Impacts of feral deer on threatened ecological communities in south-eastern New South Wales

By Heather Burns

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Candidate's Declaration

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university. To the best of the author's knowledge, it contains no material previously published or written by another person, except where due reference is made in the text.

Heather Burns Date:

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Abstract

Feral deer populations and their associated ecological impacts are variously increasing across Australia. High density deer populations can cause extensive damage through intensive browsing of native plant species, rubbing or ringbarking trees, thrashing saplings, exacerbating soil erosion and reducing water quality. Despite the ecological risks that deer theoretically pose, relatively little quantitative research has so far assessed realised impacts, particularly in Australia. Additionally, the existing limited research has generally focused on high-density populations, while areas with emerging low-density deer populations are often overlooked. As a result, many land managers primarily rely on anecdotal evidence to make decisions about management and whether to apply control measures. The South Coast of New South Wales is a prime example: anecdotal evidence suggests deer populations are expanding, but land managers lack key information needed to create effective, targeted management plans.

In this thesis I aimed to (1) predict the current distribution of deer in the South Coast region to be used as a baseline; and (2) establish the extent to which deer are a current and likely future threat to the conservation of threatened ecological communities within the region. My study examined the area from the coastal city of Nowra, south to the Victorian border, and extending inland to approximately the coastal escarpment. Fieldwork underpinning my studies was conducted between April and July 2019.

To address the first aim, I compiled over 500 presence-only records of feral deer (including 76 records of fallow deer and 60 records of sambar deer) from online databases, knowledgeable National Parks and Wildlife Service staff, and field-based surveys. Maxent species distribution software was used to combine these data and a selection of environmental variables and model the probability of occurrence of deer across my study area. Using these methods I produced three models: one for all species of deer, and two species-specific models for fallow and sambar deer. Two bioclimatic variables, mean temperature of the coldest quarter and precipitation of the driest quarter, had a high contribution to all three models. The predicted probability of occurrence of deer decreased with increasing distance from non-woody vegetation across all models. The geographic representations of all three models agreed with anecdotal reports of deer distribution across this area. However, these models were derived from opportunistic observations only, and they should therefore be interpreted with due caution.

To address my second aim, I focused on the threat that browsing by deer currently poses to plant species within threatened ecological communities in my study area. Three hundred fifty-six transects were surveyed across 89 sites representing eight threatened ecological communities, and browse data was collected for both woody species and non-woody life forms. Data on the presence or absence of deer sign and the abundance of macropod pellets was collected at the same time. Generalised linear mixed models (GLMMs) were developed to predict: (1) the average browse

intensity and (2) proportion of individuals browsed for woody species and non-woody life forms. Sites where deer were present had a higher average intensity and proportion of non-woody life forms browsed, and a higher proportion of woody species browsed. Where deer were present, there was increased browsing pressure on rushes, cycads, sedges, and grass life forms. In contrast, average browse intensity of woody species was not affected by the presence of deer. I also used GLMMs to predict whether average browse intensity and proportion of individuals browsed for woody species and non-woody life forms varies in different threatened ecological communities. Where deer were present, there was increased average browse intensity of non-woody plants in Coastal Saltmarsh, Freshwater Wetland, and Littoral Rainforest threatened ecological communities. The comparison of my models indicates that plant life form (or species) is a better predictor of deer browsing pressure than threatened ecological community.

Although the deer population across this region currently appears to be at a low density, their presence is nevertheless resulting in changes to the browsing impacts on a range of plant species and life forms. My results can be used by land managers to prioritise where deer control should occur, and as a baseline for ongoing monitoring of deer impacts in threatened ecological communities across the South Coast of New South Wales. My results also contribute to a growing body of research about the impacts of feral deer in Australia.

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List of acronyms and abbreviations

BSF- Bangalay Sand Forest

CSM- Coastal Saltmarsh

FWL- Freshwater Wetland

GLM- Generalised linear model

GLMM- Generalised linear mixed model

LRF- Littoral Rainforest

NPWS- National Parks and Wildlife Service

NSW- New South Wales

RFE- River-flat Eucalypt

SOF- Swamp Oak Floodplain Forest

SSF- Swamp Sclerophyll Forest

TEC- Threatened ecological community

THL- *Themeda* Grasslands

Chapter 1: Introduction

Australian land managers have long struggled with the disruptive and destructive consequences of introduced pest species. Along with longstanding issues associated with foxes and rabbits (Cowan and Tyndale-Biscoe, 1997), feral deer are becoming increasingly problematic (Moriarty, 2004a). Since their introduction in the 1800s, six species of deer have successfully established viable populations and expanded their range (Bentley, 1978). Existing literature from around the world shows that high density deer populations can cause significant ecological damage (Côté *et al.*, 2004; Davis *et al.*, 2016). Types of damage include intensive browsing of native species, rubbing or ringbarking trees, thrashing saplings, exacerbating soil erosion and reducing water quality through creation of wallows (Davis *et al.*, 2016; McDowell, 2007).

Despite the ecological risks posed by deer, relatively little quantitative research has examined factors influencing their distribution or realised impacts. While some published distribution and habitat suitability models exist in the literature (Potts *et al.*, 2015; Gormley *et al.*, 2011; Yamada *et al.*, 2003; Forsyth *et al.*, 2009), parts of Australia with emerging deer populations are not included in these models. Moriarty (2004a) produced habitat suitability models across Australia for each established species of feral deer, but these models are at a coarse scale that is not suitable for local management initiatives. In particular, the South Coast region of New South Wales (NSW) is poorly studied in relation to deer, and current models classify the distribution of deer in much of the area as 'absent/unknown' (NSW Department of Primary Industries, 2016). However, anecdotal reports suggest that several species of deer are present throughout the South Coast and are variously increasing their population size and distribution.

My research aims to (1) predict the current distribution of deer in the South Coast region to be used as a baseline; and (2) establish the extent to which deer are a current and likely future threat to the conservation of threatened ecological communities (TECs) within the region. As my research covers a large geographic area and many sites, I had to limit on-ground measures to observations of presence or absence of deer as obtaining estimates of population density (Forsyth, 2005) was too arduous. As deer can impact vegetation communities in a variety of ways, the scope of my second research aim was limited to the threat of browsing by deer in TECs. This aim will be addressed by answering three key and related questions:

1) In which TECs is there a high probability of occurrence of deer?
2) How does browsing by deer affect non-woody vegetation within TECs?
3) How does browsing by deer affect woody vegetation within TECs?

My research provides a significant contribution to the emerging field of feral deer biology by providing land managers on the South Coast with empirical data that can be used to create more informed management strategies. The resolution of the distribution models will provide land managers with information about the probability of occurrence of deer at an ecological community level. These models can also be used as a baseline, and the methods I used can be repeated in future to monitor changes in deer distribution across the South Coast of NSW. Browse data will provide a baseline for average browse intensity in areas where deer are absent or scarce, and indicate which TECs and plant species are at risk from browse damage in areas with significant deer presence. Such an approach contributes to the creation of a repeatable, best-practice monitoring methodology to track changes in future deer distribution and impacts.

This thesis contains six chapters, including this introductory chapter. Chapter 2 examines current literature and discusses ecological impacts and accepted monitoring methodologies for deer populations in both an international and domestic context. It also discusses the current legislation regarding deer classification and management within Australia. Chapter 2 culminates in a discussion of the current knowledge gaps and the focus of my research.

In Chapter 3 I discuss methods I chose to (1) predict the current distribution of deer in the South Coast region of NSW and (2) determine the impact of deer on threatened ecological communities. This includes both the methods I used in field surveys to collect data as well as methods for statistical analysis.

Chapter 4 contains key results of my modelling and statistical analysis. This includes a model that predicts probability of occurrence for all species of feral deer, and two species-specific models for fallow (*Dama dama)* and sambar (*Cervus unicolor)* deer. Chapter 4 also includes the results from my browse analysis which demonstrate the impact deer have on woody and non-woody vegetation, as well as TECs.

In Chapter 5 I discuss how the environmental variables used in my models impact the probability of occurrence of deer throughout my study area, as well as how my models might be used to focus local land management activities for deer. I also discuss the limitations of these models, and how they could be improved with varied inputs and improved sampling effort. Chapter 5 also includes a discussion of how the presence of deer and macropods impact browse across a range of non-woody life forms, woody species, and TECs. I conclude this chapter by discussing future avenues of research to improve the current understanding of feral deer both within my study area and elsewhere across Australia.

In Chapter 6 I summarise the findings of my research and how they could be used to effect local approaches to land management within the South Coast, and more generally introduced species control within Australia.

Chapter 2: Literature Review

In this chapter I will examine current literature and discuss ecological impacts and accepted monitoring methodologies for deer populations in both an international and domestic context. I will also discuss current legislation regarding deer classification and management within Australia. This review of the literature will lead to a discussion of current knowledge gaps, and the focus of my research.

2.1 Overview of distribution

The early 1800s brought the introduction of deer to Australia and, in the 200 years since, six species have successfully established populations in the wild. Populations were established through three main avenues: acclimatisation societies, commercial deer farms, and illegal translocations (Moriarty, 2004a). Acclimatisation societies introduced deer for aesthetics and sport in an attempt to bring some aspects of England to the Australian continent (Bentley, 1978). These herds are the longest-established of all wild populations and have not only remained viable, but expanded their distributions (Moriarty, 2004a). In the past century deer farms have also become a source for wild populations with individuals escaping through worn or broken fences (Bentley, 1978). Accidental escapes were relatively low until the 1990s, when a crash in the commercial deer market led to farm owners deliberately releasing hundreds of individuals (Low, 2002). Releases from acclimatisation societies and commercial farms have now reduced, and illegal translocation of deer has become the most prevalent source of newly-established populations (Moriarty, 2004a). To create more game opportunities, hunters have begun illegally translocating deer into previously unoccupied areas. This continues to accelerate the spread of feral deer and goes beyond the predictable trends of range expansion for each species.

Deer populations in Australia are far from stable, with most species increasing in population size and distribution. Moriarty (2004a) estimated the population of wild deer in Australia to be approximately 200,000. However, this estimate is now 15 years old and populations have likely increased. Moriarty (2004a) also examined the bioclimatic ranges of each wild species, and predicted most species have a large potential for range expansion. Some species have been introduced or released into regions of Australia that do not contain their optimal habitat (Table 2-1). Hog deer are a prime example, having been released in southern Victoria while their ideal habitat is predicted to be similar to that of northern Australia (Moriarty, 2004a). Sambar deer are already widespread within their current range, however, they too are predicted to be better suited to the climate in northern Australia (Moriarty, 2004a). Should species such as these eventually spread to their ideal bioclimatic range, their impacts may increase beyond what has been previously recorded.

Table 2-1: The distribution (Moriarty, 2004a) and preferred habitat (Claridge, 2016a) of the six species of deer that have successfully established wild populations in Australia.

Deer Species	Geographic Distribution	Preferred Habitat Type
Fallow Deer (*Dama dama*)	Widespread along the coast from southern Queensland to Victoria Parts of South Australia and Tasmania	Open forest and woodland, and adjacent grasslands
Red Deer (*Cervus elephus*)	Along the coast from southern Queensland to Victoria Parts of South Australia and Tasmania	Open forest and woodland, and adjacent grasslands
Rusa Deer (*Cervus timorensis*)	Coastal areas from Queensland to South Australia	Grassy clearings Heathlands, woodlands, forests and rainforest
Sambar Deer (*Cervus unicolor*)	New South Wales, Victoria, and the Australian Capital Territory	Dry forest and rainforest, woodland Peripheral farmland Heathland Presence of water is an important factor
Hog Deer (*Axis porcinus*)	Concentrated in Gippsland, Victoria Scattered herds in coastal Victoria and New South Wales	Freshwater and saltwater marshes Heathland, woodland, and forests
Chital Deer (*Axis axis*)	Sparsely distributed from Queensland to South Australia along the coast	

2.2 Ecological impacts

In Australia the ecological impacts caused by feral deer are largely unrecorded. International research, in areas with both native and introduced deer populations, points to the potential for significant negative impacts on ecological communities. The main impacts associated with high densities of deer are browsing damage, ringbarking and rubbing of trees, creation of wallows, and the spread of weeds.

2.2.1 Browsing

Of significant concern to land managers around Australia is the potential impact of deer browsing on native vegetation. In Europe and America, high densities of deer have caused significant reductions in plant species diversity (Gill and Beardall, 2001) and created simplified community structures (Côté *et al.*, 2004). Although Australian native plants are not a natural part of their diet, deer are opportunistic feeders and their broad diet has adapted to include a wide range of species (Claridge, 2016a; Davis *et al.*, 2008; Forsyth and Davis, 2011; Keith and Pellow, 2005).

High-intensity browsing can affect the recruitment and age structure of vegetation communities. The survival of young shoots and seedlings is key to recruitment for many plants, however, shoots and seedlings are highly palatable and often browsed by deer (Tanentzap *et al.*, 2009). This stunts regeneration and limits the proportion of individuals reaching reproductive maturity (Côté *et al.*, 2004). Individuals that do reach the mature stage may be limited in their potential for seedling dispersal and establishment by opportunistic browsing on reproductive organs (Peel *et al.*, 2005). A study in New Zealand found that it can take decades for vegetation communities to recover from the effects of intense browsing, and even low densities of deer can stifle regeneration and recruitment (Tanentzap *et al.*, 2009).

Selective browsing by deer can also shift the composition and physical structure of vegetation. Although deer have broad diets, certain plant species are preferable to others due to their increased palatability. This hierarchy leads to preferential browsing of certain species, and can greatly reduce the abundance of palatable species in areas of high deer density (Gill and Beardall, 2001). With continued browsing, communities can shift towards non-palatable species and eventually lead to the emergence of a new stable state (Côté *et al.*, 2004). Such community shifts can make it difficult for palatable species to re-establish even in the event of reduced browsing intensity (Côté *et al.*, 2004; Tanentzap *et al.*, 2009). Browsing preferentially on certain functional groups may also shift communities in a specific direction. Gill and Beardall (2001) found that preferential grazing on grasses, and the subsequent dispersal of their seeds, shifted communities into more grass-dominated systems. However, as most deer species are browsers and grazers they have the potential to impact both woody and non-woody vegetation.

The complete range of plants consumed by any introduced deer species has not been recorded or catalogued in Australia. Although deer have broad diets, their browsing preferences vary slightly between species and locations. Internationally and within Australia hog deer are widely known to be grazers, however Davis *et al.* (2008) found hog deer in Wilsons Promontory expanded their diet to include browse. Sambar deer have highly plastic diets and in Australia can range from grazers to browsers depending on the season (Forsyth and Davis, 2011). Red deer in Australia have slightly different diets to conspecifics in their native range (Latham *et al.*, 1999; Roberts *et al.*, 2015). Roberts *et al.* (2015) also observed red deer diets varying by gender, with females acting as grazers and males acting as browsers.

Keith and Pellow (2005) examined rusa deer in Royal National Park. Similar to previous international studies, vegetation in deer-inhabited sites showed signs of browsing on shoots, foliage, reproductive material, and bark (Keith and Pellow, 2005). Australian native species were among those heavily impacted, with 69 of 78 species surveyed showing signs of browsing (Keith and Pellow, 2005). As concluded by other authors, Keith and Pellow (2005) found that such intense browsing will likely have a negative impact on the long-term regeneration and recruitment of the local native vegetation.

2.2.2 Ringbarking and rub trees

Ringbarking and bark stripping by deer can cause mortality in areas of high deer density. Trees are often rubbed as a form of signposting for dominant males or to assist in the removal of antler velvet (Claridge, 2016a). In some instances bark removal is considered more damaging than browsing, because the impacts are not limited to the understory and can affect mature individuals (Akashi and Nakashizuka, 1999). National Park rangers in the Australian Alps reported mortalities from rubbing by sambar, and reduced regeneration in yellow box (*Eucalyptus melliodora*) as a result of rubbing by fallow deer (Claridge, 2016b).

Trees at certain age or diameter classes may be more vulnerable to ringbarking or rubbing. In Australia, deer primarily select rub trees that have reached a height of over two metres (Claridge, 2016a). A study of sambar in their native range in Taiwan found a correlation between ringbarking frequency and diameter at breast height (DBH) (Yen *et al.*, 2015). Trees with smaller DBH (5-30 cm) were preferentially selected, with the highest frequency of rubbed trees around 20 cm DBH (Yen *et al.*, 2015). As with browsing, preferentially rubbing smaller or younger individuals could impact forest regeneration and structure.

Two key case studies of the impacts of rubbing have been undertaken in Australia. Bilney (2013) found a high prevalence of rubbing by sambar deer in yellow-wood rainforest in eastern Victoria. Trees that were severely rubbed often had reduced foliage cover, with attempts at regeneration from shoots destroyed (Bilney, 2013). Of the ecological communities of note in the yellow-wood rainforest, threatened Littoral Rainforest was severely damaged by this behaviour. Bennett and Coulson (2010) observed both thrashing of young saplings and rubbing of mature trees in the threatened shiny nematolepis (*Nematolepis wilsonii*). Thrashing of saplings primarily occurred along game trails used by dominant males, most likely a result of territorial marking (Bennett and Coulson, 2010). High rates of mortality in saplings severely limited regeneration and threatened the persistence of shiny nematolepis, especially when coupled with severe rubbing and bark removal in mature trees (Bennett and Coulson, 2010). These consequences are likely not limited to one species, and high sapling mortality could be an issue for all species subject to thrashing.

2.2.3 Weed dispersal

As browsers with broad diets, deer have the potential to spread large amounts of seed. Davis *et al.* (2010) found that a population of hog deer in Victoria had the potential to disperse 130,000 viable seeds per day. When coupled with their ability to travel large distances (Davis *et al.*, 2010), deer become potential vectors for large-scale weed dispersal (Claridge *et al.*, 2016).

Seeds spread by deer most likely include both native and introduced plant species. A study by Claridge *et al.* (2016) compared seedlings arising from scat of native eastern grey kangaroos (*Macropus giganteus*) and fallow deer. While both distribute introduced seeds, deer scat contained significantly more viable species than kangaroo when grown under the same conditions (Claridge *et al.*, 2016). Introduced species, especially those that have been recognised as a significant threat, can cause large scale problems if they are aided in their dispersal by herbivores. Although Claridge *et al.* (2016) did not find evidence of nationally significant weeds in fallow deer scat, a related survey of National Park staff in the Australian Alps (Claridge, 2016b) revealed that rangers believed deer were spreading blackberries, among other introduced weeds. Contrary to these findings, Davis *et al.* (2010) found that hog deer in Victoria spread primarily native species. Differences in these results may have been due to slight variations in browsing preferences between species, as well as variation in the abundance of introduced plant species present in each study area.

Deer behavioural traits may further facilitate the spread of weeds. Heavy browsing by deer reduces both the viability of native plants and competitive pressure, allowing weeds to establish (Peel *et al.*, 2005). Habitat use patterns may also direct seed dispersal via scat to communities favoured by deer (Davis *et al.*, 2010). For example, deer may browse on a weed species in one location, then travel elsewhere to where the weed has not yet spread and deposit viable seeds via scat. Ecological communities frequented by deer are at higher risk from this form of weed exposure than those that are not.

2.2.4 Wallows and scrapes

Wallows created by deer are typically constructed near waterways or drainage lines and can negatively impact the water quality of a catchment. Deer form wallows by scraping a section of a stream or other riparian area, disturbing sediment and facilitating downstream nutrient transport (McDowell, 2007) (Figure 2-1). To mark their territory and create scent markers, deer often urinate and defecate in wallows before coating themselves in mud (Claridge, 2016a). This is particularly common during the mating season (Semiadi *et al.*, 1994), and provides an additional source of contamination. In a study of catchments containing deer farms, McDowell (2007) observed that catchments directly connected to wallows registered levels of contaminants above State-recommended guidelines. Fencing wallows and preventing deer access to these sites resulted in reduced contaminant concentrations and suspended sediments (McDowell, 2008). In feral deer populations, survey responses collected by Claridge (2016b) provided anecdotal evidence for reduced water quality in the Australian Alps in areas associated with deer wallows. Trampling and wallowing by sambar deer was also anecdotally reported as a threat to alpine and subalpine bogs, as well as riparian and wetland ecosystems (Claridge, 2016b).

Figure 2-1: Deer wallow near Freshwater Wetland in Ben Boyd National Park (photo: Heather Burns)

2.2.5 Soil erosion

Erosion caused by feral deer in Australia has been noted as an impact in several anecdotal reports. In Heathcote National Park, feral deer reportedly cause erosion and soil slip (NPWS, 2000). Keith and Pellow (2005) also measured significant erosion at sites frequently visited by rusa deer in Royal National Park. In one section where deer were frequently fed by locals, 0.6 metres depth of soil was lost over a period of four years (Keith and Pellow, 2005).

2.3 Current management

2.3.1 Legal status of deer

Australian State and Commonwealth legislation varies widely in its classification of feral deer. The Australian Capital Territory, New South Wales, Northern Territory, Queensland, South Australia, and Western Australia classify deer as pests (Davis *et al.*, 2016). Herbivory and environmental degradation caused by feral deer has also been declared a Key Threatening Process (KTP) in NSW under the *Biodiversity Conservation Act* (2016) . The state of Victoria classifies deer as game, and hunting is permitted year-round for most species. At the opposite end of the spectrum, Tasmania has classified feral deer as partly protected wildlife under the *Nature Conservation Act* (2002). Hunting is heavily restricted in Tasmania, with specified seasons and bag limits for each species. At the Commonwealth level, deer are classified as pests but are not listed as a KTP under the *Environmental Protection and Biodiversity Conservation Act* (1999) .

Legislation in New South Wales has recently changed to reflect the increasing threat posed by feral deer. On 6 September 2019 deer were de-listed as a game animal and obtained pest status under the NSW *Game and Feral Animal Control Act* (2002) . Private landholders no longer require a game license to shoot deer on their property, and can manage deer in the same way as any other feral animal. This means that landholders are no longer confined by bag limits, restricted hunting seasons, or bans on night hunting.

2.3.2 Current management approaches

The process of successful establishment and subsequent management of invasive species has recently become a heavily researched topic. Blackburn *et al.* (2011) created a unified framework for both plant and animal invasions, identifying distinct stages of invasion, the associated barriers, and appropriate management strategies. Management strategies include prevention, eradication, containment, and mitigation (Blackburn *et al.*, 2011). Managing the various stages of invasion can be time and resource intensive, therefore the Victoria Department of Primary Industries (2017) uses a model to assess the most effective management strategy based on the area occupied by an invasive species (Figure 2-2). Depending on the species and region of Australia, feral deer are in either the establishment or spread stage of invasion. Where deer are still in the establishment stage of invasion, prevention or eradication may still be viable options. In areas where deer are established at relatively high densities, the best strategies may be containment or asset protection.

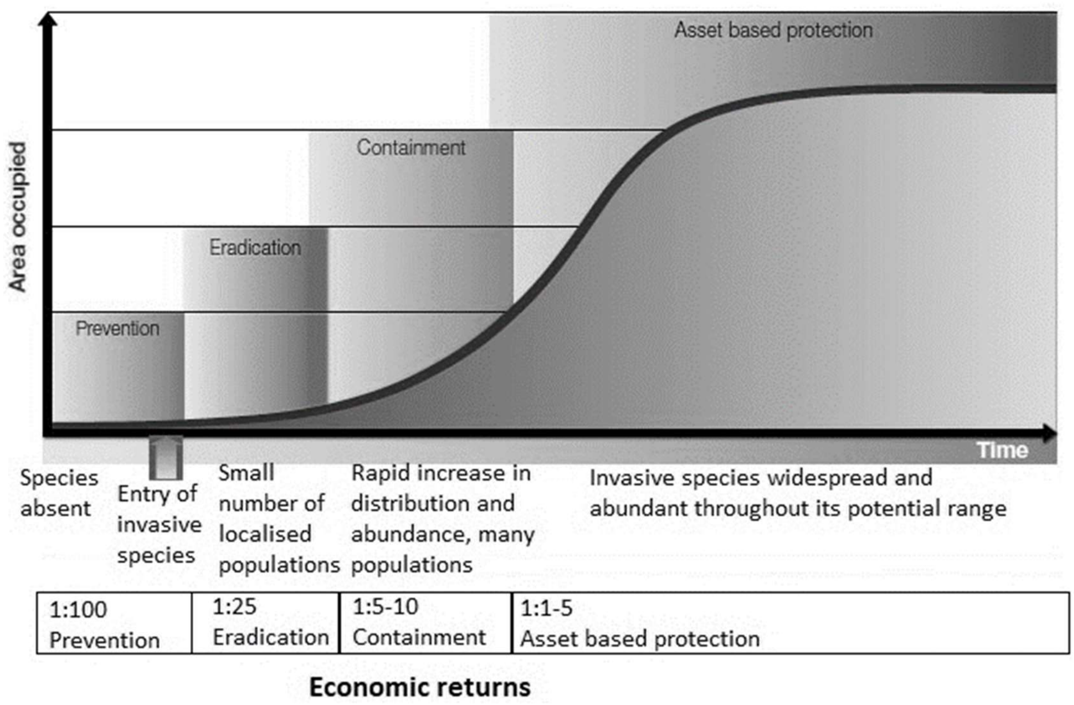

Figure 2-2: The invasion curve, with appropriate management strategies for each stage (Victoria Department of Primary Industries, 2017)

To effectively implement this framework, Australian land managers require quantitative data related to deer abundance, impacts, and appropriate control methods. Unfortunately this information is not always readily available. For example, the Australian Alps region has a well-established feral deer population. However, due to a lack of available data they rely on anecdotal and not science-based evidence to track changes in population size and density (Claridge, 2016b). Increasing the amount of research on deer impacts and distribution should provide the data land managers around Australia need to implement successful scientific-based management plans (Moriarty, 2009).

2.4 Monitoring methodology

Establishing an appropriate methodology, to answer a defined question(s), is a key consideration for any research. When it comes to understanding the distribution of deer, there are many viable options. The National Workshop on Deer Management (2016) called for the creation of a best-practice guide to be the standard for deer monitoring in Australia (Forsyth *et al.*, 2016). In the case of establishing a long-term monitoring program, deciding on a single methodology and sticking to it is critical (Forsyth *et al.*, 2011). When methodologies change over time it is difficult to directly compare current populations to past records.

Creating a feasible methodology in a management context relies on the application of a few key principles. Gaines *et al.* (1999) discuss how monitoring starts with an initial survey to establish what is present, and then continues over time to detect changes. A key component in this process is the use of repeatable methods (Cassey and Blackburn, 2006). Using repeatable methodology across spatial and temporal scales means results can be compared, and used to inform management actions. Methodologies must also be time and resource efficient. Spending too much time at a single site can mean that a smaller percentage of the overall management area is surveyed.

2.4.1 Camera traps

Camera traps are becoming an increasingly popular method for capturing a range of information about a population (Meek *et al.*, 2014b; Moseby and Read, 2014; Urlus *et al.*, 2014). Initially designed for hunters to monitor their prey, camera traps have recently become more common in ecological monitoring. Although they have been trialled in studies of small mammals, cameras are ideally suited to large mammals such as deer (Urlus *et al.*, 2014). Species with individually-identifiable markings can be "marked and recaptured" on camera traps to estimate population density. If individuals cannot be uniquely identified, data collection is limited to presence/absence information and limited occupancy modelling.

The methodology may seem rather simple, but understanding how camera traps work and avoiding pitfalls provides the most accurate results (Meek *et al.*, 2014a). Camera traps are subject to false negatives (Newey *et al.*, 2015), failure to capture an image when an animal is present, which can reduce the probability of detection. Meek *et al.* (2015) identified the time lag between detection of an animal and capturing an image as a key factor in false negatives or blank images, particularly when the animal is moving quickly across the detection zone. The occurrence of false negatives can be reduced by conducting a pilot study to optimise camera settings for the desired study species.

If used correctly, camera traps can fit the requirements of monitoring methodology for management purposes. Maintaining consistent camera settings, models, and deployment lengths is key when conducting repeatable camera trap surveys (Meek *et al.*, 2014a). Although the initial investment in cameras may be expensive, the cost per survey day is relatively low if they are used over long study periods (Lyra-Jorge *et al.*, 2008). Using camera traps requires multiple visits to deploy, maintain, and collect each camera. These visits are often quick and cameras can be left for long periods of time between visits. Therefore, camera traps can be used in a way that meets the key criteria of a successful monitoring methodology. However, if the study species is not individually identifiable there may be limitations to the range and quality of data that can be obtained.

2.4.2 Faecal pellet surveys

Faecal pellet surveys encompass a wide range of methodologies used to estimate absolute abundance of deer populations. Two broad categories in this survey type are faecal standing crop and faecal accumulation rate. Faecal standing crop involves using an experimentally determined decay rate and counts of complete pellet groups to determine density estimates in a given area (Amos *et al.*, 2014). Faecal accumulation rate does not require the determination of decay rates, and instead estimates density based on the accumulation of pellet groups over a set period of time (Amos *et al.*, 2014). Conducting a small-scale pilot study prior to surveying the entire study area is important in both types of survey (Alves *et al.*, 2013), making this method extremely labour intensive (Amos *et al.*, 2014). In both study types, the accuracy of the estimates increases with the number of study sites used (Alves *et al.*, 2013).

Although faecal pellet counts produce relatively precise estimates, they may not always be ideal. Aside from recording mere presence/absence pellet data, no methodology exists for determining population density that is not considerably time-intensive. Detection of pellets also varies by habitat type, with decreased pellet visibility in dense vegetation. This makes pellet counts poorly suited for broad-scale data collection, which is often required for management purposes.

2.4.3 Distance sampling

Distance sampling provides a relatively simple method for measuring the absolute abundance of deer populations. Transects are walked for a set distance and when groups of deer are sighted their distance from the transect and group size are recorded (Amos *et al.*, 2014). Although simple, this methodology relies on four key assumptions: every individual on the transect is detected, the distance between observer and subject is accurately measured, transects are randomly selected, and the individuals being studied do not move before they are detected (Buckland *et al.*, 2015). When using distance sampling in a study, it is important to test that these assumptions have been accounted for (Focardi *et al.*, 2005). Distance sampling is suitable for use by multiple observers. However, Koenen *et al.* (2002) recommends using a set calibration course to ensure there is minimal variability between observers. Studies using this methodology have been successful, and Focardi *et al.* (2005) concluded that distance sampling produces results suitable for use in management decisions.

In an Australian context, this methodology has had limited application. A study in Queensland by Amos *et al.* (2014) found distance sampling to be repeatedly precise but labour-intensive. However, in many parts of Australia deer are known to be elusive and avoid contact with humans. Dense vegetation also presents problems for this methodology, as many individual deer go undetected. A pilot study in the survey area before using this methodology is required to ensure that herds of deer are easily detectable and do not flee.

2.4.4 Aerial surveys

Aerial surveys involve observations taken from a helicopter and are often used to monitor large mammals. Aerial surveys have been used to measure the abundance of macropods in the rangelands (Short and Hone, 1988) and horses in the Australian Alps (Dawson and Miller, 2008). Amos *et al.* (2014) used the aerial survey technique with feral deer in Queensland and found it to have a high detection probability, produce relatively precise estimates and require minimal labour (Amos *et al.*, 2014). However, aerial surveys are limited by their expense and have reduced precision when sampling dense vegetation (Amos *et al.*, 2014).

2.4.5 Spotlight counts

Spotlight counts are controversial in the literature with regards to their precision and relevance in monitoring deer populations. Spotlight methodology typically involves driving down an accessible track and using a strong light to identify and count groups of deer. When trialling methods in Queensland, Amos *et al.* (2014) found spotlighting to be sufficiently precise for the amount of labour and monetary investment involved. Garel *et al.* (2010) and Fafarman and DeYoung (1986), however, found that spotlighting population estimates were consistently below those found using other methods. Collier *et al.* (2007) also compared population estimates using thermal imaging and spotlight counting, and found that spotlight counts detected only half the number of individuals as thermal imaging. These examples suggest that spotlight counting is an ineffective method of estimating absolute abundance. However, Garel *et al.* (2010) did find that spotlight counting may be appropriate for land managers to estimate relative abundance and track population changes over time.

2.4.6 Sand plots

Sand plots are primarily used to capture the tracks of animals passing through a site (Bider, 1968). Although they have been used on their own, they are increasingly being used in conjunction with camera traps (Lyra-Jorge *et al.*, 2008). Camera traps are known to produce false negatives, and sand plots placed nearby can capture the tracks of individuals not caught on camera (Espartosa *et al.*, 2011). Additionally, sand traps require minimal investment of time and money, giving them a higher detection efficiency than camera traps (Lyra-Jorge *et al.*, 2008). However, sand plots are limited in their application as they only provide presence/absence rather than abundance data.

2.4.7 Signs

Various signs can be used to identify the presence or absence of deer in an area. Deer alter their habitat in ways that can be easily identified such as scrapes, wallows, preaching trees, ringbarking, and rub trees (Claridge, 2016a; Akashi and Nakashizuka, 1999; Bilney, 2013; McDowell, 2007). Rangers in the Australian Alps reported that such signs were common and easily detectable in areas where deer were present, making them ideal indicators of deer presence (Claridge, 2016b). The application of sign data is limited to determining the presence of deer in an area, and cannot be used to estimate abundance or density.

2.4.8 Ecological indicators

Ecological indicators lie at the intersection of monitoring and management, to challenge the need for absolute estimates of population size and density. Practitioners argue that the size of a population matters less than it's impacts, and focusing on impact-related goals may be more beneficial than simply trying to reduce population size (Braysher *et al.*, 2012). Shifting thinking towards a more species-habitat system that can be altered and adjusted using adaptive management may be best for reaching the ideal community state (Morellet *et al.*, 2007). As demonstrated elsewhere, estimating absolute population densities is highly resource and time intensive and often impractical. Morellet *et al.* (2007) emphasise the need for establishing a link between population density and impact, and developing indicators that accurately reflect these links.

In relation to deer populations and their impact, browsing indices have been identified as a key ecological indicator. Rawinski (2017) found that the evenness with which a section of vegetation was browsed indicated the intensity of browsing pressure by deer. Similarly Moore *et al.* (1999) used growth and scars on terminal and lateral shoots of browsed plants to determine the intervals between browsing activity. Longer intervals between browsing indicated a lower intensity than patches with shorter intervals. LaGory *et al.* (1985) compared total available browse at each site, which was based on canopy cover, to the amount of vegetation that was actually browsed to create an index for browse intensity. Beals *et al.* (1960) monitored impacts on a stand scale and compared values for relative browse of each species, relative categories of browse intensity, and the minimum density of browsed individuals. Morellet *et al.* (2001) created a browsing index that measured the number of woody plants with >5% total volume browsed over the winter season. There is no singular ideal index, however the range of methodologies available allows researchers to choose the most appropriate option for a given set of conditions and aims.

2.4.9 Assessing browse damage

Australia presents a unique challenge when it comes to quantifying browsing impacts of deer. Unlike other countries where deer are present, deer in Australia occupy the same browsing niche as native herbivores such as kangaroos and wallabies (Davis *et al.*, 2016). Park rangers in the Australian Alps believe deer are causing more damage than native herbivores, however, this claim is anecdotal (Claridge, 2016b). Apart from costly and time-intensive exclusion plots (Bennett and Coulson, 2008; Moriarty, 2004b), no standardised or effective method has yet emerged to distinguish browse impacts between species.

2.5 Knowledge gaps and research aims

The single largest issue for Australian deer monitoring and management is the lack of quantitative data available to land managers. Moriarty (2004a) and Claridge (2016b) both report significant and large-scale negative impacts of deer, however, the majority of their evidence is anecdotal. Claridge (2016b) reported that 90% of survey responses from land managers identified anecdotal evidence as the basis for understanding deer populations and distributions. Anecdotal reports cannot be used quantitatively to track changes over time, a key feature for assessing the success of management actions (Likens and Lindenmayer, 2018). Improved understanding of deer population demographics was flagged by both Davis *et al.* (2016) and the National Workshop on Deer Management (Forsyth *et al.*, 2016) as a key priority for future research.

Despite limitations, anecdotal reports can still be useful and should not be overlooked. Published examples from overseas prove just how damaging deer can be to a variety of ecosystems and outcomes are unlikely to be different in Australia. Though difficult, quantifying browsing impacts nevertheless would provide valuable information for assessing the threat deer pose to various communities.

Although deer are fairly widespread across the eastern coast of Australia, very limited research has been done on how deer impact TECs (Davis *et al.*, 2016). As a result, local land managers lack information to effectively manage deer impacts at these sites. In Royal National Park, grazing and browsing by feral deer is reportedly threatening an endangered population of black cypress pine (*Callitris endlicheri*) (Department of Environment and Climate Change NSW, 2007). Areas of Littoral Rainforest, sandstone gully forests, and sandstone heath showed significant damage in the presence of high deer densities (NPWS and Royal National Park Deer Working Group, 2005). Land managers near Port Macquarie have also reported significant deer impacts in littoral rainforest, which is classified as a TEC in New South Wales. Despite these reports, most of these negative impacts remain unquantified and are therefore difficult to respond to. Quantitative information on the presence and impact of deer in these areas can help develop informed adaptive management strategies.

The existing habitat suitability and population distribution models for deer in Australia are often at a large scale and have not been generated frequently enough to characterise the rapidly changing conditions. Large scale models make it difficult to create effective management plans for individual parks or vegetation communities. Effective management plans require regularly updated habitat suitability and distribution models at fine-scale outputs. Land managers in regions with little information on feral deer, such as the South Coast region of New South Wales (NSW), would greatly benefit from accurate, fine-scale models of the distribution of feral deer. Despite anecdotal reports of a growing deer presence in the area, the distribution of deer in the South Coast has been classified as ‘absent/unknown’ in current models (NSW Department of Primary Industries, 2016).

In this thesis I aim to: (1) predict the current distribution of feral deer in the South Coast region to provide a baseline for monitoring, and (2) establish the extent to which deer are a current and likely future threat to the conservation of threatened ecological communities in the region. My research is confined to current and future threats deer pose to threatened ecological communities through browse damage. This will be addressed by answering the following questions:

1) In which TECs is there a high probability of occurrence of deer?
2) How does browsing by deer affect non-woody vegetation within TECs?
3) How does browsing by deer affect woody vegetation within TECs?

I hypothesise that deer will have an impact on both woody and non-woody browse within TECs over and beyond that of native herbivores. That is, sites where deer are present will have increased levels of browsed vegetation compared to sites where deer are absent. I also hypothesise that browsing pressure by deer will vary between TEC types.

Chapter 3: Methods

3.1 Study area

My study area is in the South Coast of NSW, lying between the coastal city of Nowra (34.88° S, 150.60° E) in the north and the Victorian border (37.49° S, 149.97° E) in the south, extending inland to approximately the top of the coastal escarpment (Figure 3-1). The annual average rainfall ranges from 600-1000 mm/yr in the southern half of my study area to 1000-1500 mm/yr in the northern half. The mean annual temperature for this region is 12–15 °C.

The complete geographic extent of my study area will be used to address my first aim: predicting the current distribution of deer to create a baseline for monitoring. My study area is dominated by wet and dry sclerophyll forest, and also contains a variety of TECs. Eight of these TECs are the focus for my second research aim: establishing the extent to which deer are a current and likely future threat to the conservation of TECs within the region (Table 3-1).

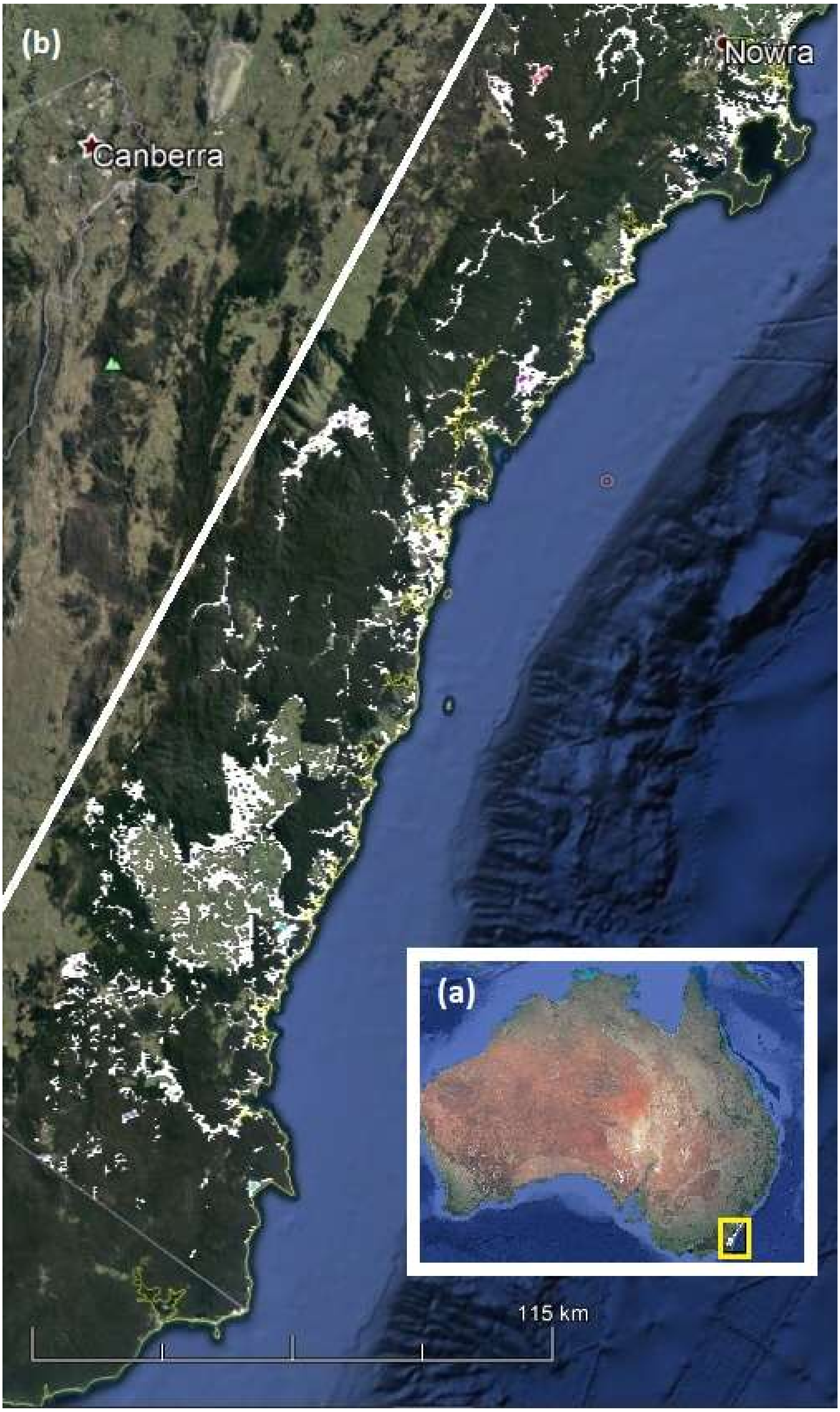

Figure 3-1: Map of survey area and predicted distribution of TECs

(a) Location of my study area within Australia (b) The section of the South Coast I will be studying, ranging from Nowra to the Victorian border (Map data: Google Earth Pro (2018)). The predicted distribution of TECs (New South Wales National Parks and Wildlife Service, 2019) is represented in white on the map. The bold white line represents the top of the coastal escarpment.

Table 3-1: The eight TECs that are the focus of the second aim of my thesis.

Threatened Ecological Community	Description
Littoral Rainforest (LRF)	This community is most common within 2 km of the coast, and is likely to be more common in the northern region of my study area. Vegetation composition is characterised by a high percentage of rainforest species and an abundance of vines in the canopy layer.
Swamp Oak Floodplain Forest (SOF)	This community can be found throughout my study area. Dominant tree species varies with location; north of Bermagui *Casuarina glauca* is dominant, and south of Bermagui *Melaleuca ericifolia* is dominant.
Swamp Sclerophyll Forest on Coastal Floodplains (SSF)	Paperbarks and eucalypt species, in varying densities, are most common in this community. Small trees and shrubs also occur, along with a diverse groundcover of grasses, ferns, forbs, and sedges.
River Flat Eucalypt Forest on Coastal Floodplains (RFE)	This community is characterised by a single or mixed-species open eucalypt forest with some smaller trees and shrubs in the understory. These forests are located on river floodplains in the coastal region.
Bangalay Sand Forest (BSF)	Bangalay (*Eucalyptus botryoides*) and coast banksia (*Banksia integrifolia)* of varying densities can be found in this community. Shrub species are largely sclerophyllous. Groundcover may be sparse in areas where the canopy or shrub layer is dense.
Coastal Saltmarsh (CSM)	This community occurs at the edge of coastal estuaries or lagoons. Trees and shrubs are very scarce and salt-tolerant grasses, sedges, and herbs dominate the groundcover.
Freshwater Wetlands on Coastal Floodplains (FWL)	This community is characterised by the presence of occasional or semi-permanent standing water. Sedges, herbs, and reeds are most common in the ground layer, and the occurrence of trees and shrubs is very rare.
Themeda Grassland on Seacliffs and Coastal Headlands (THL)	This closed tussock grassland community is dominated by *Themeda australis.* In areas where shrub or heath occur in low densities, grassland species will occupy gaps in the matrix.

3.2 Predicting the current distribution of deer

I addressed the first aim of my study, predicting the current distribution of deer in the South Coast, using two key steps. First, I collected presence-only data for all species of deer across the study area from all resources available. Second, I conducted field surveys to determine the presence (or absence) of deer across a representative set of TECs within conservation reserves across the region. My methods broadly follow those utilised by Gormley *et al.* (2011) in a study of habitat use by sambar deer in Victoria.

3.2.1 Existing databases

Existing records of deer signs or sightings were collected from a range of sources. Data for the NSW region were collected from the Atlas of Living Australia, NSW BioNet, and NatureMapr. I also worked with key individuals from the NSW Department of Planning, Industry, and Environment to collect records of deer that had not yet been published in existing online databases. With the assistance of National Parks and Wildlife Service (NPWS) staff, I circulated maps throughout the major branch offices in the region and asked field staff to mark locations in their local area where they had observed signs of deer. Maps were printed in a format similar to those commonly used by NPWS, thus avoiding some of the issues experienced by Yamada *et al.* (2003) when eliciting expert knowledge. Data from these maps were later transcribed and recorded as sets of GPS coordinates. Using these combination of methods, I obtained 678 records of deer presence in my study area (Table 3-3).

3.2.2 Field surveys

Field surveys were conducted only within NPWS estate and were limited to the eight TECs listed in Table 3-1. These TECs were nominated by NPWS because they are poorly represented in current research about the ecological impacts of deer, and anecdotal evidence suggests the impact of deer in these areas is increasing. The study area was further stratified into three geographic regions of roughly equal area: north (areas between Nowra and Bateman's Bay), middle (areas between Batemans Bay and Bermagui), and south (areas between Bermagui and the Victorian border). This stratification ensured that sites were distributed equally across the study area. I used ArcMap (Esri, 2019) to randomly allocate sites within the predicted distribution of TECs (New South Wales National Parks and Wildlife Service, 2019) and NPWS estate (State Government of NSW and Department of Planning Industry and Environment, 2017). I attempted to distribute sites equally between the 24 strata (eight TECs x three geographic regions). However, sites were excluded if they were not easily accessible by 4WD vehicle or did not contain the predicted TEC. In total, 89 sites were surveyed.

At each site field surveys were conducted to assess the presence (and absence) of deer using searches for sign along transects. At each site a 400 metre transect was walked with the aim of detecting the following signs of deer within 1 metre of the transect on either side: faecal pellet groups, rubbed trees (Figure 3-3), tracks (Figure 3-2), and cast antlers (Gormley *et al.*, 2011). Whenever possible, transects were aligned with areas where the probability of detection for sign was likely to be highest, such as near waterways and along game trails (Gormley *et al.*, 2011). Along each transect, the abundance of each type of sign was counted.

Figure 3-2: An example of tracks that were used to detect deer presence (photo: Heather Burns)

Figure 3-3: An example of antler marks on a tree, another sign used to detect deer presence (photo: Heather Burns)

This methodology was carefully selected given the size of my study area and the time frame available. Although Gormley *et al.* (2011) found that using camera traps increased the probability of detection for sambar deer, a brief pilot study indicated that this method was too time consuming for my study. By using a simple search for signs of deer presence rather than focusing on gathering density data (i.e. using faecal pellet counts or camera traps) I sampled a large number of sites across my study area rather than focusing in detail on a select few. Searching for signs of deer along transects is relatively quick and easily repeatable between sites.

3.2.3 Predictive modelling

I predicted and mapped the probability of occurrence of deer across the study area using Maxent (Philips *et al.*, 2006). Maxent uses presence-only data and the spatial variation of biophysical characteristics to determine the probability of occurrence for each cell within a defined geographic range using a modelling technique called maximum entropy (Jaynes, 1957). This technique assumes that the best possible representation of conditions defined by a given dataset is the one with greatest entropy. Creating accurate habitat suitability models without presence/absence data can be challenging, however Maxent's machine-learning software has been highly successful using presence-only data (Ward, 2007).

For this study, eight biophysical variables (Table 3-2) were chosen as potential explanatory variables for habitat suitability based on discussions with experts and previous research (Forsyth *et al.*, 2009; Gormley *et al.*, 2011; Potts *et al.*, 2015). Using ArcGIS (Esri, 2019), all explanatory variables were resampled to the same cell size and clipped to the same spatial extent. When all rasters have identical spatial extents and cell sizes, Maxent is able to accurately compare data across all potential explanatory variables. An (X,Y) grid size of (0.000373°, 0.000373°) [41 metres x 41 metres] was chosen because it was the original resolution of the distance to water raster. The presence of perennial water sources was identified as an important factor for habitat suitability by experts (Forsyth *et al.*, 2009), and therefore this was selected as a key factor for determining resolution. Due to the abundance of perennial regional water sources, resampling the distance to water layer to a coarser resolution would result in a smaller range of potential distance values for each cell and reduce the model's ability to discriminate between sites. Maxent produces an output in the same resolution of the original input data, therefore using a fine resolution for my inputs not only retained valuable information for the model to use, but also created an output at an equally fine resolution. Had I used the traditional methodology and resampled all my inputs to the coarsest resolution (0.002°, 0.002°) [217 metres x 217 metres], both the quality of data used in the model and model output quality would be poorer. Some explanatory variables had an original resolution finer than the distance to water raster and were therefore resampled to a coarser resolution (Table 3-2). The original resolutions for these layers were within 0.0001° [11 metres] of the final resolution, therefore it is unlikely that a large amount of detail was lost in this process. Some explanatory variables had an original resolution coarser than the distance to water raster (Table 3-2) and the data were interpolated to produce a finer resolution. If a pair of explanatory variables was highly correlated (>0.7), one variable in the pair was eliminated.

Using the methods above, three models were created. The first model combined all presence points to create a model for all deer species in the study area, while the second and third models used records at the species level to predict the probability of occurrence for fallow and sambar deer, respectively. All presence data that were beyond the spatial extent of the bioclimatic variables, as well as points that lacked data for one or more bioclimatic variables, were excluded from the model. This reduced the number of presence-only points used to 561 for the all-species model, 76 for the fallow model, and 60 for the sambar model. While each species-specific model may have fewer than 100 points, Maxent has the ability to produce strong models with relatively little data (West *et al.*, 2016). In each model, known presence points were combined with 10,000 randomly selected points in Maxent to calculate the predicted probability of occurrence (0-1) for each cell.

Table 3-2: Potential explanatory variables used in my Maxent model, along with the source of spatial data for each variable.

Potential Explanatory Variable	Source	Original Resolution (degree units)
Vegetation formation	Sharing and Enabling Environmental Data NSW	(0.002, 0.002)
Topographic wetness index	CSIRO Data Access Portal	(0.00027, 0.00027)
Distance to non-woody vegetation	Sharing and Enabling Environmental Data NSW	(0.00039, 0.00039)
Mean annual rainfall	ANUCLIM 6.1	(0.00083, 0.00083)
Mean annual temperature	ANUCLIM 6.1	(0.00083, 0.00083)
Slope	Robinson *et al.* (2014)	(0.00083, 0.00083)
Distance to perennial water source	Geoscience Australia (2016)	(0.000373, 0.000373)
Mean precipitation of driest quarter	ANUCLIM 6.1	(0.00083, 0.00083)
Minimum temperature coldest quarter	ANUCLIM 6.1	(0.00083, 0.00083)
Land Use Class	Sharing and Enabling Environmental Data NSW	(0.000373, 0.000373)

Table 3-3: The number of presence data available for each species

Species	Presence Data
Unspecified	459
Dama dama	168
Cervus unicolor	66
Cervus timorensis	4
Cervus elaphus	1

Once the Maxent model was created, the outputs were used to identify the risk deer pose to each of the eight TECs. Using predicted probabilities of occurrence, minimum thresholds of 0.5, 0.75 and 0.9 were set. A habitat suitability score of 0.5 was selected as a minimum threshold to represent all areas that are more likely than not to be suitable for deer. This minimum threshold included a large proportion of TECs, and therefore additional minimum thresholds of 0.75 and 0.9 were chosen to identify areas with a high probability of supporting deer.

3.3 The impact of deer on threatened ecological communities

As discussed in Chapter 2, deer can cause damage to vegetation communities in a variety of ways (i.e., ringbarking, browsing, seed dispersal). My thesis seeks to establish the extent to which browsing by deer is a threat to the conservation of TECs. My methodology involved using data collected from 89 sites in my deer presence/absence survey as well as additional surveys of browse damage on these sites. All sites were randomly distributed across 24 strata (eight TECs x three geographic regions). Browse was measured for non-woody life forms and woody species using two metrics: the proportion of individual plants browsed and average browse intensity. I recorded signs of presence and abundance of other sympatric herbivores to try and separate browsing effects of deer from other species.

3.3.1 Field surveys

Browse damage was sampled at each site using four transects 50 metres in length. In total 356 transects were surveyed (89 sites x 4 transects per site). Transects were located along game tracks at each site where present, and randomly located where game tracks were absent. Each transect was sampled using the step-point method (Evans and Love, 1957) with points established at every metre (a total of 50 points per transect). The following data were recorded at each point: (a) species of woody vegetation present; (b) if woody vegetation was present, the browsing intensity (from 0-100%); (c) life form of non-woody vegetation; (d) if non-woody vegetation was present, the browsing intensity was recorded (from 0-100%). Browsing intensity was calculated as the percentage of available foliage that had been browsed for each individual (Figure 3-4). If an individual plant intersected the transect at multiple points, data were recorded for each point. The vegetation surveyed was limited to individuals with growth below 3 metres to restrict the sample to only vegetation within the browsing range of deer (Peel *et al.*, 2005).

Deer occupy a similar browsing niche to some native and introduced herbivores (Davis *et al.*, 2008). To account for this at each site, I counted scats for macropods (i.e. swamp wallaby (*Wallabia bicolor*), eastern grey kangaroo (*Macropus giganteus*), and red-necked wallaby (*M. rufogriseus*)), the common wombat (*Vombatus ursinus*), and European rabbit (*Oryctolagus cuniculus*). The abundance of scats, and associated species of sympatric herbivores was recorded simultaneously during the presence/absence survey of deer.

Figure 3-4: An example of estimating browsing intensity (photo: Heather Burns)
The browsed portions of this cycad (circled in red) are estimated to be approximately 20% of the total available browse for this individual. Therefore the browsing intensity for this cycad would be recorded as 20%.

3.3.2 Statistical Analysis

I used the data collected in my field surveys to test the hypothesis that browsing is greater in areas where deer are present. As the abundance of deer sign recorded at each positive site was uniformly low, a 0 (absent) or 1 (present) was instead recorded for the occurrence of deer at each site. In contrast, macropods were present at every site, and the abundance of faecal pellet groups (the most abundant sign type) was used as an index of relative population density across sites.

Table 3-4: Measures of browse used in my analysis, and their respective levels.

Measure of browse	Levels
Proportion of non-woody life forms browsed & Average browse of non-woody life forms	Grass, Rush, Forb, Sedge, Cycad, Fern, Vine
Proportion of woody species browsed & Average browse of woody species	*Acacia dealbata* *Acacia implexa* *Acacia longifolia* *Acronychia oblongifolia* *Babingtonia virgata* *Banksia spinulosa* *Bursaria spinosa* *Callistemon citrinus* *Casuarina cunninghamiana* *Dodonaea triquetra* *Leucopogon parviflorus* *Phyllanthus gunnii*

Using R Studio (R Core Development Team, 2018) four generalised linear mixed models (GLMMs) were generated. Each model explored the relationship between one of four measures of browse (proportion of non-woody life forms browsed, average browse intensity of non-woody life forms, proportion of woody species browsed, and average browse intensity of woody species) and four explanatory variables: TEC, deer presence, macropod pellet abundance, and life form (for non-woody plants) or species (for woody plants). Non-woody life forms were added as a fixed effect with seven levels: grasses, sedges, rushes, forbs, ferns, cycads, and vines. Moss and samphire life forms were excluded because they were represented by a very small number of individuals. Woody species were simplified from the original 125 species (see Appendix 1) recorded during my study to 12 species based on their average browse (> 5%) and abundance (>10 individuals) (Table 3-4), because there were not enough degrees of freedom to run models with more species. In each model, site was fitted as a random effect because browse was recorded separately for multiple life forms (or species) at each location. For models predicting the proportion of plants browsed I assumed a binomial distribution; for models predicting the average browse intensity I assumed a Gaussian distribution. In each GLMM I specified a range of models with varying combinations of TEC, deer presence, macropod pellet abundance, and/or life form (or species) as fixed effects and ranked them using Akaike's Information Criterion for small samples (AICc) to select the best model. TEC and life form (for non-woody browse) or species (for woody browse) were not included in the same model because the two variables were highly correlated (i.e., some plant life forms or species are more likely to occur in certain TECs).

I also used data collected during my field surveys to test the hypothesis that browsing pressure varies across TECs. Using R Studio (R Core Development Team, 2018) four generalised linear mixed models (GLMMs) were generated. Each model explored the relationship between one of four measures of browse (calculated using the same methods as previous models) and three potential explanatory variables: TEC, deer presence, and macropod pellet abundance. Macropod abundance and deer presence (or absence) was recorded in the same way as the previous analysis. In each model, site was fitted as a random effect because browse was recorded separately for multiple life forms (or species) at each location. In each GLMM I specified a range of models with varying combinations of TEC, deer presence, and/or macropod pellet abundance as fixed effects and ranked them using Akaike's Information Criterion for small samples (AICc) to select the best model. These models aimed to determine relationships between TEC and browse, therefore woody species and non-woody life form were not included as potential explanatory variables due to their high correlation with TEC.

Chapter 4: Results

4.1 Predicting the current distribution of deer

4.1.1 Modelling the probability of occurrence of all species

Through combined data from field surveys (4%), online databases (70%), and expert knowledge (26%), 560 records of deer were gathered (Figure 4-1) and combined with eight biophysical characteristics to predict the probability of occurrence of all deer species across the South Coast using Maxent software (Figure 4-2). The Area Under Curve (AUC) value (0.73) indicated this model had high discriminatory power, and was able to distinguish suitable from unsuitable habitat. AUC values are an index of model performance: 0.5 indicates the model performs no better than random, while larger values indicate higher model performance. The minimum temperature of the coldest quarter and distance to non-woody vegetation made the highest relative contribution (55% and 20% respectively) to this model (Table 4-1). The predicted probability of occurrence of deer increased as topographic wetness index increased, and decreased as distance from non-woody vegetation (Figure 4-3), distance from water (Figure 4-3), and slope increased. The predicted probability of occurrence of all deer species had a varied relationship with precipitation of the driest quarter (Figure 4-4) and minimum temperature of the coldest quarter (Figure 4-4). Conservation areas and urban land uses had the highest probability of occurrence of deer (Figure 4-5). Freshwater wetlands and forested wetlands had the highest probability of occurrence of deer across all vegetation formations (Figure 4-6). The tip of the Greencape Peninsula (near Eden) was not included in this and similar models (see below) because the land use dataset did not include data for this area. Due to the high relative contribution of the land use variable in these models, I chose not to re-run these models without this dataset.

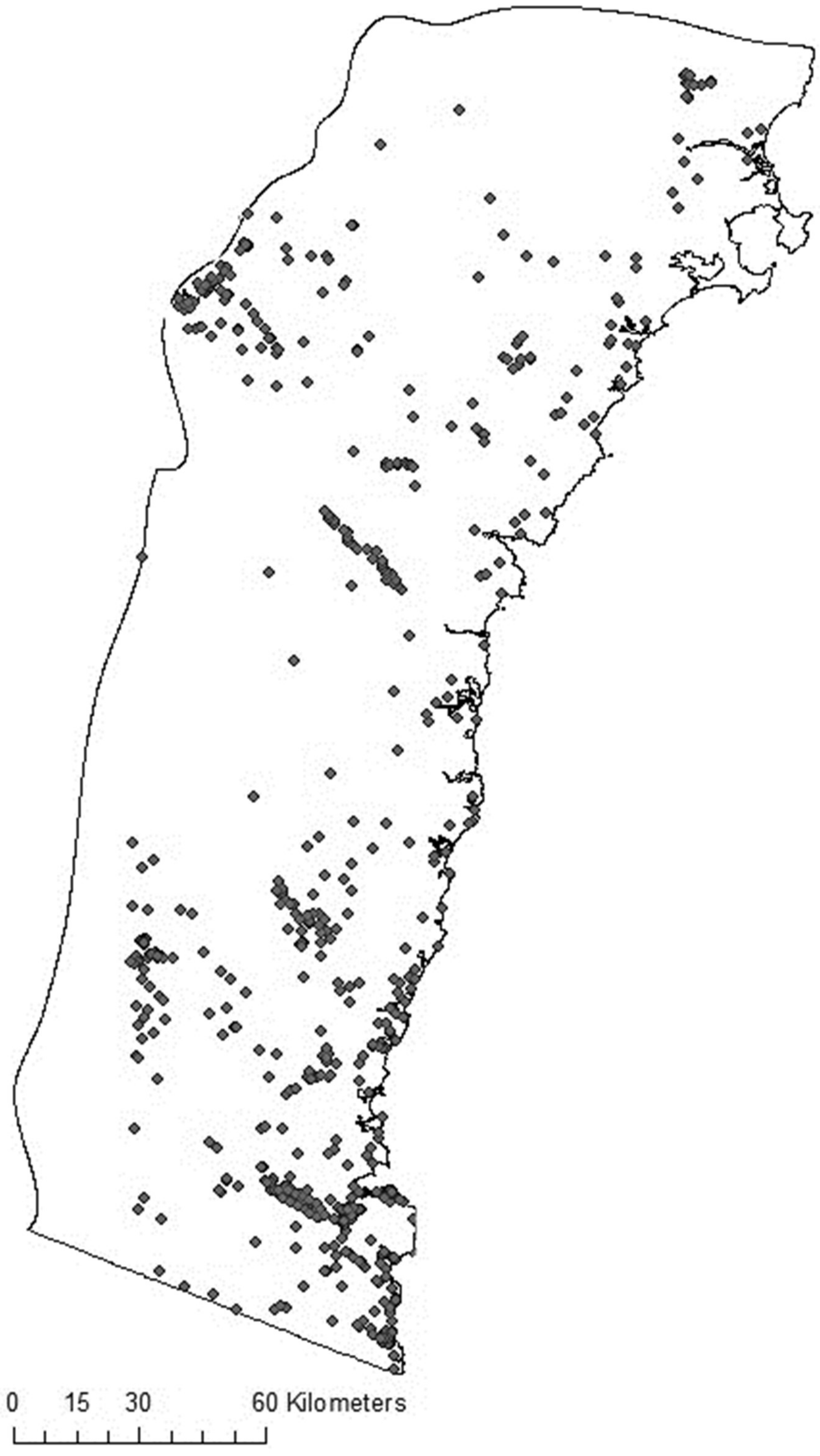

Figure 4-1: The distribution of 560 points used in the Maxent model for all species of deer

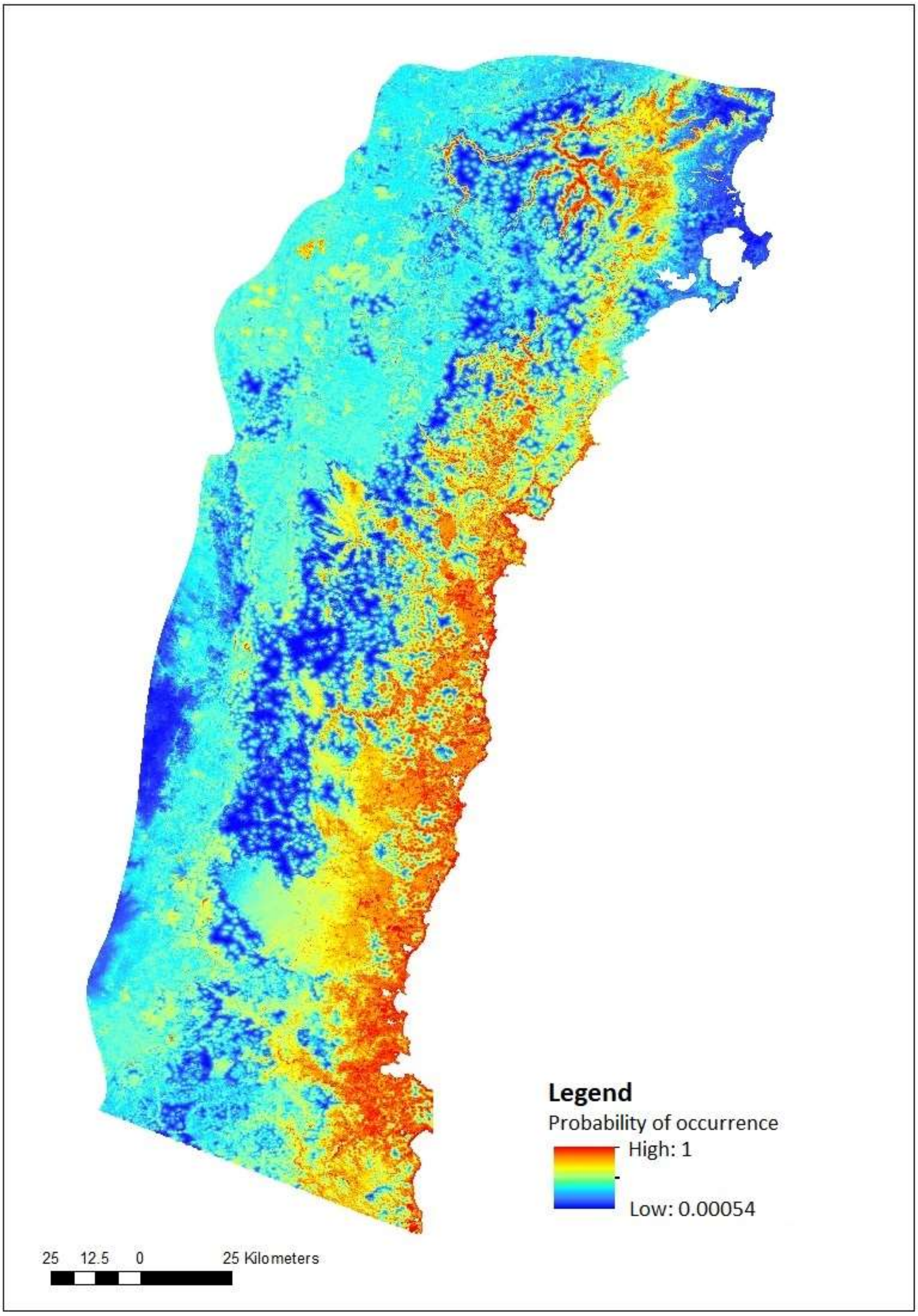

Figure 4-2: The probability of occurrence of all deer species in my study area as predicted by Maxent modelling.

Table 4-1: The relative contribution (%) of eight environmental variables used in Maxent modelling for all species of deer.

Environmental variable	Relative contribution (%)
Minimum temperature of the coldest quarter	54.8
Distance to non-woody vegetation	20.1
Precipitation of the driest quarter	9.3
Vegetation formation	6.9
Land use class	5.9
Slope	1.5
Topographic wetness index	1.1
Distance to water	0.3

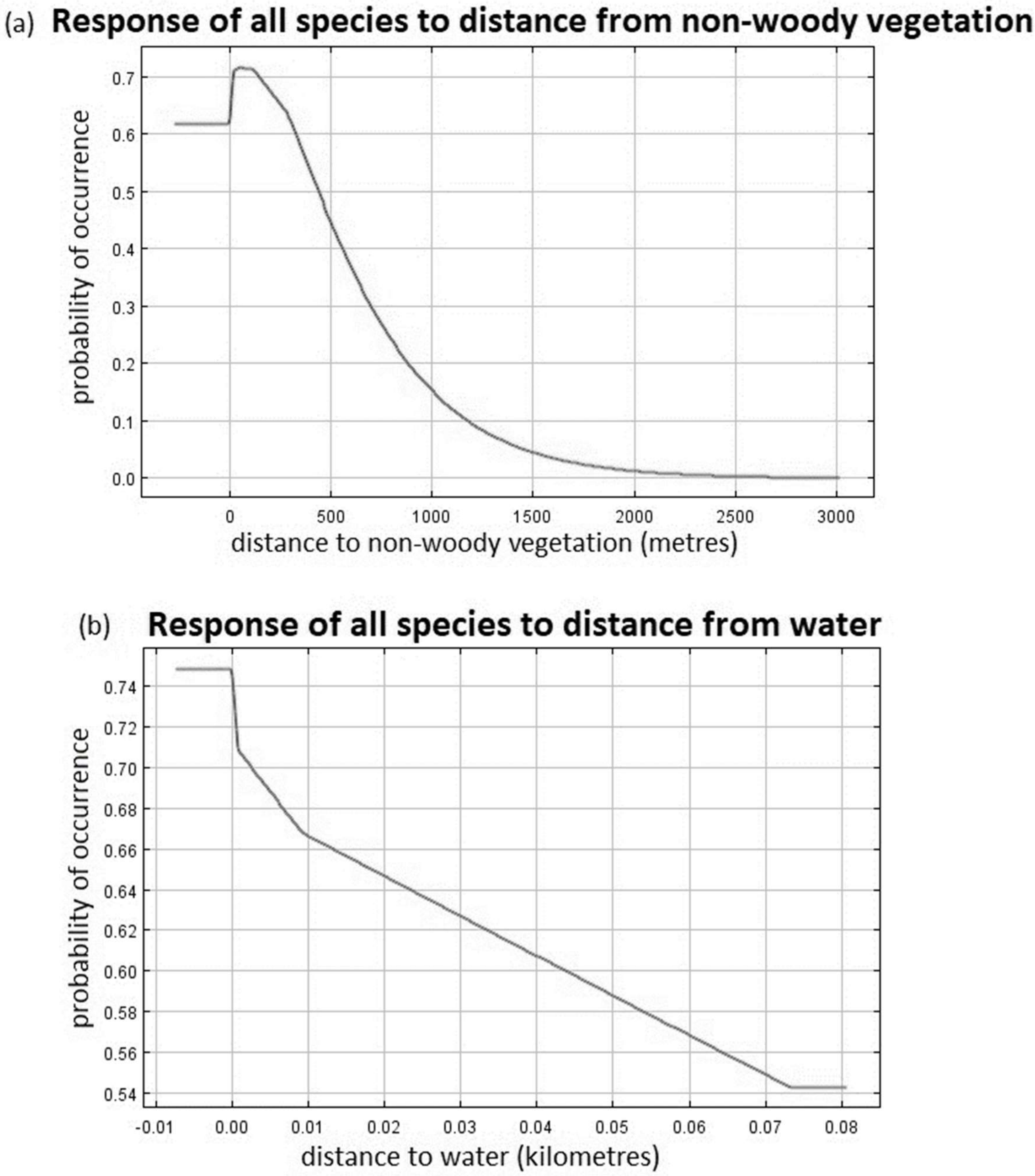

Figure 4-3: The predicted probability of occurrence of all deer species in response to (a) distance from non-woody vegetation (m), and (b) distance from water (km)

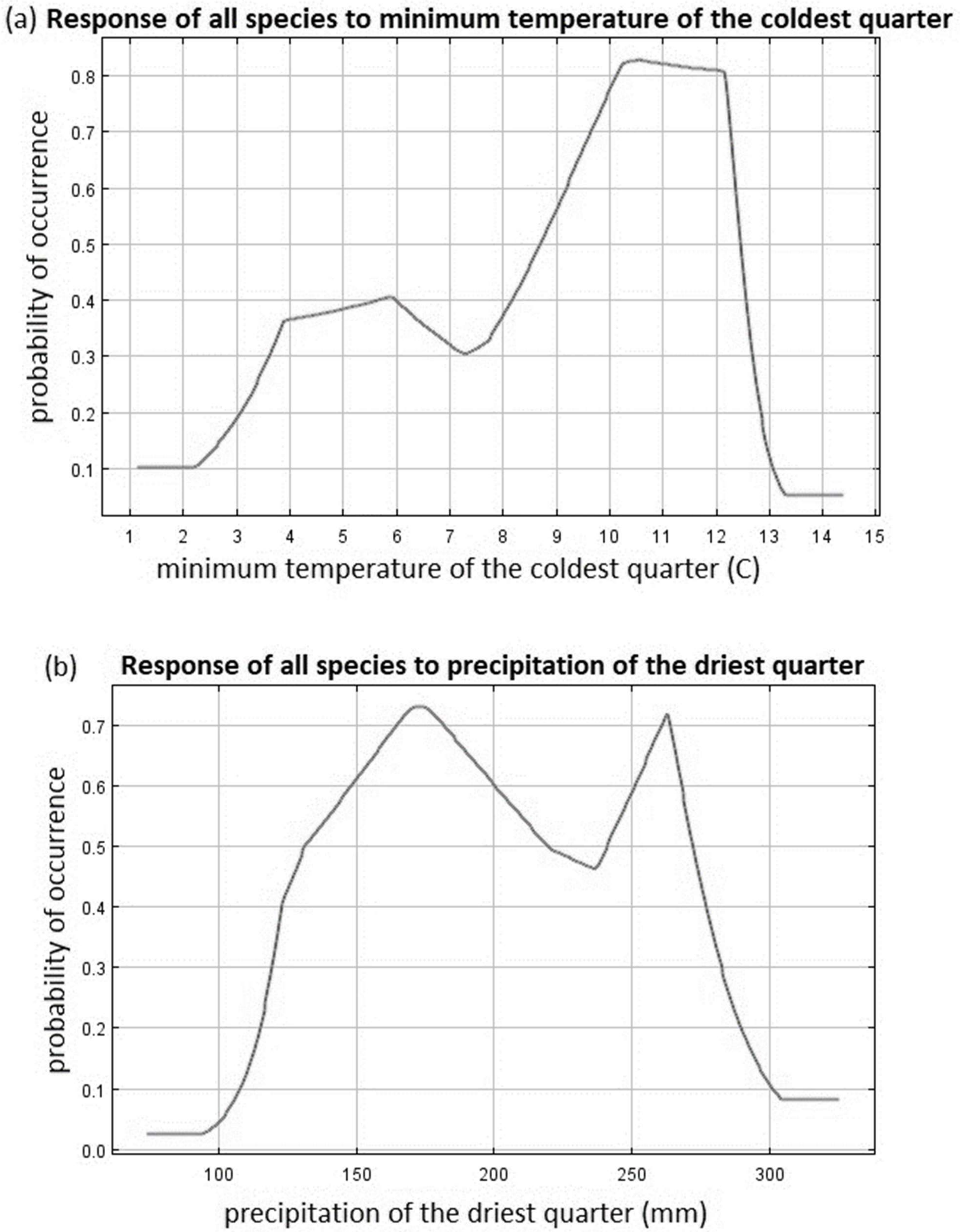

Figure 4-4: The predicted probability of occurrence of all deer species in response to (a) minimum temperature of the coldest quarter (°C), and (b) precipitation of the driest quarter (mm)

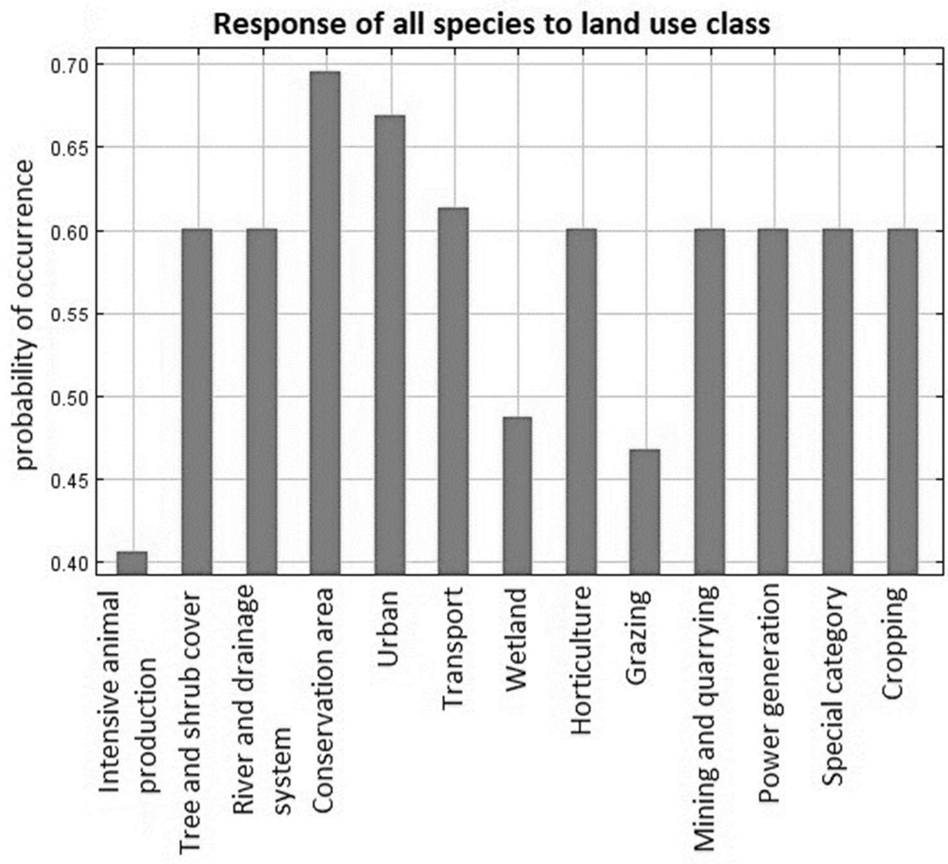

Figure 4-5: The predicted probability of occurrence of all deer species in response to land use class

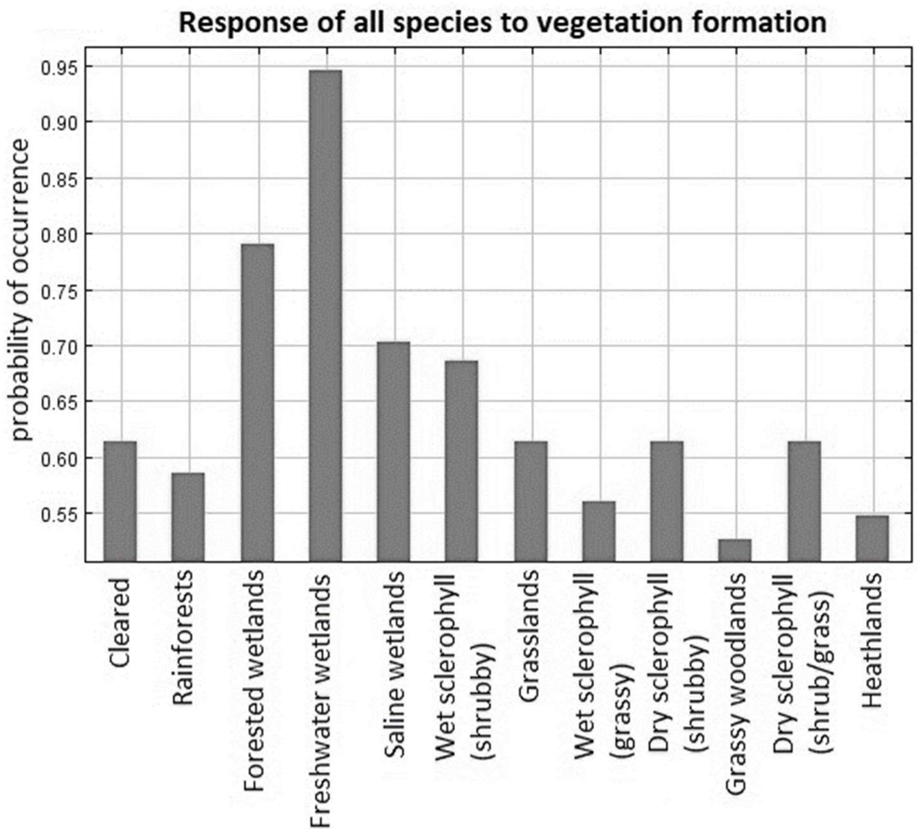

Figure 4-6: The predicted probability of occurrence of all deer species in response to vegetation formation

The output resolution of the all-species Maxent model (41 m x 41 m) made it possible to calculate the probability of occurrence at TEC level. Three thresholds of 0.5, 0.75, and 0.9 were established to determine the proportion of each TEC with moderate, high, and very high probability of occurrence for feral deer (Table 4-2). *Themeda* Grassland, Freshwater Wetland and River-flat Eucalypt have the highest proportion of area with a very high probability of occurrence of feral deer (Table 4-2). Swamp Sclerophyll Floodplain Forest has the lowest proportion of area with a very high probability of occurrence of feral deer (Table 4-2).

Table 4-2: Proportion of each TEC at three thresholds of predicted probability of occurrence.

The proportion of each TEC with a moderate (above 0.5 threshold), high (above 0.75 threshold), and very high (above 0.9 threshold) predicted probability of occurrence of deer (all species). Proportions were calculated using total area of each TEC across the study area, and again for the area of each TEC within NPWS estate. Abbreviations are as follows: Bangalay Sand Forest (BSF), Coastal Saltmarsh (CSM), Freshwater Wetlands (FWL), Littoral Rainforest (LRF), River-flat Eucalypt (RFE), Swamp Oak Floodplain Forest (SOF), Swamp Sclerophyll Forest (SSF), and *Themeda* Grassland (THL).

	Across study area			Within NPWS estate		
TEC	**Moderate (0.5)**	**High (0.75)**	**Very High (0.9)**	**Moderate (0.5)**	**High (0.75)**	**Very High (0.9)**
SSF	0.204	0.133	0.051	0.207	0.098	0.049
CSM	0.527	0.460	0.156	0.341	0.274	0.219
LRF	0.548	0.386	0.185	0.649	0.398	0.195
BSF	0.554	0.363	0.154	0.531	0.292	0.149
SOF	0.636	0.414	0.161	0.589	0.266	0.134
RFE	0.836	0.585	0.267	0.823	0.525	0.291
FWL	0.841	0.686	0.321	0.665	0.568	0.402
THL	1.00	1.00	0.892	1.00	1.00	0.959

4.1.2 Modelling probability of occurrence of fallow deer

Of the 560 combined records, 76 were fallow deer (Appendix 2; Figure 4-7). This subset was used to predict probability of occurrence of the species using Maxent software (Figure 4-8). The AUC value (0.90) indicated this model had high discriminatory power. Precipitation of the driest quarter, minimum temperature of the coldest quarter and vegetation formation had the highest relative contribution (combined 83.1%) (Table 4-3). Probability of occurrence decreased with increasing distance from non-woody vegetation (Figure 4-9), slope, and topographic wetness index. Wet sclerophyll (grassy subformation) had the lowest probability of occurrence of all vegetation formations (Figure 4-10). Urban and grazing land use classes had the lowest probability of occurrence, while all other land use classes were predicted as having relatively equal occurrence of fallow deer (Figure 4-10).

Figure 4-7: A fallow deer captured by a camera trap near Yadboro Flats campground (photo: Heather Burns)

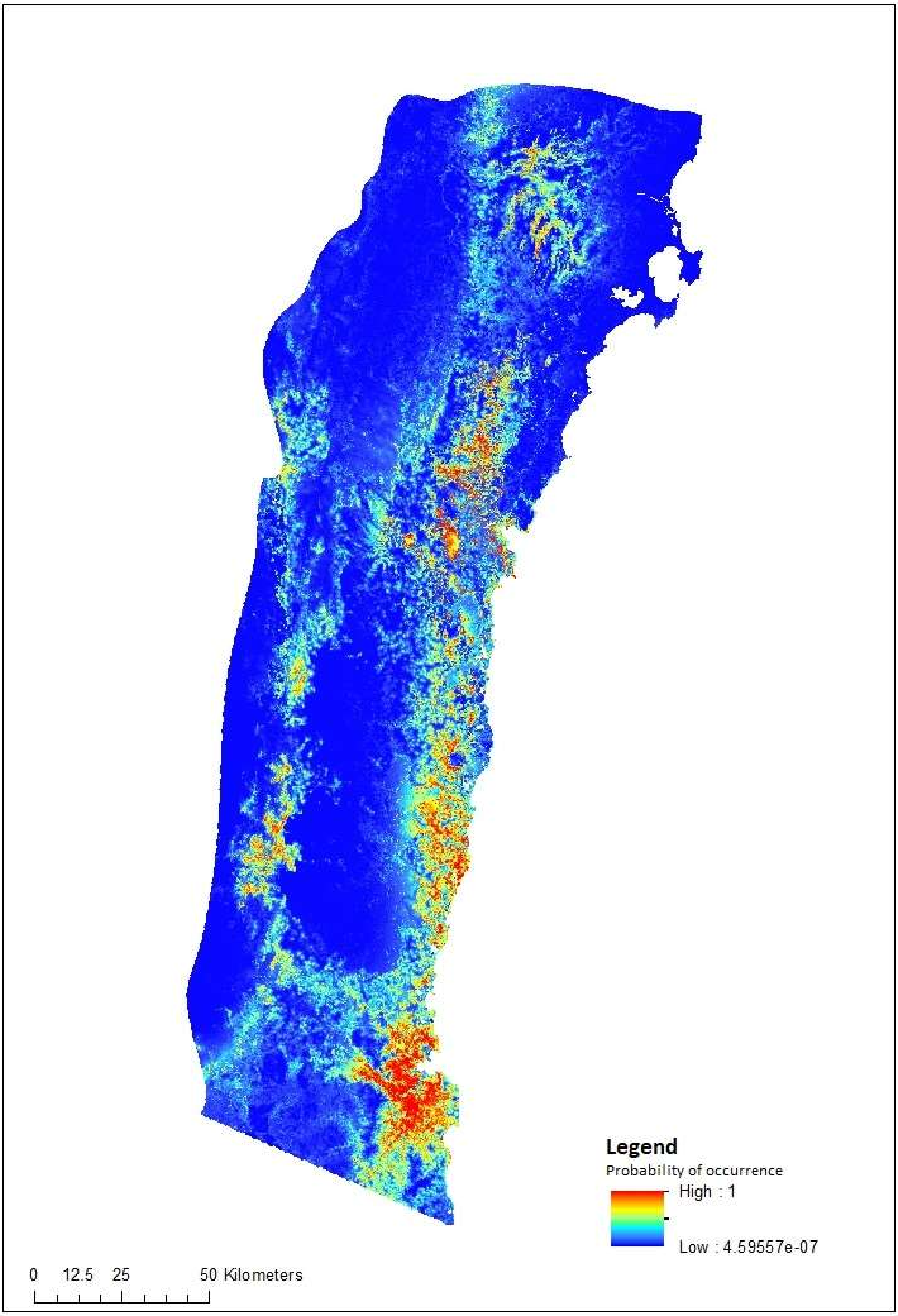

Figure 4-8: Probability of occurrence of fallow deer in my study area as predicted by a species-specific Maxent model.

Table 4-3: Relative contribution (%) of eight environmental variables used in the Maxent model to predict probability of occurrence of fallow deer.

Environmental variable	Relative contribution (%)
Precipitation of the driest quarter	39.5
Minimum temperature of the coldest quarter	27.7
Vegetation formation	15.9
Distance to non-woody vegetation	10.8
Land use class	5.2
Distance to water	0.5
Slope	0.3
Topographic wetness index	0.2

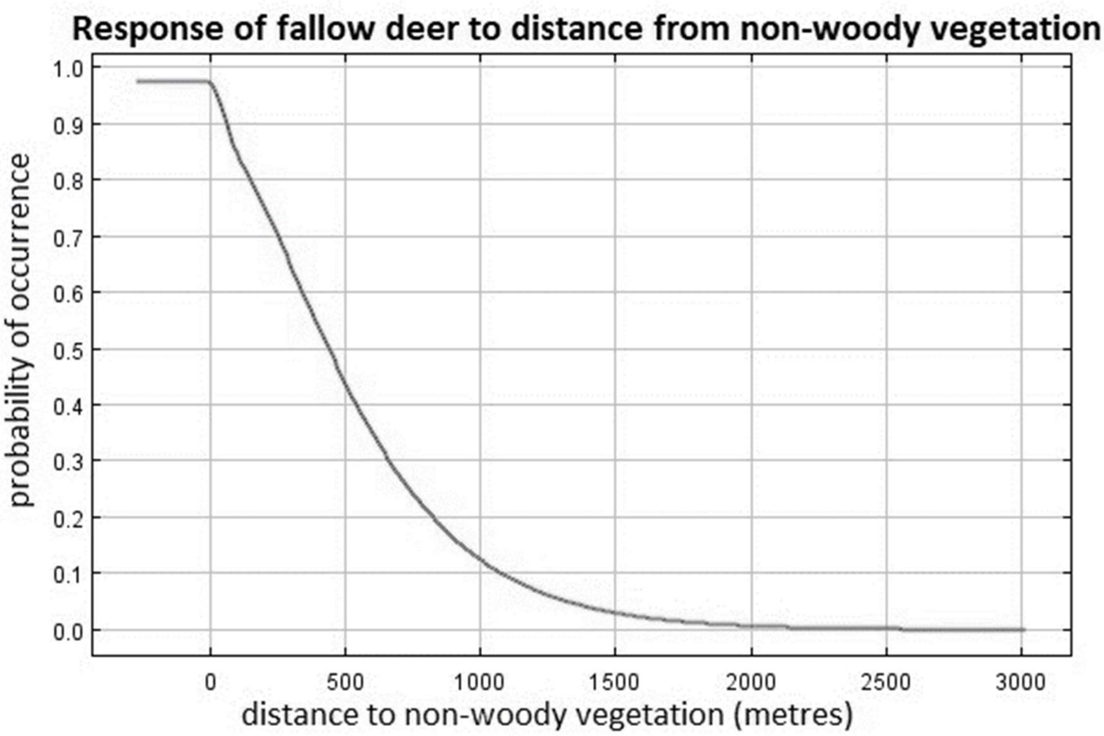

Figure 4-9: The predicted probability of occurrence of fallow deer in response to distance from non-woody vegetation (m)

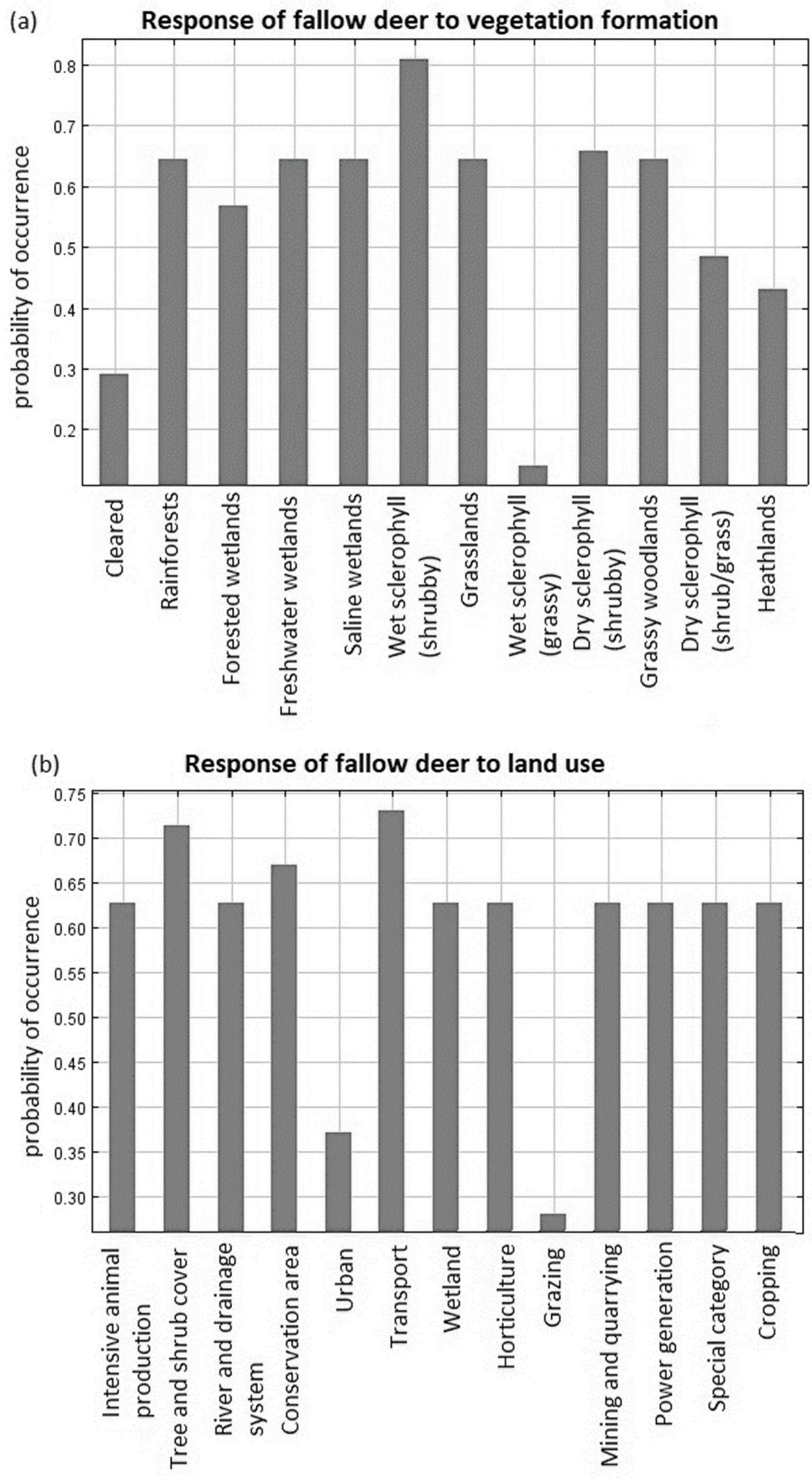

Figure 4-10: The predicted probability of occurrence of fallow deer in response to (a) vegetation formation, and (b) land use

4.1.3 Modelling the probability of occurrence of sambar deer

Of the 560 combined presence points, 60 were sambar deer (Appendix 3). These were used to predict the probability of occurrence of the species using Maxent software (Figure 4-11). The AUC value (0.93) indicated this model had high discriminatory power. Minimum temperature of the coldest quarter, land use class, and precipitation of the driest quarter had the highest relative contribution (combined 74.2%) (Table 4-4). The predicted probability of occurrence increased with increasing distance to water (Figure 4-12) and topographic wetness index, and decreased with increasing distance from non-woody vegetation and slope. Forested wetland and freshwater wetland vegetation formations had the highest predicted probability of occurrence, while cleared land and wet sclerophyll (grassy subformation) had the lowest predicted probability of occurrence (Figure 4-13). Conservation reserves and river and drainage system land use classes had the highest predicted probability of occurrence, while all other land use classes were predicted as having low levels of occurrence of sambar (Figure 4-13).

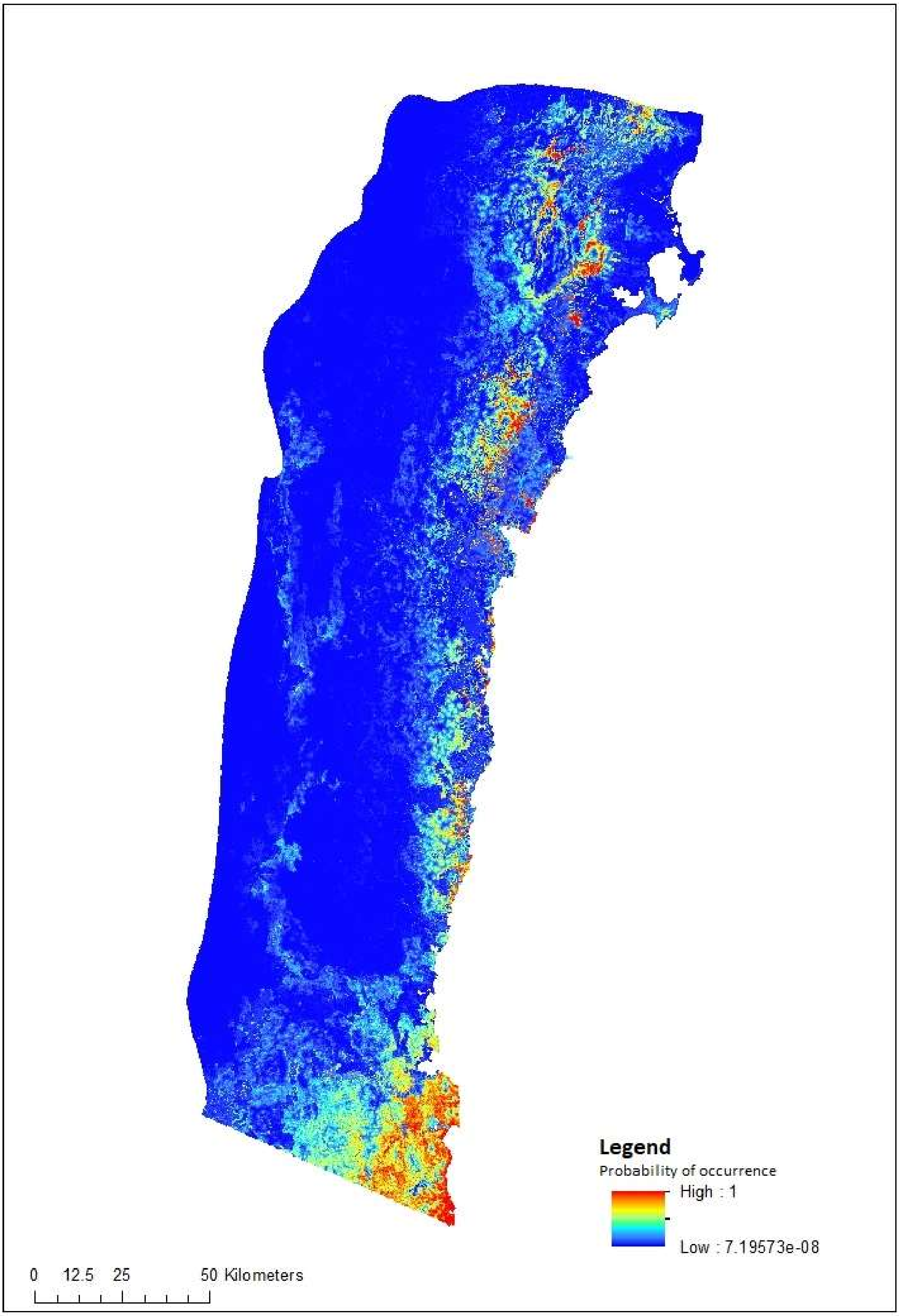

Figure 4-11: Probability of occurrence of sambar deer in my study area as predicted by a species-specific Maxent model.

Table 4-4: The relative contribution (%) of eight environmental variables used in the Maxent model to predict the probability of occurrence for sambar deer

Environmental variable	Relative contribution (%)
Minimum temperature of the coldest quarter	33.8
Land use class	26.8
Precipitation driest quarter	13.6
Vegetation formation	11.2
Distance to non-woody vegetation	8.2
Slope	5.3
Topographic wetness index	1
Distance to water	0.1

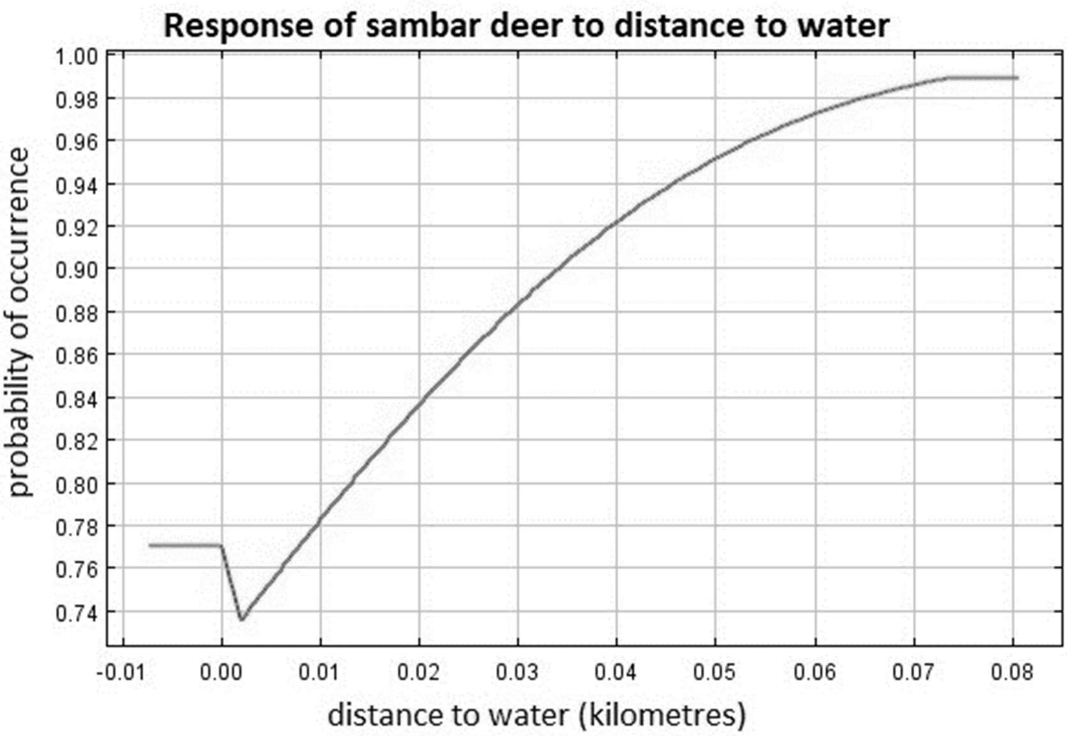

Figure 4-12: The predicted probability of occurrence of sambar deer in response to distance to water (km)

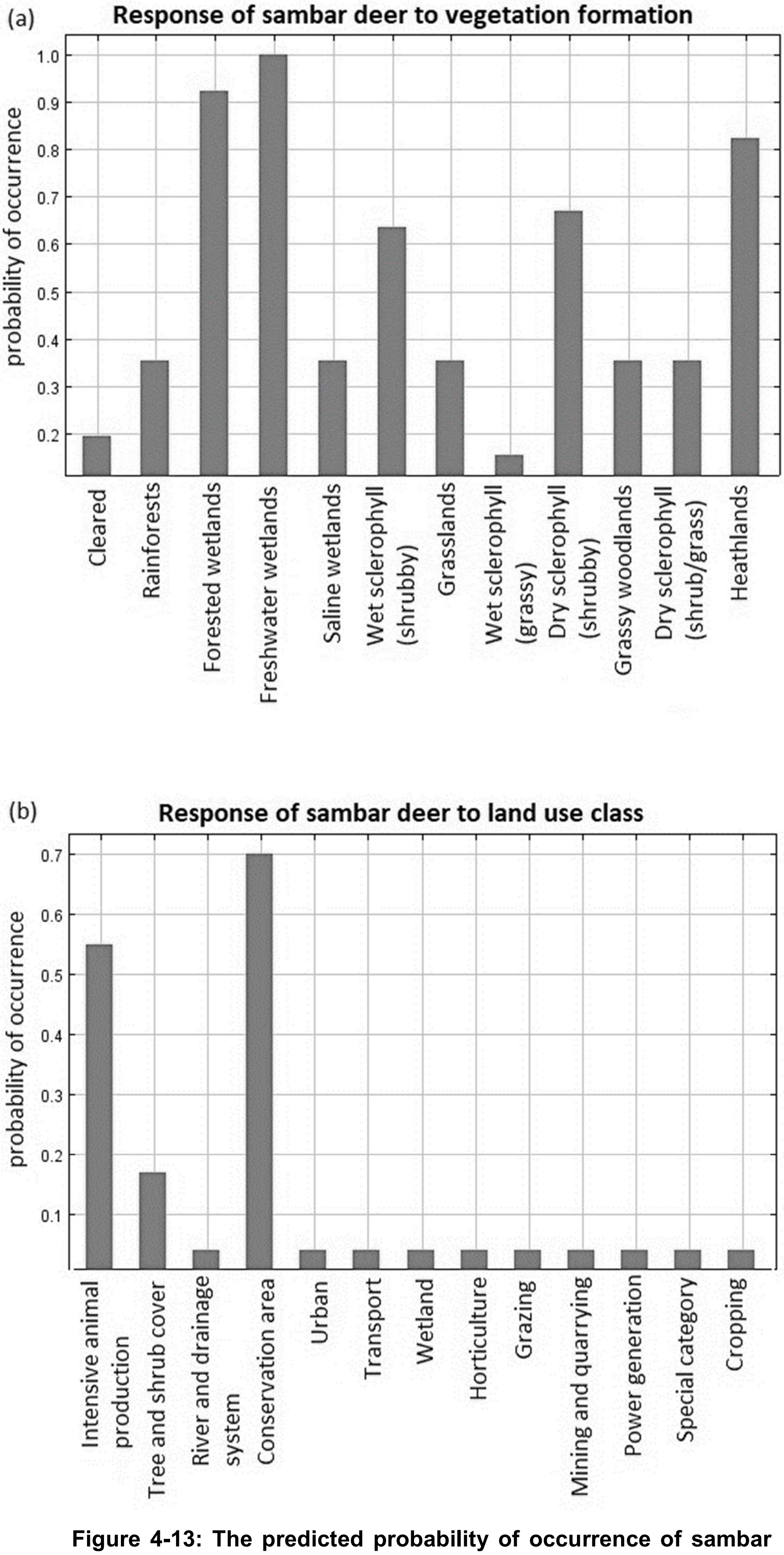

Figure 4-13: The predicted probability of occurrence of sambar deer in response to (a) vegetation formation, and (b) land use class

4.2 Impacts of browsing by deer

A total of 89 sites across eight TECs were surveyed across my study area (Table 4-5). Browse damage to both woody and non-woody vegetation was assessed from 356 transects (89 sites x 4 transects per site). Bangalay Sand Forest was the most widely distributed TEC, and so was sampled at a greater frequency (Table 4-5). *Themeda* Grasslands were only found in small patches of Eurobodalla National Park and therefore were sampled only twice (Table 4-5). Signs of deer were recorded at 19 of the 89 (21.3%) sites (Table 4-5, Table 4-6). Macropod scat was present at 86 of 89 sites, and was used as an index for macropod abundance. Macropod scat ranged from 0-181 pellet groups per site (mean = 19.8 pellet groups).

Table 4-5: Distribution of sites across eight TECs.
Area (km²) represents the total predicted area of each TEC within NPWS estate. The abbreviations used for each TEC in the statistical analysis are shown in parentheses.

Threatened Ecological Community	Area (km²)	Sites	Sites with deer present
Bangalay Sand Forest (BSF)	36.46	28	1
Coastal Saltmarsh (CSM)	7.19	13	3
River Flat Eucalypt (RFE)	3.57	11	7
Swamp Sclerophyll Floodplain Forest (SSF)	2.31	11	0
Freshwater Wetland (FWL)	31.73	9	6
Swamp Oak Floodplain Forest (SOF)	19.38	9	0
Littoral Rainforest (LRF)	11.16	7	1
Themeda Grassland (THL)	0.25	2	0

Table 4-6: Types of sign used to identify deer presence, and their respective counts.

Sign Type	Number of sites
Pellets	12
Tracks	6
Rubbed Trees	7
Wallow	1

4.2.1 Impacts of deer on non-woody vegetation

Nine non-woody life forms were identified in my browse survey (Table 4-7). Seven of those (78%) experienced some level of browsing. The impact of deer on non-woody vegetation was assessed using two measures of browsing pressure: average browse intensity of seven non-woody life forms and the proportion of non-woody individuals browsed. Nine generalised linear mixed models (GLMMs) were tested to predict the average browse intensity of non-woody life forms (Table 4-8) and seven GLMMs were tested to predict the proportion of non-woody individuals browsed (Table 4-9). GLMM7 was selected as the best model for predicting average browse intensity of non-woody life forms (AICc= 565.6, weight=0.774). This model contained 'deer presence', 'non-woody life form', and an interaction between deer presence and non-woody life form as fixed effects (Table 4-8). Where deer were present, average browse intensity was greater for grass, rush, sedge, and cycad life forms (Figure 4-14). GLMM7 was selected as the best model for predicting the proportion of non-woody individuals browsed (AICc=1759.2, weight=0.877). The model contained 'macropod pellet', 'deer presence' and 'non-woody life form' as fixed effects (Table 4-9). The proportion of non-woody plants browsed was higher where: deer were present, for certain life forms (cycads, ferns, and sedges), and in areas with a higher abundance of macropod pellets (Figure 4-15). TEC was not included as a fixed effect in the best model of average browse intensity nor proportion of individuals browsed because of correlations between the occurrence of some non-woody life forms and certain TECs.

Table 4-7: Average proportion of browsed individuals by life form. Proportion of browsed individuals for each life form was calculated first for each site, and then averaged across sites.

Life Form	Average of proportion of browsed individuals	Count
Moss	0	11
Samphire	0	38
Forb	0.02	152
Vine	0.04	66
Sedge	0.11	92
Rush	0.12	130
Grass	0.19	281
Fern	0.24	140
Cycad	0.24	29

Table 4-8: Generalised linear mixed models for predicting average browse intensity of non-woody life forms ranked by AICc values.

GLMM7, containing deer presence, non-woody life form, and the interaction between deer presence and non-woody life form as independent variables, has a 77% chance of being the best model.

Model	Random Effect	Fixed Effects	AICc	Delta	Weight
GLMM 7	Site number	Deer presence + Non-woody life form + Deer presence: Non-woody life form interaction	565.6	0.00	0.774
GLMM5	Site number	Deer presence + Non-woody life form	569.8	4.17	0.096
GLMM9	Site number	Deer presence + TEC + Deer presence: TEC interaction	570.7	5.07	0.061
GLMM8	Site number	Deer presence + Macropod pellet + Non-woody life form + Deer presence: Non-woody life form interaction	572.3	6.64	0.028
GLMM3	Site number	Deer presence + TEC	572.6	7.02	0.023
GLMM2	Site number	Deer presence	574.1	8.46	0.011
GLMM6	Site number	Deer presence + Macropod pellet + Non-woody life form	575.6	9.97	0.005
GLMM10	Site number	Deer presence + TEC + Macropod pellet + Deer presence:TEC interaction	580.3	14.69	0.000
GLMM1	Site number		580.8	15.18	0.000
GLMM4	Site number	Deer presence + TEC + Macropod pellet	582.1	16.47	0.000
GLMM11	Site number	Macropod pellet	587.5	21.88	0.000

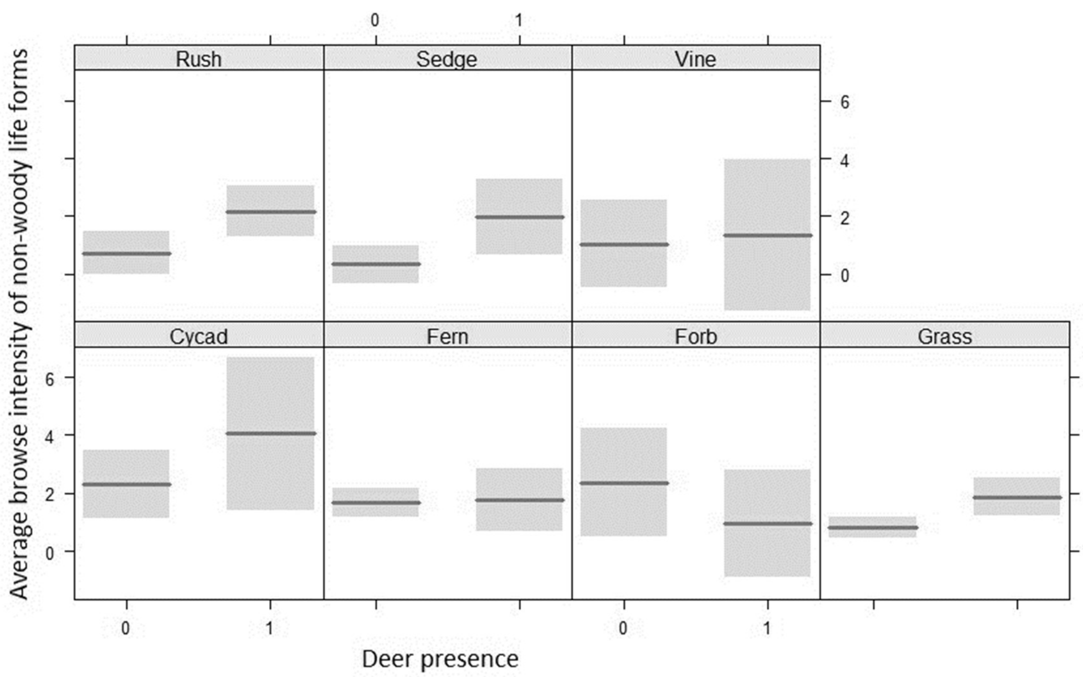

Figure 4-14: The predicted average browse intensity by non-woody life form where deer are present (1) or absent (0) (± 95% confidence interval), based on data from GLMM7.

Table 4-9: Generalised linear mixed models for predicting the proportion of non-woody individuals browsed ranked by AICc values.

GLMM7, containing macropod pellet, deer presence, and non-woody life form as independent variables, has an 88% chance of being the best model.

Model	Random Effect	Fixed Effects	AICc	Delta	Weight
GLMM 7	Site number	Macropod pellet + Deer presence + Non-woody life form	1759.2	0.00	0.877
GLMM6	Site number	Deer presence + Non-woody life form	1763.1	3.94	0.123
GLMM2	Site number	Macropod pellet	2034.6	275.37	0.000
GLMM3	Site number	Deer presence	2035.0	275.79	0.000
GLMM1	Site number		2035.9	276.68	0.000
GLMM5	Site number	Macropod pellet + Deer presence + TEC	2037.8	278.59	0.000
GLMM4	Site number	Deer presence + TEC	2039.5	280.27	0.000

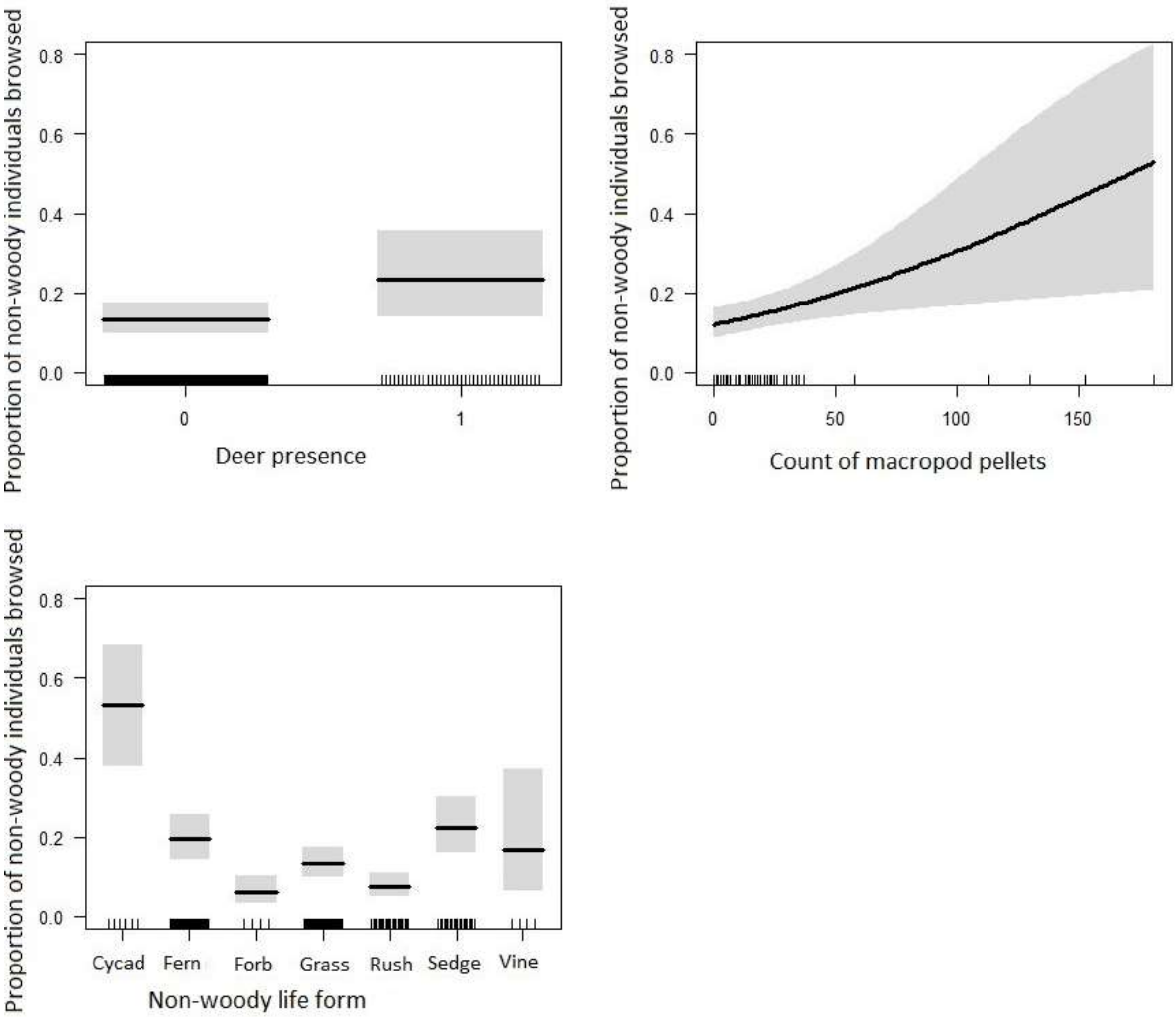

Figure 4-15: The predicted proportion of non-woody individuals browsed where deer are present (1) vs. absent (0), as a function of the count of macropod pellets, and by life form (± 95% confidence interval), based on data from GLMM7.

Marks along the x-axis represent the number of sites sampled for different levels of each variable.

4.2.2 Impacts of deer on woody vegetation

Of the 121 woody species identified in the browse survey, 74 species (61.1 %) experienced some level of browsing. Of this subset, 31 had an average browse intensity at or above 5% (Table 4-10). Acacia dealbata (silver wattle), Acacia implexa (hickory wattle), Acacia longifolia (Sydney golden wattle), Acronychia oblongifolia (white aspen), Babingtonia virgata (heath myrtle), Banksia spinulosa (hairpin banksia), Bursaria spinosa (sweet bursaria), Callistemon citrinus (crimson bottlebrush), Casuarina cunninghamiana (river she-oak), Dodonaea triquetra (common hop bush), Leucopogon parviflorus (coast beard-heath), and Phylanthus gunnii (scrubby spurge) have both high average browse intensity (> 5%) and abundance (n > 10), and analysis of browsing pressure on woody vegetation was limited to these 12 species. The impact of deer on woody vegetation was assessed using two measures of browse: average browse intensity by species and proportion of woody individuals browsed. Seven GLMMs were tested to predict the average browse intensity of woody species (Table 4-11) and seven GLMMs were tested to predict the proportion of woody individuals browsed (Table 4-12). GLMM5 was selected as the best model for predicting average browse intensity of woody species (AICc=320.2, weight=0.76). This model contained 'deer presence' and 'woody species' as fixed effects (Table 4-11). Average browse intensity for woody species was higher where deer are absent and was higher for some species (Figure 4-16). GLMM7 was selected as the best model for predicting the proportion of woody individuals browsed (AICc= 205.7, weight=0.935). The model contained 'woody species', 'macropod pellet', and 'deer presence' as fixed effects (Table 4-12). The best-fit model indicated that the proportion of woody individuals browsed is higher for some species, increases with increasing macropod pellet abundance, and is marginally higher where deer are present (Figure 4-17). TEC was not included as a fixed effect in the best model of average browse intensity or proportion of individuals browsed because of correlations between the occurrence of some woody species and TECs.

Table 4-10: Woody species with an average browse intensity > 5%
Species with both high abundance (count n >10) and average browse intensity >5% are listed in bold.

Species Name	Average of Browse Intensity	Count
Acacia melanoxylon	5	1
Acacia terminalis	5	1
Ficus rubiginosa	5	1
Marsdenia rostrata	5	4
Bursaria spinosa	**5.20**	**74**
Acronychia oblongifolia	**5.54**	**37**
Casuarina paludosa	5.71	7
Acacia dealbata	**5.93**	**27**
Acacia implexa	**6.67**	**27**
Persoonia linearis	7.14	7
Casuarina cunninghamiana	**7.35**	**83**
Dodonaea triquetra	**7.56**	**78**
Acacia longifolia	**7.61**	**71**
Babingtonia virgata	**8**	**10**
Angophora costata	10	2
Callistemon citrinus	**11.18**	**17**
Indigofera australis	11.25	4
Myoporum boniense	11.25	4
Helichrysum diosmifolium	12.86	7
Cissus antarctica	13.33	6
Lantana camara	13.33	3
Banksia spinulosa	**15.42**	**12**
Phyllanthus gunnii	**16.92**	**13**
Breynia oblongifolia	20	2
Hibbertia aspera	22.5	2
Backhousia myrtifolia	26.67	3
Phytolacca octandra	26.67	3
Leucopogon parviflorus	**35.45**	**44**
Pomaderris lanigera	40	3
Acacia silvestris	50	2
Clematis glycinoides	70	1
Sigesbeckia orientalis	95	1

Table 4-11: Generalised linear mixed models for predicting average browse intensity of woody species ranked by AICc values.

GLMM5, containing deer presence and woody species as the independent variables, has a 76% chance of being the best model.

Model	Random Effect	Fixed Effects	AICc	Delta	Weight
GLMM 5	Site number	Deer presence + Woody species	320.2	0.00	0.760
GLMM6	Site number	Deer presence + Woody species + Macropod pellet	322.5	2.32	0.239
GLMM3	Site number	Deer Presence + TEC	333.1	12.92	0.001
GLMM4	Site number	Deer Presence + TEC + Macropod pellet	336.7	16.49	0.000
GLMM2	Site number	Deer Presence	366.0	45.79	0.000
GLMM7	Site number	Macropod pellet	369.8	49.54	0.000
GLMM1	Site number		371.6	51.39	0.000

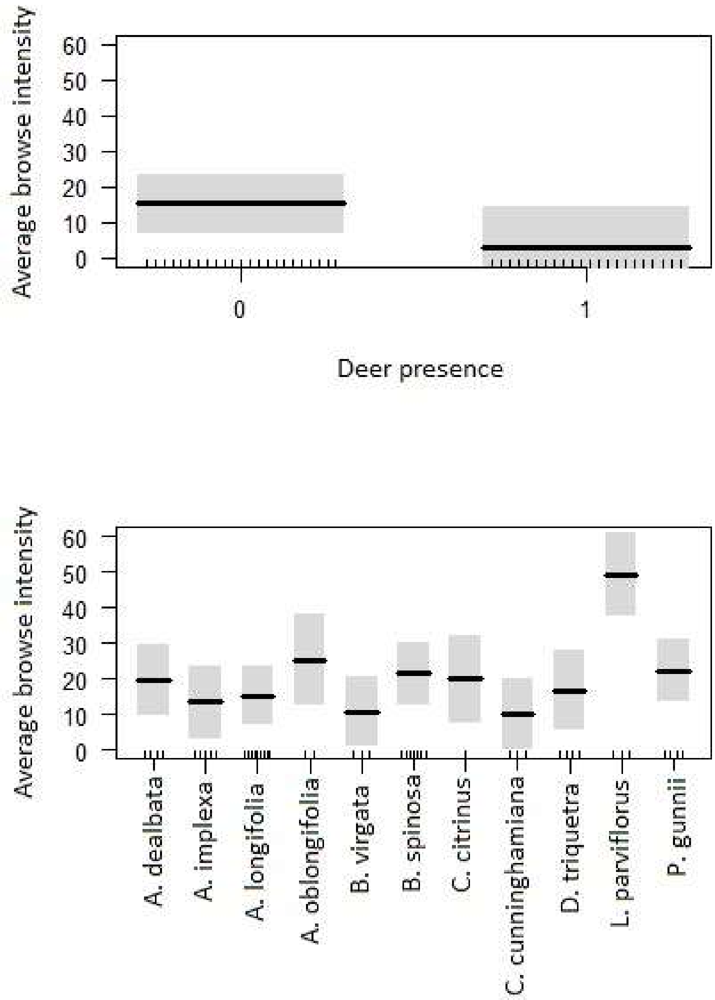

Figure 4-16: The predicted average browse intensity where deer are present (1) or absent (0), and by species (± 95% confidence interval), based on data from GLMM5.

Marks along the x-axis represent the number of sites sampled for different levels of each variable.

Table 4-12: Generalised linear mixed models for predicting proportion of woody individuals browsed by AICc values.

GLMM7, containing macropod pellets, deer presence, and woody species as independent variables, has a 94% chance of being the best model.

Model	Random Effect	Fixed Effects	AICc	Delta	Weight
GLMM 7	Site number	Macropod pellet + Deer presence + Woody species	205.7	0.00	0.935
GLMM6	Site number	Deer presence + Woody species	211.0	5.34	0.065
GLMM5	Site number	Macropod pellet + Deer presence + TEC	245.1	39.40	0.000
GLMM4	Site number	Deer presence + TEC	246.0	40.35	0.000
GLMM2	Site number	Macropod pellet	251.9	46.25	0.000
GLMM1	Site number		255.1	49.39	0.000
GLMM3	Site number	Deer presence	256.7	51.01	0.000

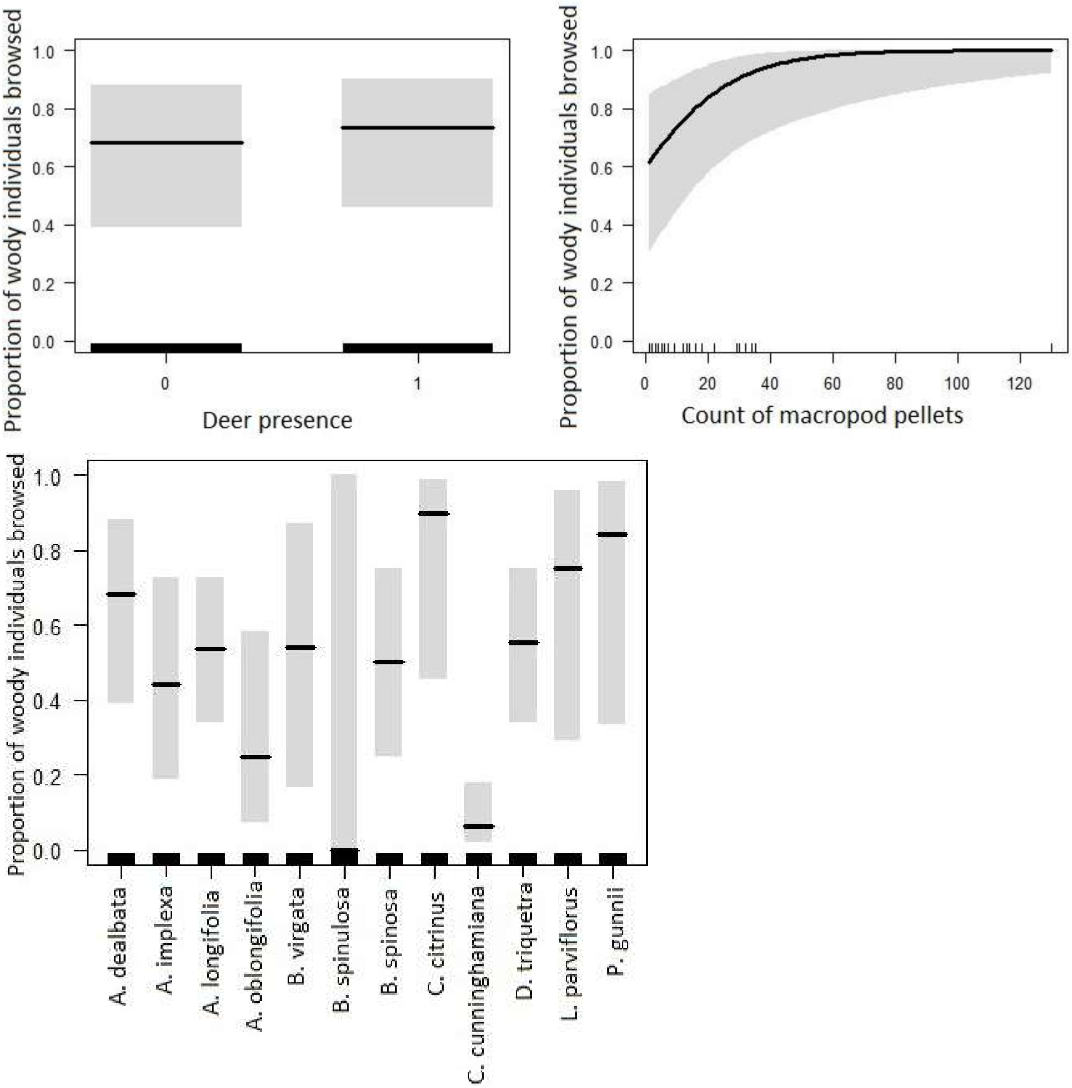

Figure 4-17: The predicted proportion of woody individuals browsed where deer are present (1) and absent (0), as a function of macropod pellet abundance, and by species (± 95% confidence interval), based on data from GLMM7.

Marks along the x-axis represent the number of sites sampled for different levels of each variable.

4.2.3 Impacts of deer on threatened ecological communities

Eight TECs were sampled across 89 sites over the study area. TECs had varying geographic extents and were sampled proportionately (Table 4-5). Four individual analyses were conducted using GLMMs to predict the average browse intensity and proportion of individuals browsed for both woody species and non-woody life forms across the different TECs. In contrast to the models in sections 4.2.1 and 4.2.2, these models did not include 'woody species' or 'non-woody life form' as potential fixed effects because of correlation between some life forms and species and certain TECs. Seven GLMMs were tested to predict the average browse intensity of non-woody life forms (Table 4-13). GLMM5 was selected as the best model for predicting the average browse intensity of non-woody life forms (AICc=570.7, weight=0.633). This model contained 'deer presence', 'TEC', and an interaction between deer presence and TEC as fixed effects (Table 4-13). GLMM5 contained an interaction between deer presence and TEC, therefore only TECs with recorded deer presences were plotted with data from this model (Figure 4-18). Where deer were present, average browse intensity for non-woody life forms increased for Coastal Saltmarsh, Freshwater Wetland, and Littoral Rainforest TECs (Figure 4-18). *Themeda* Grassland was sampled only twice, and therefore the results have very large confidence intervals and should be interpreted with caution.

Table 4-13: Generalised linear mixed models for predicting average browse intensity of non-woody life forms by AICc values.
GLMM5, containing deer presence, TEC, and a deer presence:TEC interaction as independent variables, has a 63% chance of being the best model.

Model	Random Effect	Fixed Effects	AICc	Delta	Weight
GLMM 5	Site number	Deer presence + TEC + Deer presence:TEC interaction	570.7	0.00	0.633
GLMM3	Site number	Deer presence + TEC	572.6	1.95	0.239
GLMM2	Site number	Deer presence	574.1	3.39	0.116
GLMM6	Site number	Deer presence + TEC + Macropod pellet + Deer presence:TEC interaction	580.3	9.62	0.005
GLMM1	Site number		580.8	10.11	0.004
GLMM4	Site number	Deer presence + TEC + Macropod pellets	582.1	11.40	0.002
GLMM7	Site number	Macropod pellets	587.5	16.81	0.000

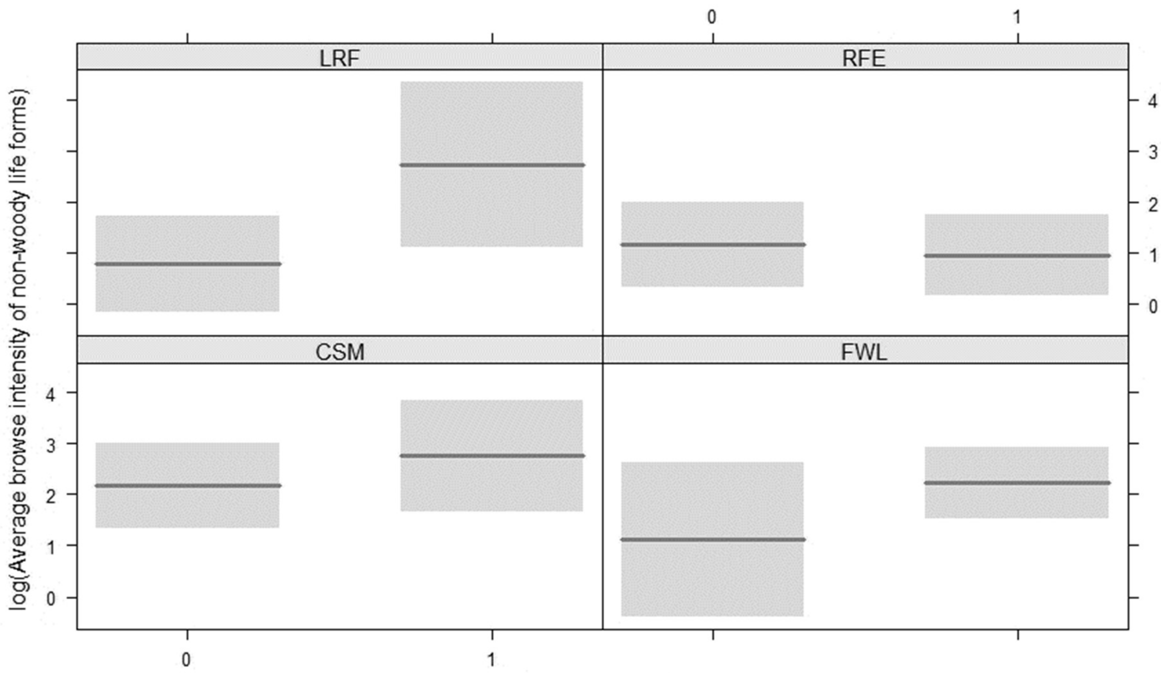

Figure 4-18: The predicted average browse intensity of non-woody life forms, by TEC, where deer are present (1) or absent (0) (±95% confidence interval), based on data from GLMM5.

Chapter 5: Discussion

In this thesis I aimed to: (1) predict the current distribution of feral deer across the South Coast of NSW, and; (2) establish the extent to which browsing by deer is a current and likely future threat to the conservation of threatened ecological communities within the region. Using presence-only records and a selection of environmental variables, I created three Maxent models that predicted the probability of occurrence of deer across the study area. These models comprised one prediction for all species of deer, and two species-specific predictions for fallow and sambar deer, respectively. Two bioclimatic variables, minimum temperature of the coldest quarter and precipitation of the driest quarter, had high contributions to all three models. Distance to non-woody vegetation had a uniformly negative relationship with probability of occurrence across all three models. Using predictions from this model, TECs at greatest threat were *Themeda* Grassland, Freshwater Wetland, and River-flat Eucalypt. The results corresponding to my second aim indicate that sites where deer were present had a higher average intensity and proportion of non-woody life forms browsed, and a higher proportion of woody species browsed. Where deer were present, there was increased browsing pressure on rushes, cycads, sedges, and grass life forms. In contrast, the average browse intensity of woody species was not affected by presence of deer. My results also indicate that, within TECs, the average browse intensity of non-woody life forms browsed is higher where deer are present, while all other measures of browse currently have no measurable effect from deer presence. Specifically, the average browse intensity of non-woody plants was higher in Coastal Saltmarsh, Freshwater Wetland, and Littoral Rainforest TECs where deer were present.

5.1 Estimating the current distribution of feral deer

5.1.1 Explanatory variables affecting probability of occurrence in all models

The results from Maxent modelling indicate that bioclimatic variables such as temperature and precipitation appear to be important variables for predicting the probability of occurrence of deer across the study region (Table 4-1, Table 4-3, Table 4-4). The importance of these variables was likely increased by my decision to use extreme representations of each: minimum temperature of the coldest quarter and precipitation of the driest quarter. This creates maximum contrast from north to south and east to west. Both bioclimatic variables produced similar relationships with predicted probability of occurrence across all three models. The upward trend in probability of occurrence with increasing minimum temperature indicates that deer are more likely to occur in warmer areas of my study area such as the coast (Figure 4-4). These results may reflect characteristics of the areas sampled in my study rather than the true climatic ranges of feral deer, many species of which are known to occur in cooler areas such as the Australian Alps (Claridge, 2016b). The predicted probability of occurrence is relatively stable within a range of values for precipitation of the driest quarter, indicating that deer are suited to all but extremely dry and extremely wet areas (Figure 4-4). These patterns are predicated on the records of deer presence I assembled being a representative sample of the true (current) distribution of these mammals.

Across all three models, probability of occurrence increased as distance to non-woody vegetation decreased (Figure 4-3, Figure 4-9). This is unsurprising, as multiple species of deer are highly suited to habitat that contains open grassland or grassy clearings (Claridge, 2016a). Specifically, red, rusa, and fallow deer rely on the interface between woodlands and grassy areas for shelter and food (Claridge, 2016a; Apollonio *et al.*, 1998; Godvik *et al.*, 2009). However, the relative contribution of distance to non-woody vegetation varied across models. In the pooled species model, distance to non-woody vegetation contributed twice as much to predicting probability of occurrence compared to the species-specific models. In both species-specific models, the relative contribution from distance to non-woody vegetation decreased, while relative contribution from vegetation formation increased (Table 4-3, Table 4-4). The decreased importance of distance to non-woody vegetation for fallow deer is surprising, considering the importance of grassy clearings for grazing for this species (Claridge, 2016a; Thirgood, 1995; Apollonio *et al.*, 1998).

In the pooled species model, forested wetlands and freshwater wetlands had probabilities of occurrence marginally higher than all other vegetation classes (Figure 4-6). The higher probability of occurrence in wetland ecosystems may be because they combine desirable habitat characteristics, such as the presence of water and non-woody vegetation. This is also supported by the species-specific model for sambar deer, where freshwater wetlands and forested wetlands had the highest overall predicted probabilities of occurrence (Figure 4-13). This increased probability of occurrence may result in increased risk to these systems; in a survey conducted by Claridge (2016b), rangers in the Australian Alps reported degradation of wetland and riparian ecosystems by sambar deer.

These conclusions should be interpreted carefully, as the higher predicted probabilities of occurrence in wetland vegetation formations and open areas may reflect a sampling bias towards areas where deer are highly visible. Conducting targeted sampling for the presence of sambar deer in closed habitats would test whether these results are truly representative or the result of sampling bias.

In the model predicting probability of occurrence for fallow deer, wet sclerophyll forests (shrubby subformation) had the highest predicted probability of occurrence across all vegetation formations (Figure 4-10). These results contradict established habitat preferences of fallow deer, which often select for grassy woodlands or grasslands (Apollonio *et al.*, 1998; Thirgood, 1995). However, these results may also be due to sampling bias as records of fallow deer are primarily concentrated along the Towamba River, an area that is predominantly wet sclerophyll forest. A more uniform sampling effort across the study area aimed at detecting fallow deer may be more representative.

In the pooled species model, wetlands and urban areas had the highest probability of occurrence of deer (Figure 4-5). The high probability of occurrence in the wetland land use class is likely linked to the high probability of occurrence of deer in wetland vegetation formations in this model as discussed above. The high probability of occurrence in urban areas is more surprising. However, in other areas of Australia deer have been successful in suburban and urban areas (Burgin *et al.*, 2015). Urban areas also have a higher volume of human traffic, therefore increasing the chances of deer being opportunistically observed. This is an inherent bias present in opportunistic sampling and incidental reporting, which can artificially skew the distribution of an observed species towards areas nearer human activity.

Land use has a significantly higher contribution to predicting probability of occurrence for sambar deer compared to all other models (Table 4-4). In the sambar deer model conservation area and river and drainage system classes have probability of occurrence values above 0.5, while all other land use classes have values below 0.2 (Figure 4-13). The importance of river and drainage areas is likely because the presence of perennial water sources is an important factor in habitat suitability for sambar (Forsyth *et al.*, 2009; Gormley *et al.*, 2011). Conservation areas are likely to be highly suitable because they contain vegetation communities that are known to support sambar, such as heathland, wet eucalypt forests, rainforests, and dry eucalypt forests (Claridge, 2016a; Forsyth *et al.*, 2009; Gormley *et al.*, 2011).

Distance to water had a much lower than expected contribution to all models (Table 4-1, Table 4-3, Table 4-4). The presence of perennial water sources are an important factor in habitat suitability for all deer species in my study area (Claridge, 2016a; Forsyth *et al.*, 2009; Gormley *et al.*, 2011), and I expected it to be a limiting factor in the landscape. For the pooled species model and the model of fallow deer, probability of occurrence did decrease with increasing distance from water, but only marginally (Figure 4-3). The model of sambar deer produced a weak positive relationship between probability of occurrence and distance to water. These unexpected responses may be the result of an abundance of perennial water sources in my study area. Whereas distance to non-woody vegetation ranged from 0 to 2,742 metres, distance to water ranged from just 0 to 73 metres. Due to the large home range of deer (approximately 2 km^2) (Chatterjee *et al.*, 2014; Lewis *et al.*, 1990), even a distance of 73 metres would be well within the average home range of an individual. If perennial water sources are within the home range of deer regardless of their location, it is unlikely to be an important factor in predicting probability of occurrence in this area.

5.1.2 Comparing model outputs with existing models and anecdotal reports

There are multiple geographic trends in the predicted distribution of deer in my study area. The highest probabilities of occurrence are concentrated in the near-coastal areas from approximately Eden to Bateman's Bay, and then shift further inland towards the escarpment to the north of Bateman's Bay. Major river and drainage systems, such as the Shoalhaven River in the north and the Towamba River in the south, align with corridors of high probability of occurrence. Areas with low probability along the escarpment are highly correlated with areas of densely forested conservation reserves. In these areas pockets of non-woody vegetation are uncommon, likely contributing to the lower predicted probabilities in these areas. Only a small proportion of the overall study area is predicted to have near-zero probability of occurrence. This model conforms with anecdotal reports and informal conversations I had with NPWS field staff about the presence of deer in their local area. I was expecting slightly higher probability of occurrence values for areas along the escarpment (J Bentley, D McCreery, pers. comm. 2019), however, these areas are poorly represented in my dataset due to a lack of sampling and the limited accessibility of the area.

Anecdotal evidence suggests that fallow deer are the most widespread species in my study area and present throughout the South Coast of NSW. However, these claims are not reflected in the available records (see Appendix 2). The fallow deer model was created with presence data that were primarily from the far south of my study area, representing only a fraction of the alleged range of the species. Despite the geographic bias in my input data, my model nevertheless indicates that there are areas with moderate and high probabilities of occurrence throughout the South Coast. Conducting additional surveys targeted at detecting fallow deer may be necessary to produce more comprehensive future models.

Although my model covers a different geographic extent, the results can be compared to a Maxent model created by Gormley *et al.* (2011) to predict the probability of occurrence of sambar deer across Victoria. Slope had a low contribution to both models, likely because slope is correlated with a variety of other variables (e.g. land use, vegetation formation, etc.) and key relationships between slope and probability of occurrence would be represented in these datasets. Distance to water had a much higher contribution in the model created by Gormley *et al.* (2011) (6.7%) than in my model (0.1%), likely the result of differences in the distribution of perennial water sources in the two study areas. The study area used by Gormley *et al.* (2011) had distance to water values ranging from 0-1500 meters, whereas in my study area the maximum distance to water was 73 meters. Therefore, the range of values for distance to water across Victoria is larger than the range in my study area, creating more contrast between sites and a greater impact on the overall probability of occurrence. Gormley *et al.* (2011) also found that the percent cover of wet sclerophyll forest in each grid cell had the highest contribution to the predicted probability of occurrence of sambar deer. My model separates wet sclerophyll forest into two subformations, grassy and shrub. The shrub subformation has a moderate probability value, whereas the grassy subformation has the lowest probability value of all vegetation formations (Figure 4-13). The stark difference in probability values between the two types of wet sclerophyll forest may indicate that not all wet sclerophyll forests are equal as habitat for sambar. The dominance of wetland vegetation formations in my model (Figure 4-13) which were not included in the Victorian model, may also indicate that sambar deer prefer these instead of wet sclerophyll forests.

The geographic trends represented in my model (Figure 4-8) conform with anecdotal reports that suggest sambar deer are primarily concentrated in the south of my study area. However, my model indicates that there may be areas with highly suitable habitat further north: particularly along the escarpment north of Bateman's Bay and near the Shoalhaven River.

5.1.3 Assessing the risk to threatened ecological communities

Results from my analysis of the probability of occurrence of deer within TECs indicates that some TECs currently appear to be more suitable for deer than others. *Themeda* Grassland, Freshwater Wetland, and River-flat Eucalypt TECs had the largest proportion of area with high probabilities of deer (Table 4-2). *Themeda* Grassland is likely to have a high proportion of suitable land because this is an open, grassy ecosystem typically adjacent to dense woody vegetation that can provide cover. Freshwater Wetlands are similarly dominated by non-woody vegetation and occur close to dense woody vegetation. River-flat Eucalypt communities are often found immediately surrounding perennial water sources, and typically occur in areas with higher rainfall. The combination of a low distance to water and higher levels of precipitation in the driest quarter result in a large proportion of this TEC having a high probability of supporting deer. Littoral Rainforest also has a substantial proportion of overall area above the moderate and high probability thresholds. Swamp Sclerophyll Floodplain Forest has the lowest proportion of any TEC in each of the three probability categories.

These findings provide important information for land managers. TECs with a large proportion of area above the high or very high probability thresholds of supporting deer are likely to be at higher risk from the ecological impacts of deer. Although there is limited research into the impacts of feral deer on threatened communities in Australia, a study in Royal National Park found that deer had a significant impact on browse in endangered Littoral Rainforest communities (Moriarty, 2004b). My results indicate that Littoral Rainforest, as well as *Themeda* Grassland, Freshwater Wetlands, and River-flat Eucalypt communities, may be at high risk of damage by feral deer (Table 4-2). Using this information, land managers can prioritise monitoring and control efforts towards those communities at highest risk.

5.1.4 Model limitations

Despite producing three relatively strong models, there are limitations associated with each. There are multiple ways to represent each environmental variable. For example, Gormley *et al.* (2011) used the amount of grassland, the amount of wet sclerophyll forest, and amount of native grassland or shrubland to represent important vegetation types, while I chose to use vegetation formation. My decision allowed me to test across a range of vegetation types, while Gormley *et al.* (2011) chose to test against a select few. These differences in approach can be justified given the given specific aims of the research, but must be interpreted accordingly. All interpretations of the results of this model, and the relative importance of each environmental variable to predicting probability of occurrence, are limited to the appropriate context of the original input datasets.

The applications of these models are limited to probability of occurrence only. These models cannot be used to assume abundance of deer at a given location. The expansion of existing deer populations is limited by several factors (e.g. population growth, habitat connectivity, etc.), and some areas with a high probability of occurrence may never be occupied. Conversely, relocation of deer by hunters (Moriarty, 2004a) may mean that deer will occupy areas currently with moderate to low probability of occurrence. This model is simply a prediction; it should be used to optimise the effectiveness of monitoring programs, and not as a replacement for further field-based studies.

The relationship between the geographic extent of model inputs and the geographic extent of the study area being tested can influence the model output. When using Maxent to model fallow deer in Tasmania, Potts *et al.* (2015) found that output varied depending on whether the geographic extent of inputs was limited to Tasmania or included Tasmania and mainland Australia. As fallow deer had not reached the full extent of their potential range in Tasmania, the model using only inputs from Tasmania reflected the habitat characteristics of locations where populations had been introduced rather than the true climatic range of the species. By incorporating environmental variables and presence points from the mainland as well, the model included a wider range of values for each variable and the output better reflected the true climatic range of fallow deer (Potts *et al.*, 2015). This has potential application to my study area, which is bounded by two large and expanding populations of deer: dense rusa deer populations to the north in the Illawarra region and Royal National Park, and expanding sambar deer populations spreading north from Victoria. The habitat occupied by sambar in Victoria is similar to that in the southern end of my study area, and there are a sufficient number of sampling points to ensure these characteristics are well represented in my model. However, the northern end of my model has a much lower sampling effort and is poorly populated with records of deer (Figure 4-1). By including presence points from Royal National Park and the Illawarra region, the habitat characteristics that are suitable for deer in these areas would be included in my model and may create a more accurate representation of the probability of occurrence of feral deer in the north.

Although I used 560 presence points in my model, there are pockets within my study area that are under sampled for deer (Figure 4-1). Maxent assumes that all locations within the given geographic extent have an equal probability of being sampled and that all environmental factors are sampled proportionately (Merow *et al.*, 2013). My field methods ensured that all TECs within my study area were sampled proportionately and had a relatively equal probability of being sampled. However, this represents only a small proportion of the overall region. Achieving an equal probability of sampling from a dataset comprised of incidental sightings is rare, particularly when parts of the study area (such as the coastal escarpment) have limited accessibility. Results from my model therefore have to be interpreted cautiously. An increased sampling effort targeted at a few key areas might dramatically improve the accuracy of my model. These include the area north of Jervis Bay, the escarpment between approximately Bega and Moruya, and all areas immediately to the west of the coastal escarpment. Due to under-sampling in these regions, it is possible that predicted probabilities of occurrence for these areas in my model are lower than true values.

Moriarty (2004a) found that escapes from deer farms are a key point source for new feral populations. Within my study area, an abandoned deer farm in Bimberamala National Park has been recognized as the source of a local fallow deer population (D Cunningham, pers. comm. 2019). I intended to include the location of current or historic deer farms as a potential explanatory variable in my Maxent models. However, I found very few records of deer farms within my study area. Including these data in the future would strengthen the model and represent a key explanatory factor in the distribution of feral deer in Australia.

5.2 Browse damage by deer

My methods and analysis considered two metrics of browse: proportion of individuals browsed and average browse intensity of each species or life form. These two metrics are calculated differently and should be interpreted differently. The proportion of individuals browsed indicates the evenness of browse for each species or life form; differentiating between plants with just a few individuals browsed and those with the majority of individuals browsed. Average browse intensity indicates whether plants were browsed lightly or at a potentially damaging level. Species or life forms with both a high proportion of individuals browsed and high average browse intensity are likely to be most at risk from browse damage.

5.2.1 Browse damage to non-woody vegetation

Results from my GLMM indicate that the predicted proportion of non-woody individuals browsed is higher where deer are present and for certain life forms (Figure 4-15). Cycads, ferns, and sedges had the highest proportion of browse across all non-woody life forms. This indicates that herbivores generally prefer these life forms over others such as rushes or grasses. However, because there is no interaction between life form and the presence of deer, I cannot infer from my results whether deer are directly increasing the proportion of browse of any specific life form. Previous studies of feral deer (Davis *et al.*, 2008; Forsyth and Davis, 2011; Moriarty, 2004b) and macropod (Davis *et al.*, 2008; Moriarty, 2004b) diets, with techniques such as pellet and rumen content analysis, indicate that ferns and sedges are within the normal dietary ranges of both herbivores. Cycads have not been frequently recorded as components of either deer or macropod diets, making it unclear which group is responsible for the recorded browsing pressure.

The presence of deer, along with macropod density, increased proportion of non-woody life forms browsed (Figure 4-15). This indicates that deer have an additional impact on the level of browse above background rates. Research within Australia (Davis *et al.*, 2016; Davis and Coulson, 2010; Moriarty, 2004b) and internationally (Côté *et al.*, 2004; Rooney and Waller, 2003; Augustine and McNaughton, 1998) repeatedly shows that deer can create large-scale changes to vegetation communities with the quantity or quality of browse they consume. For example, a study in Royal National Park estimated that a single deer consumes as much browse as 3.88 swamp wallabies on a daily basis (Moriarty, 2004b), which means even a low density deer population could raise the proportion of browse above background rates.

In contrast to the model predicting the proportion of non-woody life forms browsed, the average browse intensity model included an interaction between life form and deer presence (Figure 4-14). This interaction allows for analysis of differences in average browse intensity for each life form in locations where deer are present or absent. Rushes, cycads, sedges, and grasses had higher average browse values at sites where deer were present compared to sites where deer were absent. It is reasonable to infer then that deer are preferentially browsing rushes, sedges, cycads, and grasses at higher intensities than other life forms. Deer have highly versatile diets, consuming a wide range of vegetation. Rushes, sedges, and grasses have been frequently recorded in the diets of deer throughout eastern Australia (Davis *et al.*, 2008; Forsyth and Davis, 2011; Moriarty, 2004b). Cycads, as previously mentioned, have not been recorded. However, anecdotal reports indicate that cycads may be more heavily browsed at sites where deer are present (J Miles, pers. comm. 2019).

Differences in results between the proportional and average browse models for non-woody life forms could have multiple implications. Browsing by deer is responsible for an increase in average browse intensity for some life forms, while the proportion of non-woody individuals browsed indicates this increased browsing pressure is currently focused on a small proportion of plants. Anecdotal evidence suggests that the density of deer across most of my study area is still low compared to other areas of south-eastern Australia, and it is likely that there are only low numbers of deer at each of the sites I recorded presences. A small number of deer might be heavily browsing a small number of individuals of each of the preferred life forms. As the density of deer increases, the proportion of browse and total volume of browse is likely to increase as well. These possibilities underscore the need for an established monitoring program to track changes in deer density, and their associated impacts, over time.

My results also indicate that both the proportion of non-woody life forms browsed and average browse intensity increases with increasing macropod pellet abundance. Macropod pellet abundance can be used as an index for macropod density (Hill, 1981). Sites with a higher pellet abundance are likely to have denser populations than sites with lower pellet abundance. Research and logic support the conclusion that sites with a higher density of macropods will have higher levels of browsing pressure (Dexter *et al.*, 2013). However, browse predictions for sites with extremely high pellet counts (n > 50) have high confidence limits (due to the low number of sites with high pellet counts) and should be interpreted with caution (Figure 4-15).

5.2.2 Browse damage to woody vegetation

The model predicting the proportion of browse of woody species similarly included deer presence, macropod pellets, and woody species (Table 4-12). The proportion of woody species browsed was only slightly higher where deer were present (Figure 4-15). As with non-woody life forms, deer appear to marginally increase the proportion of plants being browsed, but the effect of deer on the proportional browse of specific species cannot be determined. The proportion of woody species browsed increases with increasing macropod pellets (Figure 4-15), likely for the same reasons that this relationship existed with non-woody life forms. Research has shown that the swamp wallaby (*Wallabia bicolor*), a common macropod in my study area, can browse on a range of both woody and non-woody species, explaining the positive relationship between pellet counts and all types of browse (Davis *et al.*, 2008; Moriarty, 2004b).

The average level of browse of woody species was lower where deer were present (Figure 4-16). This is in direct contradiction with quantitative and anecdotal evidence which suggest deer not only consume woody vegetation, but do so at a higher rate than native herbivores (Claridge, 2016b; Moriarty, 2004b). However, much of that research was conducted in areas with high density deer populations. The current density of deer in my study area may not be high enough yet to significantly affect browsing pressure on woody vegetation. Deer can also be seasonal browsers, shifting between woody and non-woody vegetation depending on their relative availabilities. Moriarty (2004b) found that rusa deer in Royal National Park consumed more woody material in the winter when grasses are scarce, but shifted to more grasses and herbs in the summer when those plant forms were more abundant. Most of my field work was conducted in autumn when grasses and other non-woody vegetation were still available. My data therefore may not have captured the maximum period of woody browse for deer in the area. Resultant models may represent an underestimate of the true browsing pressure for woody species.

The 12 woody plant species I chose to examine for browsing pressure had varying proportions of individuals browsed and average browse intensities (Figure 4-16, Figure 4-17).

Callistemon citrinus, Leucopogon parviflorus, and *Phyllanthus gunnii* had the highest proportions of individuals browsed, while *Banksia spinulosa* and *Casuarina cunninghamiana* had the lowest. Average browse intensity was relatively similar across all species, except for *Leucopogon parviflorus* which was significantly higher than all other species (Figure 4-16). Many of the woody species used in my analysis have been linked to browsing by deer in previous research. Using rumen content analysis of sambar deer in Victoria, Forsyth and Davis (2011) identified traces of *Acacia dealbata* and *Banksia spinulosa* along with other unidentified *Acacia* species. Keith and Pellow (2005) also concluded rusa deer were likely browsing on *Leucopogon parviflorus, Phyllanthus gunnii, Acacia longifolia, Acacia implexa,* and *Acronychia oblongifolia* in Royal National Park. Anecdotal evidence also suggests that browsing on *Acronychia oblongifolia* is also higher in areas where deer are present (J Miles, pers. comm, 2019). Without an interaction between woody species and deer presence, deer cannot be specifically linked to heightened browse of any one species. However, the research by Keith and Pellow (2005) and Forsyth and Davis (2011) demonstrates deer are potentially browsing on a majority of the species in my model.

5.2.3 Browse in threatened ecological communities

I chose to conduct separate analyses of browsing pressure within threatened ecological communities even though TEC was not included as a factor in any of the best GLMMs. Non-woody life form and woody species could not be included in the same model as TEC due to high correlation between the two variables. Thus, only the variable with greater predictive power was included in the best model. The exclusion of TEC, and subsequent inclusion of non-woody life form or woody species, indicates that vegetation species or life form is a more important factor than TEC in determining browsing pressure. However, understanding browsing at a community level may be easier for land managers to respond to than statistics about browse at a species or life form level.

Results from my GLMMs indicated that within TECs, the average browse intensity of non-woody life forms was higher where deer were present, while all other measures of browse currently have no measurable effect from deer presence. Where deer were present, there was increased average browse intensity of non-woody plants in Coastal Saltmarsh, Freshwater Wetland, and Littoral Rainforest TECs (Figure 4-18). The increased average browse intensity of non-woody life forms in Coastal Saltmarsh and Freshwater Wetland where deer were present may have been influenced by the high proportion of grasses and rushes in these communities (earlier analysis indicated that the average browse intensity of these life forms was higher when deer were present). These results indicate that Coastal Saltmarsh, Freshwater Wetland, and Littoral Rainforest communities may be at a higher risk of damage by feral deer.

5.2.4 Limitations

My analysis included only presence or absence of deer at each site and did not include an index of their density or abundance. As such, results presented here cannot be used to infer a relationship between deer density and browse damage. Rather, they should be used to support anecdotal evidence that the presence of deer does have an additional impact on browse for some vegetation types.

Browsing niche overlap creates one of the largest limitations to determining the impact of browsing by deer. Deer have highly versatile diets and have been recorded to browse on a wide range of native and non-native vegetation (Davis *et al.*, 2008; Forsyth and Davis, 2011; Moriarty, 2004b). This often includes varying degrees of overlap with the diets of sympatric macropods (Moriarty, 2004b; Davis *et al.*, 2008), making it difficult to attribute browse damage to a particular species. The only way to provide more certainty about the impacts of browse by specific herbivore groups is to conduct experiments with long-term exclusion plots (Goetsch *et al.*, 2011; Moriarty, 2004b), which I did not have the time nor resources to complete in the limited timeframe.

Another limitation to my research is the extent of my surveying effort. Although I attempted to sample vegetation communities at a rate proportional to their available area, some TECs were harder to access and therefore remained under-sampled. *Themeda* Grassland, Swamp Oak Floodplain Forest, Littoral Rainforest and Freshwater Wetland each had less than ten survey sites. These limited sample sizes mean that the conclusions I have drawn about these TECs may not be as wholly representative of these communities. Additionally, my surveys were only conducted in TECs and sampled a small portion of the existing vegetation communities. Deer occur across a wide range of vegetation communities throughout the NSW South Coast and their impacts are likely not limited to the eight TECs in my study. Expanding research on browsing of plants by deer to include a representative sample of all vegetation types might provide a clearer representation of overall impacts.

5.3 Implications for future management and research

5.3.1 Species distribution and habitat suitability

These models represent a baseline for the likely current distribution of deer in my study area, and can be repeated to track future changes in the distribution of feral deer over time. The invasion curve established by Blackburn *et al.* (2011), and the corresponding recommended management strategies (Victoria Department of Primary Industries, 2017) (Figure 2-2), can be used in conjunction with my models to optimise management of feral deer. My analysis indicates in the South Coast of NSW *Themeda* Grassland, Freshwater Wetland, and River-flat Eucalypt have the highest probability of occurrence of deer (Table 4-2) and should be prioritised for asset protection. Freshwater Wetland should be a particular priority because browse damage is also higher in this TEC where deer are present (Figure 4-18). The application of my model can also be expanded beyond TEC, to include the protection and management of a range of assets important to land managers. Areas that my model identifies as having a high probability of occurrence, but are not yet occupied by deer, could be identified as sites where the occupancy of deer could be prevented.

Maxent software provides a repeatable and relatively simple method for predicting the probability of occurrence of feral deer. The methods I used are not limited to my study area, and could be replicated to produce models for any region in Australia. Producing a similarly fine-scale model for probability of feral deer occurrence would be useful for land managers looking to manage established populations or get ahead of emerging threats. The literature also provides examples of Maxent successfully predicting the probability of occurrence of a range of species. This would likely be a successful technique for modelling the potential spread of invasive plant and animal species across Australia.

5.3.2 Impacts of deer browsing

The statistics on browsing pressure on plants presented throughout this thesis are simply a snapshot across the broader South Coast of NSW. In their simplest state, my results provide support for anecdotal evidence that deer have an additional browsing effect on some types of vegetation. This provides immediate justification for effective control programs and an investment in further, more detailed research into the effects of deer on vegetation communities.

My research does not provide data about the relationship between deer density and browsing impact on plants. However, this may not be necessary to inform management. Morellet *et al.* (2007) put forward the concept of using ecological indicators rather than absolute measures of density or abundance to justify managing deer and other pest species. Browse indices, such as the ones used in my research, are ecological indicators which measure impacts of a species on their ecosystem. In their paper, Morellet *et al.* (2007) argue that ecological indicators are not only more efficient than methods used to measure absolute abundance, but do a better job of answering key management questions. For example, measuring the density of deer across the South Coast is only relevant to management if there is a known relationship between density and impact. Undertaking the research necessary to understand these interrelationships would be both time and resource intensive, and delay the implementation of any management plans. Therefore I recommend that land managers continue to monitor browsing and other impacts of deer on vegetation, and adjust control methods to maintain a desired level of impact rather than a desired population density.

Creating effective management plans for feral deer throughout the South Coast is important because my results indicate that deer presence measurably increases browsing pressure on both woody and non-woody vegetation, and some TECs (Figure 4-14, Figure 4-15). This is particularly evident for rush, sedge, grass, and cycad life forms. Although the effect of deer on woody vegetation is low, previous research suggests it will increase as the deer population expands and local populations increase in density (Côté *et al.*, 2004; Moriarty, 2004b).

Certain TECs should be immediately prioritised for feral deer management and control. *Themeda* Grassland, Freshwater Wetland, and River-flat Eucalypt TECs have the highest proportion of area with a high probability of occurrence of deer (Table 4-2). Additionally, Coastal Saltmarsh, Freshwater Wetland, and Littoral Rainforest TECs have increased average browse intensity of non-woody life forms where deer are present (Figure 4-18). Freshwater Wetland is likely at the highest risk of damage by feral deer due to the combination of a high proportion of suitable habitat and increased average browse intensity where deer are present. *Themeda* Grassland is also at an increased risk due to a combination of highly suitable habitat and browse composition. *Themeda* Grassland communities are comprised primarily of grasses, a life form with increased average browse intensity where deer are present. Therefore, if deer were present in these communities, impacts would likely be significant.

The methods I used in this research were designed to be easily repeatable and require minimal training. The integration of these monitoring programs, both for species distribution and browsing pressure, with new or existing citizen science programs to maximise the amount of data that can be collected, would be ideal. Citizen science is a useful tool that can help researchers and management groups increase the amount of data collected for a range of conservation projects (Devictor *et al.*, 2010; McKinley *et al.*, 2017; Kobori *et al.*, 2016; van Strien *et al.*, 2013). In the South Coast of NSW there is a wealth of anecdotal information held by local landholders and members of the public about where deer are in the landscape. By promoting existing reporting tools and/or creating a new system, anecdotal information has the potential to be transformed into quantitative, usable data. Even though the surveys of browsing pressure are slightly more complicated, there are a range of applications that allow users to design surveys and upload photos and data from their mobile phones. This method would still require some level of verification from experts, but the overall efficiency of data collection would still be improved.

Chapter 6: Conclusion

The first aim of my thesis was to predict the current distribution of deer in the NSW South Coast, based on available presence data. My models indicate that deer are likely to be widespread throughout the South Coast of NSW, although areas closer to the coast have the highest predicted probability of occurrence. *Themeda* Grassland, Freshwater Wetland, and River-flat Eucalypt TECs have the highest proportion of area likely to be occupied by deer. I recommend that these models are improved upon and used as the basis of a continuing research effort to model habitat suitability and occupancy throughout the region.

My second aim was to establish the extent to which deer are a current and likely future threat to TECs within the South Coast region. I found that where deer are present, the proportion of non-woody life forms and woody species browsed increases, as does the average browse intensity of non-woody life forms. Specifically, deer increase the average browse intensity of grasses, sedges, rushes, and cycad life forms. Plant life form (or species) was found to be more important than TEC for predicting browsing pressure by deer. However, my results indicate that deer also increase the average browse intensity of non-woody life forms in Freshwater Wetland, Coastal Saltmarsh, and Littoral Rainforest TECs.

Combining the results from both aims, I conclude that Freshwater Wetland and *Themeda* Grassland TECs are at greatest potential risk from impacts of feral deer. Freshwater Wetland communities are threatened by a combination of two factors: a high proportion of area likely to be occupied by deer, and increased browsing pressure on non-woody life forms when deer are present. Similarly, *Themeda* Grassland communities have a high proportion of area likely to be occupied by deer and are dominated by grasses, a life form with increased browsing pressure where deer are present.

The predictive models I created can provide the data necessary for land managers in the South Coast of NSW to monitor and manage feral deer. These models represent a baseline for the likely current distribution of deer in my study area, and can be repeated in future to track changes in the distribution of feral deer over time. Using these models in conjunction with the results from my browse surveys will allow land managers to target specific at-risk geographic areas or vegetation communities for deer control. At a broader scale, the results from my studies of browsing pressure can be used to help estimate the impacts of feral deer populations elsewhere across south-eastern Australia. Additionally, the methodology I used to create the Maxent models can be repeated across a range of introduced species to further inform invasive species management in Australia.

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Appendix 1: Complete list of woody species browsed

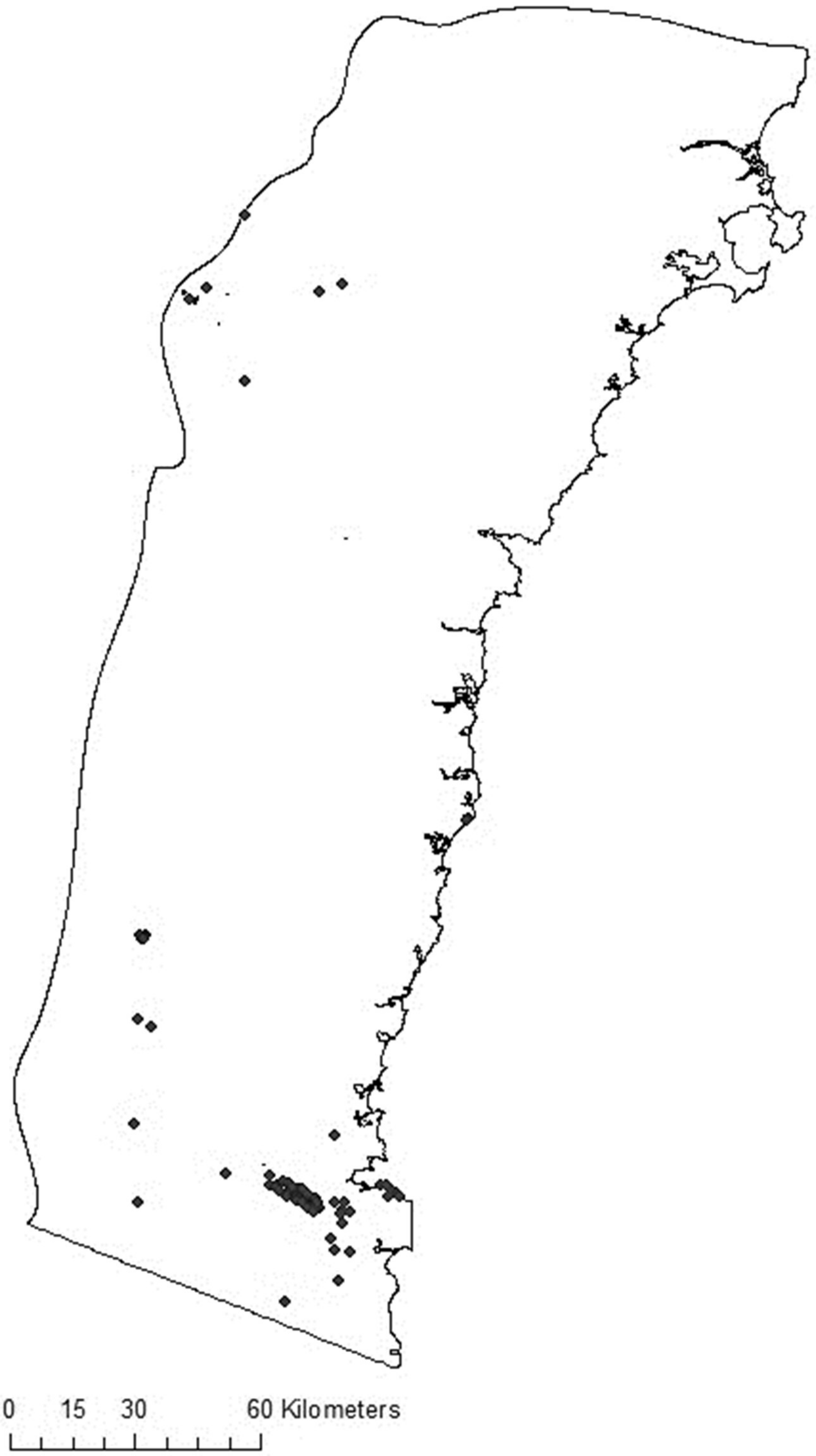
Species name	Average of browse intensity	Count
Acacia binervata	0	2
Acacia myrtifolia	0	2
Acacia ulicifolia	0	2
Avicennia marina	0	14
Baeckea virgata	0	1
Banksia marginata	0	33
Beyeria lasiocarpa	0	1
Callicoma serratifolia	0	8
Cassinia aculeata	0	2
Chrysanthemoides monilifera	0	10
Cissus hypoglauca	0	1
Coprosma hirtella	0	5
Dillwynia glaberrima	0	2
Doryphora sassafras	0	7
Eupomatia laurina	0	2
Eustrephus latifolius	0	1
Glochidion ferdinandi	0	11
Goodia lotifolia	0	1
Grevillea parviflora	0	1
Hakea dactyloides	0	4
Hakea sericea	0	14
Helichrysum dendroideum	0	2
Leptospermum attenuatum	0	43
Leptospermum emarginatum	0	21
Leptospermum flavescens	0	36
Leucopogon juniperinus	0	25
Melaleuca armillaris	0	3
Melaleuca ericifolia	0	121
Monotoca scoparia	0	3
Persoonia levis	0	1
Persoonia mollis	0	1
Petrophile pedunculata	0	10
Pimelea ligustrina	0	1
Pimelea linifolia	0	1
Pittosporum revolutum	0	20

Platysace lanceolata	0	6
Pomaderris cinerea	0	5
Pomaderris virgata	0	2
Senecio minimus	0	1
Smilax glyciphylla	0	4
Solanum mauritianum	0	2
Solanum stelligerum	0	1
Stellaria flaccida	0	17
Syncarpia glomulifera	0	1
Tristaniopsis laurina	0	11
Zieria arborenscens	0	3
Zieria smithii	0	1
Banksia serrata	0.1	50
Kunzea ambigua	0.3	50
Leptospermum lanigerum	0.34	29
Banksia integrifolia	0.43	35
Lomatia myricoides	0.48	73
Pomaderris aspera	0.5	10
Ricinocarpos pinifolius	0.53	19
Rhagodia baccata	0.59	51
Hedycarya angustifolia	0.67	15
Eucalyptus botryoides	0.83	6
Myoporum acuminatum	0.83	6
Correa reflexa	0.91	22
Pittosporum undulatum	0.98	382
Acmena smithii	1	85
Hymenanthera dentata	1	5
Hibertia scandens	1.19	66
Notelaea venosa	1.20	83
Epacris paludosa	1.25	8
Leucopogon lanceolatus	1.25	4
Leptospermum phylicoides	1.30	27
Eriostemon trachyphyllus	1.39	18
Rapanea howittiana	1.39	18
Leptospermum juniperinum	1.61	31
Elaeocarpus reticulatus	1.67	21
Lambertia formosa	1.83	30
Ficus coronata	1.88	16
Platylobium formosum	2	5
Monotoca elliptica	2.22	18

Synoum glandulosum	2.26	93
Acacia suaveolens	2.31	39
Goodenia ovata	2.33	15
Leptospermum brevipes	2.42	31
Casuarina littoralis	2.48	161
Exocarpos cupressiformis	2.78	9
Solanum prinophyllum	3	5
Casuarina glauca	3.17	115
Cassinia trinerva	3.33	3
Polyscias sambucifolia	3.33	3
Acacia mabelliae	4.03	62
Acacia floribunda	4.17	36
Acacia melanoxylon	5	1
Acacia terminalis	5	1
Ficus rubiginosa	5	1
Marsdenia rostrata	5	4
Bursaria spinosa	5.20	74
Acronychia oblongifolia	5.54	37
Casuarina paludosa	5.71	7
Acacia dealbata	5.93	27
Acacia implexa	6.67	27
Persoonia linearis	7.14	7
Casuarina cunninghamiana	7.35	83
Dodonaea triquetra	7.56	78
Acacia longifolia	7.61	71
Babingtonia virgata	8	10
Hibbertia scadens	8.33	3
Angophora costata	10	2
Callistemon citrinus	11.18	17
Indigofera australis	11.25	4
Myoporum boniense	11.25	4
Helichrysum diosmifolium	12.86	7
Cissus antarctica	13.33	6
Lantana camara	13.33	3
Banksia spinulosa	15.42	12
Phyllanthus gunnii	16.92	13
Breynia oblongifolia	20	2
Hibbertia aspera	22.5	2
Backhousia myrtifolia	26.67	3
Phytolacca octandra	26.67	3

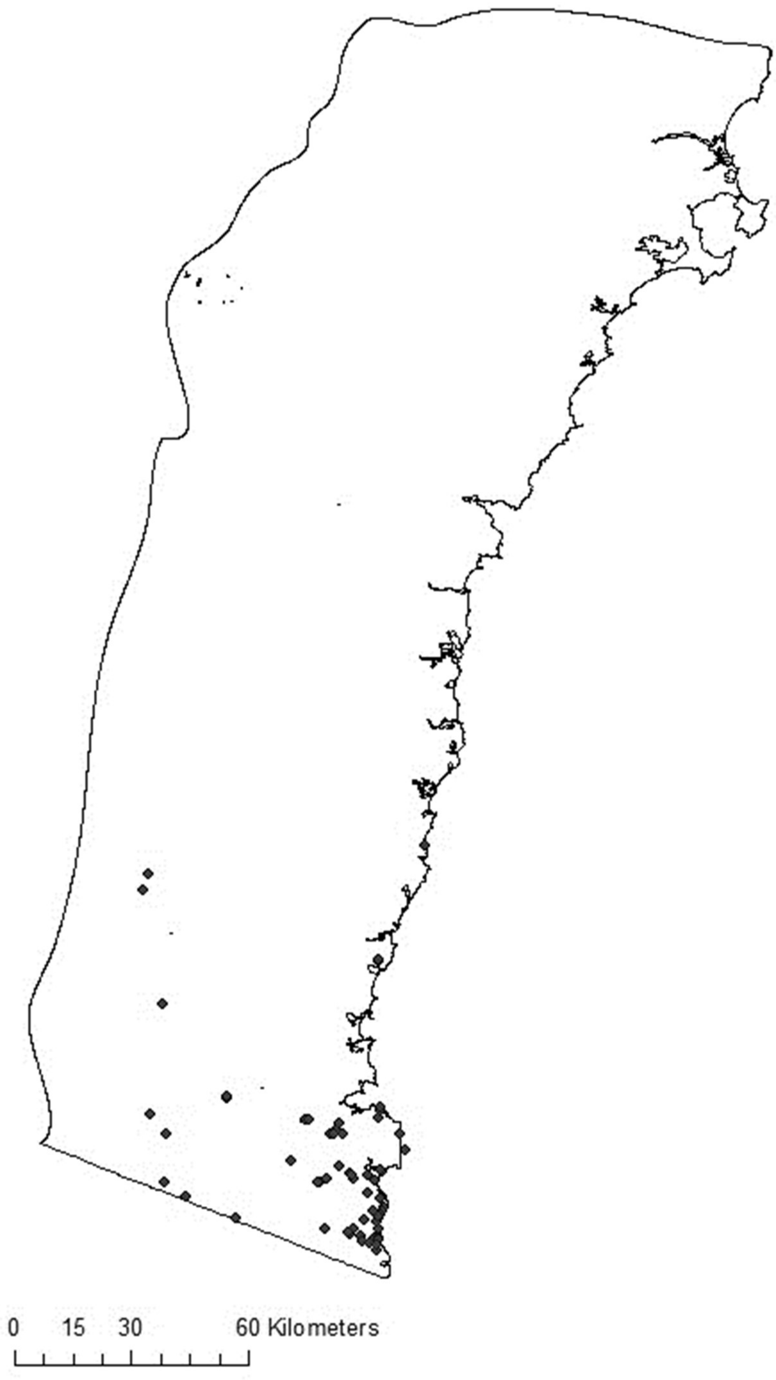
Leucopogon parviflorus	35.45	44
Pomaderris lanigera	40	3
Acacia silvestris	50	2
Clematis glycinoides	70	1
Sigesbeckia orientalis	95	1

Appendix 2: Distribution of presence-only data used in fallow species Maxent model



This map shows the distribution of 76 data points used in the Maxent model predicting the probability of occurrence for fallow deer.

Appendix 3: Distribution of presence-only data used in sambar species Maxent model



This map shows the distribution of 60 data points used in the Maxent model predicting the probability of occurrence for sambar deer.

Appendix 4: Scientific License

NSW | NSW National Parks & Wildlife Service
Office of Environment & Heritage

SCIENTIFIC LICENCE

Biodiversity Conservation Act 2016

Name and postal address of principal licensee

Miss Heather Burns

AUSTRALIAN NATIONAL UNIVERSITY

24 Cole Street

DOWNER ACT 2602

Your licence number is: SL102234

This licence is valid from: 18 April 2019 **This licence will expire on:** 30 April 2020
Additional authorisations:

Project Title: Evaluating the threat of feral deer to threatened ecological communities on the NSW South Coast

This class of biodiversity conservation licence granted under Part 2 of the *Biodiversity Conservation Act 2016* authorises the following activities: Pick plants for identification, establish survey plots, place camera traps and conduct research on NPWS estate.

This licence authorises the principal licensee and any associates named in **Attachment A** to conduct those activities authorised above, to those species, communities or materials listed in **Attachment B**, at the locations specified in **Attachment C** of this licence.

This licence also authorises the principal licensee to conduct research on National Park estate under clause 23 of the *National Parks and Wildlife Regulation 2009 (*NPW Reg*),* where this forms part of a project approved by a delegated officer of the Office of Environment and Heritage (OEH).

This licence is granted subject to the provisions of *Biodiversity Conservation Act 2016*, Biodiversity Conservation Regulation 2017, the general conditions listed below, any special conditions as may be notified in writing to the licensee by the Environment Agency Head of the Department of Planning and Environment or a 'delegated officer' of OEH ("delegated officer") and the OEH "Scientific Licensing Policy".

…………………………………………… ..……Heather Burns……..

Signature of Delegated Officer **Signature of Principal Licensee***

Date: 07 June 2019 Date:..…18 June 2019…………..

* This licence is not valid unless it is signed by the principal licensee. By signing this licence the licensee agrees that they have read, understood and agree to comply with all of the conditions listed on the licence.

LICENCE CONDITIONS

Specific

a) The licensee may only undertake works within NPWS managed lands with the prior written approval of the relevant area manager.
b) The licensee must contact the local NPWS Area Office at least 7 days prior to work on National Park estate and comply with any restrictions or limitations imposed by NPWS staff. No entry is to occur on total fire ban days or Public Holidays.
c) The licensee must comply with any restrictions or conditions imposed by the NPWS local area office.
d) The following conditions apply specifically for works undertaken in the Shoalhaven Area:
    i) The licensee must provide two weeks' notification prior to cameras being installed and provide a location map for distribution to area staff.
    ii) Notification prior to field work must be provided to npws.shoalhaven@environment.nsw.gov.au
    iii) iii) A copy of results and any reports must be provided to the local NPWS area office.
e) The following conditions apply specifically for works undertaken in the Sapphire Coast Area:

  i) The licensee must contact Jo Vincent (Ranger) Jo.Vincent@environment.nsw.gov.au at least seven days prior to undertaking work.

f) The licensee may only conduct animal survey activities that do not require the complementary approval of an animal care and ethics committee (ACEC). This condition may be reviewed on production of a relevant ACEC permit.
g) Activities are to be undertaken in accordance with the NPWS Guidelines for the Collection of Voucher Specimens and Plant Material for Identification.
h) Clean, sharp secateurs are to be used to sample plants.
i) Phytophthora and frog hygiene protocols must be followed, and all tools and equipment must be thoroughly cleaned between sites and visits.
j) During collection, all associates are to wear apparel marked with 'RESEARCH' for clear identification to any public and NPWS staff i.e. high visibility vests.

General

1. Only the person/s named on the licence, or authorised to operate under the terms and conditions of the licence, may undertake the work. This licence is not transferable except with written confirmation from the Wildlife Team ("WT").
2. The principal licensee may vary the associated parties authorised during the term of the licence only by maintaining a signed and dated register of the associates. A copy of the register must be provided to the WT at renewal or on request by an authorised officer.
3. The licensee must carry this licence at all times whilst work is being undertaken in the field. Where multiple parties are listed, photocopies will suffice provided some other proof of identity can be provided e.g. Driver's licence.
4. The licensee must provide other parties authorised to conduct the specified activities with a copy of this licence.
5. The licensee must obtain the permission of the owner, manager or occupier of lands upon which research is conducted **(for persons working on NPWS lands see also conditions 18-20).**
6. Specimens or samples taken under this licence must not be sold, bartered, given, lent or promised to others without the prior written approval of the Environment Agency Head or delegate.
7. Collections or research shall, as far as is possible, be carried out away from the view of the public.
8. The licensee shall indemnify and keep indemnified, so far as the law allows, Her Majesty Queen Elizabeth II, the Minister administering the *Biodiversity Conservation Act 2016*, the Government of New South Wales, the Environment Agency Head of the Department of Planning and Environment, and the National Parks and Wildlife Service and its servants, agents or contractors (herein jointly and severally referred to as "OEH"), FROM AND AGAINST all lawful suits, claims, demands, proceedings, costs, (including solicitor - client costs) and expenses of any nature whatsoever which the OEH may suffer or incur in connection with loss of life, personal injury or damage to property from an occurrence in connection with any land, premises, vehicle or other mode of conveyance or other item under the care, control or management of the OEH, and arising either directly or indirectly from any negligent or wrongful act or omission of the licensee in the course of an operation or activities pursuant to the licence or otherwise.

Reporting requirements

9. The licensee undertaking survey, research or other biodiversity assessment works must provide a full report of the work carried out under this licence online via Bionet using the most recent version of the Atlas data sheet available at www.environment.nsw.gov.au/resources/atlas/AtlasDatasheet.xls.

10. The licensee must ensure that all coordinates provided as part of the data submitted to OEH include a measurement of the accuracy of those coordinates. Coordinate accuracy should not be greater than zero but no greater than **100m**.
11. The licensee must submit reports online using a secure login acquired from OEH Biodiversity Information Systems. Contact bionet@environment.nsw.gov.au for account details and guidelines.
12. Licensees undertaking work that cannot be supplied in the above format must provide a report to the OEH specifying:
    a. Title of the project
    b. A precise description of the locality including geographic coordinates where practical c. Results of the project

13. The licensee may also be required to complete a metadata proforma for works on NPWS estate.

14. Licensees undertaking permanent/semi-permanent marking, banding or tagging must provide marking details (e.g. tag number, date, location, species) to WT with any renewal application.

15. The licensee must provide a copy of any final report and/or any scientific papers relating to this work to the Environment Agency Head (marked "attention Wildlife Team") when the study is completed.

Additional reporting requirements for consultants

16. Licences granted to consultants and consulting companies for survey and assessment purposes are required to provide a list of the sites where work was conducted and a list of the reports produced. A copy of these reports may be requested.
17. Reports in accordance with licence conditions 9. to 16. must be provided annually, from the "valid from" date of the licence.

Projects undertaken on NPWS managed land

18. The licensee may only undertake works in NPWS managed lands with the prior written approval of the relevant Area Manager and comply with any imposed restrictions or conditions.
19. The licensee must maintain regular contact with the NPWS Area office throughout the project as park management activities and other events may affect access to research locations. Access to reserves may be restricted during management activities or while the reserve is closed for other reasons.
20. The licensee must only use vehicles on public roads unless otherwise approved by an authorised officer.

It is an offence under the *Biodiversity Conservation Act 2016* to breach any of the conditions of this licence, issue any false receipt, make a false entry in any record, or otherwise keep a false record or provide false or misleading records or information.

Records, notifications and inquiries should be directed to:

Wildlife Team Phone: 02 9585 6406

Office of Environment and Heritage Fax: 02 9585 6401

PO Box 1967 Email: scientific.licensing@environment.nsw.gov.au

Hurstville NSW 1481

Additional Information for licence holders

It is the licence holder's responsibility to ensure they are familiar with any other relevant statutory or regulatory provisions relevant to this licence such as the ***National Parks and Wildlife Regulation 2009****,* particularly with respect to activities undertaken on NPWS managed lands, the ***Firearms Act 1999,*** any local council, building and health requirements and codes of practice under the ***Prevention of Cruelty to Animals Act 1979***, as well as specific requirements under the ***Animal Research Act 1985***. On the expiration of your permit the onus is on you to renew. While OEH forwards renewal notices to permit holders, it will not be responsible for the non-receipt of such a notice.

It is the licensee's responsibility to inform themselves of any likely hazards and ensure that appropriate risk management and emergency procedures are developed and in place for works undertaken on NPWS managed lands. The risk management and emergency procedures will also extend to cover OEH staff and any other third parties which may be impacted by the licensee's works. OEH accepts no responsibility for any event which results in the licensee suffering any loss. The licensee will be held liable for any damages resulting from their works which have impacted on OEH staff or any other third party.

Attachment A

Other parties

In addition to the principal licensee identified above, the following parties are also authorised under this licence:

Title	Name
Dr	Philip Gibbons

Attachment B

Licence Class

Class Name	Class Start Date
Research	18/04/2019

Focus of work

This project authorises the licensee to Harm, Pick, collect or otherwise interact with the following species, communities or materials as described on this licence in the listed quantities:

Species Type	Family	Genus	Species	Species Code	Common Name
FL	ALL FLORA				ALL FLORA
FA	Cervidae	Axis	axis	1524	CHITAL
FA	Cervidae	Dama	dama	1523	FALLOW DEER
FA	Cervidae	Cervus	porcinus	1525	HOG DEER
FA	Cervidae	Cervus	elaphus	1526	RED DEER
FA	Cervidae	Cervus	timorensis	1528	RUSA DEER
FA	Cervidae	Cervus	unicolor	1527	SAMBAR
TEC					Littoral Rainforest
TEC					River-Flat Eucalypt Forests
TEC					Swamp Oak Floodplain Forest
TEC					Swamp Sclerophyll Forest
TEC					Bangalay sand Forest

Attachment C

Project location

This project is authorised in the following areas:

NPWS Estate

Tenure Type	Branch	Region	Area	Park
NPWS Estate	Coastal	South Coast	Nowra	Conjola National Park
NPWS Estate	Coastal	South Coast	Ulladulla	Budawang National Park
NPWS Estate	Coastal	South Coast	Ulladulla	Bimberamala National Park
NPWS Estate	Coastal	South Coast	Ulladulla	Meroo National Park
NPWS Estate	Coastal	South Coast	Ulladulla	Murramarang National Park
NPWS Estate	Coastal	Far South Coast	North	Monga National Park
NPWS Estate	Coastal	Far South Coast	North	Deua National Park
NPWS Estate	Coastal	Far South Coast	North	Eurobodalla National Park
NPWS Estate	Coastal	Far South Coast	Central	Kooraban National Park
NPWS Estate	Coastal	Far South Coast	Central	Gulaga National Park
NPWS Estate	Coastal	Far South Coast	Central	Biamanga National Park
NPWS Estate	Coastal	Far South Coast	Merimbula	Bournda National Park
NPWS Estate	Coastal	Far South Coast	Merimbula	Ben Boyd National Park
NPWS Estate	Coastal	Far South Coast	Merimbula	South East Forest National Park
NPWS Estate	Coastal	Far South Coast	Merimbula	Mount Imlay National Park

Other

Tenure Type	State Forests	LLS Region	LGA	Lot Sec DP	Other Location
Other					Non-NPWS estate with land manager approval