducing wildlife damage (Apfelbach *et al.* 2005; Breitenmoser *et al.* 2005; Woodroffe *et al.* 2005a; Schakner and Blumstein 2013).

For a wide variety of devices used as deterrents, anecdotal and testimonial reports are often the only evidence of their success (Koehler *et al.* 1990; Mason 1998; Barlow *et al.* 2010). Even where devices have been quantitatively field tested, studies often produce conflicting results. Roadside reflectors, for instance, effectively reduced deer-vehicle collisions in some studies (Gladfelter 1984; Schafer and Penland 1985) but were ineffective in others (Reeve and Anderson 1993; D'Angelo *et al.* 2006). Similarly, the use of reflective tape to repel birds from agricultural crops has provided mixed results (Bruggers *et al.* 1986; Dolbeer *et al.* 1986; Conover and Dolbeer 1989). Variations in experimental design, sampling periods, data analyses, and measures of repellence can all produce differing results and make it difficult to compare studies. Numerous uncontrollable factors in field experiments, also influence the manner in which animals detect and respond to deterrents (Lima and Dill 1990; Frid and Dill 2002; Schakner and Blumstein 2013). Distance to safety, for instance, can influence animals' responses to deterrents (Bonenfant and Kramer 1996; Lima 1998), as observed in yellow-bellied marmots (*Marmota flaviventris*), whose perceptions of risk increased when further from the safety of a burrow (Monclus *et al.* 2015). Individual variation in fitness, based on factors such as age can affect the level of perceived threat and responses to it (Chivers *et al.* 2000; Borowski 2002; Lind and Cresswell 2005; Ramp *et al.* 2005). Juvenile eastern quolls (*Dasyurus viverrinus*), for example, exhibited stronger responses to predator vocalisations than less vulnerable and more experienced adults (Jones *et al.* 2004). A greater understanding of the manner in which animals respond to threats and the factors influencing them is needed to guide the development of deterrents.

Even where devices successfully deter animals, they often have a limited duration of success. Chemical substances can decompose or become diluted over time, requiring constant reapplication (Bytheway *et al.* 2013). Chilli rope used to deter elephants (*Loxodonta africana*) from crop raiding is less effective in areas of high rainfall, as the chilli constantly washes away (Chelliah *et al.* 2010). Habituation can also be a major limiting factor in the long-term success of deterrents, with animals becoming less wary of devices that are used continuously, at low intensities, in predictable patterns, or when unaccompanied by negative reinforcements (Koehler *et al.* 1990; Gilsdorf *et al.* 2002; Shivik *et al.* 2003; Breitenmoser *et al.* 2005). For example, gulls (*Larus* spp.) became habituated to acoustic and visual stimuli within a week of their constant use (Soldatini *et al.* 2008). Using devices with moving components, altering their positions, using them sparingly, or in combination with other deterrents may prolong their effectiveness (Koehler *et al.* 1990; Belant *et al.* 1996; Mason 1998; Shivik *et al.* 2003).

Damage to agricultural land by southern hairy-nosed wombats (*Lasiorhinus latifrons*) is widespread throughout South Australia. *Lasiorhinus latifrons* provide grazing competition for stock and consume crops (St John 1998; Stott 1998). The burrowing activity of *L. latifrons* can pose safety risks for farm personnel, and damage roads, fences, and machinery, placing large financial burdens on landholders (St John and Saunders 1989; Stott 1998; Sparrow *et al.* 2011). Conflicts are predominantly managed by culling animals, under a permit system monitored by the Department of Environment, Water, and Natural Resources (*National Parks and Wildlife Act 1972* (SA)). Culling often fails to provide long-term relief from conflict, and its effects on *L. latifrons* populations are poorly understood, raising ethical and conservation concerns for the species (St John and Saunders 1989; Tartowski and Stelmann 1998; Sparrow *et al.* 2011). The use of deterrents could provide an alternative means for reducing *L. latifrons* damage and may enhance co-existence with the agricultural sector. Consultation with landholders identified two potential deterrents that were thought to affect *L. latifrons*: compact discs (CDs) and blood and bone (BB), a plant fertiliser (Brunnings Pty Ltd). Blood meal, bone meal, and bone oil-based products have been found to deter mammals from desired resources (Atkinson and Macdonald 1994; Witmer *et al.* 2000; Kimball *et al.* 2009), whereas reflective devices have produced mixed reports of success (Schafer and Penland 1985; Ramp and Croft 2006; Kaplan and O'Riain 2015). This study aimed to evaluate the effectiveness of CDs and BB in deterring free-living *L. latifrons* from using their burrows and gain insight into the mechanisms involved in threat detection by evaluating changes in *L. latifrons* behaviour before and after treatment application.

4.3 Methods

4.3.1 Study site

This study was conducted on a sheep grazing property in Swan Reach (34.34 ºS, 139.36 ºE), located within the Murraylands of South Australia, approximately 100 km northeast of Adelaide. The region has a semi-arid environment characterised by low and unpredictable rainfall (approx. 300 mm annually), and is frequently subjected to periods of drought (Finlayson *et al.* 2005). The vegetation consists predominantly of grazing land with remnant patches of mallee eucalypts, interspersed with saltbush (*Atriplex* spp.) and blue bush (*Maireana* spp.). The understorey is dominated by introduced weed species, such as thread iris (*Moraea setifolia)* and ward's weed (*Carrichtera annua*), with some remnant native grasses (S*tipa* spp.). The soils consist predominantly of alluvial clay, supported by layers of sheet limestone (calcrete).

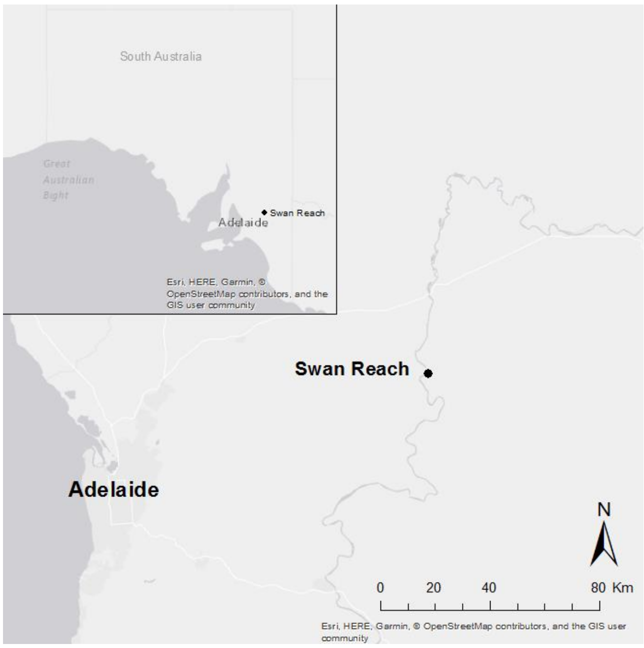

Figure 4.1. The study site, Swan Reach, located approximately 140 km north east of Adelaide in the Murraylands of South Australia.

4.3.2 Experimental design

The effects of BB and CDs on *L. latifrons* activity and behaviour were examined in spring 2013. Ten warrens > 100 m apart and each containing a mixture of 20–30 active/inactive burrows were selected for the study to ensure potentially displaced *L. latifrons* had alternative warrens/burrows to retreat to within their home range. Warrens were located in clay soil and had similar surrounding vegetation to control for environmental factors that may affect an animal's threat perceptions (Lima and Dill 1990). Within each warren, three active burrows located ≥ 6 m apart were monitored continuously throughout trials, using ScoutGuard KG680V and Reconyx HC600 motion sensor cameras. The cameras were attached to star pickets 40–50 cm from the ground and positioned 2–3 m from the burrow entrance (Figure 4.2). They were programmed to take three photographs per trigger, with a 1 s interval between sequential triggers. The activity and behaviour of *L. latifrons* were monitored 2 weeks before and 2 weeks after treatments were applied to burrows. The three burrows being monitored were randomly selected to receive one of three treatments, consisting of a continuous control with no treatment, a CD fixed to a garden stake, or BB, a plant fertilizer made of animal bones, slaughterhouse waste products, and animal manure. The CDs were positioned 15 cm above the ground, parallel to the track leading into the burrow, as wombats have better lateral than frontal vision, because of the positioning of their eyes (Smythe 1975; Wells 1989). A metric cup (250 ml) of BB was dispersed evenly across the soil. All treatments were applied 30–40 cm from the burrow entrance. The cameras were checked on a weekly basis to download data and replace batteries throughout the trials. This research was performed under University of Adelaide animal ethics permit s2011-197C, and a scientific research permit A25828-1 obtained from the Department of Environment and Water.

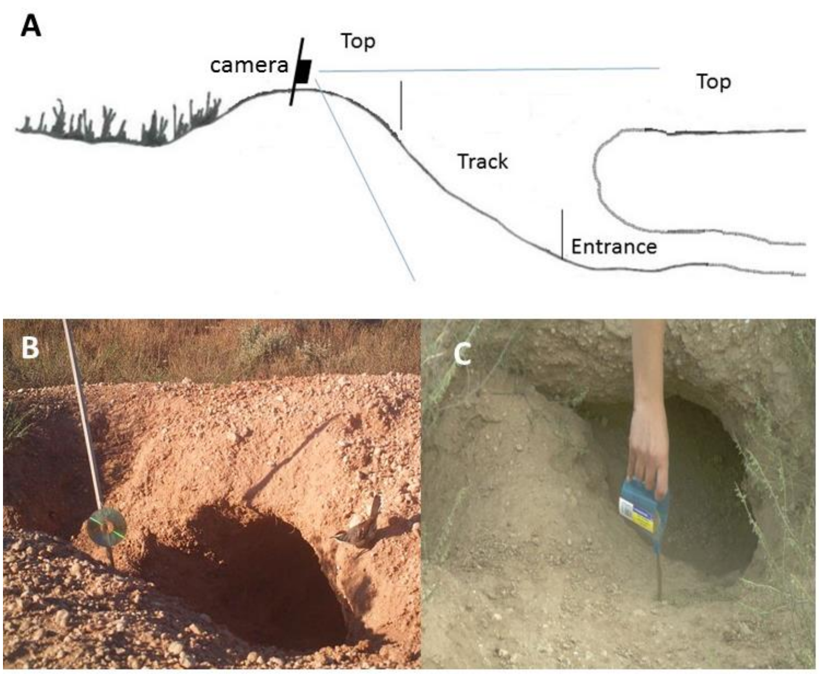

Figure 4.2. A) A cross-section of a *L. latifrons* burrow, showing the location of the camera and the demarcation of the three locations used to analyse space use, B) A CD positioned in front of a burrow, and C) Blood and bone being applied to the entrance of a burrow.

Table 4.1. Ethogram of *L. latifrons* behaviours.

Behaviour	**Classification**
Vigilant	Sitting, lying, or standing with head up in alert position or scanning of the head
Resting	Sitting, lying, or standing, awake and relaxing, scratching, rubbing, or rolling
Social	Interacting with another wombat in a friendly manner
Travelling	Walking or running at a constant gait without stopping
Exploratory	Digging/scratching at dirt or sniffing the ground or air
Defensive	Blocking the burrow entrance with backside, or chasing another wombat away
Unknown	Behaviour unable to be discerned, wombat not in full view

4.3.3 Photo analyses

Due to the large number of photos acquired, a subset of the data totalling 5 nights immediately before and 5 nights immediately after treatment application were analysed. Although *L. latifrons* are predominantly nocturnal with peak activity between 6 pm and 6 am, they are known to be active during the day. Thus, for the purpose of analysis, nights constituted a 24-hour period, beginning and ending at 6 am. For each night of the trial, the number and duration(s) of visits made to the burrow were recorded. Visits were considered the same if consecutive photos were ≤ 15 s apart, unless the *L. latifrons* was identified as a different individual. Where possible, individuals were identified based on their fur patterns, ear markings and scars (Figure 4.3) and classified as adult or juvenile (a quarter the size of an adult) age classes, to account for individual variations in responses. The proportion of time *L. latifrons* were observed in seven main behaviours (Table 4.1) was recorded during each visit, as behaviours are known to change when animals react to threats (Apfelbach *et al.* 2005). Behaviours were analysed in 1 s intervals, as vigilance in common wombats (*Vombatus ursinus*) has been observed to last for as little as 1 s (Favreau *et al.* 2009). The proportion of time *L. latifrons* were observed at the top of the burrow, on the track into the burrow, and at the entrance of the burrow were also recorded for each visit, to determine if the application of odours resulted in a shift in space use. To distinguish treatment effects from natural temporal fluctuations in *L. latifrons* behaviour, moon phase, nightly rainfall, and minimum overnight temperature, were extracted from the Australian Bureau of Meteorology's (ABM's) nearest weather stations, in Swan Reach and Nuriootpa (54 km away) (ABM 2013).

Figure 4.3. Samples of images captured with the motion sensor cameras, showing different individuals, identified based upon fur patterns, scars, and ear markings.

4.3.4 **Statistical analyses**

Mixed models were used to examine the effects of treatments on *L. latifrons* behaviour, in *R-3.3.2* (R Core Team 2014). Burrows with low levels of activity (≤ 5 visits before treatment application) and visits by juvenile *L. latifrons* were excluded from the analysis, because of insufficient data. As *L. latifrons* behaviours could not be classified in visits ≤ 2 s, and to reduce the effects of delayed triggers failing to record total visit duration, only visits > 2 s were included in the analysis. Rainfall data were not included in the models, as rainfall was negligible throughout the trial. Individual variations in responses were not taken into account, as *L. latifrons* were unable to be identified in a large proportion of the visits. This may violate the assumptions of independence, as multiple interactions from one individual may have occurred within the same treatment over time, potentially biasing the results by responses of one of a few individuals. However, burrows were predominantly used by the same individuals, therefore including the warren by burrow interaction as a random effect would account for some of the individual variability in responses.

Variation in the duration of visits to the burrows between trial phases (before/after treatment application) was analysed using linear mixed-effects models in the *Lme4* package (Bates *et al.* 2015). Visit duration was log transformed to meet the assumptions of normality. All models were fitted with the fixed interaction between treatment by trial phase and the random interaction effects of warren by burrow, warren by burrow by night, and night, to account for repeated observations within warrens and burrows across multiple nights. To determine if *L. latifrons* became less wary of the treatments, time (night of the trial phase (1–5)) was fitted as an interaction term with treatment by trial phase. The effects of weather on *L. latifrons* behaviour, was assessed by fitting minimum overnight temperature and moon phase as fixed factors. Models were compared using Akaike's Information Criterion (AIC), Akaike weights (w*i*), and behaviour of model residuals. Post hoc differences between the trial phases (before and after treatment application) for each treatment were assessed using planned comparisons of model means in the *multcomp* package (Hothorn *et al.* 2008).

The number of visits to burrows was compared between trial phases using generalised linear mixed-effects models in the *Lme4* package (Bates *et al.* 2015). All models were fitted with the fixed interaction between treatment by trial phase and the random interaction effect of warren by burrow, to account for repeated measures within warrens and burrows. Additional models included the fixed factors of minimum overnight temperature and moon phase, to distinguish between treatment effects and natural temporal fluctuations in *L. latifrons* behaviour. Model comparisons and post hoc tests were done using the same approach described for the visit duration analysis.

Among the seven behaviours (Table 4.1) observed throughout the trials, there were too few observations of resting and defensive behaviour to enable statistical analysis. The proportion of time *L. latifrons* spent in the remaining five behaviours, and in each location was compared between trial stages for each treatment using mixed-effects beta regression models in the *glmmADMB* package (Fournier *et al.* 2012). Each model contained the fixed interaction between treatment by trial phase, and the random interaction effect of warren by burrow, to account for repeated measures within warrens and burrows. To determine if *L. latifrons* became less wary of the treatments, time (1–5) was fitted as an interaction with treatment by trial phase. Additional models included the fixed factors of minimum overnight temperature and moon phase. Model comparisons and post-hoc tests were done using the same approach described for the visit duration analysis.

4.4 Results

Thirty burrows within 10 warrens were monitored over the duration of the experiment. Approximately 59,800 photos of *L. latifrons* were collected during the trials. Sufficient data were obtained from 15 burrows; four treated with CDs, seven with BB and four controls. Throughout the study, 422 visits by adult *L. latifrons* of ≥ 3 s in duration were made to the burrows. Fifty-two individual *L. latifrons* were identified from 215 of the visits, leaving 207 visits in which individuals were unable to be identified. Throughout the trial, mean nightly temperature lows were 6.37 ± 3.57 ^{o}C, moon phase ranged between a quarter and full moon and rainfall was negligible.

4.4.1 Visit duration

The best performing model for the duration of time *L. latifrons* spent visiting burrows was the intercept-only model, D0 (Table 4.2). The duration of visits did not differ significantly between trial phases for the treatments of BB, CDs and the control ($P = 0.41$, $P = 1.00$, $P = 1.00$ respectively, Figure 4.4).

Table 4.2. Comparison of linear-mixed effects models used to assess differences in the duration of *L. latifrons* visits to burrows, between trial phases. All models were fitted with the random effects of warren by burrow and warren by night. Fixed factors included trial phase (Tp), treatment (Tr), time (T), minimum overnight temperature (Mt), and moon phase (Mp). ΔAIC represents the difference in AIC from the model of best fit, highlighted in bold, and wi is the Akaike weight of the model.

Model	Linear form	*df*	Loglik	AIC	ΔAIC	w*i*
D0	**1**	**4**	**-577.28**	**1162.56**	**0.00**	**0.68**
D1	Tp*Tr	9	-573.99	1165.98	3.42	0.12
D2	Tp*Tr/T	15	-572.61	1175.22	12.66	0.00
D3	Tp*Tr + Mt	10	-573.26	1166.52	3.96	0.09
D4	Tp*Tr + Mp	11	-573.13	1167.62	5.06	0.05
D5	Tp*Tr + Mt + Mp	11	-572.85	1167.71	5.15	0.05

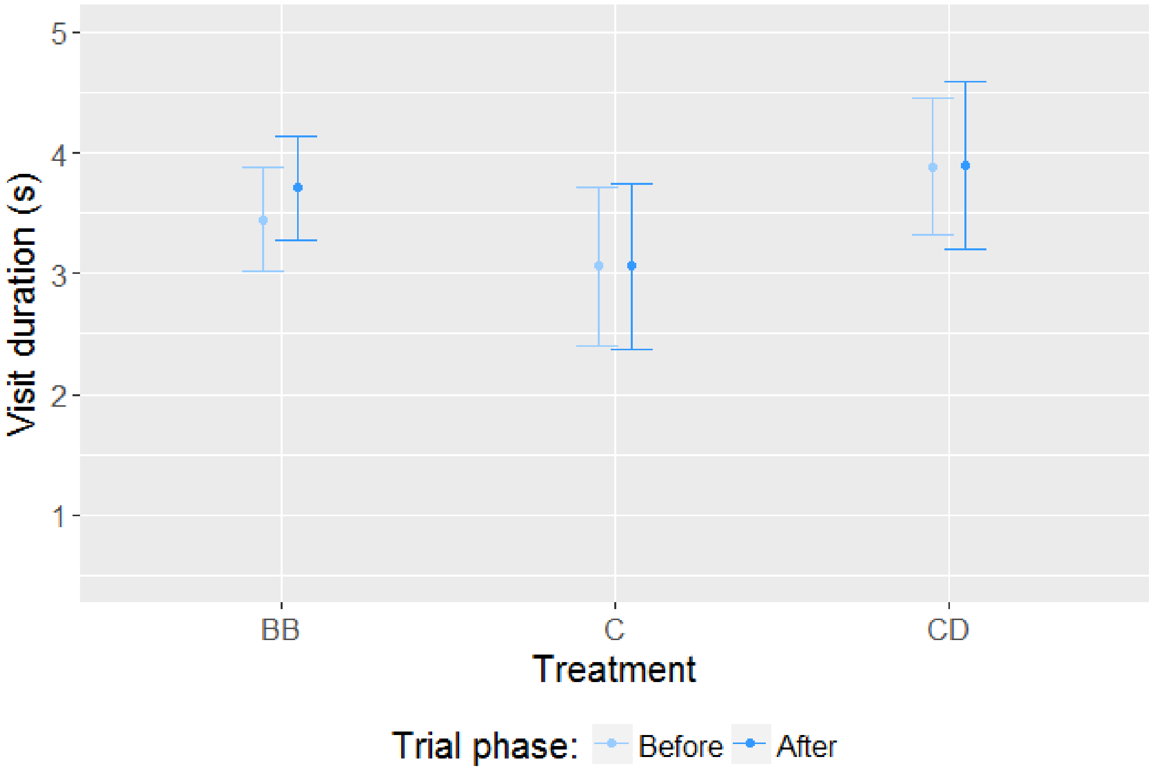

Figure 4.4. Comparison of the duration of visits (mean and 95% confidence intervals) *L. latifrons* made to the burrows before and after the treatments of blood and bone (BB) (*n* = 7), compact discs (CD) (*n* = 4) and the control (C) (*n* = 4), derived from the model D1 (see Table 4.2). Differences in visit duration between trial phases were not significant for all treatments.

4.4.2 Number of visits

When considering the number of *L. latifrons* visits to the burrows, the top performing model was V1, which contained the treatment by trial phase interaction (Table 4.3). Post hoc comparisons revealed a significant decrease in the number of visits to the burrows following the application of CD's (*P* = 0.02). No significant difference was found in the number of visits to the burrows between the trial phases, for either the BB or control treatments (*P* = 0.81, *P* = 0.41 respectively, Figure 4.5).

Table 4.3. Comparison of the generalised linear mixed-effects models used to determine the differences in the number of visits *L. latifrons* made to burrows between trial phases. All models were fitted with a negative binomial distribution and the random interaction effect of warren by burrow. The fixed variables included trial phase (Tp), treatment (Tr), time (T), minimum overnight temperature (Mt), and moon phase (Mp). ΔAIC represents the difference in AIC from the model of best fit, highlighted in bold, and *wi* is the Akaike weight of the model.

Model	Linear form	*df*	Loglik	AIC	ΔAIC	*wi*
V0	1	2	-225.20	454.40	1.48	0.17
V1	**Tp*Tr**	**7**	**-219.46**	**452.92**	**0.00**	**0.35**
V2	Tp*Tr +Mt	8	-218.50	452.99	0.07	0.34
V3	Tp*Tr + Mp	8	-219.20	454.80	1.80	0.14

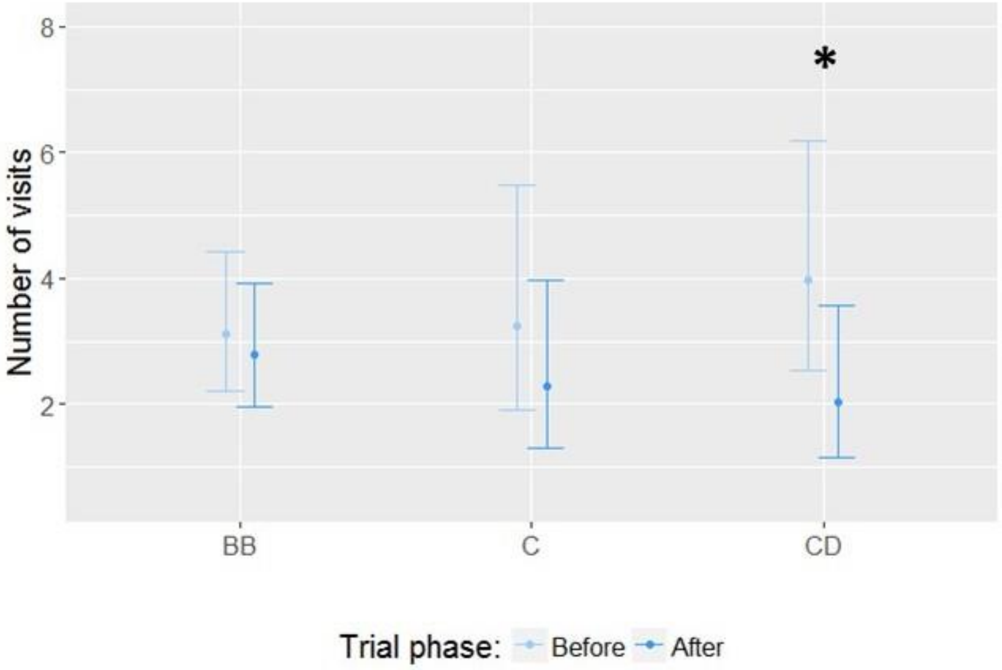

Figure 4.5. Comparisons between the number of visits (mean and confidence intervals) *L. latifrons* made to the burrows before and after the treatments of blood and bone (BB), compact discs (CD) and the control (C) were applied. Asterisks indicate significant differences between the trial phases derived from generalised linear mixed-effects models and post hoc tests.

4.4.3 **Behaviours**

The top performing model for exploratory behaviour, E1, contained the treatment by trial phase interaction (Table 4.4). Post hoc comparisons of the top model revealed *L. latifrons* spent more time in exploratory behaviour following the application of BB (*P* = 0.01, Figure 4.6). There was no significant difference in the proportion of time *L. latifrons* were observed in exploratory behaviour, between trial phases for the CD or control treatments (*P* = 0.99, *P* = 0.89 respectively, Figure 4.6). For travelling behaviour, the model of best fit, T1, contained the treatment by trial phase interaction (Table 4.4). Post hoc comparisons revealed *L. latifrons* spent significantly less time engaged in travelling behaviour following the application of BB (*P* = 0.05, Figure 4.6). There was no significant difference in the proportion of time *L. latifrons* spent in travelling behaviour, between trial phases for the CD or control treatments (*P* = 0.95, *P* = 0.93 respectively, Figure 4.6).

The top model for vigilant behaviour was the null model, V0 (Table 4.4). There was no significant difference in the proportion of time *L. latifrons* were observed in vigilant behaviour between trial phases for the treatments of BB, CD's or the control (*P* = 0.99, *P* = 0.82, *P* = 0.94 respectively, Figure 4.6). For resting behaviour, the best performing model was the null model, R0 (Table 4.4). There was no significant difference in the proportion of time *L. latifrons* were observed in resting behaviour between trial phases for BB, CD or control treatments (*P* = 0.99, *P* = 0.98, *P* = 0.86 respectively, Figure 4.6). The top model for unknown behaviour was the null model, U0 (Table 4.4). There was no significant difference in the proportion of time *L. latifrons* were observed in unknown behaviour between trial phases for the BB, CD or control treatments (*P* = 1.00, *P* = 0.41, *P* = 0.96 respectively, Figure 4.6).

Table 4.4. Comparison of the mixed-effects beta regression models used to determine the differences in the proportion of time *L. latifrons* spent in individual behaviours between trial phases. All models included the random effect of warren by burrow. The fixed factors included trial phase (Tp), treatment (Tr), time (T), minimum overnight temperature (Mt), and moon phase (Mp). ΔAIC represents the difference in AIC from the models of best fit, are highlighted in bold, and *wi* is the Akaike weight of the model.

Model	**Linear form**	***df***	**Loglik**	**AIC**	**ΔAIC**	***wi***
Exploratory						
E0	1	3	857.39	-1708.78	0.69	0.29
E1	**Tp*Tr**	8	**862.74**	**-1709.47**	**0.00**	**0.42**
E2	Tp*Tr /T	14	864.30	-1700.59	8.88	0.01
E3	Tp*Tr + Mp	10	863.06	-1706.12	3.35	0.08
E4	Tp*Tr + Mt	9	862.97	-1707.93	1.54	0.19
Travelling						
T0	1	3	760.38	-1514.77	1.06	0.28
T1	**Tp*Tr**	8	**765.92**	**-1515.83**	**0.00**	**0.47**
T2	Tp*Tr /T	14	766.49	-1504.98	10.85	0.00
T3	Tp*Tr + Mp	10	766.08	-1512.16	3.67	0.08
T4	Tp*Tr + Mt	9	765.93	-1513.85	1.98	0.17
Vigilant						
V0	**1**	3	**917.62**	**-1829.24**	**0.00**	**0.96**
V1	Tp*Tr	8	918.84	-1821.67	7.57	0.02
V2	Tp*Tr /T	14	920.03	-1812.05	17.19	< 0.00
V3	Tp*Tr + Mp	10	919.37	-1818.73	10.51	0.01
V4	Tp*Tr + Mt	9	919.52	-1821.04	8.20	0.02
Resting						
R0	**1**	3	**1118.08**	**-2230.16**	**0.00**	**0.98**
R1	Tp*Tr	8	1118.50	-2220.90	9.26	0.01
R2	Tp*Tr /T	14	1118.98	-2209.96	20.20	< 0.00
R3	Tp*Tr + Mp	10	1118.60	-2217.26	12.90	0.00
R4	Tp*Tr + Mt	9	1119.20	-2220.44	9.72	0.01
Unknown						
U0	**1**	3	**1074.56**	**-2143.12**	**0.00**	**0.89**
U1	Tp*Tr	8	1076.90	-2137.88	5.24	0.06
U2	Tp*Tr /T	14	1079.52	-2131.04	12.08	0.00
U3	Tp*Tr + Mp	10	1077.80	-2135.54	7.58	0.02
U4	Tp*Tr + Mt	9	1077.10	-2136.22	6.90	0.03

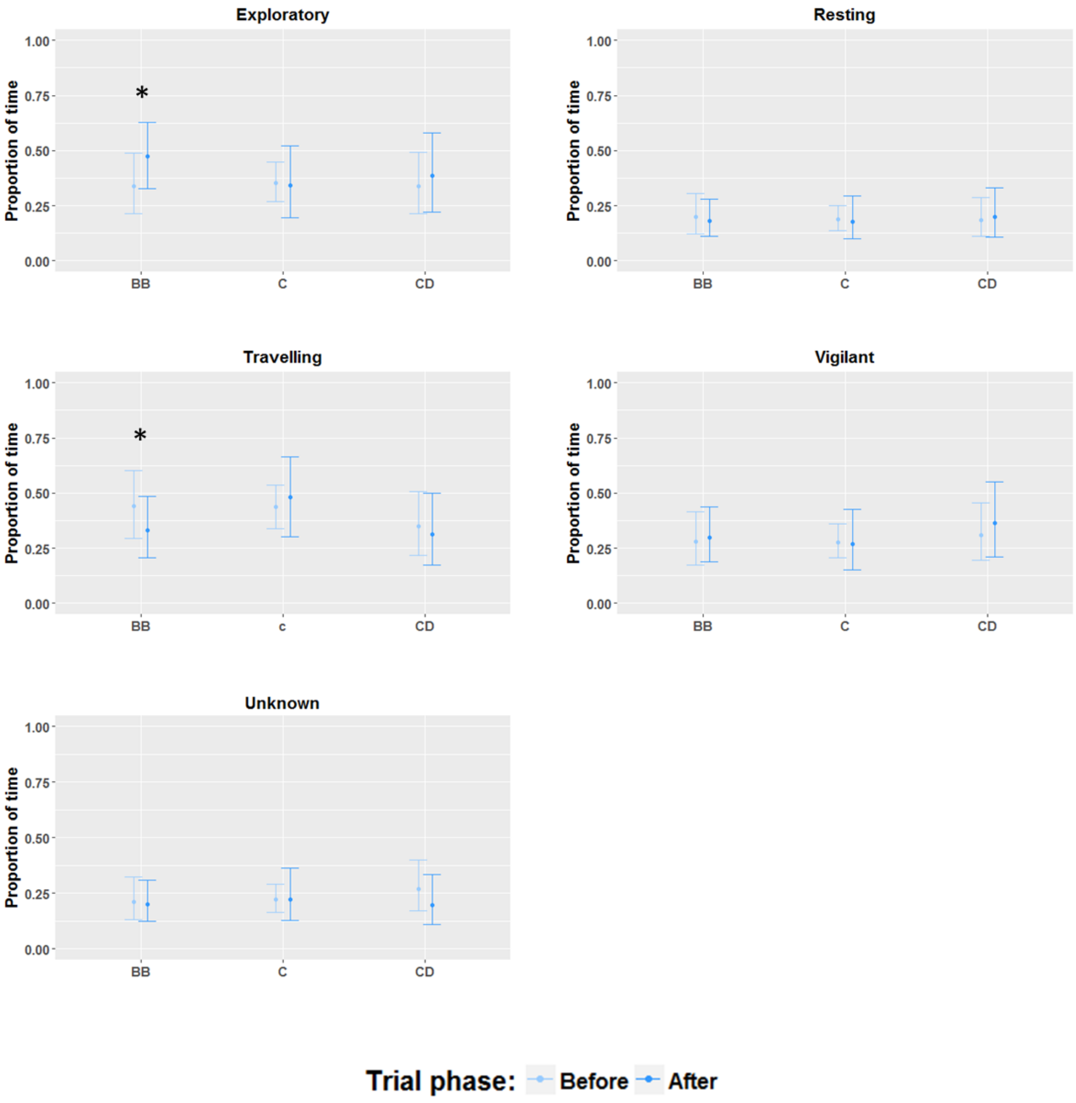

Figure 4.6. Comparison of the proportion of time (mean and confidence intervals) *L. latifrons* spent in exploratory, resting, travelling, vigilant, and unknown behaviour before and after the treatments of blood and bone (BB), compact discs (CD) and the control (C) were applied. Asterisks indicate significant differences between the trial phases derived from beta models and post hoc tests.

4.4.4 Locations

The top performing model for the proportion of time *L. latifrons* spent at the top of the burrow was the null model, LT0 (Table 4.5). There was no significant difference in the proportion of time *L. latifrons* were observed at the top of the burrow between trial phases for the treatments of BB, CDs or the control (*P* = 0.94, *P* = 0.72, *P* = 0.99 respectively, Figure 4.7). The null model, LU0, was the top performing model for the proportion of time *L. latifrons* spent on the track of the burrow (Table 4.5). There was no significant difference in the proportion of time *L. latifrons* were observed on the track of the burrow between trial phases for the BB, CD, or control treatments (*P* = 0.85, *P* = 0.77, *P* = 0.91 respectively, Figure 4.7). The top model for the proportion of time *L. latifrons* were observed at the entrance to the burrow was the null model, LE0 (Table 4.5). There was no significant difference in the proportion of time *L. latifrons* were observed at the entrance of the burrow between trial phases for the BB, CD, and control treatments (*P* = 0.99, *P* = 0.35, *P* = 0.91 respectively, Figure 4.7).

Table 4.5. Comparison of the mixed-effects beta regression models used to determine the differences in the proportion of time *L. latifrons* spent in individual locations between trial phases. All models included the random effect of warren by burrow. The fixed factors included trial phase (Tp), treatment (Tr), time (T), minimum overnight temperature (Mt), and moon phase (Mp). ΔAIC represents the difference in AIC from the models of best fit, which are highlighted in bold, and *wi* is the Akaike weight of the model.

Model	Linear form	*df*	Loglik	AIC	ΔAIC	*wi*
Top						
LT0	**1**	**3**	**849.46**	**-1686.92**	**0.00**	**0.833**
LT1	Tp*Tr	8	849.40	-1682.79	4.13	0.105
LT2	Tp*Tr /T	14	850.99	-1673.98	12.94	0.001
LT3	Tp*Tr + Mp	10	849.71	-1679.42	7.50	0.019
LT4	Tp*Tr + Mt	9	849.43	-1680.85	6.07	0.040
Track						
LU0	**1**	**3**	**824.23**	**-1642.45**	**0.00**	**0.935**
LU1	Tp*Tr	8	826.11	-1636.22	6.23	0.041
LU2	Tp*Tr /T	14	827.03	-1626.07	16.38	< 0.001
LU3	Tp*Tr + Mp	10	826.37	-1632.74	9.71	0.007
LU4	Tp*Tr +Mt	9	826.15	-1634.29	8.16	0.015
Entrance						
LE0	**1**	**3**	**877.97**	**-1749.93**	**0.00**	**0.604**
LE1	Tp*Tr	8	881.99	-1747.97	1.96	0.226
LE2	Tp*Tr /T	14	884.70	-1741.39	8.54	0.008
LE3	Tp*Tr + Mp	10	882.63	-1745.26	4.67	0.058
LE4	Tp*Tr + Mt	9	882.19	-1746.37	3.56	0.101

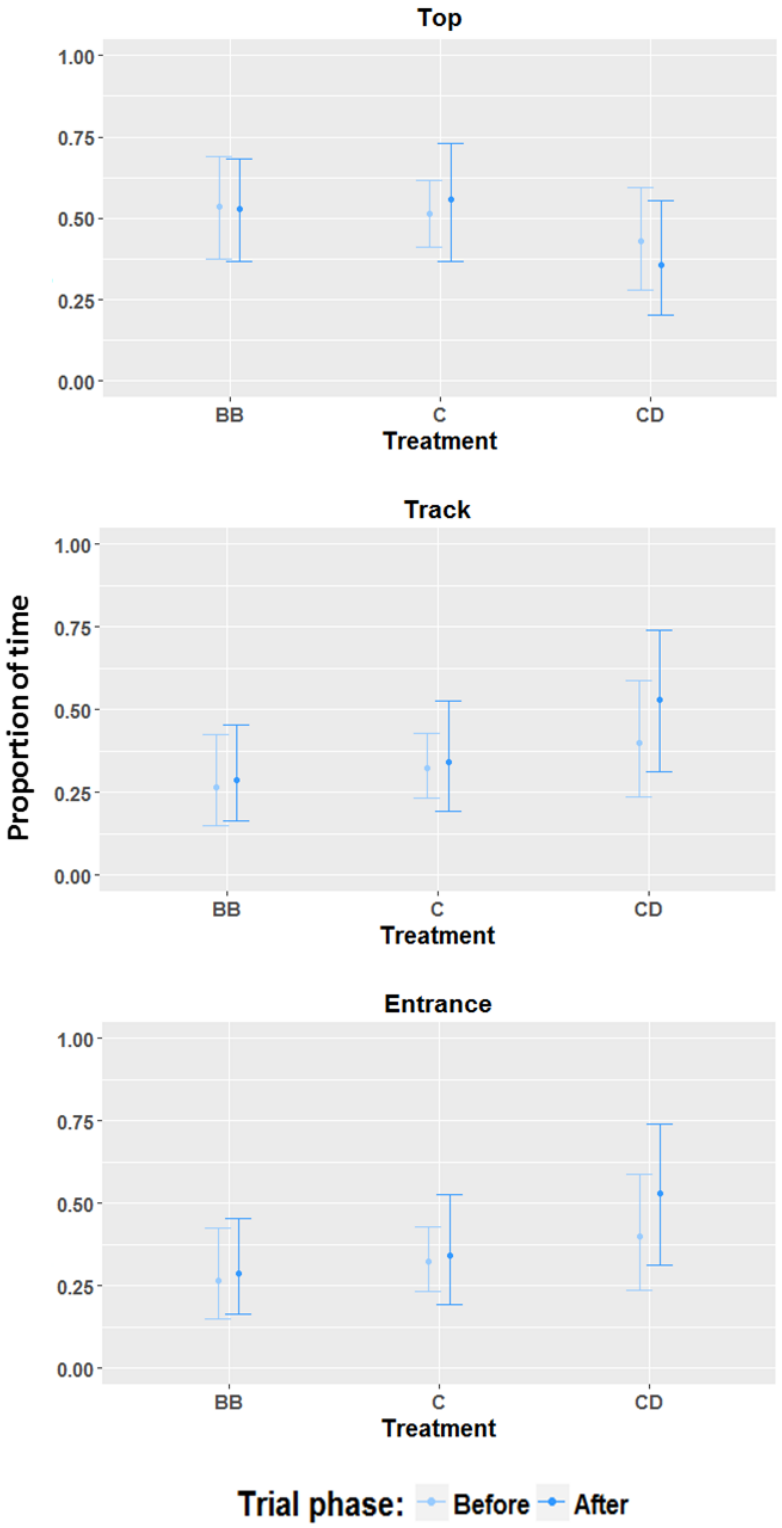

Figure 4.7. Comparisons between the proportion of time (mean and 95% confidence intervals) *L. latifrons* spent at the top, track, and entrance of the burrow, before and after the treatments of blood and bone (BB), compact discs (CD) and the control (C) were applied.

4.5 Discussion

Interest in the use of deterrents designed to discourage species from utilising agricultural resources is increasing, yet the effectiveness of many devices remains questionable (Mason 1998; Breitenmoser *et al.* 2005). Quantitative field data on the effectiveness of devices marketed as deterrents are needed to guide their use and improve HWC management. Anecdotal evidence suggests BB and CDs deter *L. latifrons*. This study aimed to quantify the effectiveness of these devices in deterring *L. latifrons* from their burrows and gain a greater understanding of the behavioural mechanisms involved in threat detection. Burrows were monitored with motion sensor cameras before and after treatments were applied, to assess changes in *L. latifrons* activity and behaviour. Blood and bone was not effective in deterring *L. latifrons* from their burrows, but CDs were. This suggests *L. latifrons* may respond more readily to visual rather than olfactory cues presented at the burrow, as they are more easily detected from further away.

4.5.1 Blood and bone

Blood and bone was not found to deter *L. latifrons* from their burrows but did result in increased exploratory and decreased travelling behaviour. Similarly, captive *L. latifrons* responded to the presentation of faeces from conspecifics and dingoes (*Canis lupus dingo*) with increased investigation of the odour source (Descovich *et al.* 2012). Increased investigation of threats may seem counterintuitive, but it is well documented in many taxa and can provide animals with the benefit of obtaining more information about potential threats (Caine and Weldon 1989; Dugatkin and Godin 1992; Fishman 1999). In contrast to this study, blood meal, bone meal, and bone oil-based products have been found to deter other species from desired resources (Harris *et al.* 1983; Atkinson and Macdonald 1994; Witmer *et al.* 2000; Kimball *et al.* 2009). DeerBuster's^{TM} sachets, a meat meal based area repellent, for instance, reduced black-tailed deer (*Odocoileus hemionus*) browsing on seedlings (Wagner and Nolte 2001). The disparity between these studies may reflect species-specific or context-dependent responses to treatments.

Rather than expend energy fleeing, avoidance measures such as reduced activity or altered space use may be more beneficial for species with effective concealment from threats (Ydenberg and Dill 1986; Merkens *et al.* 1991; Kats and Dill 1998). This was observed in badgers (*Meles meles*), which sought shelter in burrows when faced with a threat (Butler and Roper 1995). Field observations suggest *L. latifrons* respond similarly, fleeing to a burrow when faced with a physical threat (C. O'Brien, pers. obs.). Captive *L. latifrons* increased their hiding behaviour following the presentation of faeces from conspecifics and dingos (Descovich *et al.* 2012). However, their response may be heightened by their limited options for escape. It is possible that *L. latifrons* in this study responded to BB with increased hiding, or reduced ranging behaviour in a manner that was not evident from the camera data.

The application of odours inside the burrows or in open areas further from the safety of a burrow may heighten risk perceptions, resulting in greater avoidance responses. This has been observed in yellow-bellied marmots (*Marmota flaviventris*), whose perceptions of risk increased when further from the safety of a burrow (Monclus *et al.* 2015). European moles (*Talpa europaea*), avoided areas within their burrows treated with a bone oil-based repellent every square metre, for up to 28 days (Atkinson and Macdonald 1994). The more widespread distribution of the bone oil repellent may have also increased the level of risk, thereby producing a greater avoidance response. In contrast, the single application of BB at burrow entrances may have been perceived as low risk and the use of a greater volume and repeated or more widespread application of BB may elicit greater avoidance responses. This was observed in white-tailed deer (*Odocoileus virginianus*), which showed greater avoidance of predator urines when presented in larger volumes or with repeated weekly application (Swihart *et al.* 1991).

4.5.2 Compact discs

Stationary CDs elicited avoidance responses in *L. latifrons*, with a decrease in the number of visits to burrows following their application. Similarly, visual stimuli have produced avoidance responses in many species, most likely due to an immediate fear response (neophobia) of an unknown object within their environment (Bruggers *et al.* 1986; Harris and Knowlton 2001; Davidson-Nelson and Gehring 2010). Fladry, for instance, was found to deter wolves (*Canis lupus*) from depredating livestock for up to 60 days (Musiani *et al.* 2003), while reflective tape deterred blackbirds (*Agelaius phoeniceus*) from consuming crops (Dolbeer *et al.* 1986). Habituation to visual stimuli is a major limiting factor to their long-term use in a variety of bird and mammal species, particularly when unaccompanied by negative reinforcement (Koehler *et al.* 1990; Gilsdorf *et al.* 2002). This was observed in fallow deer (*Dama dama*), which quickly habituated to wildlife warning reflectors following repeated exposure (Ujvari *et al.* 1998). Similarly, a variety of bird species have been found to habituate to lasers within days to weeks of their use (Blackwell *et al.* 2002b). Though habituation to CDs was not observed in this study, and anecdotal accounts suggest it does not occur in response to moving CDs, only 5 nights of data were analysed. Further analysis of additional nights are required to determine if *L. latifrons* habituate to the presence of CDs. Though these preliminary investigations are encouraging, responses to stimuli can be context dependent. The use of mylar ribbons to deter blackbirds, for instance, is more effective when they are spaced closer together than further apart (Dolbeer *et al.* 1986). The size of novel objects was found to influence coyotes (*Canis latrans*), with larger objects producing longer avoidance responses (Windberg 1996). Further research is needed to evaluate *L. latifrons* responses to CDs in varying environmental conditions such as distance to cover, and with different methods of application and placement, to determine the optimal conditions for success.

4.5.3 Summary

Based on the preliminary findings of this study, CD's deterred *L. latifrons* from their burrows, but BB did not. Even though CDs reduced *L. latifrons* visits to burrows, they may be limited in their use due to habituation. Habituation to visual stimuli has been observed in numerous bird and mammal species, particularly when unaccompanied by negative reinforcement (Ujvari *et al.* 1998; Blackwell *et al.* 2002b; Gilsdorf *et al.* 2002). Further analysis is required to determine if *L. latifrons* habituate to the CDs. *Lasiorhinus latifrons* may have responded more readily to visual cues as they are more easily detected from further away than are olfactory cues, which require closer investigation. In addition, visual cues may represent more imminent danger than olfactory cues, which can represent both past and present danger (Biedenweg *et al.* 2011). Though BB failed to facilitate the avoidance of burrows by *L. latifrons*, it does not mean it is ineffective as a deterrent. Perceptions of risk and responses to them vary greatly and are dependent on the manner and environment in which threats are applied (Koehler *et al.* 1990; Lima and Dill 1990; Frid and Dill 2002; Schakner and Blumstein 2013). The single application of BB at the burrow entrance may not have produced a sufficient enough threat to warrant an avoidance response. Before deterrents can be recommended to landholders, a better understanding of the factors influencing *L. latifrons* perceptions of threats and responses to them is needed. Further research is needed to evaluate *L. latifrons* responses to CDs and BB in different environmental conditions such as varying distance to cover, and with different methods of application and placement, to determine the optimal conditions for success. In addition, future studies should consider conducting trials in a closed system with known individuals or capturing and marking individuals prior to commencing trials for ease of identification so individual variation in responses can be taken into consideration.

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Chapter 5. Evading the enemy: do dingo (*Canis lupus dingo*) odours deter southern hairy-nosed wombats (*Lasiorhinus latifrons*) from their burrows?

One of the dingos from Cleland Wildlife Park

5.1 Statement of Authorship

Title of paper	Evading the enemy: Do dingo (*Canis lupis dingo*) odours deter southern hairy-nosed wombats (*Lasiorhinus latifrons*) from their burrows?			
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5.2 Abstract

As human–wildlife conflicts escalate worldwide, negatively affecting humans and wildlife, there is an increasing need to develop management strategies that reduce damages and enhance co-existence. The use of predator odour deterrents has been recognised as a means of deterring pest species from areas of conflict. However, the anti-predator behaviour of many species including the southern hairy-nosed wombat (*Lasiorhinus latifrons*), is poorly understood. Conflicts between *L. latifrons* and the agricultural sector have been ongoing for decades, primarily due to damage caused by their burrowing behaviour. This study assessed the effectiveness of predator odours in deterring free-living *L. latifrons* from their burrows. The behaviour of *L. latifrons* was monitored using remote cameras, before and after dingo (*Canis lupus dingo*) urine and faeces were applied to burrow entrances. Dingo odours did not reduce *L. latifrons* use of their burrows. This does not mean they are ineffective deterrents, as responses to threats can be context dependent. Further research into the use of deterrents in differing contexts is needed to gain a better understanding of *L. latifrons* threat perceptions and guide the use of deterrents.

Key Words: damage mitigation, human–wildlife conflict, *Lasiorhinus latifrons*, repellent, wildlife management, wombat.

5.3 Introduction

As human–wildlife conflicts (HWCs) escalate worldwide, negatively affecting humans and wildlife (Conover 2002; Barua *et al.* 2013), there is a growing need to develop management strategies that reduce conflicts and promote co-existence with wildlife. The use of deterrents designed to exploit prey's natural avoidance of predators is a potential means of reducing wildlife damage. Predator odours alone elicit avoidance responses in many species (Nolte *et al.* 1994; Apfelbach *et al.* 2005). Rabbits (*Oryctolagus cuniculus*) were found to avoid warrens treated with synthetically derived lion (*Panthera leo*) faeces for up to 5 months (Boag and Mlotkiewicz 1994), while bank voles (*Microtus oeconomus*) reduced their home range in response to the odours of weasel (*Mustela nivalis*) (Borowski 1998). However, not every predator cue encountered by a prey animal produces a strong repellent effect. White-footed mice (*Peromyscus leucopus*) for example, did not alter their foraging behaviour in response to a variety of predator odours (Orrock and Danielson 2009).

Perceptions of predation threats and responses to them vary greatly, because of variation in evolutionary, life history and ecological characteristics (Lima and Dill 1990; Apfelbach *et al.* 2005; Ramp *et al.* 2005). Prey often respond more readily to sympatric predators with whhich they have a co-evolutionary history (Jedrzejewski *et al.* 1993; Barreto and Macdonald 1999). This was observed in bettongs (*Bettongia lesueur*), which avoided models of dingos (*Canis lupus dingo*) more readily than foxes (*Vulpes vulpes*) with which they have a shorter evolutionary history (Atkins *et al.* 2016). The availability of cover, predator densities, and individual fitness can strongly influence prey responses to predator cues (Merkens *et al.* 1991; Chivers *et al.* 2000; Borowski 2002; Lind and Cresswell 2005). Woodchucks (*Marmota monax*), for instance, decreased their flight initiation distance when closer to the protection of a burrow (Bonenfant and Kramer 1996). Brushtail possums' (*Trichosurus vulpecula*) anti-predator behaviour declined in areas of lower predator densities (Hollings *et al.* 2015). Juvenile eastern quolls (*Dasyurus viverrinus*) responded more strongly to auditory predator cues than did less vulnerable adults (Jones *et al.* 2004).

Even where predator odours have successfully deterred pest species, their effects are often limited in duration. The evaporation, dilution, or decomposition of odours over time can result in reduced repellency (Sullivan *et al.* 1985; Swihart 1991; Bytheway *et al.* 2013). The defensive behaviour of Brandt's voles (*Lasiopodomys brandtii*) for instance, diminished in response to ageing cat (*Felis catus*) faeces (Hegab *et al.* 2014). The effectiveness of odours can also be limited by habituation, a process by which animals learn that the odour is non-threatening when used repeatedly, in predictable patterns, or when unaccompanied by negative reinforcement (Breitenmoser *et al.* 2005). Repeated exposure to tiger (*Panthera tigris*) faecal odours caused habituation in kangaroos (*Macropus giganteus)* (Cox *et al.* 2010). Species with innate avoidance responses may not habituate to predator cues (Muller-Schwarz 1974; Apfelbach *et al.* 2005). In contrast, repeated exposure to bobcat (*Lynx rufus*) urine did not cause habituation in woodchucks (Swihart 1991). A greater understanding of prey animal's threat perceptions in natural environments is needed to guide the development of predator deterrents.

As one of the largest fossorial herbivores in the world, southern hairy-nosed wombats (*Lasiorhinus latifrons*) create large warrens, with multiple burrow entrances. Warrens hinder agricultural production, undermining infrastructure, such as roads, dams, and fences, threatening human safety, and can damage heavy machinery (St John and Saunders 1989; Stott 1998). The resulting damage can cause substantial financial losses (Chapter 2). Damages are alleviated by culling *L. latifrons*, under a permit system (*National Parks and Wildlife Act 1972*(SA)*)*, but recolonisation of burrows following the removal of animals is an ongoing problem (Stott 1998). Predator odour deterrents may reduce damage and prolong time to recolonisation of vacant burrows. While olfaction is known to play an important role in social communication *L. latifrons* (Taggart and Temple-Smith 2008; Triggs 2009), little is known about its role in threat perception. The voids (faeces and urine) of dingoes, a known predator of wombats (Newsome *et al.* 1983), have been found to elicit avoidance responses in captive *L. latifrons* (Descovich *et al.* 2012) and in the field following the deliberate collapsing of burrows (Sparrow *et al.* 2016). However, captive and control–impact studies like the one used by Sparrow *et al.* (2016) may lead to a misconception of the effectiveness of treatments (Koehler *et al.* 1990; Parsons and Bondrup-Nielsen 1996; Rytwinski *et al.* 2016). The failure of control–impact designs to consider differences in pre-existing activity levels between treatment sites makes them less likely to correctly detect effectiveness than those including a before–after–control–impact design (Rytwinski *et al.* 2016). Furthermore, burrow collapsing is time intensive and cost prohibitive on a small landholder scale and raises ethical concerns for *L. latifrons*. Therefore, this study aimed to assess the effectiveness of dingo odours in deterring free-living *L. latifrons* from using un-collapsed burrows and to provide insight into the mechanisms involved in threat detection through detailed behavioural analysis with a before–after–control–impact design.

5.4 Methods

5.4.1 Study site

Predator scent trials were conducted on a sheep grazing property, in Swan Reach (34.34ºS, 139.36ºE), South Australia. The property is located approximately 400 km within the dog proof fence, a 5,600 km fence erected in the 1980s to exclude dingoes from sheep grazing land (South Australian Wild Dog Advisory Group (SAWDAG) 2016). This provided a controlled environment in which to conduct experiments on the effects of dingo scents in the field, as dingo incursions were unlikely that far within the fence (SAWDAG 2016). Swan Reach is in a semi-arid region characterised by unpredictable low rainfalls (~ 300 mm annually) and frequently subjected to periods of drought (Finlayson *et al.* 2005). The vegetation consists predominantly of saltbush (*Atriplex* spp.) and blue bush (*Maireana* spp.), interspersed with patches of remnant mallee eucalypts. The understory is dominated by introduced forbs, such as thread iris (*Moraea setifolia)* and ward's weed (*Carrichtera annua*), with minimal native grasses (*Stipa* spp.). The soils consist predominantly of alluvial clay, supported by layers of sheet limestone (calcrete).

Figure 5.1. The location of Swan Reach in semi-arid Australia, positioned 400 km within the dog proof fence, which is depicted by the red line.

5.4.2 Experimental design

The effects of dingo odours on *L. latifrons* activity and behaviour were examined in two consecutive replicates during winter 2013. In each replicate, seven warrens located > 100 m apart containing 20–30 occupied and empty burrows were selected for monitoring. This ensured *L.latifrons* had alternative burrows to retreat to within their home range. Warrens were located in clay soil, with similar surrounding vegetation, to control for environmental variables that may influence an animal's perceived predation risk (Lima and Dill 1990). Within each warren, three active burrows, located ≥ 6 m apart were monitored using ScoutGuard KG680V and Reconyx HC600 motion sensor cameras. The cameras were attached to star pickets and positioned 40–50 cm from the ground 2–3 m from the burrow entrance (Figure 5.2). The cameras monitored *L. latifrons* activity and behaviour 2 weeks before, and 2 weeks after treatments were applied. The three burrows within each warren were randomly selected to receive one of three treatments, a continuous control with no treatment, 40 ml of dingo urine, or 20 g of dingo faeces. Treatments were applied directly to the soil, 30–40 cm from the burrow entrance. The volume/mass of the dingo odours used represents a typical dingo void (Parsons and Blumstein 2010a). The cameras were checked weekly throughout the study, to download photos and replace the batteries.

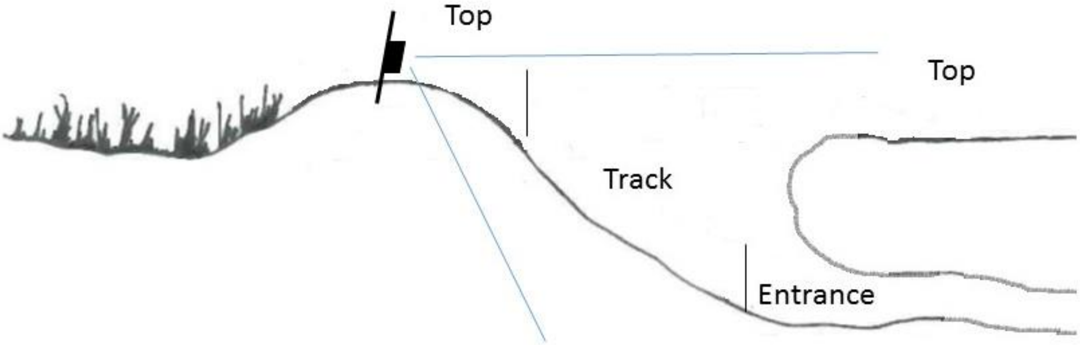

Figure 5.2. A cross-section of a *L. latifrons* burrow, showing the locations of the top, track and entrance of the burrow and the positioning of the camera.

Dingo urine was collected from purebred dingoes at Cleland Wildlife Park, and the Australian Dingo Conservation Association (ADCA) in Oberon, NSW. Urine was collected at Cleland Wildlife Park by inciting dingos to scent mark on plastic sheeting that had been sprayed with 1 ml of dog urine obtained from a local vet (Figure 5.3). The dog urine was allowed to dry before the sheeting was placed inside the dingo enclosure, to minimise cross-contamination. Once a dingo had urinated on the sheet, its void was collected, stored in a sterile container and frozen at -4 °C until use. Urine from the ADCA was refrigerated at 2 °C and shipped within 5 days of collection. Upon receipt of the urine, it was stored at 2 °C until application the following day. Fresh faeces was collected from Cleland and Urimbirra Wildlife Parks in South Australia. A fresh supply of voids could not be maintained because of the large volume required, thus sll samples were frozen at -4 °C until use to prevent the effectiveness of odours diminishing with age (Bytheway *et al.* 2013; Hegab *et al.* 2014). Although freezing samples has been reported to affect the stability of chemical messages and species responses to them (Hoffmann et al. 2009) frozen samples have elicited avoidance responses in rodents (Hayes *et al.* 2006; Russell and Banks 2007). This research was performed under University of Adelaide animal ethics permit s2011-197C, and a scientific research permit A25828-1 obtained from the Department of Environment and Water.

Figure 5.3. Dingos at Cleland Wildlife Park interacting with plastic garden sheeting scented with dog urine.

5.4.3 Photo analyses

Because of the large number of photos obtained, a subset of the data, totalling 10 consecutive nights, five in each trial phase (before and after treatment application) was analysed. For the purpose of analysis, a ‘night’ constituted a 24-hour period, beginning and ending at 6 am, because although *L. latifrons* are predominantly nocturnal with peak activity between 6 pm and 6 am, they are still known to be active during the day. For each night (1–10) of the trial, the number and duration of visits to burrows were recorded. Visits were considered the same if consecutive photos were ≤ 15 s apart unless the *L. latifrons* could be identified as a different individual. Individuals were identified based on their fur patterns, ear markings and scars (Figure 5.4), and classified into adult or juvenile (quarter the size of an adult) age classes, to account for individual variation in responses. The proportion of time *L. latifrons* spent in seven main behaviours (Table 4.1) was recorded during each visit, as behaviours change when animals react to risks (Apfelbach *et al.* 2005). Behaviours were analysed in 1 s intervals, as vigilance in common wombats (*Vombatus ursinus*) is known to last for as little as 1 s (Favreau *et al.* 2009). The proportion of time *L. latifrons* spent at the top of burrows, on the track into burrows, and at the entrances of burrows were recorded to determine if the application of odours resulted in a shift in space use. Nightly rainfall, moon phase, and minimum overnight temperature were extracted from the Australian Bureau of Meteorology’s (ABMs) nearest weather stations, in Swan Reach and Nuriootpa (54 km away) (ABM 2013) to assess their effects on *L. latifrons* activity.

Figure 5.4. A sample of images captured with the motion sensor cameras, showing different individuals, identified based on fur patterns, scars, and ear markings.

Table 5.1. Ethogram of *L. latifrons* behaviour

Behaviour	**Classification**
Vigilant	Sitting, lying, or standing with head up in alert position or scanning of the head
Resting	Sitting, lying, or standing, awake and relaxing, scratching, rubbing, or rolling
Social	Interacting with another wombat in a friendly manner
Travelling	Walking or running at a constant gait without stopping
Exploratory	Digging/scratching at dirt or sniffing the ground or air
Defensive	Blocking the burrow entrance with backside, or chasing another *L. latifrons* away
Unknown	Behaviour unable to be discerned, *L. latifrons* not in full view

5.4.4 Statistical analyses

Linear mixed-effects models (LMM), generalised linear mixed-effects models (GLMMs) and mixed-effects beta regressions were used to investigate the responses of *L. latifrons* to the three treatments. No significant differences were found in *L. latifrons* activity and behaviour between replicates; thus replicates were combined for analysis. Only visits ≥ 3 s were included in the analysis to reduce the effects of delayed triggers failing to record the entire visit duration (Driessen *et al.* 2017) and the difficulty of classifying *L. latifrons* behaviours in visits < 3 s. Burrows with less than five visits before treatment application and visits by juvenile *L. latifrons* were excluded from analyses, because of insufficient data. Individual *L. latifrons* could not be identified in approximately half the visits to the burrows. Therefore, individual variation in responses were not accounted for in analyses. This may violate the assumptions of independence, as multiple visits from one individual could have occurred within the same treatment over time. As burrows were predominantly used by the same individuals, the effects of non-independence were reduced by including the warren by burrow interaction as a random effect. All analyses were conducted in *R-3.3.1* (R Core Team 2014).

Differences in the duration of visits to burrows before and after treatment application were analysed using LMM in the *lme4* package (Bates *et al.* 2015). The duration of visits was log transformed to meet the assumptions of normality. All models were fitted with the fixed interaction between the explanatory variables treatment by trial phase. Warren by burrow and warren by burrow by night interactions were fitted as random effects in all models, to account for repeated observations within warrens and burrows across multiple nights. Time (night of the trial phase (1–5)) was fitted as an interaction with treatment by trial phase, to determine if *L. latifrons* habituated to the treatments. In addition, the weather parameters of rainfall, moon phase and minimum overnight temperature, were fitted as fixed factors. Models were evaluated using Akaike's Information Criterion (AIC), Akaike weights (*wi*), and behaviour of model residuals. Post-hoc differences between trial phases (before and after treatment application) for each treatment were assessed using planned comparisons of model means. Differences in responses between trial phases over time were estimated at the mid-point of the time period in the before and after trial phases for each treatment. Planned comparisons were conducted using the *multcomp* package (Hothorn *et al.* 2008).

Differences in the number of visits made to burrows before and after treatment application were analysed using GLMMs in the *lme4* package (Bates *et al.* 2015). Preliminary analyses with Poisson models revealed overdispersion, so models were re-fitted using a negative binomial distribution, with an additional parameter to represent overdispersion. All models included the explanatory variable of treatment by trial phase and the random intercept interaction of warren by burrow, to account for repeated measures within warrens and burrows. Additional models included the fixed factors of rainfall, minimum overnight temperature, and moon phase. Model selection and post hoc comparisons were conducted using the same approach described for the analysis of visit duration.

Seven behaviours (Table 5.1) were observed throughout the trials. There were too few observations of resting and defensive behaviour for statistical analysis. The effects of treatments on the proportion of time *L. latifrons* spent in the remaining five behaviours, and in each location were analysed using mixed-effects beta regression models in the *glmmADMB* package (Fournier *et al.* 2012). Each model contained the fixed interaction of treatment by trial phase, and the random interaction effect of warren by burrow, to account for repeated measures within warrens and burrows. To determine if *L. latifrons* became less wary of the treatments, time (1–5) was fitted as an interaction with treatment by trial phase. Additional models included the fixed factors of minimum overnight temperature, rainfall, and moon phase. Models selection and post hoc comparisons were conducted using the same approach described for the visit duration analysis.

5.5 Results

Forty-two burrows within 14 warrens were monitored over the duration of the experiment, from which > 105,000 photos of *L. latifrons* were collected. Sufficient data was obtained from 29 burrows, eight treated with urine, 12 with faeces, and nine controls. Throughout the study, 1,269 visits by adult wombats of ≥ 3 s in duration were made to the burrows. One hundred and ninety-one individual *L. latifrons* were identified from 670 visits. In the remaining 599 visits, individuals could not be identified. Total rainfall averaged 2.6 ± 0.81 mm. There was little variation in minimum nightly temperature, averaging 10.85 ± 2.12 °C, and moon phase ranged between no to half-moon.

5.5.1 Visit duration

The top performing model for the duration of visits to the burrows was D5 (Table 5.2). It contained the fixed interaction of treatment by trial phase over time and rainfall. Post-hoc comparisons revealed no significant difference in the duration of visits to the burrows between trial phases for the control, faeces or urine treatments ($P = 0.75$, $P = 0.56$, $P = 0.35$ respectively). There was no significant difference in the duration of visits between trial phases over time, for the control, faeces, and urine treatments ($P = 0.65$, $P = 0.64$, $P = 0.21$ respectively, Figure 5.5). Visit duration decreased with increasing rainfall ($P = 0.05$).

Table 5.2. Comparison of the LMMs models used to assess differences in the duration of visits to the burrows between trial phases. All models were fitted with the random effects of warren by burrow and warren by burrow by night. Fixed factors included trial phase (Tp), treatment (Tr), time (T), minimum overnight temperature (Mt), rainfall (R) and moon phase (Mp). ΔAIC represents the difference in AIC from the model of best fit, which is highlighted in bold, and *wi* is the Akaike weight of the model.

Model	Linear form	*df*	Loglik	AIC	ΔAIC	*wi*
D0	1	4	-1920.87	3849.74	3.33	0.04
D1	Tp*Tr	9	-1918.37	3854.74	8.33	0.00
D2	Tp*Tr/T	15	-1909.13	3848.26	1.85	0.09
D3	Tp*Tr/T + Mt	16	-1909.13	3850.26	3.85	0.03
D4	Tp*Tr/T + Mp	16	-1908.19	3848.39	1.98	0.09
D5	**Tp*Tr/T + R**	**16**	**-1907.21**	**3846.41**	**0.00**	**0.23**
D6	Tp*Tr/T + Mt + Mp	17	-1907.96	3849.92	3.51	0.04
D7	Tp*Tr/T + R + Mt	**17**	-1907.14	3848.29	1.88	0.23
D8	Tp*Tr/T + R + Mp	**17**	-1906.23	3846.46	0.05	0.09
D9	Tp*Tr/T + R + Mt + Mp	18	-1905.62	3847.24	0.83	0.15

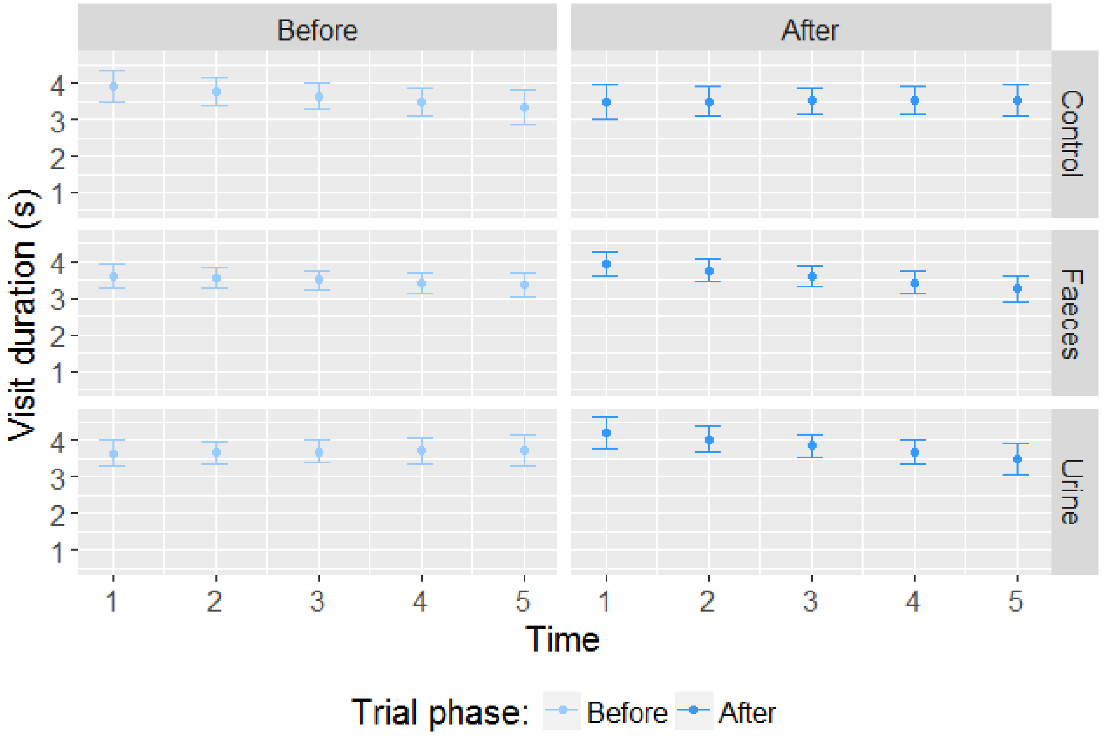

Figure 5.5. Comparison of the duration of visits (mean and 95% confidence intervals) *L. latifrons* made to burrows before and after the treatments of dingo urine, faeces, and the control, derived from LMMs.

5.5.2 Number of visits

The best performing model for the number of visits to the burrows was V1, which contained the fixed interaction of treatment by trial phase (Table 5.3). Post-hoc comparisons revealed no significant difference in the number of visits to the burrows, between the trial phases for the urine, faeces, or control treatments (*P* = 0.33, *P* = 0.14, *P* = 1.00 respectively, Figure 5.6).

Table 5.3. Comparison of the GLMMs that assessed differences in number of visits *L. latifrons* made to burrows between trial phases. All models were fitted with a negative binomial distribution and the random interaction effect of warren by burrow. The fixed variables included trial phase (Tp), treatment (Tr), time (T), minimum overnight temperature (Mt), rainfall (R) and moon phase (Mp). ΔAICc represents the difference in AIC from the model of best fit, highlighted in bold, and *wi* is the Akaike weight of the model.

Model	Linear form	*df*	Loglik	AIC	ΔAIC	*wi*
V0	1	2	-631.96	1267.92	51.09	< 0.00
V1	**Tp*Tr**	**8**	**-600.42**	**1216.83**	**0.00**	**0.31**
V2	Tp*Tr + Mt	9	-600.41	1218.83	2.00	0.12
V3	Tp*Tr + R	9	-599.71	1217.42	0.59	0.23
V4	Tp*Tr + Mp	9	-599.83	1218.78	1.95	0.12
V5	Tp*Tr + R + Mp	10	-599.70	1219.42	2.59	0.09
V6	Tp*Tr + R + Mt	10	-599.70	1219.41	2.58	0.09
V7	Tp*Tr + Mp + Mt	10	-600.37	1220.74	3.91	0.04

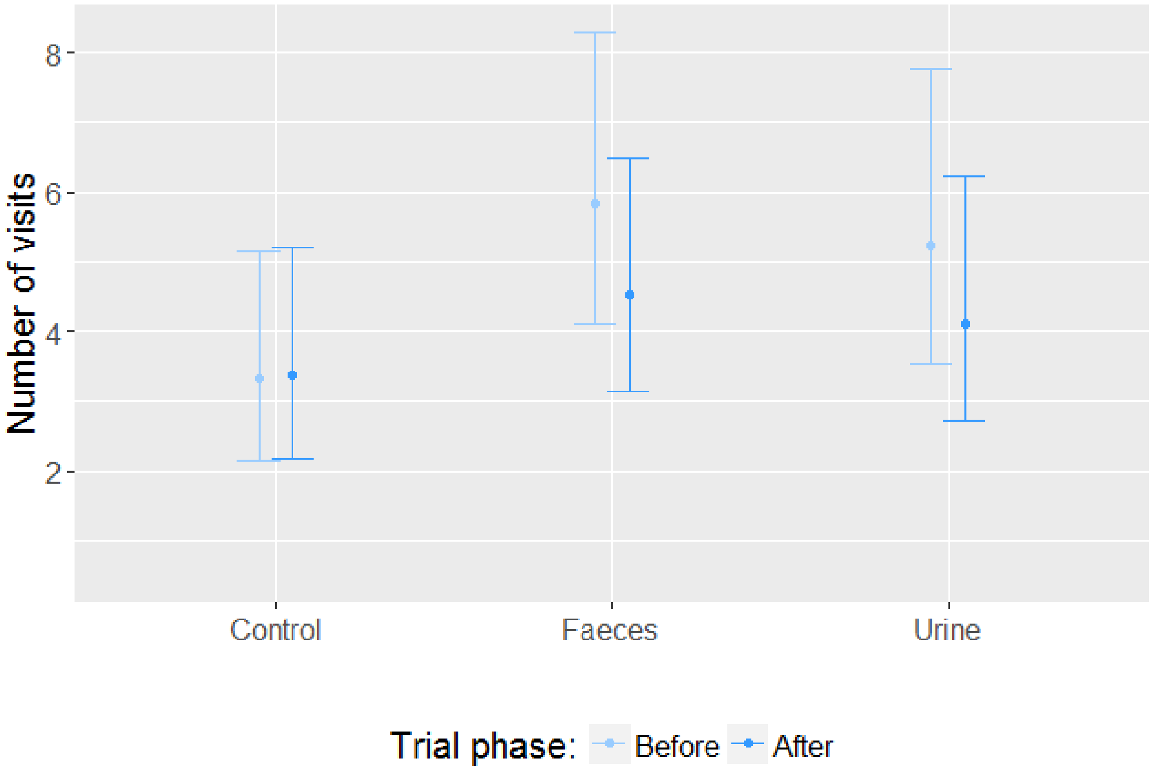

Figure 5.6. Comparison of the number of visits (mean and 95% confidence intervals) *L. latifrons* made to the burrows before and after treatments were applied, derived from the model of best fit.

5.5.3 **Behaviours**

The top performing model for exploratory behaviour, E1, contained the treatment by trial phase interaction (Table 5.4). Post-hoc comparisons revealed a significant increase in the proportion of time *L. latifrons* were observed in exploratory behaviour following the application of urine (*P* = 0.01, Figure 5.7). There was no significant difference in the proportion of time *L. latifrons* spent in exploratory behaviour between trial phases for the faeces and control treatments (*P* = 0.34, *P* = 0.96 respectively, Figure 5.7). For resting behaviour, the top performing model was the intercept-only model, R0 (Table 5.4). There was no significant difference in the proportion of time *L. latifrons* were observed in resting behaviour between trial phases for the urine, faeces, and control treatments (*P* = 0.92, *P* = 1.00, *P* = 0.95, respectively, Figure 5.7). The top performing model for travelling behaviour was the intercept-only model, R0 (Table 5.4). There was no significant difference in the proportion of time *L. latifrons* were observed in travelling behaviour between trial phases for the urine, faeces, and control treatments (*P* = 0.83, *P* = 0.54, *P* = 0.96 respectively, Figure 5.7). For vigilant behaviour, the top performing model was the intercept-only model, V0 (Table 5.4). There was no significant difference in the proportion of time *L. latifrons* were observed in vigilant behaviour between trial phases for the urine, faeces, or control treatments ($P = 0.39$, $P = 0.98$, $P = 0.12$ respectively, Figure 5.7). The top performing model for unknown behaviour was the intercept-only model, U0 (Table 5.4). There was no significant difference in the proportion of time *L. latifrons* were observed in unknown behaviour between trial phases for the urine, faeces, and control treatments ($P = 0.99$, $P = 0.89$, $P = 0.73$ respectively, Figure 5.7).

Table 5.4. Comparison of the mixed-effects beta regression models used to assess differences in the proportion of time *L. latifrons* spent in individual behaviours between trial phases. All models included the random effect of warren by burrow. The fixed factors included trial phase (Tp), treatment (Tr), time (T), minimum overnight temperature (Mt), rainfall (R) and moon phase (Mp). ΔAICc represents the difference in AIC from the models of best fit (highlighted in bold), and *wi* is the Akaike weight of the model.

Model	**Linear form**	***df***	**Loglik**	**AIC**	**ΔAICc**	***wi***
Exploratory						
E0	1	3	3853.40	-7700.80	3.90	0.07
E1	**Tp*Tr**	**8**	**3860.35**	**-7704.70**	**0.00**	**0.47**
E2	Tp*Tr /T	14	3863.55	-7699.10	5.60	0.03
E3	Tp*Tr + R	9	3860.40	-7702.80	1.90	0.18
E4	Tp*Tr +Mp	12	3861.94	-7699.88	4.82	0.04
E5	Tp*Tr + Mt	9	3860.58	-7703.16	1.54	0.22
Resting						
R0	**1**	**3**	**5586.53**	**-11167.06**	**0.00**	**0.97**
R1	Tp*Tr	8	5587.42	-11158.84	8.22	0.016
R2	Tp*Tr /T	14	5587.85	-11147.70	19.36	< 0.00
R3	Tp*Tr + R	9	5587.50	-11157.00	10.06	0.01
R4	Tp*Tr + Mp	12	5587.56	-11151.12	15.94	< 0.00
R5	Tp*Tr + Mt	9	5587.44	-11156.88	10.18	0.01
Travelling						
T0	**1**	**3**	**3650.49**	**-7294.98**	**0.00**	**0.79**
T1	Tp*Tr	8	3653.29	-7290.58	4.40	0.08
T2	Tp*Tr /T	14	3653.59	-7285.18	9.80	0.01
T3	Tp*Tr + R	9	3654.15	-7290.30	4.68	0.08
T4	Tp*Tr + Mp	12	3654.86	-7285.72	9.26	0.01
T5	Tp*Tr + Mt	9	3653.31	-7288.62	6.36	0.03
Vigilant						
V0	**1**	**3**	**4071.09**	**-8136.18**	**0.00**	**0.65**
V1	Tp*Tr	8	4074.69	-8133.38	2.80	0.16
V2	Tp*Tr /T	14	4075.61	-8123.22	12.96	0.00
V3	Tp*Tr + R	9	4074.74	-8131.48	4.70	0.06
V4	Tp*Tr +Mp	12	4074.89	-8125.78	10.40	0.00
V5	Tp*Tr + Mt	9	4075.39	-8132.78	3.40	0.12
Unknown						
U0	**1**	**3**	**4938.89**	**-9871.78**	**0.00**	**0.973**
U1	Tp*Tr	8	4939.67	-9863.34	8.44	0.014
U2	Tp*Tr /T	14	4942.14	-9856.28	15.50	< 0.001
U3	Tp*Tr + R	9	4939.81	-9861.62	10.16	0.006
U4	Tp*Tr + Mp	12	4940.73	-9857.46	14.32	< 0.001
U5	Tp*Tr + Mt	9	4939.69	-9861.38	10.40	0.005

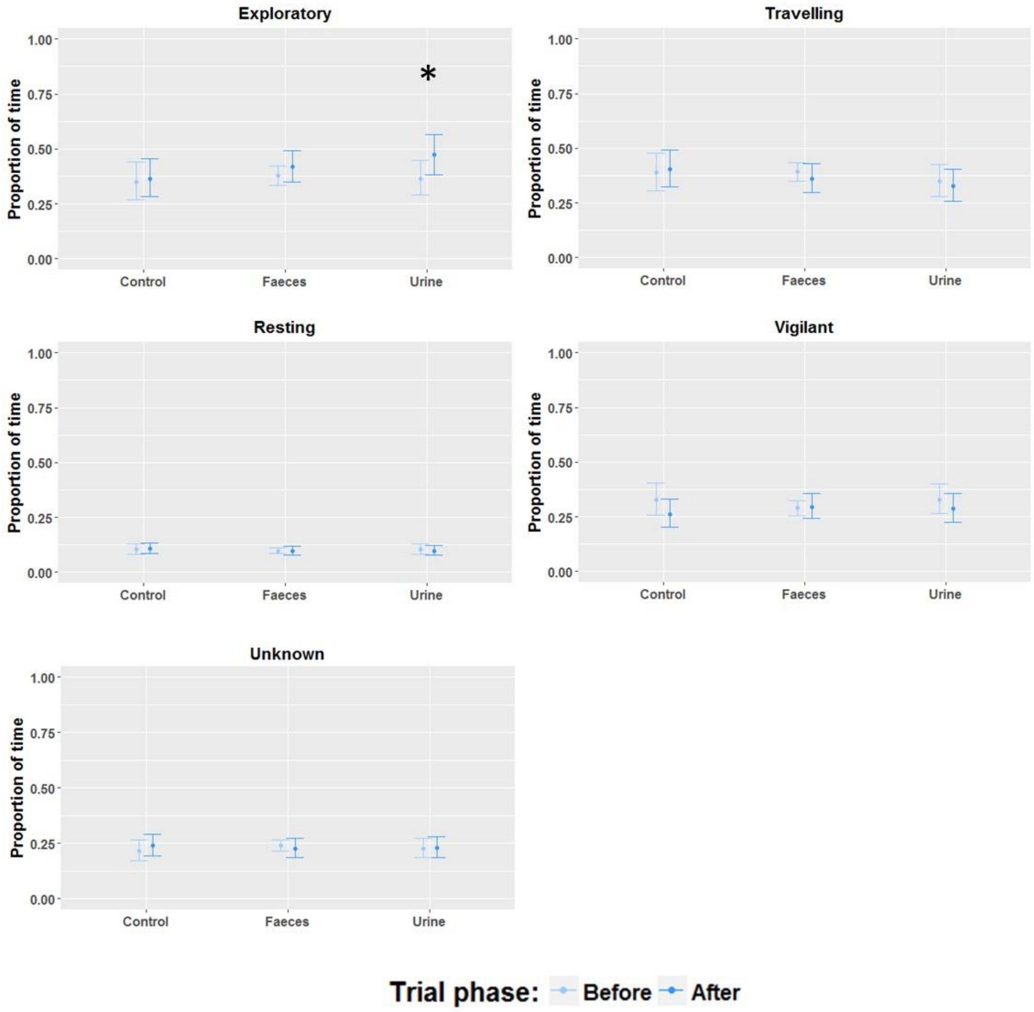

Figure 5.7. Comparison of the proportion of time (mean and 95% confidence intervals) *L. latifrons* spent in exploratory, vigilant, resting, travelling and unknown behaviours before and after the treatment application.

5.5.4 **Locations**

The top performing model for the proportion of time *L. latifrons* were observed at the top of burrows, L1, contained the treatment by trial phase interaction (Table 5.5). There was no significant difference in the proportion of time *L. latifrons* were observed at the top of burrows between trial phases for the urine and faeces treatments (*P* = 0.08, *P* = 0.97 respectively, Figure 5.8). The proportion of time *L. latifrons* were observed at the top of burrow increased significantly in the after phase of the control treatment (*P* = 0.03, Figure 5.8). The best fit model for the proportion of time *L. latifrons* were observed on the track of the burrow was the null model, LT0 (Table 5.5). The proportion of time *L. latifrons* were observed on the track of the burrow between trial phases was not significant, for the urine, faeces, and control treatments (*P* = 0.23, *P* = 0.34, *P* = 0.68 respectively, Figure 5.8). The best fit model for the proportion of time *L. latifrons* were observed at burrow entrances was the intercept-only model, LE0 (Table 5.5). The proportion of time *L. latifrons* were observed at the burrow entrances between trial phases was not significant for the urine, faeces, and control treatments (*P* = 0.99, *P* = 0.56, *P* = 0.31 respectively, Figure 5.8).

Table 5.5. Comparison of the mixed-effects beta regression models used to assess the differences in the proportion of time *L. latifrons* spent in locations between trial phases. All models included the random effect of warren by burrow. The fixed factors included trial phase (Tp), treatment (Tr), time (T), minimum overnight temperature (Mt), rainfall (R) and moon phase (Mp). ΔAICc represents the difference in AIC from the models of best fit, highlighted in bold, and *wi* is the Akaike weight of the model.

Model	**Linear form**	***df***	**Loglik**	**AIC**	**ΔAICc**	***wi***
Top						
L0	1	3	3855.38	-7704.76	3.56	0.08
L1	**Tp*Tr**	**8**	**3862.16**	**-7708.32**	**0.00**	**0.47**
L2	Tp*Tr /T	14	3863.13	-7693.34	14.98	0.00
L3	Tp*Tr + R	9	3862.36	-7706.72	1.60	0.21
L4	Tp*Tr + Mp	12	3864.23	-7704.46	3.86	0.07
L5	Tp*Tr + Mt	9	3862.16	-7706.32	2.00	0.17
Track						
LT1	**1**	**3**	**3611.32**	**-7216.64**	**0.00**	**0.34**
LT2	Tp*Tr	8	3616.29	-7216.58	0.06	0.33
LT3	Tp*Tr /T	14	3617.85	-7207.70	8.94	0.00
LT4	Tp*Tr + R	9	3616.53	-7215.06	1.58	0.16
LT5	Tp*Tr + Mp	12	3617.92	-7211.84	4.80	0.03
LT6	Tp*Tr + Mt	9	3616.38	-7214.76	1.88	0.13
Entrance						
LE0	**1**	**3**	**3993.66**	**-7981.32**	**0.00**	**0.89**
LE1	Tp*Tr	8	3995.92	-7975.84	5.48	0.06
LE2	Tp*Tr /T	14	3998.20	-7968.40	12.92	0.00
LE3	Tp*Tr + R	9	3995.98	-7973.96	7.36	0.02
LE4	Tp*Tr + Mp	12	3997.12	-7970.24	11.08	0.00
LE5	Tp*Tr + Mt	9	3996.21	-7974.42	6.90	0.03

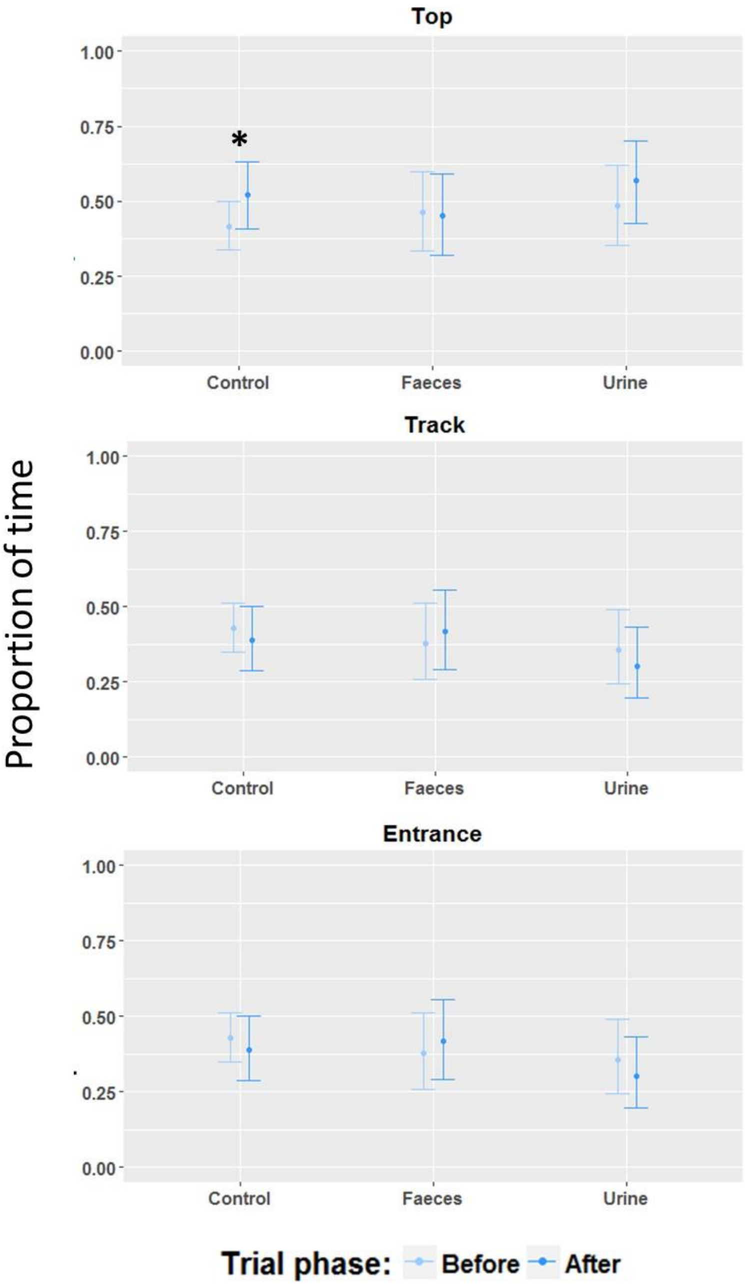

Figure 5.8. Comparison of the proportion of time (mean and 95% confidence intervals) *L. latifrons* spent at the top of the burrow, on the track leading into the burrow and at the burrow entrance before and after treatments were applied, derived from the models.

5.6 Discussion

The presentation of dingo urine and faeces at burrow entrances in this study did not deter *L. latifrons* from using them. Conversely, Sparrow *et al.* (2016) reported *L. latifrons* showed greater avoidance and prolonged time to recolonisation of collapsed warrens treated with dingo odours than control warrens. Whereas captive *L. latifrons* increased their exploratory behaviour in response to dingo faeces placed inside burrows, followed by increased use of burrows and concealed locations (Descovich *et al.* 2012). Increased exploratory behaviour following the application of dingo urine was also observed in the current study. The inspection of predator cues may seem counterintuitive, but is well documented in many taxa and can provide prey with more information about the nature of a threat (Caine and Weldon 1989; Dugatkin and Godin 1992; Fishman 1999). The differing results of these studies indicate *L. latifrons* responses to dingo odours may be context dependent, but disparities in experimental design and analyses make it difficult to determine which factors are influencing threat perceptions of *L. latifrons*.

The environment in which prey resides can have a strong influence on their threat perceptions (Kats and Dill 1998; Verdolin 2006). Species with effective concealment from predators often reduce the probability of an encounter by decreasing their movement or altering their space use towards increased cover rather than expend energy fleeing and potentially drawing the attention of a predator (Merkens *et al.* 1991; Kotler 1997; Borowski 1998; Lima 1998). Burrows provide *L. latifrons* with refuge from the threat of predation; thus, rather than fleeing they may have reduced their ranging behaviour to be in closer proximity to the protection of a burrow in a manner that was not evident from the camera data. Increased burrow use was observed in captive *L. latifrons* following exposure to dingo faeces (Descovich *et al.* 2012). Field observations suggest *L. latifrons* respond similarly, fleeing towards the protection of a burrow when faced with an imminent physical threat (C. O’Brien, pers. obs.). Odours encountered inside burrows or in open areas further from the safety of a burrow could prove more effective at deterring animals, as predation risk may be heightened by their limited options for escape (Boag and Mlotkiewicz 1994; Monclus *et al.* 2015). This was observed in marmots (*Marmota flaviventris*) whose risk perceptions increased when further from the safety of a burrow (Monclus *et al.* 2015). The greater avoidance of dingo odours reported by Sparrow *et al.* (2016) following the collapsing of warrens may be due to *L. latifrons* increased vulnerability to predation in open areas further from the safety of a burrow.

The importance of the area being treated may reflect the effectiveness of deterrents, with animals being more difficult to deter from important resources that are in limited supply, or areas where they are well established (Koehler *et al.* 1990; Gilsdorf *et al.* 2002). Mylar ribbons, for example, failed to deter herring gulls (*Larus argentatus*) from nesting sites to which they displayed strong fealty, but not loafing areas that were of lesser importance (Belant and Ickes 1997). Burrows require a large amount of energy to construct and are critical to the survival of *L. latifrons* in harsh semi-arid environments (Shimmin *et al.* 2002). *Lasiorhinus latifrons* are known to use multiple warrens within their home range, but they display strong fealty towards 1-2 warrens (Finlayson *et al.* 2005). Thus, though the potential exists for *L. latifrons* to retreat to an untreated burrow within their home range, they may be more difficult to deter from burrows where they are well established. The greater avoidance response to dingo odours following the removal of *L. latifrons* burrows (Sparrow *et al.* 2016) suggests burrows have an important role to play in threat perceptions. Further research is required to determine the threshold between staying and leaving the burrow.

Variation in predator numbers can function as a level of risk, with the threat of capture increasing as predator density increases (St Juliana *et al.* 2011). The single application of dingo voids at burrows may indicate low predator densities and therefore low risk. Repeated or more widespread application of odours may produce greater avoidance responses. This was observed in dabbling ducks (*Anatinae* spp.), which showed greater avoidance towards nest sites treated with a higher density of artificial fox urine (Eichholz *et al.* 2012). It is also possible that the low probability of encountering a dingo or similar native mammalian predator in Swan Reach has resulted in a relaxation of *L. latifrons* antipredator behaviour. Loss of anti-predator behaviour has been observed in numerous species including possums (Hollings *et al.* 2015), wallabies (Blumstein *et al.* 2004), toads (Kraaijeveld-Smit *et al.* 2006), and invertebrates (Stoks *et al.* 2003) following the removal of predators from the environment. Reinforcement of predator cues through physical exposure or simulated attacks may be required to increase avoidance responses (Griffin *et al.* 2001; Vilhunen *et al.* 2005). This was observed in predator naive bettongs, which displayed greater avoidance of cats following 18 months of exposure to predators at low densities (West *et al.* 2018). In contrast, some species display innate avoidance and do not need continued or reinforced exposure to predators to maintain avoidance behaviour (Calder and Gorman 1991; Russell and Banks 2007). The avoidance of dingo faeces by captive *L. latifrons* with no prior experience of dingos (Descovich *et al.* 2012), indicates this may be the case.

Another explanation may be that odours presented in low volumes or concentrations may not be sufficient to elicit an overt avoidance response (Mirza and Chivers 2003). This was observed in white-tailed deer (*Odocoileus virginianus*) whose avoidance response declined with lower volumes of predator urine (Swihart *et al.* 1991). The small volume of dingo voids presented in this study may not have posed a threat sufficient to warrant an overt avoidance response. However, captive *L. latifrons* exhibited avoidance responses when presented with a smaller volume of dingo faeces than was used in this study (Descovich *et al.* 2012). Similarly, pademelons (*Thylogale billardierii*) exhibited avoidance responses following exposure to 40 ml of dingo urine (Parsons and Blumstein 2010b), the same volume used in this study. The use of frozen samples in the current study may have resulted in reduced repellency, as the chemical composition of voids may break down following freezing, altering the stability of the messages contained within them (Schultz *et al.* 2000; Hoffmann *et al.* 2009). However, frozen voids have produced avoidance responses in captive *L. latifrons* (Descovich *et al.* 2012) and rodents (Hayes *et al.* 2006; Russell and Banks 2007). Further research is required to determine the impact of using different volumes or concentrations of voids on *L. latifrons* behaviour.

5.6.1 **Summary**

Dingo urine and faeces did not reduce the use of burrows by *L. latifrons*, but this does not mean they are ineffective deterrents. Burrows are vital to the survival of *L. latifrons*, providing them with effective protection from threats. Thus, rather than expending energy fleeing, *L. latifrons* may reduce their home range to be in closer proximity to the safety of a burrow. The presentation of odours inside burrows or open areas where *L. latifrons* are yet to establish territories may prove more effective in deterring the species. It is also possible that the manner in which the odours were applied did not pose a sufficient threat to warrant fleeing. A more widespread application or use of a larger volume of odours may alter *L. latifrons* threat perceptions. Further research is required to understand the effects of distance to cover, volume and concentration of dingo voids, and the manner of odour application on *L. latifrons* threat perceptions.

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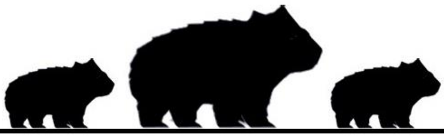
Chapter 6. General discussion

Jackhammering for a GPS collar that slipped off a *L. latifrons*



Few human land uses have as great an impact on wildlife resources or as much contact with wildlife as agriculture (Ramankutty and Foley 1999; Foley *et al.* 2005). Large scale land clearance for agricultural production is increasingly restricting biodiversity to small fragmented patches, threatening species conservation (Woodroffe *et al.* 2005b; Tanentzap *et al.* 2015). If the global trend in human population growth reaches the projected 9.8 billion people by 2050, agricultural production is expected to more than double (Tilman *et al.* 2011; United Nations Department of Economics and Social Affairs Population Division (UNDESAPD) 2017). This will increase the potential for conflicts with wildlife to arise, as they compete with humans for limited resources. Wildlife damage to agricultural resources affects food security and the economic viability of farms (Conover *et al.* 1995; Barua *et al.* 2013). Humans typically respond with lethal control, which can have devastating impacts on wildlife populations (Treves and Naughton-Treves 2005). Consequently, the management of human–wildlife conflicts (HWC) on agricultural land will become increasingly important to ensure environmental and production sustainability. Effective HWC management requires an interdisciplinary approach, incorporating the ecological and human dimensions of the problem. Such an approach will enable wildlife managers to gain an understanding of the conditions and human perspectives that determine whether interventions will be successful for humans and wildlife, thereby strengthening our ability to resolve conflicts. For many conflict species, like the southern hairy-nosed wombat (*Lasiorhinus latifrons*), there is a lack of scientific data on the human dimensions of conflicts and the effectiveness of conflict mitigation measures to successfully manage conflicts.

The aim of this thesis was to examine landholder perceptions of *L. latifrons* and their management and quantify the effectiveness of potential damage mitigation strategies including translocation and deterrent use, to direct future management efforts for the species.

6.1 Landholder perceptions of *L. latifrons*

Lasiorhinus latifrons are largely distributed across private agricultural land (St John and Saunders 1989). Thus, co-operation from agriculturalists will be critical in ensuring the effective conservation and management of this species. Acknowledging and understanding landholder perceptions of *L. latifrons* and the problems associated with them is, therefore, an important first step in determining how to best manage conflicts on farmland (Sillero-Zubiri and Laurenson 2001; Madden 2004). If landholders feel their views have not been taken into consideration or the needs of wildlife are given priority over their own, they will be less likely to co-operate and support management decisions (Kellert 1981; Zinn *et al.* 1998). Consequently, the perceptions of landholders living throughout the range of *L. latifrons* were surveyed, to gain insight into their attitudes towards the species and its management. The information gained provides insight into the drivers of conflict and where to direct research efforts to reduce it. However, it is important to note that the low response rate may have resulted in significant non-response bias, which could not be assessed because of the anonymity of the surveys. Thus, the findings presented may not represent the wider population. Future surveys should consider personalising envelopes, using pre-contact forms, or providing monetary incentives to improve response rates.

6.1.1 Causes of conflict

The majority of respondents with *L. latifrons* on their property reported damage, indicating widespread perception of conflict. The most prevalent and problematic form of damage reported resulted from the burrowing behaviour of *L. latifrons*. Burrowing damage included the undermining of infrastructure, such as fences, roads, buildings, and dams, and damage to heavy machinery. Similarly, the burrowing behaviour of badgers (*Meles meles*) causes conflicts with people when they excavate setts close to infrastructure or on agricultural land (Moore *et al.* 1999). Reducing the damage caused by *L. latifrons* burrowing behaviour should therefore be a priority for wildlife managers, to reduce conflicts on agricultural land. Consumption of crops, competition for stock, erosion, and weed infestation were also widely reported forms of *L. latifrons* damage. The financial impact of damages was perceived to be substantial, with the majority of respondents reporting ≥ $10,000 in losses per annum, though estimates ranged between $0–100,000 in damage. The types and costs of damage reported varied both within and between regions, most likely because of differences in environmental and land use characteristics, *L. latifrons* abundance, and human perceptions. This highlights the need for locally adaptive management plans to target area specific problems.

Though *L. latifrons* are perceived to cause substantial problems, landholder estimates of damage are often disproportionate to actual damages. This was observed in prairie dogs (*Cynomys ludovicianus*), which were perceived to reduce the carrying capacity of agricultural land, even though studies showed they improve grassland and forage quality (Reading *et al.* 2005). Similarly, greater sandhill cranes (*Grus canadensis tabida*) were perceived to cause significant damage to barley fields, but actual damage was ≤ 3% of total crop yields (McIvor and Conover 1994a). Perceptions of damage are often influenced by financial dependence on stock or crops, misinformation, fear of damage, catastrophic events, or social, and cultural influences (Siex and Struhsaker 1999; Knight 2000; Naughton-Treves and Treves 2005; Dickman 2010). Financial dependence on properties was not found to influence reports of damage in this study, but there may be other factors affecting people's perceptions of *L. latifrons*. Regardless of whether the damage is real or perceived, understanding landholder perceptions is important as a baseline for investigating antagonism towards wildlife, as landholder attitudes influence the success of wildlife management programs (Madden 2004; Treves *et al.* 2006). Thus, a lack of quantitative data on *L. latifrons* damage does not detract from the value of these results. Nonetheless, it remains important to identify if the damage is real or perceived to determine the best management approach.

6.1.2 **Management of conflicts**

Despite landholder perceptions of damage, their opinions towards *L. latifrons* were largely positive, with 85.8% of those surveyed supporting the species conservation and 83.5% believing co-existence between the species and agricultural production was possible. Support for conservation and co-existence declined among those who had *L. latifrons* on their property and/or were financially dependent upon their property. Decreased support for co-existence with *L. latifrons* from respondents who are financially dependent on their property is understandable, as the impact of resource damage would be intensified by lack of other assets (Dickman 2010). Interestingly, support for conservation and co-existence was not influenced by landholders' experience with damage. This is inconsistent with other studies, where landholders who had suffered damage were more likely to have negative views towards the species (Schumann *et al.* 2012). Support for conservation was largely driven by the view that *L. latifrons* are an iconic Australian species and have an important role in the environment. Landholders who did not support *L. latifrons* conservation thought it was not necessary as the species would never become extinct or caused too much damage. Reducing the damage caused by *L. latifrons*, particularly for those at the greatest financial risk, and improving knowledge about their ecological role and conservation status may therefore result in increased support for conservation and co-existence with the species.

Culling, predominantly via shooting, is the most widely used control method for reducing conflicts with *L. latifrons* and the agricultural sector, regardless of region. Culling was used most prevalently in the Far West, which may reflect differences in the intensity of conflicts, land use, human perceptions, or the demographics sampled. Regardless of region, culling was more likely to be used by landholders who had experienced damage and/or were dependent on their property as a main source of income. Despite the widespread use of culling, only half of the landholders who used it perceived it to be effective. Dissatisfaction with the permit system may be due to an inability to accurately estimate *L. latifrons* numbers and set culling quotas that resolve conflicts. This often leads to landholder frustration and indiscriminate culling resulting in inaccurate data on the number of animals killed, and the scale of conflicts on agricultural land (Taggart *et al.* 2008). Official government registers of *L. latifrons* mortality are considered an underestimate of the true numbers culled (St John and Saunders 1989; Taggart *et al.* 2008). To successfully manage conflicts, it is clear that the culling permit system requires improvement, but a lack of information on its impacts on *L. latifrons* population demographics and viability, and accurate methods of estimating the species abundance makes this difficult.

Few landholders trialled non-lethal conflict mitigation strategies, possibly because of a lack of effective interventions. The blocking of burrows was the only non-lethal method reported to effectively reduce conflict. Electric fencing was perceived as ineffective by landholders, despite being found to reduce *L. latifrons* damage to the dog proof fence (St John and Saunders 1989). Even where effective non-lethal alternatives are available, culling is typically favoured by landholders as a damage control measure as it is immediately satisfying, convenient, and cost effective (Horton and Craven 1997; Zinn *et al.* 1998). Landholder perceptions of success may not be influenced solely by damage reduction, but also the time and costs associated with implementing and maintaining control measures. Despite this, half of Murraylands and a quarter of Far West respondents suggested developing non-lethal control measures to reduce conflicts.

6.2 Reducing *L. latifrons* damage

Reducing the damage *L. latifrons* cause to farmland is a complex and challenging problem. Wildlife managers are under increasing pressure from the public to implement non-lethal damage mitigation strategies. Landholders have also shown support for the implementation of non-lethal damage interventions. As landholders' lives are already very labour intensive (Australian Bureau of Statistics (ABS) 2011), management strategies need to be time and cost effective. There is currently a lack of quantitative evidence on the effectiveness and humaneness of non-lethal damage interventions on which to base policy and advice to landholders. Given the highly variable nature of conflicts, no single management strategy is likely to alleviate all conflicts. Damage mitigation strategies need to be tailored to the particular circumstances in which problems occur.

6.2.1 **Translocation**

Translocation is typically perceived as a humane, non-lethal alternative for reducing HWC, despite a growing body of research that suggests otherwise (Massei *et al.* 2010). Translocation has been widely suggested as a damage mitigation strategy for *L. latifrons* (Chapter 2) presenting an opportunity to reduce conflicts and aid in the species conservation, by moving animals from conflict zones to regions where the species has declined. However, the translocation of problem *L. latifrons* proved to be problematic in this study. The capture of *L. latifrons* was time intensive as animals evaded traps by digging around them or remaining underground. Two *L. latifrons* died as a result of heat stress during capture and handling, an uncommon occurrence in this species (D. Taggart pers. comm.). Although no post-release mortality was observed, many *L. latifrons* went missing shortly after release. Translocated animals commonly disappear, as a result of mortality, dispersal or collar loss and failure (Fritts *et al.* 1984; Dickens *et al.* 2009; Matthews *et al.* 2013; Germano *et al.* 2015). The survival of gray squirrels (*Sciurus carolinensis*) translocated from urban to forested areas was low, with 97% either dying or disappearing soon after release (Adams *et al.* 2004), and 92% of translocated nine-banded armadillo (*Dasypus novemcinctus*) dispersed from the release site with the first few days after release (Gammons *et al.* 2009). Following extensive search efforts, half of the missing *L. latifrons* were relocated, the majority of which had lost or failed collars. Translocated *L. latifrons* whose locations were known displayed high release-site fidelity and settled into their new locations within 3 months post-release. One translocated female recaptured 7 months post-release had a pouch young, providing preliminary evidence that translocation did not have negative long-term health effects. However, these results should be treated with caution because of the small sample size.

Though *L. latifrons* responded well to translocation conflicts were not reduced at the source site, because neighbouring *L. latifrons* quickly recolonised vacant burrows. It is possible that translocating a larger number of animals could provide more sustained effects, but given the time-intensive nature of catching them, this may not be feasible. In addition, as *L. latifrons* are largely distributed across agricultural land, identifying suitable release locations may prove problematic over time. The translocation of animals into areas with already well-established populations could result in lower survival or a failure to establish territories, due to competition with conspecifics (Germano *et al.* 2015; Sullivan *et al.* 2015). The introduction of northern hairy-nosed wombats into part of their former range resulted in the deaths of two males, as a result of competition over territories (Department of Environment and Heritage Protection 2017). Furthermore, translocating *L. latifrons* requires trained personnel to capture, handle and monitor animals, as well as identify suitable release sites. The use of translocation in conjunction with other non-lethal control measures could prove fruitful; however, given the time-intensive nature of translocation it is unlikely to be favoured by landholders. Efforts may be better expended researching other less time-intensive measures for reducing human–wombat conflict, such as the use of deterrents.

6.2.2 **Deterrents**

Deterrents may provide a cost and time effective method for reducing conflicts with *L. latifrons*. Despite recent research, the manner in which *L. latifrons* detect and respond to threats is still poorly understood. This study examined the effectiveness of four treatments: blood and bone (BB), compact discs (CDs), dingo (*Canis lupus dingo*) urine, and dingo faeces in deterring *L. latifrons* from their burrows. The findings show that olfactory cues did not deter *L. latifrons* from their burrows. However, dingo urine and BB produced increased exploratory behaviour. Increased investigation of threats may seem counterintuitive but may in fact provide animals with the benefit of obtaining more information about potential threats (Caine and Weldon 1989; Dugatkin and Godin 1992; Fishman 1999). The presentation of CDs resulted in a significant decrease in *L latifrons* visits to burrows. Visual cues may have been more effective as they are more easily detected from further away, whereas olfactory cues require closer inspection. However, visual cues are likely to be limited in their effectiveness, because of habituation and may only provide short-term relief from conflict (Koehler *et al.* 1990; Mason 1998). Numerous bird species, for instance, habituate to lasers within minutes to days of their use (Blackwell *et al.* 2002b). Although olfactory cues were not effective in this study, this does not mean they are ineffective deterrents. Responses to devices may be context dependent. Species with effective concealment from threats may increase their hiding efforts, or reduce their ranging behaviour to be in closer proximity to shelter, rather than expend energy fleeing (Ydenberg and Dill 1986; Lima 1998). Rats (*Rattus norvegicus*), for example, decreased their use of surface areas and increased burrow use following exposure to a cat (*Felis catus*) (Blanchard and Blanchard 1989). Although alterations in hiding and ranging behaviour could not be determined in this study, captive *L. latifrons* have been found to increase their use of burrows and concealed locations when presented with dingo faeces (Descovich *et al.* 2012). Dingo odours presented following the collapse of warrens were found to prolong the time taken for *L. latifrons* to re-establish them (Sparrow *et al.* 2016). Differences in the design and analyses of these studies makes it difficult to determine the factors influencing threat perceptions in *L. latifrons*. Before deterrents can be made readily available to landholders, further research into the context and manner in which devices are applied is required to determine optimal methods of application.

6.3 Management implications and future research

The management of human–wombat conflicts is a complex and challenging issue facing wildlife managers, as it involves balancing conflicting views from a wide range of stakeholders. This study focused on surveying landholder perceptions of *L. latifrons* and their management, as landholder co-operation will be crucial to ensuring the successful management of this species across the agricultural land on which it is largely distributed (St John and Saunders 1989). However, numerous stakeholders influence management decisions, and landholders are increasingly being asked to incorporate differing community views into wildlife management actions. This often results in conflict among stakeholder groups over how best to manage the species. Surveying the public’s knowledge and views towards *L. latifrons* and their management will give wildlife managers a better understanding of disparities among stakeholder groups and how best to reconcile them. A more participatory approach between landholders, government departments, and conservation groups is recommended for the continued development of effective management strategies. Participatory planning will ultimately result in greater ownership and acceptance of the outcomes; enhance program credibility, and realization of long-term wildlife management goals (Hewitt and Messmer 1997).

The results of this study suggest *L. latifrons* are perceived to cause substantial damage to agricultural properties. However, landholder perceptions of damage are often disproportionate to actual damages (Siex and Struhsaker 1999; Dickman 2010). A key area for further research will be to quantify the types and costs of damage *L. latifrons* cause to agricultural properties. Future surveys should consider ground-truthing landholder reports of damage. More detailed knowledge of the timing, intensity, types, and costs of damage will empower wildlife managers to focus research efforts on conflict interventions to where they are needed most. Improving awareness about realistic levels of damage will empower landholders to better manage and deal with conflicts, thereby improving their tolerance towards the species.

Regardless of whether damage is real or perceived, addressing landholder concerns towards the species will be crucial for ensuring the species is effectively managed on private land. Perceptions of damage varies within and between the regions in which *L. latifrons* occurs. Given the diversity of circumstances in which conflicts arise, no single intervention can effectively reduce all conflict. A variety of mitigation strategies need to be developed and adapted depending on landholder views, types of damage and land use. Tolerance towards *L. latifrons* was lowest among those who were most vulnerable to damage. Focusing research efforts on damage mitigation strategies towards those who are most vulnerable to *L. latifrons* damage, may help to improve tolerance towards the species.

Culling is currently landholders preferred method of reducing the damage caused by *L. latifrons.* Reliance on the use of culling will likely continue as effective, economically viable, non-lethal alternatives are not yet available for *L. latifrons.* Where effectively managed, culling can reduce damage without detrimental effects on species conservation. However, only half of the landholders who used culling as a management tool perceived it to be effective in reducing damages. This may be a result of reinvasion by new individuals (Stott 1998) or dissatisfaction with the permit system and culling quotas (St John and Saunders 1989; Taggart *et al.* 2008). Inaccurate estimates of *L. latifrons* abundance and a poor understanding of the impact of culling on the species can cause the permit system to be ineffective for conflict management and conservation (Tartowski and Stelmann 1998; Taggart *et al.* 2008). A more in-depth analysis of landholder perceptions of the culling permit system may help to identify problems and improve its implementation. Further research into accurate methods of estimating *L. latifrons* abundance, population fluctuations in relation to climatic conditions and habitat types, and the impact of culling on recruitment is needed to improve the culling permit system and ensure conflicts are reduced without threatening *L. latifrons* conservation. Increased public awareness of the problems that arise from living with *L. latifrons* and the costs of damages may help to improve people's understanding and tolerance towards the use of culling (Reiter *et al.* 1999). This in turn could assist in reducing conflict among people over how best to manage *L. latifrons*.

To reduce landholder reliance on culling, effective and economically viable non-lethal damage mitigation strategies need to be identified. Although the olfactory devices used in this study failed to deter *L. latifrons* from using their burrows, this does not mean they are ineffective deterrents. Responses to the devices may have been dependent on the environment or manner in which they were applied. Conversely, CDs significantly reduced *L. latifrons* use of burrows over 5 days, but the usefulness of visual deterrents can be limited due to habituation (Ujvari *et al.* 1998; Gilsdorf *et al.* 2002). Further research over a longer period is required to determine if *L. latifrons* habituate to CDs. In addition, research on these treatments in different contexts, such as by manipulating distance to cover, volume, concentration, or mode of application, and with repeated and widespread application, may yield more promising results.

There is, however, the possibility that effective deterrents (should they be identified) may shift conflicts to new areas or have negative effects on *L. latifrons*. The use of deterrents over large tracts of land may exclude *L. latifrons* from critical resources and have detrimental effects on populations. Therefore, the impacts of deterrents on conflict reduction and *L. latifrons* health and ranging behaviour need to be investigated to determine if their use is appropriate. Until such questions are answered, landscape-scale management of *L. latifrons* should ensure that suitable alternative habitat is available within a 200 m radius of where individuals are excluded from vulnerable agricultural zones. This strategy will ensure the needs of landholders and of *L. latifrons* are met.

Conflict management should focus not only on reducing the damage *L. latifrons* cause but also increasing landholder tolerance of the species. Landholders who view pest species as having an ecological role to play are more likely to be tolerant of them (Schumann *et al.* 2012). In this study, support for *L. latifrons* conservation was driven by the view that they are an iconic Australian species that has an important role to play in the environment.

Although burrowing animals are known to be important ecosystem engineers (Eldridge and James 2009; Davidson *et al.* 2012), the ecological role of *L. latifrons* is poorly understood. Improving knowledge on the benefits of *L. latifrons* to the environment may enhance landholders' tolerance of them.

Although small-scale translocations proved to be ineffective for reducing human–wombat conflict, this study provided some promising preliminary results to suggest translocation could be effective for restoring declining populations of *L. latifrons*. Translocated *L. latifrons* that could be located adapted well to their new environment. Further research using a larger sample size is required to determine the best methods for translocating the species (e.g. number of animals to relocate, time of year, and habitat requirements).

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Appendices

Appendix A. Southern hairy-nosed wombat landowner opinion survey

1. Gender: Male ☐ Female ☐

2. Age: 18-24 ☐ 25-34 ☐ 35-44 ☐ 45-54 ☐ 55-64 ☐ 65+ ☐

3. In what region do you reside (please circle)?

Murraylands **Yorke Peninsula** **Eyre peninsula**

Gawler ranges **Nullarbor Plain** **Adelaide** **Other**

4. What is your opinion of wombats?

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5. Do you think it is important to conserve wombats? YES NO

If yes, where do you think they should live?

………………………………………………………………………………………

………………………………………………………………………………………

If no, why not?

………………………………………………………………………………………

………………………………………………………………………………………

6. Where do you think they should live?

………………………………………………………………………………………

………………………………………………………………………………………

………………………………………………………………………………………

7. Do you think it matters if wombats become extinct? YES NO

Why?

....

....

....

8. In your opinion do you think wombats and primary producers can co-exist?

YES NO

Why? ....

....

....

9. Do you have wombats on your property? YES NO

If yes: a) Is this property your main source of income? YES NO

b) Would you like to see wombats completely eradicated from your property?

YES NO NOT APPLICABLE

c) Can you tell the difference between an active and inactive wombat burrow?

YES NO

If yes, please explain...

....

....

d) Do wombats cause damage on this property? YES NO

Please tick boxes to specify damage caused (tick as many as appropriate). Place a number in each ticked box to rank the impact on your property - start from # 1 = most important, to least important

	Digging under fence lines		Undermining tanks		Damage to grave sites
	Digging under roads		Damage to machinery		Damage to dams
	Grazing competition		Digging under buildings		Weed infestation on warrens
	Erosion		Feeding on crops		Other (specify) ....

10. Have you ever applied for a destruction permit from DENR? YES NO

If yes: a) Was your application successful? YES NO

b) How many wombats were you permitted to cull?

In the year 2010 ………… Average annual number..............

c) In relation to the number of the wombats on the permit, how many do you cull?

MORE LESS EQUAL

d) Were you satisfied with the number of wombats approved on the permit?

YES NO

11. Have you culled wombats without a permit in the past? YES NO

If yes, why?..

....

....

a) Did this culling occur on (please circle):

YOUR PROPERTY ANOTHER PROPERTY CONSERVATION PARK

OTHER (please specify)…………..……………

b) Was this a "one-off" incident? YES NO

If no, how long have you been culling wombats without a permit? ....years.........months

12. Did culling reduce the impacts caused by wombats? Yes / No

Why/why not?..

....

13. What time of year did you cull?

Summer Spring Autumn Winter

Why? ....

....

14. In your opinion, what percentage of the total number of wombats on the property needs to be culled to keep numbers at a manageable level? Why?

....

....

....

15. Approximately how much of your time is spent annually:

a) Controlling wombat numbers? …………………………………

b) Repairing damaged property? …………………………………

16. How much money would you lose as a result of damage to property from wombats?

In an average year $…………….. In 2010 $........................

17. What method have you used to cull wombats? (please circle)

FIREARMS (Type? .... Calibre? ....)

RIPPING WARRENS EXPLOSIVES FUMIGATION

OTHERS (please specify) ……………… ……...................... ....

Of these methods, which do you think is the most humane? ....

18. Have you used non-lethal methods for controlling wombat numbers? YES/NO

a) If yes, what methods were effective, and what was not? Why?

……………………………………………………………………………………………

……………………………………………………………………………………………

....

……………………………………………………………………………………………

b) How long did you trial these methods for?

....

19. Do you have any ideas about non-lethal wombat control that could be researched and tested for effectiveness?

……………………………………………………………………………………………

……………………………………………………………………………………………

……………………………………………………………………………………………

……………………………………………………………………………………………

……………………………………………………………………………………………

....

20. Please fill in the following table:

Abbreviations
Landowners (yourself and neighbours) = LO
Department for Environment and Natural Resource Management = DENR
Regional Natural Resource Management Board = NRM
National Parks and Wildlife Rangers = NPW
Wombat Awareness Organisation = WAO
Conservation Groups (e.g. Conservation Ark / Zoos SA) = CG
Farmers Federation = FF

Tick the appropriate box/boxes to answer each of the following questions:	**LO**	**DENR**	**NRM**	**NPW**	**WAO**	**CG**	**FF**	**Other (please specify)**
Who is responsible for the management of wombats?								
Who would you approach to get information on wombat management?								
Who do you think listens to, and acts upon, concerns regarding wombat management?								
Who do you think is best equipped to help you with the management of wombats?								

21. What do you think could be done to improve wombat management (include any comments on the current permit system, community workshops, field days, field trials of control techniques, role/involvement of conservation/volunteer groups)?

…………………………………………………………………………………………

…………………………………………………………………………………………

…………………………………………………………………………………………

…………………………………………………………………………………………

22. What are your concerns for the future of your property?

…………………………………………………………………………………………

…………………………………………………………………………………………

…………………………………………………………………………………………

…………………………………………………………………………………………

…………………………………………………………………………………………

23. Do you have any concerns for the conservation of wombats in the future?

....

....

....

....

....

24. We would like to find out your basic knowledge of wombats – please answer the following TRUE/FALSE questions about wombats. Even if you are not sure of the answer, please guess. Where possible more detail in the response would be appreciated.

There is only one species of wombat in Australia	TRUE/FALSE
Wombats are solitary animals	TRUE/FALSE
Wombats prefer to eat native grass rather than crops	TRUE/FALSE
Southern hairy-nosed wombats can be found nation-wide	TRUE/FALSE
The southern hairy-nosed wombat conservation status is vulnerable	TRUE/FALSE
More than one wombat lives in a burrow	TRUE/FALSE
An individual wombat uses more than one burrow	TRUE/FALSE
Wombats share burrows with other species	TRUE/FALSE
Wombats have multiple young at one time	TRUE/FALSE
Wombats do not breed during times of drought	TRUE/FALSE

Thank you very much for your time!

Appendix B. Respondent demographics for all 5 regions sampled

Table 6.1. The demographics of respondents from all five regions sampled.

	Murraylands	**Yorke Peninsula**	**Nullarbor**	**Eyre Peninsula**	**Gawler Ranges**
Age (yr):					
≤ 44	23	3	-	4	2
45–64	68	1	3	12	8
≥ 65	26	-	2		1
Gender:					
Male	44	4	2	10	3
Female	50	-	2	4	8
Financial dependence on property:					
Dependent	25	3	6	14	2
Independent	39	-	-	2	5
***L. latifrons* on property:**					
Present	35	3	6	15	5
Absent	66	1	-	1	6

Appendix C. The chi squared statistics for differences in responses to binary questions between gender and age groups

Results for Gender:

Is it important to conserve wombats?

Chi-Square Tests

Region		Value	df	Asymptotic Significance (2-sided)	Exact Sig. (2-sided)	Exact Sig. (1-sided)
Murraylands	Pearson Chi-Square	2.742[a]	1	.098		
	Continuity Correction[b]	1.789	1	.181		
	Likelihood Ratio	2.827	1	.093		
	Fisher's Exact Test				.120	.090
	Linear-by-Linear Association	2.716	1	.099		
	N of Valid Cases	108				
Far West	Pearson Chi-Square	1.167[b]	1	.280		
	Continuity Correction[b]	.292	1	.589		
	Likelihood Ratio	1.213	1	.271		
	Fisher's Exact Test				.596	.298
	Linear-by-Linear Association	1.125	1	.289		
	N of Valid Cases	28				

a. 0 cells (.0%) have expected count less than 5. The minimum expected count is 5.40.

b. 2 cells (50.0%) have expected count less than 5. The minimum expected count is 2.00.

Can landholders and wombats co-exist?

Chi-Square Tests

Region		Value	df	Asymptotic Significance (2-sided)	Exact Sig. (2-sided)	Exact Sig. (1-sided)
Murraylands	Pearson Chi-Square	.007[a]	1	.936		
	Continuity Correction[b]	.000	1	1.000		
	Likelihood Ratio	.007	1	.936		
	Fisher's Exact Test				1.000	.581
	Linear-by-Linear Association	.006	1	.936		
	N of Valid Cases	100				
Far West	Pearson Chi-Square	4.052[b]	1	.044		
	Continuity Correction[b]	2.154	1	.142		

	Likelihood Ratio	5.573	1	.018		
	Fisher's Exact Test				.100	.067
	Linear-by-Linear Association	3.896	1	.048		
	N of Valid Cases	26				

a. 0 cells (.0%) have expected count less than 5. The minimum expected count is 6.86.

b. 2 cells (50.0%) have expected count less than 5. The minimum expected count is 1.85.

Do wombats cause damage?

Chi-Square Tests

Region		Value	df	Asymp. Sig. (2-sided)	Exact Sig. (2-sided)	Exact Sig. (1-sided)
Murraylands	Pearson Chi-Square	3.703[a]	1	.054		
	Continuity Correction[b]	2.364	1	.124		
	Likelihood Ratio	3.639	1	.056		
	Fisher's Exact Test				.119	.063
	Linear-by-Linear Association	3.613	1	.057		
	N of Valid Cases	41				
Far West	Pearson Chi-Square	.240[c]	1	.624		
	Continuity Correction[b]	.000	1	1.000		
	Likelihood Ratio	.236	1	.627		
	Fisher's Exact Test				1.000	.517
	Linear-by-Linear Association	.230	1	.632		
	N of Valid Cases	23				

a. 1 cells (25.0%) have expected count less than 5. The minimum expected count is 3.51.

b. Computed only for a 2x2 table

c. 2 cells (50.0%) have expected count less than 5. The minimum expected count is 1.57.

Have you culled wombats?

Chi-Square Tests

Region		Value	df	Asymptotic Significance (2-sided)	Exact Sig. (2-sided)	Exact Sig. (1-sided)
Murraylands	Pearson Chi-Square	.903[a]	1	.342		
	Continuity Correction[b]	.368	1	.544		
	Likelihood Ratio	.922	1	.337		
	Fisher's Exact Test				.496	.274
	Linear-by-Linear Association	.879	1	.348		
	N of Valid Cases	39				
Far West	Pearson Chi-Square	2.424[b]	1	.119		
	Continuity Correction[b]	1.224	1	.269		
	Likelihood Ratio	2.431	1	.119		
	Fisher's Exact Test				.187	.135
	Linear-by-Linear Association	2.314	1	.128		
	N of Valid Cases	22				

a. 0 cells (.0%) have expected count less than 5. The minimum expected count is 5.38.

b. 2 cells (50.0%) have expected count less than 5. The minimum expected count is 3.27.

Results for Age:

Is it important to conserve wombats?

Chi-Square Tests

Region		Value	df	Asymptotic Significance (2-sided)
Murraylands	Pearson Chi-Square	2.446[a]	2	.294
	Likelihood Ratio	2.377	2	.305
	Linear-by-Linear Association	2.320	1	.128
	N of Valid Cases	114		
Far West	Pearson Chi-Square	2.323[b]	2	.313
	Likelihood Ratio	3.590	2	.166
	Linear-by-Linear Association	2.058	1	.151
	N of Valid Cases	31		

a. 2 cells (33.3%) have expected count less than 5. The minimum expected count is 2.22.

b. 4 cells (66.7%) have expected count less than 5. The minimum expected count is .58.

Is it possible to co-exist with wombats?

Chi-Square Tests

Region		Value	df	Asymptotic Significance (2-sided)
Murraylands	Pearson Chi-Square	.350[a]	2	.839
	Likelihood Ratio	.355	2	.837
	Linear-by-Linear Association	.346	1	.556
	N of Valid Cases	106		
Far West	Pearson Chi-Square	.357[b]	2	.836
	Likelihood Ratio	.327	2	.849
	Linear-by-Linear Association	.261	1	.610
	N of Valid Cases	29		

a. 2 cells (33.3%) have expected count less than 5. The minimum expected count is 2.64.

b. 5 cells (83.3%) have expected count less than 5. The minimum expected count is .62.

Do wombats cause damage?

Chi-Square Tests

Region		Value	df	Asymptotic Significance (2-sided)	Exact Sig. (2-sided)	Exact Sig. (1-sided)
Murraylands	Pearson Chi-Square	.448[a]	2	.799		
	Likelihood Ratio	.432	2	.806		
	Linear-by-Linear Association	.181	1	.670		
	N of Valid Cases	44				
Far West	Pearson Chi-Square	1.949[b]	2	.377		
	Likelihood Ratio	2.030	2	.362		
	Linear-by-Linear Association	1.795	1	.180		
	N of Valid Cases	25				

a. 3 cells (50.0%) have expected count less than 5. The minimum expected count is 2.05.

b. 4 cells (66.7%) have expected count less than 5. The minimum expected count is .32.

Have you culled wombats?

Chi-Square Tests

Region		Value	df	Asymptotic Significance (2-sided)
Murraylands	Pearson Chi-Square	3.150[a]	2	.207
	Likelihood Ratio	3.674	2	.159
	Linear-by-Linear Association	1.980	1	.159
	N of Valid Cases	45		
Far West	Pearson Chi-Square	1.887[b]	2	.389
	Likelihood Ratio	2.429	2	.297
	Linear-by-Linear Association	1.773	1	.183
	N of Valid Cases	25		

a. 2 cells (33.3%) have expected count less than 5. The minimum expected count is 3.33.

b. 4 cells (66.7%) have expected count less than 5. The minimum expected count is .64.

Appendix D. The Cohens kappa results for the open ended survey questions

Question 4. Why do you think it is important to conserve/or not conserve wombats

Symmetric Measures

		Value	Asymp. Std. Error[a]	Approx. T[b]	Approx. Sig.
Measure of Agreement	Kappa	.770	.025	39.937	.000
N of Valid Cases		341			

Question 5. Where do you think *L. latifrons* should live?

Symmetric Measures

		Value	Asymp. Std. Error[a]	Approx. T[b]	Approx. Sig.
Measure of Agreement	Kappa	.800	.022	35.704	.000
N of Valid Cases		450			

Question 8. Why do you think co-existence with *L. latifrons* is or is not possible?

Symmetric Measures

		Value	Asymp. Std. Error[a]	Approx. T[b]	Approx. Sig.
Measure of Agreement	Kappa	.713	.024	35.945	.000
N of Valid Cases		455			

Question 19. Do you have suggestions for the control of wombats that could be researched?

Symmetric Measures

		Value	Asymp. Std. Error[a]	Approx. T[b]	Approx. Sig.
Measure of Agreement	Kappa	.962	.012	28.512	.000
N of Valid Cases		451			

Question 22. What do you think could be done to improve *L. latifrons* management?

Symmetric Measures

		Value	Asymp. Std. Error[a]	Approx. T[b]	Approx. Sig.
Measure of Agreement	Kappa	.815	.021	33.982	.000
N of Valid Cases		456			

Appendix E. The Chapter 5 model results for only known individuals

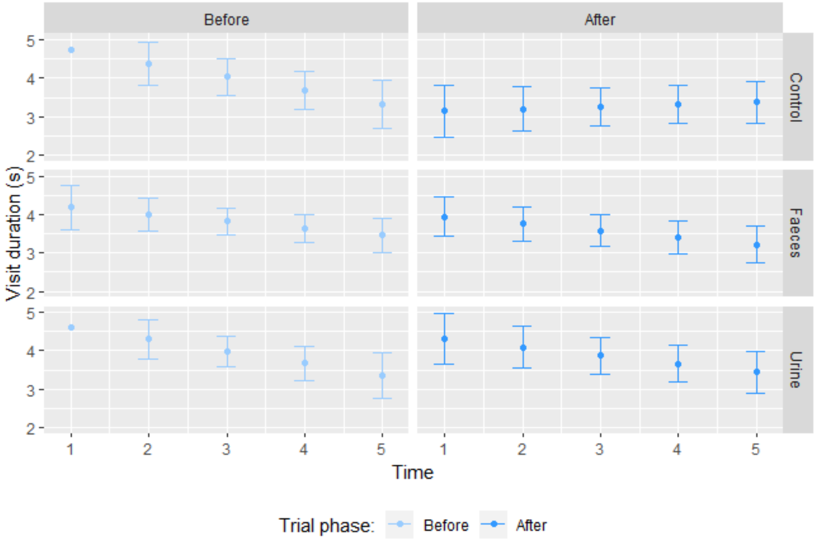
Visit duration

The model of best fit for the duration of visits was D9; it included the treatment by trial phase over time interaction and the weather parameters of rainfall, minimum overnight temperature and moon phase. There was no significant difference in the duration of visits between trial phases over time for the faeces or urine treatments (*P* = 0.99, *P* = 0.94). There was a significant decrease in the duration of visits to the burrows in the ‘after phase’ of the trial for the control treatment (*P* = 0.03). Visit duration declined significantly with decreasing temperature (*P* = 0.05) and with increasing moon visibility (*P* = 0.05).

Comparison of the LMMs used to assess differences in the duration of visits to the burrows between trial phases. All models were fitted with the random effects of warren by burrow and warren by burrow by night. Fixed factors included trial phase (Tp), treatment (Tr), time (T), minimum overnight temperature (Mt), rainfall (R) and moon phase (Mp). ΔAIC represents the difference in AIC from the model of best fit, highlighted in bold. The Akaike weight of the model is *wi*.

Model	Linear form	*df*	Loglik	AIC	ΔAIC	*wi*
D0	1	5	-1021.33	2052.67	7.21	0.01
D1	Tp*Tr	10	-1018.31	2056.62	11.16	0.00
D2	Tp*Tr/T	16	-1007.55	2047.10	1.64	0.11
D3	Tp*Tr/T + Mt	17	-1006.83	2047.66	2.2	0.08
D4	Tp*Tr/T + Mp	17	-1006.81	2046.48	1.02	0.09
D5	Tp*Tr/T + R	17	-1006.24	2047.61	2.15	0.16
D6	**Tp*Tr/T + Mt + Mp**	**18**	**-1004.73**	**2045.46**	**0**	**0.27**
D7	Tp*Tr/T + R + Mt	18	-1005.42	2046.83	1.37	0.13

D8	Tp*Tr/T + R + Mp	18	-1005.44	2046.88	1.42	0.13



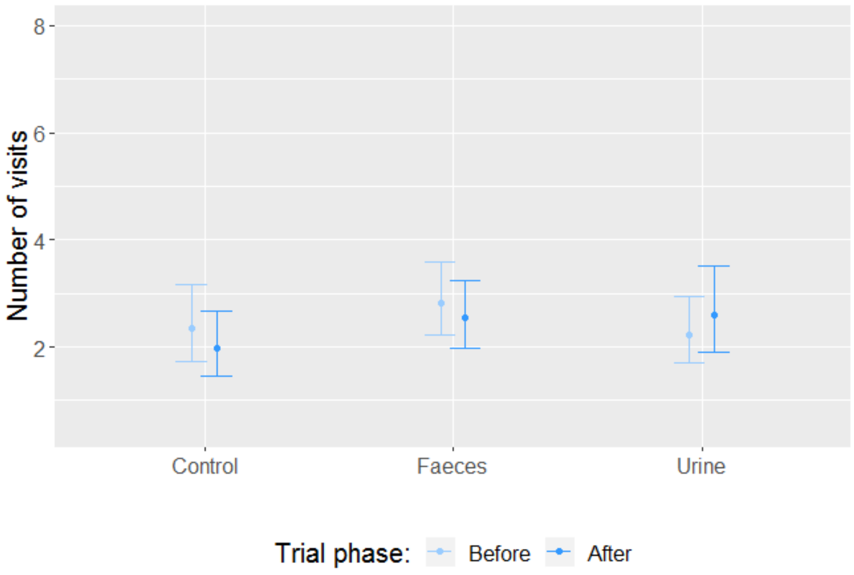
Comparison of the duration of visits (mean and 95% confidence intervals) *L. latifrons* made to the burrows before and after treatment with dingo urine, faeces, and the control, derived from LMMs.

Number of visits

The model of best fit for the number of *L. latifrons* visits to the burrows was the null model. There was no significant difference in the number of visits to burrows between trial phase for the faeces, urine or control treatments (*P* = 0.81, *P* = 0.84, *P* = 0.90).

Comparison of the GLMMs that assessed differences in number of visits *L. latifrons* made to burrows between trial phases. All models were fitted with a negative binomial distribution and the random interaction effect of warren by burrow. The fixed variables included trial phase (Tp), treatment (Tr), time (T), minimum overnight temperature (Mt), rainfall (R) and moon phase (Mp). ΔAICc represents the difference in AIC from the model of best fit, highlighted in bold, and *wi* is the Akaike weight of the model.

Model	Linear form	*df*	Loglik	AIC	ΔAIC	*wi*
V0	**1**	**2**	**-479.09**	**964.18**	**0.00**	**0.81**
V1	Tp*Tr	8	-476.89	969.77	5.59	0.05
V2	Tp*Tr + Mt	9	-476.56	971.11	6.93	0.04
V3	Tp*Tr + R	9	-476.80	971.60	7.42	0.03
V4	Tp*Tr + Mp	9	-476.01	970.03	5.85	0.02
V5	Tp*Tr + R + Mp	10	-475.75	971.49	7.31	0.02
V6	Tp*Tr + R + Mt	10	-475.70	971.41	7.23	0.02
V7	Tp*Tr + Mp + Mt	10	-476.56	973.12	8.94	0.01



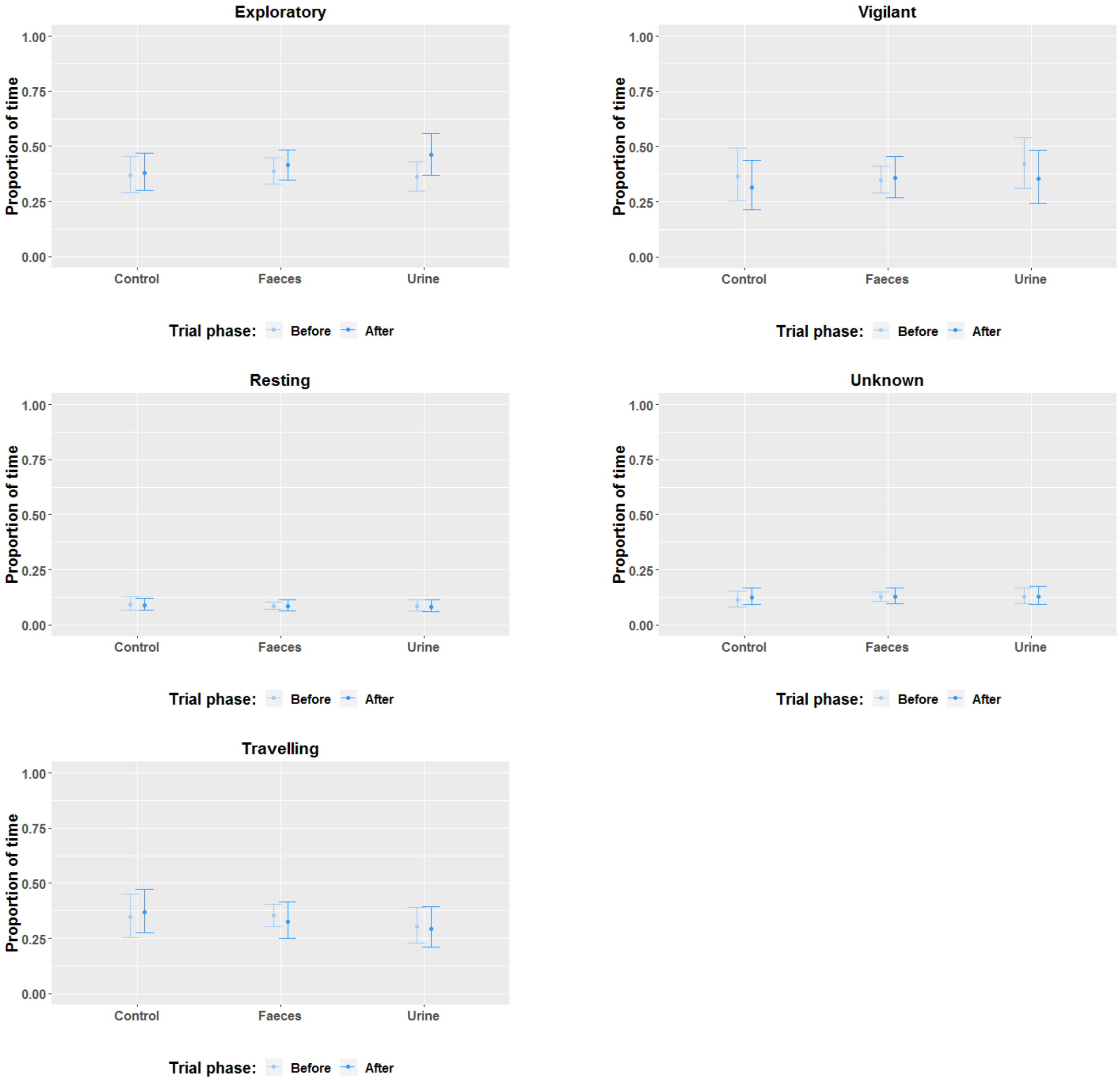
Comparison of the number of visits (mean and 95% confidence intervals) *L. latifrons* made to the burrows before and after treatments were applied, derived from the model V1.

Behaviour

The model of best fit for all behaviours was the null model. There was no significant difference in the proportion of time *L. latifrons* spent in each behaviour between trial phases for the faeces, urine, or control treatments (exploratory: *P* = 0.99, *P* = 0.98, *P* = 0.99; resting: *P* = 0.82, *P* = 0.99, *P* = 0.96; travelling: *P* = 0.85, *P* = 0.14, *P* = 0.99; vigilant: *P* = 0.99, *P* = 0.46, *P* = 0.64; unknown: *P* = 1.00, *P* = 1.00, *P* = 0.92, respectively).

Comparison of the mixed-effects beta regression models used to assess differences in the proportion of time *L. latifrons* spent in individual behaviours between trial phases. All models included the random effect of warren by burrow. The fixed factors included trial phase (Tp), treatment (Tr), time (T), minimum overnight temperature (Mt), rainfall (R) and moon phase (Mp). ΔAICc represents the difference in AIC from the models of best fit (highlighted in bold), and *wi* is the Akaike weight of the model.

Model	**Linear form**	***df***	**Loglik**	**AIC**	**ΔAICc**	***wi***
Exploratory						
E0	**1**	**4**	**1619.26**	**-3225.76**	**5.24**	**0.86**
E1	Tp*Tr	9	1616.88	-3220.52	0.00	0.06
E2	Tp*Tr /T	15	1620.24	-3210.48	10.04	< 0.00
E3	Tp*Tr + R	10	1619.71	-3219.42	1.10	0.04
E4	Tp*Tr +Mp	13	1620.94	-3215.88	4.64	0.01
E5	Tp*Tr + Mt	10	1619.65	-3219.30	1.22	0.03
Resting						
R0	**1**	**4**	**2593.54**	**-5179.08**	**0.00**	**0.98**
R1	Tp*Tr	9	2593.92	-5169.84	9.24	0.01
R2	Tp*Tr /T	15	2594.42	-5158.84	20.24	0.00
R3	Tp*Tr + R	10	2593.97	-5167.94	11.14	< 0.00
R4	Tp*Tr + Mp	13	2594.40	-5162.80	16.28	< 0.00
R5	Tp*Tr + Mt	10	2594.23	-5168.46	10.62	0.00
Travelling						
T0	**1**	**4**	**1595.69**	**-3179.68**	**0.00**	**0.92**
T1	Tp*Tr	9	1593.84	-3173.38	6.30	0.03
T2	Tp*Tr /T	15	1598.15	-3166.30	13.38	0.00
T3	Tp*Tr + R	10	1596.14	-3172.28	7.40	0.02
T4	Tp*Tr + Mp	13	1597.07	-3168.14	11.54	0.00
T5	Tp*Tr + Mt	10	1595.83	-3171.66	8.02	0.01
Vigilant						
V0	**1**	**4**	**1548.89**	**-3089.78**	**0.00**	**0.86**
V1	Tp*Tr	9	1551.22	-3084.44	5.34	0.06
V2	Tp*Tr /T	15	1551.82	-3073.64	16.14	< 0.00
V3	Tp*Tr + R	10	1551.28	-3082.56	7.22	0.02
V4	Tp*Tr +Mp	13	1551.75	-3077.50	12.28	0.00
V5	Tp*Tr + Mt	10	1552.02	-3084.04	5.74	0.05
Unknown						
U0	**1**	**4**	**2432.21**	**-4856.42**	**0.00**	**0.98**
U1	Tp*Tr	9	2432.75	-4847.50	8.92	0.01
U2	Tp*Tr /T	15	2433.74	-4837.48	18.94	< 0.00
U3	Tp*Tr + R	10	2432.79	-4845.58	10.84	0.00
U4	Tp*Tr + Mp	13	2433.17	-4840.34	16.08	< 0.00
U5	Tp*Tr + Mt	10	2432.75	-4845.50	10.92	0.00



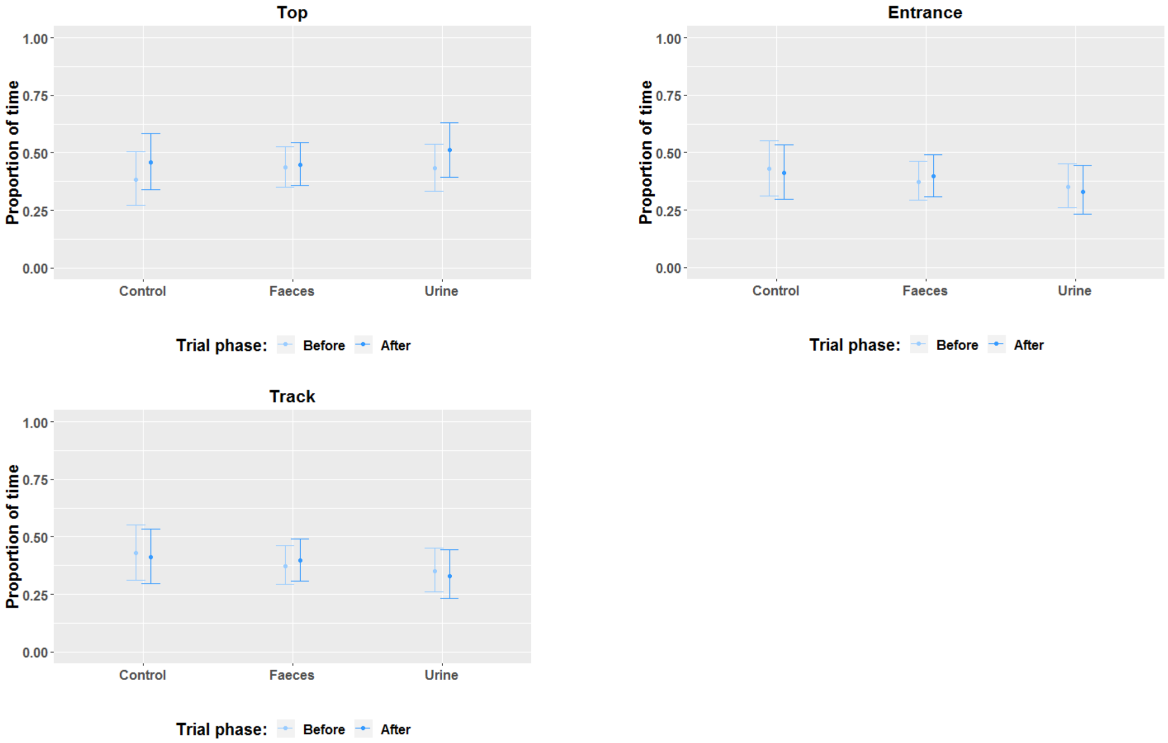
Comparison of the proportion of time (mean and 95% confidence intervals) *L. latifrons* spent in exploratory, vigilant, resting, travelling and unknown behaviours before and after the treatment application.

Location

The model of best fit for all three locations was the null model. There was no significant difference in the proportion of time *L. latifrons* were observed in each location between trial phases for the faeces, urine or control treatments (top: *P* = 0.37, *P* = 0.98, *P* = 0.33; track: *P* = 0.98, *P* = 0.90, *P* = 0.96; entrance: *P* = 0.64, *P* = 0.42, *P* = 0.93, respectively).

Comparison of the mixed-effects beta regression models used to assess the differences in the proportion of time *L. latifrons* spent in locations between trial phases. All models included the random effect of warren by burrow. The fixed factors included trial phase (Tp), treatment (Tr), time (T), minimum overnight temperature (Mt), rainfall (R) and moon phase (Mp). ΔAICc represents the difference in AIC from the models of best fit, highlighted in bold, and *wi* is the Akaike weight of the model.

Model	**Linear form**	***df***	**Loglik**	**AIC**	**ΔAICc**	***wi***
Top						
L0	**1**	**4**	**1658.98**	**-3309.96**	**0.00**	**0.84**
L1	Tp*Tr	9	1661.41	-3304.82	5.14	0.06
L2	Tp*Tr /T	15	1661.75	-3293.50	16.46	< 0.00
L3	Tp*Tr + R	10	1662.47	-3304.94	5.02	0.07
L4	Tp*Tr + Mp	13	1662.03	-3298.06	11.90	0.00
L5	Tp*Tr + Mt	10	1661.41	-3302.82	7.14	0.02
Track						
LT1	**1**	**4**	**1447.43**	**-2883.88**	**0.00**	**0.94**
LT2	Tp*Tr	9	1445.94	-2876.86	7.02	0.03
LT3	Tp*Tr /T	15	1449.79	-2869.58	14.30	< 0.00
LT4	Tp*Tr + R	10	1448.21	-2876.42	7.46	0.02
LT5	Tp*Tr + Mp	13	1447.51	-2869.56	14.32	< 0.00
LT6	Tp*Tr + Mt	10	1447.78	-2875.02	8.86	0.01
Entrance						
LE0	**1**	**4**	**1592.62**	**-3177.24**	**0.00**	**0.91**
LE1	Tp*Tr	9	1594.56	-3171.12	6.12	0.04
LE2	Tp*Tr /T	15	1596.31	-3162.62	14.62	< 0.00
LE3	Tp*Tr + R	10	1594.63	-3169.26	7.98	0.01
LE4	Tp*Tr + Mp	13	1595.73	-3165.46	11.78	0.00
LE5	Tp*Tr + Mt	10	1594.99	-3169.98	7.26	0.02



Comparison of the proportion of time (mean and 95% confidence intervals) *L. latifrons* spent at the top of the burrow, on the track leading into the burrow and at the burrow entrance before and after treatments were applied, derived from the models.