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M. H. M. Nilar

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Effects of an alien vertebrate herbivore on the regeneration of endangered vegetation communities: Example of feral rusa deer impacts on the restoration of subtropical rainforest on an abandoned mine site in the Illawarra region.

Abstract

Anthropogenic land clearance is responsible for reductions in the extent of native vegetation and losses in biodiversity, however restoration ecology could reverse this damage. The Illawarra Subtropical Rainforest is an endangered ecological community currently undergoing restoration, but is threatened by alien plant species and an introduced herbivore *Cervus timorensis*.

This honours project examined the effects of vertebrate herbivores on the establishment of re-vegetated tree and shrub seedlings, and the composition of regenerating Illawarra Subtropical Rainforest. The study was performed on the deforested slopes of an abandoned coal mine on the foothills of the Illawarra Escarpment, whereby local land managers and restoration practitioners used a combination of alien plant control, followed by planting of nursery-grown native tree and shrub seedlings and herbivore exclusion with fenced plots to encourage natural regeneration.

Five key questions were addressed by this honours project: (1) how widespread is browsing damage to revegetated shrub and tree seedlings, and does such damage limit plant establishment? (2) Is there a difference in effectiveness of different guard types (fenced plots versus individual wire or corflute guards) at protecting seedlings from browsing damage? (3) Do plant species differ in their susceptibility to attack by vertebrate browsers? (4) What are the patterns of vertebrate herbivore diversity and activity across the revegetated habitat versus established patches of forest?

Plant responses to browsing were assessed by measuring proportion of stems and apical shoots that were damaged by herbivores for 900 nursery-grown and revegetated rainforest shrub and tree seedlings (54 species) in four different protective treatments: fenced plots, individual wire and corflute guards and unprotected control plants. Vegetation community responses to herbivores were assessed by comparing composition and vegetation structure between fenced and adjacent unfenced control plots. Vertebrate identity and activity were assessed with camera traps deployed across three habitats: deforested grassy areas, deforested areas that had been revegetated and establish patches of mature rainforest.

Unprotected seedlings generally experienced significantly higher (more than 40%) vertebrate herbivore damage, but varied among species and protective guard type. Corflute guards were less effective than individual wire guards or fenced plots at protecting seedlings from browsing damage. Fenced plots were most effective at protective seedlings from damage but also had the benefit of enhancing the diversity and abundance of non-planted species and the foliage cover of shrubs and ground cover vegetation compared with unfenced control plots. Unexpectedly, camera traps revealed that swamp wallabies (*W. bicolor*) were 2 times more frequently detected than rusa deer, although deer tended to spend 6 times longer foraging than swamp wallabies when detected. Occupancy and activity of both vertebrate species did not vary between different habitat types, although there was a trend toward higher activity of swamp wallabies in the cleared revegetated area.

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Degree Name

BEnviSci Hons
Effects of an alien vertebrate herbivore on the regeneration of endangered vegetation communities:

*Example of feral rusa deer impacts on the restoration of subtropical rainforest on an abandoned mine site in the Illawarra region.*

Mohamed Hussain Mudassar Nilar

A thesis submitted in part fulfillment of the requirements of the Bachelor of Advanced Environmental Science (Honours) in the School of Earth and Environmental Sciences, Faculty of Science, Medicine and Health, University of Wollongong 2017.

24th October 2017
The information in this thesis is entirely the result of investigations conducted by the author, unless otherwise acknowledged, and has not been submitted in part, or otherwise, for any other degree or qualification.

M. Hussain M. Nilar

24th October 2017
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1. Introduction

1.1 Anthropogenic landscape change and impacts on native vegetation communities.

As the human population continues to rise, the amount of land use for agriculture or urbanisation is ever growing. Of the 13,300 Million hectares (Mha) of total ice-free land area, only 4000Mha is suitable for rain-fed agriculture. Of that land, the most suitable 445Mha of land is predicted to have considerable environmental and social costs if it is converted for agricultural use, due to its incredibly rich biodiversity (Lambin and Meyfroidt 2011). Given the finite amount of land which all organisms depend on, humanity is challenged with trying to meet its own needs for land, while also trying not to compromise the loss of biodiversity through the clearing of native vegetation (Pongratz, Reick et al. 2008). Between 1700 and 1990, there was a rapid increase in the amount of agricultural land from 265 million ha to 1471 million hectares, as well as a six-fold increase in the amount of pastoral land (Goldewijk 2001) which coincided with a net decrease in the cover of all types of natural vegetation. When determining future global land use trends, models predict similar patterns. The cover of agricultural land will increase by up to 25% (Schmitz, van Meijl et al. 2014) while valuable native vegetation cover will decrease (Tilman, Fargione et al. 2001).

The clearing of native vegetation cover to make land available for human development and growth is responsible for considerable losses in biodiversity (Tilman, Fargione et al. 2001, Foley, DeFries et al. 2005, Newbold, Hudson et al. 2015). The clearing of native vegetation and habitats can influence biodiversity particularly through reducing the extent of suitable habitat for organisms (Haddad, Brudvig et al. 2015), altering the vegetation communities upon which species depend for habitat and food resources, and by increasing the susceptibility of native habitats to invasion by invasive species (Larson, Anderson et al. 2001). For example, Aronson, La Sorte et al. (2014) found that there were decreases in native bird and plant diversity in urban areas as a result of vegetation disturbance. Similarly Alroy (2017) also demonstrated that disturbed rainforest (cleared for plantations) habitats support significantly lower numbers of native plant and animal species, while facilitating alien species invasion.

Fragmentation of natural vegetation is also a resultant of land clearance, and threatens biodiversity by reducing continuous tracts of natural habitat to smaller patches of discontinuous habitat, which makes movement for animals particularly challenging and risky (Wilson, Chen et al. 2016). Fragmentation also leaves these smaller patches of habitat susceptible to invasion by alien species as a smaller area mass has relatively greater amounts
of disturbed edges that favour invasion (Honnay, Verheyen et al. 2002, Wilson, Chen et al. 2016)

Within Australia, similar patterns of landscape change have been occurring. In a review by Bradshaw (2012), Australia experiences some of the highest rates of land clearing despite only recently being colonised by European settlers. During early European settlement, grasslands occupied just seven percent of the nation, however by 1995 occupied more than 16% by total area due to deforestation of woodlands and forests for grazing and cropping (Barson, Bordas et al. 2000). Deforestation of Australia’s closed and open forests for human development has caused Australia to have the lowest amount of remaining closed forest of any other country at less than 4.6% (Sinclair, Jermyn et al. 1993).

Given that habitat loss and fragmentation are a major threat to the loss of biodiversity, there is a need to combat reductions in natural vegetation cover, if biodiversity losses are to be negated. Together with land clearance, other anthropogenic driven forces including alien plant invasion and introduced vertebrate herbivores are often associated with the reduction in biodiversity and therefore should be considered if habitat modifications and biodiversity losses are to be managed (Woodroffe and Ginsberg 1998, McKinney 2002, Cliquet, Backes et al. 2009).

1.2 Restoration ecology

Restoration ecology is proposed as a possible solution to reverse the damage to endangered communities as it aims to return a degraded or deforested ecosystem to a form that resembles a reference ecosystem in terms of number and identity of species (i.e composition) and function (Hobbs and Norton 1996). More recently, the Society for Ecological Restoration defined restoration as “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed” (SER, 2004) (http://www.ser.org/).

Hilderbrand, Watts et al. (2005) recognised that there are four assumptions for most restoration projects, which can lead to the projects demise if overlooked. These include that, (i) complete restoration is possible and that there is a clearly defined end-point, (ii) the provision of the physical structure alone for an ecosystem will facilitate the entry and establishment of crucial biotic interactions, (iii) that selective planting of given species can help accelerate the restoration of an entire community, and (iv) that a restoration procedure that proved successful in one situation will always be applicable in future situations and sites.
However, restoration projects are extremely diverse, facing unique challenges, requiring specific protocols, and occur over varying scales. For example, the extent of restoration of native communities varies considerably from localised highly degraded forest edges (Zhuang 1997) or abandoned farmlands (Aguirre, Palomeque et al. 2011), through to the restoration of entire landscapes or forest ecosystems (Lamb 1998, Rodrigues, Gandolfi et al. 2011).

“Restoration” is often used interchangeably with other terms such as rehabilitation, relocation, reconstruction or revegetation. Given this complexity in strictly defining restoration, Hobbs and Norton (1996) argue it most suitable to recognise that restoration is broadly performed for (a) the treatment of altered physical and chemical properties of soil, (b) to improve the condition of the environment to increase productivity, or (c) for purposes of increasing biological diversity (Hobbs and Norton 1996).

The restoration procedures used will depend upon the extent of disturbance to the site in relation to the desired natural state. For example, degraded ecosystems that still retain remnant vegetation and an intact soil profile and seed bank may simply require the removal of the disturbance process to enable ecosystem recovery to a desired ‘natural’ state (Morimoto, Shibata et al. , Prach, Řehounková et al. 2014). In extremely disturbed ecosystems, such as abandoned mines in which the soil and other physical properties have been degraded or often completely removed, reconstructive approaches may be required to restore the ecosystem once the original disturbance has ceased (Miao and Marrs 2000, Wong 2003, Macdonald, Landhäusser et al. 2015).

Restoration that requires soil amelioration is commonplace following the completion of mining activities, landfill, intensive agriculture or other extractive industries. In these circumstances multiple stages of restoration are required such as improving soil microbe activity, soil water retention, soil biomass, soil structure and porosity, as well as the soil seed bank, all prior to revegetation activities (Hobbs and Norton 1996, Miao and Marrs 2000, Turner, Lambin et al. 2007). While some restoration projects are focused on the simple stabilisation of pollutants in the soil through the establishment of a vegetation cover (Simmons 1999, Wong 2003), others such as that by Koch (2007) are far more comprehensive and thorough. Koch (2007) guided the restoration process of a Bauxite Ore Mine in Western Australia which detailed a series of steps including soil levelling and cultivation to improve porosity, the return of the soil seedbank from a neighbouring forest ecosystem, followed by the careful selection and planting of specific endemic plant species. In a step further, Juwarkar and Jambhulkar (2008) demonstrated that the addition of manure to heavy metal contaminated soil that occurred on a fly ash dump in India significantly
increased the biological activity, nutrient content, and soil water holding potential of the soil which enhanced the establishment and growth of planted vegetation. These studies demonstrate that while the planting of key plant species is pivotal to the restoration of exploited or contaminated landscapes, further insight into the biological activity and condition of the soil should be considered prior to any revegetation efforts performed to maximise the effectiveness of the restoration process.

The most understood purpose of restoration ecology is to improve biological values of disturbed landscapes. These biological values which are being preserved or improved include the abundance, diversity and distribution of native vegetation, the natural vegetative structure of the target community, ecosystem function, habitat amenity to fauna and ecosystem resilience to future disturbance (Hobbs and Norton 1996). While previous forms of restoration ecology restore landscapes from a particular use (i.e mining, farming), this form of restoration expands the cover of native vegetation purely maintaining or improving values of biological diversity. These qualities typically include, but are not limited to; species composition, vegetative structure, basic ecological processes, and the resilience of species (Hobbs and Norton 1996).

The process of restoration for improving the extent of natural vegetation communities requires two major steps. First, the threat which was responsible for degrading the site initially needs to be managed, or removed (i.e mining, agriculture, noxious weeds) (Miao and Marrs 2000, Reid, Morin et al. 2009, Aguirre, Palomeque et al. 2011). Afterwards, native vegetation must be established on site either passively or actively (Sun, Dickinson et al. 1995, Wehr, Fulton et al. 2006, Beltran, Kreidler et al. 2014). Passive restoration relies on the natural establishment and succession of plant communities following the removal of the disturbance, which is unlike active restoration in which plants or seeds are sown (Morrison and Lindell 2011). While passive restoration is cheaper in terms of labour and resource costs (Morrison and Lindell 2011), it is not always the most effective. Passive restoration has a lower success rate in recruiting native species compared to actively managed areas (Morrison and Lindell 2011, Ruwanza, Gaertner et al. 2013), and is also more susceptible to further degradation through secondary invasion of weeds (Claeson and Bisson 2013). Further, passive restoration assumes the presence of a viable soil seedbank, which at times may not be present (Morrison and Lindell 2011) and does not favour all species of native plant equally as some are simply better dispersers and colonizers than others which can lead to an undesirable vegetation community (Claeson and Bisson 2013, Meli, Martínez-Ramos et al. 2013). In contrast, active restoration is comparatively more expensive, however is more effective and
even economical. This is because, passively restored sites are largely unmanaged beyond removing the initial environmental stressor, which can allow unforeseen challenges (emergence of secondary weeds) to emerge which can prolong the project’s completion, or add other costs associated with managing new threats (Zahawi, Reid et al. 2014). Therefore, it is urged that the type of restoration (Active or passive) is carefully considered, taking into account the type of threats, location, the degree of degradation, the target community composition, anticipated time of completion, and resources available for management (Holl and Aide 2011).

1.3 Challenges of restoration in forest ecosystems to improve biodiversity

Restoration projects may still be threatened by the disturbances that were originally responsible for their degradation. It is crucial that the threats that face restoration projects are identified and appropriate measures taken to protect the restoration site from these threats indefinitely or until they have subsided. The major threat to the restoration of Hong Kong’s temperate forests were wildfires, (Zhuang 1997), while Zhang and Dong (2010) recognised land clearance as the biggest threat to the restoration of China’s Loss Plateau Forest, such that the removal of both threats was pivotal to the success of both restoration projects.

In some circumstances, threats to restoration cannot be removed from the site and therefore must be managed appropriately. In the case for heavy metal contaminated soils, certain types of soil microbes must be present to allow plants to establish on the restoration site (Khan 2005). Similarly, herbivory is a commonly reported phenomenon among restoration sites within south eastern Australia, and can only be managed using fences or guards to exclude herbivores (Kasel 2008, Burger, Reich et al. 2010, Florentine, Graz et al. 2013, Wevill and Florentine 2014). Likewise, invasive plant species often monopolise on the available space, light, and nutrients on restoration sites (Sun, Dickinson et al. 1995). They require constant suppressing through manual weeding, mowing, or spraying of herbicide to ensure that they do not dominate the restoration site and suppress native plant species (Wilkins, Keith et al. 2003, Cole and Lunt 2005).

During revegetation events, plants are often planted either through broadcast seeding, or by nursery grown seedlings and each method is faced with its own challenges (Doust, Erskine et al. 2008, Florentine, Graz et al. 2013). While broadcast sowing of seeds is more economical (time and money saving) (Zhuang 1997), this technique is often disadvantaged as
it often suffers from reduced rates of germination particularly for smaller seeded species (Doust, Erskine et al. 2008), has a slower rate of vegetation establishment, and a high seedling mortality (Sun, Dickinson et al. 1995, de Jong 2000). In comparison, planting of tube stock is generally more expensive and requires intensive management (watering, fertilising etc), however allows plants to grow faster and therefore achieve the targets of the restoration site of “increasing native vegetation cover” more rapidly (de Jong 2000, Wilkins, Keith et al. 2003).

Although restoration sites are faced with their unique set of challenges, they are all united in their success by time. Time of restoration strongly determines the success of a restoration site. Wilkins, Keith et al. (2003) recognised that a period of even 10 years of protection from herbivory was unable to detect significant differences in native plant diversity, despite actively sowing seeds and planting seedlings. Likewise Wevill and Florentine (2014) demonstrated by monitoring the community responses of restoration sites of varying ages, that detecting significant improvements in restoration sites was not possible for sites younger than 8 years, but was for sites aged over 12 years. Therefore, sufficient amounts of time are required to ensure the success of a restoration project.

1.3.1 The threat of weeds to restoration success

Invasive plant species pose a significant threat to restoration, before, during or after restoration activities. Alien plant species can threaten restoration by invading a restoration site during the period of revegetation and monitoring, during which weed supressing is often required (Sun, Dickinson et al. 1995, Miao and Marrs 2000, Wehr, Fulton et al. 2006). Weed removal is often the first stage of restoration and must occur before revegetation (Anderson, Dhillion et al. 1996, Holmes 2001, Erskine Ogden and Rejmánek 2005, Reid, Morin et al. 2009). Invasive species can smother native plants, prevent the establishment and recovery of native plant communities by inhibiting their growth and fitness, which ultimately leads to a decline in native species diversity (Erskine Ogden and Rejmánek 2005, Sharma and Raghubanshi 2007, Gooden, French et al. 2009, Vilà, Espinar et al. 2011). Given the complication of invasive plants on restoration efforts, it is crucial to develop a sound understanding of the role of alien plant species in restoration including their inherent qualities, impacts, and management protocols.
Invasive plants from a variety of structural forms exhibit varying levels of competition with natives for light. Invasive herbs, trees or climbers are particularly good at covering native plants, and are responsible for significantly more damage to native plant communities than alien plants of other structural forms (Gaertner, Breeyen et al. 2009). Smothering typically targets established plants and seeds. *Tradescantia fluminensis* for example, is a weed of New Zealand that forms dense mats on the forest floor which inhibits the germination of seeds and even reduces their survival into adulthood by affecting the amount of light seedlings acquire (Standish, Robertson et al. 2001). The impact that smothering has on native seedlings and the seedbank is exemplified in studies where the return of native species from seeds dispersed or stored in the seed bank, was possible only through the removal of threatening weed species (Holmes 2001, Erskine Ogden and Rejmánek 2005, Flory and Clay 2009, Flory and Clay 2010). These studies help isolate and emphasise the role that smothering by invasive plants plays in hindering germination and growth of native species particularly in restoration sites which rely on passive restoration (Claeson and Bisson 2013, Beltran, Kreidler et al. 2014).

Invasive species can also impede restoration activities through allelopathic interactions. Allelopathy is a process where plants produce and release biomolecules into the soil around them to induce suffering in, or to benefit a neighbouring plant by influencing their growth and development (Rizvi, Haque et al. 1992). Several species of plants produce specific biomolecules, and each has varying effects on their neighbouring plants. For example, *Centaurea diffusa*, *Lolium arundinaceam* and *Elaeagnus umbellate* are all noxious weeds of North America which significantly reduce the germination and growth rates of specific native tree species by leaching biomolecules into the soil (Callaway and Aschehoug 2000, Orr, Rudgers et al. 2005). While allelopathic chemicals from invasive plants can directly hinder the growth of natives used in restoration, they can also interfere with the symbiotic relationships between native plants and soil microbiota, which are necessary for the normal growth of these native plant species (Jordan, Larson et al. 2008).

In some contexts, the removal of weeds alone may initiate the passive regeneration of native vegetation (Claeson and Bisson 2013), however secondary invasion by alien plants is commonplace and which can inhibit native vegetation establishment (Erskine Ogden and Rejmánek 2005, Reid, Morin et al. 2009). While sowing a diversity of native seeds, or planting numerous seedlings in mass has been successful in restoring native vegetation at highly degraded sites where the weed has eliminated native remnant vegetation (i.e tall grass
prairies, and other grassland communities) (Blumenthal, Jordan et al. 2003, van Ruijven, De Deyn et al. 2003), it is not always successful or sufficient (Reid, Morin et al. 2009). The solution to overcoming invasive species in restoration involves using a series of steps and techniques. The land must be prepared through manual weeding techniques, and soil amelioration to help remove the alien species and improve soil for the native species (Holmes 2001, Flory and Clay 2009). Second, it is crucial to plant a diverse range of native plants as they can competitively suppress secondary weed establishment (van Ruijven, De Deyn et al. 2003, Pokorny, Sheley et al. 2005). Lastly, ensure sustained weeding of the restoration site to prevent invasive species from dominating the community (Heleno, Lacerda et al. 2010). Given that these procedures are maintained, invasive species threats can be managed during restoration activities.

1.3.2 Animal herbivory on restoration sites

There is evidence that herbivores can facilitate vegetation diversity by diminishing competitive dominance among resident plant species or facilitating the dispersal of seeds in restoration sites (Huntly 1991, Olff and Ritchie 1998, Mouissie 2004). However, such benefits vary with the location, species of plants and species of herbivores including their size, abundance and endemicity (Forsyth 2011, Takatsuki 2009). Herbivores can cause considerable damage to revegetation efforts by browsing on plants, trampling on new regrowth, or causing other physical damage (Ret and Kathleen L. Shea 1998). The damage herbivores inflict on a revegetation site is often drastically increased due to the young age of plants, and the fragility of the restoration site.

Alien vertebrate herbivores can exacerbate the impacts to vegetation because of their ecological novelty and also because the native plant species may not have co-evolved specific defences to resist impacts (Bilney 2013, Davis et al 2016). The major mechanisms by which herbivores inflict damage onto restoration sites is through the direct feeding on the plants (Ret and Kathleen L. Shea 1998). Herbivores can also threaten restoration through trampling (Crete, Ouellet et al. 2001, Mitchell, Rose et al. 2008), or being facilitators of invasion by invasive plants by dispersing seeds through their dung (Shiponeni and Milton 2006). Animals such as deer for example can introduce an additional pressure on restoration in the form of bark stripping. By rubbing their antlers on tree trunks, they can damage the plant enough to cause death (Yokoyama, Maeji et al. 2001).
Where the herbivore is non-native, an ideal solution would involve the total removal of the exotic species (Donlan, Tershy et al. 2002, Donlan, Croll et al. 2003, Beltran, Kreidler et al. 2014), however, this method is both exhaustive and expensive (Carrion, Donlan et al. 2011). Therefore, to reduce the pressure of herbivory on restoration, guards for individual plants or exclusion fences around plots of regenerating vegetation are often used (Ret and Kathleen L. Shea 1998, Sweeney, Czapka et al. 2002, Lai and Wong 2005). These herbivore deterrence’s have been used for a variety of native fauna including Sika deer (Cervus nippon) in Hong Kong (Lai and Wong 2005), macropods in eastern Australia (de Jong 2000) or snow geese (Chen caerulescens) in Texas U.S.A. Likewise, protective guards in restoration sites have been used to deter herbivores which are non-native, including deer in Hawaii (Cabin, Weller et al. 2000, Thaxton, Cole et al. 2010) and New Zealand (Stewart and Burrows 1989, Bellingham and Allan 2003, Coomes, Allen et al. 2003, Husheer, Coomes et al. 2003) or rabbits in coastal California (Donlan, Croll et al. 2003).

The effects of native herbivores on restoration have been exemplified through exclusion fences. Within NSW Australia, exclusion plots to deter native macropods, rabbits and livestock from native plant regeneration in agricultural land in NSW Australia, were found to produce plants which were significantly taller and which contained a greater number of plant recruits in comparison to areas in which native and exotic herbivores were not excluded (Spooner, Lunt et al. 2002, Allcock and Hik 2004). These immediate effects on restoration were also found in the restoration of riparian corridors of California, in which herbivory by native deer produced shorter plants, and also reduced their population density in control versus herbivore-exclusion plots (Opperman and Merenlender 2000). Herbivory by natives can also dictate the direction and endpoint of restoration activities. Lesser snow geese (Chen caerulescens) are native to the coastal marshlands of Texas U.S.A, however intense grazing by them reduced foliar cover until a mud flat was produced as the end community (Miller, Smeins et al. 1997).

Where introduced ungulates and other herbivores are excluded, there is also a general increase in both species diversity and or density of seedlings and saplings recorded, in contrast to areas in which they were not excluded (Stewart and Burrows 1989, Bellingham and Allan 2003, Husheer, Coomes et al. 2003, Casabon and Pothier 2007, Thaxton, Cole et al. 2010). Similarly, browsing by introduced herbivores consistently produced shorter plants, compared to areas where they were excluded (Zamora, Gómez et al. 2001, Palmer and Truscott 2003). While Crete, Ouellet et al. (2001) recognised that trampling of lichens by
caribou often lead to their demise, studies exploring the impact of physical activity of ungulates on the percentage cover of bare soil were vastly different. Stewart and Burrows (1989) demonstrated that the presence of deer typically lead to an increase in the amount of bare soil while Austin, Unness et al. (1994) found that deer presence often decreased the amount of bare soil by simultaneously increasing the amount of herbs present.

The presence of herbivores (native or alien) generally leads to declines in fitness of plants in restoration through browsing, or through the dispersal of alien plant species (Shiponeni and Milton 2006). It is therefore urged that guards are used to deter browsing, and that these guards be tailored to the specific species of herbivore, and the extent of the restoration activity (Ret and Kathleen L. Shea 1998, Sweeney, Czapka et al. 2002, Griscom, Griscom et al. 2011). Other studies have used individual mesh guards to protect plants from animal herbivory (Baer 1980, Montague 1993, Lai and Wong 2005), however none of my understanding have evaluated the varying levels of browse damage among different types of individual guards, to a fenced plot.

1.4. An introduction to Cervidae and their behaviour

Cervidae are amongst the most diverse group of ungulates and include all species of deer, elk, moose, reindeer and caribou. Although they are native to almost all continents, they have been deliberately introduced to alien environments including Australia and New Zealand, as well as remote islands like Hawaii (Stewart and Burrows 1989, Moriarty 2004, Davis, Coulson et al. 2008, Thaxton, Cole et al. 2010, Davis, Bennett et al. 2016). Cervidae exhibit a variety of defining morphological and behavioural characteristics which allows them to pose a significant threat to novel environments. Apart from being browsers that feed on several layers of the community and repeatedly follow common pathways, they strip bark using their antlers or teeth (Martin, Pine et al. 2011, Vaughan, Ryan et al. 2013). These behavioural qualities of cervidae allow them to be highly influential members of ecosystems particularly at high densities. Within their native range, they are responsible for altering ecological communities, species distributions and abundances, or even entire landscapes (McShea and Rappole 1992, Fortin, Beyer et al. 2005, Suzuki, Miyashita et al. 2008).

Although deer are generally nocturnal in their behaviour, being more active at dawn and at dusk (Beier and McCullough 1990), they also vary their activity spatially in
accordance to habitat type, the condition of the habitat including soil fertility, as well as their breeding status.

The type of vegetation community that deer reside in can influence their relative activity. There is emerging evidence that deer often select areas that have the greatest soil fertility, such as alluvial soils (Vospernik 2006). This finding has been supported by the significantly higher density of deer and incidence of damage to these vegetation communities that often occur on this given soil type (Vospernik 2006, Gill and Morgan 2010).

Habitats can vary considerably over space and deer often select where they spend their time. Although there is a general consensus that deer including rusa deer (Cervus timorensis) in NSW preferentially occupy open habitats such as grasslands and forest edges due to their greater abundance of food (Blymyer and Mosby 1977, Moriarty 2004, Wirth, Meyer et al. 2008), there is still some contradiction (Casabon and Pothier 2007). Generally, the preference for open area is restricted to dawn, dusk and night after which deer retreat to more closed habitats like forests (Beier and McCullough 1990). C.timorensis which were deliberately released into both NSW Australia and the sclerophyll forests of New Caledonia also demonstrate similar habitat preferences that can vary with season. C.timorensis within New Caledonia had a narrow home range of just 1042 ± 149m, and preferentially used the wet sclerophyll forests and floodplains during the driers season (Spaggiari and de Garine-Wichatitsky 2006). Likewise, C.timorensis in the Royal National Park NSW exhibited a range of between 1-8km² (Moriarty 2004), with the greatest range occurring in winter which was often attributed to a shortage of suitable forage (Beier and McCullough 1990). Despite the different areas geographically that these deer reside in, they seem to exhibit similar home ranges, preferences for habitat types, and appeared to be active in specific periods of the day.

Deer can vary their activity due to differences in their gender. Males often have a wider home range than does, however are less inclined to venture into open areas like forest edges (Beier and McCullough 1990, Moriarty 2004, Spaggiari and de Garine-Wichatitsky 2006). In contrast females often spend a greater amount of time at forest edges, and grasslands (Moriarty 2004), such that the majority of their diet can at times be comprised entirely of grass (Roberts 2012). Further, females and males can vary their activity in response to the rut and the oestrus cycle. While males typically increase their home range during the rut season to maximise their chances of finding suitable mates, female deer showed vastly different responses in their activity. In two contrasting studies, female White-
tailed deer (*Odocoileus virginianus*) were found to both increase and decrease their range significantly in the build-up to the rut season and also in response to the stage of ovulation that they were undergoing (Ozoga and Verme 1975, Holzenbein and Schwede 1989).

Given the current findings a few conclusions can be drawn regarding the activity of deer overall. Generally, it is likely that deer activity will typically increase during the winter months, which coincides with a decrease in plant growth and therefore a shortage in suitable forage. Similarly, deer will be more active during dusk, dawn and night to avoid possible predation during the day. Further, deer may exhibit different levels of activity and site fidelity depending on their gender and breeding status with males typically having a larger home range. However, deer may demonstrate the greatest amount of activity at forest edges and clear cuts given the abundance of food often provided by grass.

1.4.1 Invasive deer influence community assemblages and biotic interactions

Although deer are browsers, feeding on leaves, soft shoots, or fruits of woody plants such as shrubs and trees, they will occasionally graze if given the opportunity. Deer consume a variety of plants residing at different levels of the vegetation structure including herbs, trees, shrubs, climbers, ferns, orchids, grasses, rushes and sedges (Takatsuki 2009, Barrios-garcia, Relva et al. 2012). By browsing in different levels of the vegetation structure, introduced or overabundant deer can impose several negative impacts on plants residing within their browse height, or to the community at large. Two major ways in which deer impose negative impacts on forest communities is through the direct consumption of the plant, or through the process of ringbarking where they rub their antlers on tree trunks (Groves and Grubb 2011).

As deer exhibit a varied diet, browsing is often responsible for the decrease in total foliar cover of a variety of vascular plants that reside within the low and intermediate vegetation heights (Moriarty 2004, Bennett 2008, Relva, Nuñez et al. 2010, Crowther, Ortac et al. 2016). Deer preferentially feed on seedlings and small recruits due to their higher palatability and ease of access, which can alter the composition of plant species that occupy understory vegetation communities (Dolman and Wäber 2008). Typically, there is an increase in the abundance of unpalatable species, and a decline in the abundance of palatable species. There is also a reduction in the seedbank of native woody species which reduce ecosystem resilience and thus reduces regeneration post disturbances (DiTommaso, Morris et al. 2014).
There are also declines in the cover of understory woody plant species, and an increase in herbs as well as graminoids (specifically turf forming) and ferns which are typically more unpalatable and disturbance adapted (Nugent 1990, Smale, Hall et al. 1995, Donlan, Tershy et al. 2002, Husheer, Coomes et al. 2003, Rooney and Waller 2003, Rooney 2009).

Ring barking often leaves trees with deep lacerations. This impact is amplified by the tendency for deer to repeatedly use trees already ring barked (Welch, Staines et al. 1987), coupled together with the generally slow rate at which trees are capable of healing (Akashi and Nakashizuka 1999). Given that trees which have coevolved with the presence of deer have suffered considerable decline due to overabundant deer (Verheyden, Ballon et al. 2006), the impact invasive deer have on novel forest communities may be expected to be higher. Deer are also highly selective in their choice of plants to ring bark, and is well documented among deer within their native ranges; Sika deer (*C. nippon*) in Japan (Yokoyama, Maeji et al. 2001, Ando, Yokota et al. 2003, Takatsuki 2009), Red deer (*C. elaphus*) in Europe (Welch, Staines et al. 1987, Gill and Beardall 2001, Vospernik 2006), White tailed deer (*O. virginianus*) in North America (Rooney and Waller 2003, Rooney 2009); as well as introduced deer in New Zealand, Patagonia and Australia (Wardle, Barker et al. 2001, Moriarty 2004, Bailey, Gauli et al. 2014, Crowther, Ortac et al. 2016). In a dramatic example, ringbarking caused by overabundant Sika deer (*C. nippon*) in Japan, saw the removal of overstory species, which allowed the entry and domination of baby bamboo *Sada nipponic* (Yokoyama, Maeji et al. 2001).

Deer strip bark from trees both as a source of food and also to remove felt from their antlers. While bark stripping (as a food source) is believed to be highest in winter due to a decline in palatable plant species (Ueda, Takatsuki et al. 2002, Jiang, Ueda et al. 2005, Verheyden, Ballon et al. 2006), others report bark stripping to be highest in summer and spring (Ando, Yokota et al. 2003) indicating that bark stripping does not arise as a result of food shortage. Although there is little correlation between the nutritive content of bark and the probability of bark stripping (Ando, Yokota et al. 2003, Ando, Yokota et al. 2004, Jiang, Ueda et al. 2005), the preference for bark stripping could be attributed to the ease of bark removal, with softer barked trees being preferred (Saint-Andrieux, Bonenfant et al. 2009).

Deer are also facilitate alien plant invasion (Malo and Suárez 1998, Eyles 2002, Bartuszevige and Endress 2008, Knight, Dunn et al. 2009, Davis, Forsyth et al. 2010) by reducing the overall distribution and fitness of native plant through preferential browsing.
(Keith and Pellow 2005, Williams, Ward et al. 2008), or by avoidance of alien species due to their lower palatability and or higher resistance to browsing (Lefcort and Pettoello 2012). Deer can also facilitate invasion through their dung which can carry a high diversity invasive species, which retain a high rate of germination (Malo and Suárez 1998, Iravani, Schütz et al. 2011) as was found for Himalayan honeysuckle (*Leycesteria Formosa*) in Victoria Australia (Eyles 2002). The process deer play in the spread and proliferation of invasive species therefore involves the cooperation of numerous factors including the deer’s natural range, their diet preferences for native or alien species, and the viability of these seeds after being defecated.

Introduced deer can also exert cascading effects on native animals both by competition for resources and indirectly by habitat modification. Deer introduced into the Haida Gwaii archipelago for example, indirectly led to the decline in total diversity of understory bird species by significantly reducing the cover of understory vegetation community through browsing, trampling, bark stripping and thrashing (Chollet, Bergman et al. 2015). Similarly, in the Southern Andes, introduced Red deer (*C. elaphus*) have reduced invertebrate diversity by physically trampling suitable microhabitats, as well as by reducing the available blooms of a herbaceous plant *Alstroemeria aurea*, upon which many insect pollinators depend on. Within Australia, introduced deer consume a highly varied diet that is unlike most native herbivores such as kangaroos and wombats. In contrast, they tend share a diet that is as varied and similar in composition to swamp wallabies (*Wallabia bicolor*) due to their tendency to mostly occupy woodlands (Hamilton 1982, Duncan 1992, Davis, Coulson et al. 2008, Pedersen, Andreassen et al. 2014). Although most studies could recognise the presence of dietary overlap between invasive deer and native mammals in eastern Australia, none were able to clearly demonstrate competitive exclusion (Forsyth and Davis 2011, Pedersen, Andreassen et al. 2014). Therefore, given this lack of knowledge, it is difficult to assume that simple dietary overlap implies competitive exclusion.

1.5 Deer in the Illawarra region

Six species of deer were deliberately introduced to Australia throughout the 19th century and include fallow deer (*Dama dama*), red deer (*C. elaphus*), sambar deer (*Rusa unicolor*), chital deer (*Axis axis*), *C.timorensis* and hog deer (*Axis porcinus*). All species of deer were introduced using acclimatisation societies to enhance the aesthetic of the environment, were escapees from deer farms (Bentley 1978), or were a product of
translocation events which saw deer being placed into state forests, national parks and catchment lands for hunting purposes (McKnight 1976). Although several species of deer reside within the Illawarra State Conservation Area including *D. dama*, *C. elaphus*, and *C. unicolor*, the majority are *C. timorensis* (Dawson 2012) (Figure 1). *C. timorensis* are a medium size deer growing to 1.1m at the shoulder and are native to the Indonesian Archipelago. Female *C. timorensis* are reproductive throughout the year from 1.5 years of age. They are primarily grazers, and do not require abundant sources of water for their survival (Geist V 1998).

![Figure 1](image.png)

**Figure 1.** Current distribution and abundance of *C. timorensis* in Australia (Moriarty 2004). Inset shows the predicted bioaclimatic distributions of *C. timorensis* (M. Bomford, unpublished data).

In addition to the biodiversity hot spot in Perth, the Illawarra forms part of the second biological hot spot extending along the east coast of Australia (Mittermeier, Turner et al. 2011). The Illawarra’s unique vegetative communities are under constant stress from land clearance, agriculture, as well as invasive plant species. It is also alarming that just 45% of the Illawarra regions vegetation currently exists, with the least disturbance occurring along the escarpments steep foothills (NPWS 2002).

Given the spread and novel effects of *C. timorensis* in the Illawarra, coupled with the management they receive from the Games and Feral Animal Control Act (NSW) 2001, it becomes increasingly urgent to evaluate the possible damage to vegetation in the Illawarra region, particularly on its unique and threatened ecological communities.
1.6 Significance of study and aims

Restoration projects are currently being undertaken to expand the extent of the Illawarra’s endangered rainforest communities in areas previously cleared of mining activity and invaded by alien plants. This initiative involves the participation of landowners, volunteers, local government organisations, and State Government agencies. Restoration is mainly in the form of alien plant control followed by extensive revegetation with native tree and shrub seedlings (R. Scarborough personal communication). Plants are usually planted within one type of protective guard to resist browsing by native or invasive herbivores such as short plastic corflute guards, large individual wire guards or fenced plots. It has been observed that native and alien vertebrate herbivores regularly damage native vegetation planted in restoration areas, which may hamper vegetation recovery (R Scarborough, M Dawson personal communication). However, the extent of such herbivore damage and the relative effects of protective treatments on plant species establishment and growth remain unknown. Given the high economic and social investment in restoring vegetation across disturbed rainforest communities by local volunteer groups and government agencies, it is important to determine the most effective means of protecting seedlings against herbivore damage. Improved efficiency in vegetation establishment will enable limited resources to be forecited to revegetation over larger spatial and temporal scales.

The aim of this honours research project was to evaluate levels of vertebrate herbivore damage at an abandoned coal mine on the Illawarra Escarpment, at which endangered subtropical rainforest vegetation is being restored using a combination of weed management and revegetation techniques. In order to maximise revegetation establishment success, the restoration practitioners deployed a variety of different protective guards to reduce herbivore impacts over a five-year period: individual wire cages, individual corflute guards and large feral fenced plots to protect multiple regenerating plant species simultaneously.

This honours project was divided into three distinct research questions:

1. How does herbivore damage vary between different protective treatments? It was predicted that vegetation growing in large feral fenced plots, in which access to the vegetation by vertebrate herbivores is physically excluded, would be lower compared with individual seedling corflute guards and wire fences. I also examined whether the amount of herbivore damage varied among different rainforest tree species to
determine which species are more vulnerable to attack and require greater investment in protection.

2. What are the effects of feral fenced plots on the composition of regenerating rainforest vegetation? Feral fenced plots contained a combination of planted seedlings as well as passively regenerating plants. Vegetation in vertebrate exclusion plots were compared to adjacent revegetated areas that could be freely accessed by vertebrates. It was predicted that vegetation in feral fenced plots would have a higher richness and abundance of vegetation compared with control plots. It was also predicted that the structural complexity of vegetation in the fenced plots would be higher than non-fenced control plots, as measured by cover of multiple vegetation layers within the canopy.

3. The third aim was to use camera traps to determine the relative activity of *C. timorensis* and various native vertebrate herbivores (e.g. *W. bicolor*) across different areas on the property, including open grassy areas, revegetation zones and intact rainforests with mature woody vegetation.
2. Methods

2.1 Regional setting and site characteristics

This honours research was performed in conjunction with South East Local Land Services (SE LLS, NSW Department of Primary Industries) and Conservation Volunteers Australia (CVA) to evaluate patterns of vegetation restoration and vertebrate herbivore activity at a private property being managed for biodiversity conservation. The property is located on the lower foothills of the Illawarra Escarpment, adjacent to the township of Balgownie (34° 37’S, 150° 87’E; Figure 2). The property was surrounded by the Illawarra Escarpment State Conservation Area (IESCA) and represents a key focal point of vegetation rehabilitation in the local area given its history as a coal mine and resultant extensive disturbance to the endangered subtropical rainforest ecosystem (M Dawson, personal communications). The site is approximately 170 m above sea level at its centre, covers a total area of 210.437 m², and has a perimeter of approximately 2260 m (Figure 2). Rainfall in the region has an average of 1345 mm annually, with the highest precipitation level in late summer from January to March. Day and night time temperatures range from 25.6 and 17 °C on average annually (BOM 2017) (http://www.bom.gov.au/). Soil in the area is a product of weathering of the upper slopes of the escarpment. The escarpment consists of a Hawkesbury sandstone capping overlying the Narrabeen group of claystones and the Illawarra coal measures (Young 1979).

Two recognised plant communities occur on the property: Illawarra Escarpment Subtropical Rainforest (IESR) and Escarpment Blackbutt Forest (NPWS 2002). These communities exist in relatively intact form at the fringes of the property away from the central abandoned mine entrance. Restoration of the vegetation on the property has therefore focussed upon revegetating cleared areas with nursery grown tree and shrub tube stock seedlings sources from plants indigenous to the IESR (see section 2.1.1 for description of revegetation protocols). This community is characterised by rainforest that form a dense upper canopy and dotted with emergent species of Eucalyptus (principally E. pilularis, saligna x botryoides and quadrangulata). Common rainforest tree species on the property include Dendrocnide excelsa, Doryphora sassafras, Acmena smithii, Diploglottis australis, Toona ciliata, Ficus obliqua and Ficus rubiginosa. This rainforest is also characterised by a high diversity of woody vine species such as Parsonsia straminea, Cissus hypoglauc, Pandorea pandorana and Sarcopetalum harveyanum. The rainforest floor has a sparse groundcover of ferns (e.g Calochlaenia dubia, Pellaea falcata), liliopsids (e.g. Gymnostachys
anceps, Lomandra longifolia) and a dense leaf litter layer. Plants of the IESR occur predominantly on deep clay soils, and in microsites where rainfall is highest such as in sheltered gullies and south-east facing slopes (NPWS 2002).

2.1.1 History of the study site and revegetation activities

The study site was previously a coal mine and is the major reason for much of the forest in the vicinity to be cleared. Mining operations began in 1870 but ceased in 1986 when it was transferred to private residential ownership (J Reay and J Hudson personal communications). When the property was purchased in 2010, the current property owners opened dialogue with NSW DPI and CVA about developing a strategy to rehabilitate the deforested slopes at the face of the former coal mine that was covered in slag, mine tailings, coal wash and infested with dense thickets of the alien shrub Lantana camara (J Reay and J Hudson personal communications). Excavation and soil removal also occurred at the face of the coal mine as part of road and rail infrastructure, which severely disrupted the natural terrain and topography of the site. When the mine closed, the site was used as an unofficial dumping ground for domestic and industrial waste and later as a popular walking trail for residents in nearby suburbs (J Reay and J Hudson personal communications).
With the considerable amount of anthropogenic disturbance and deforestation that occurred on the study site over a century, alien plants have proliferated and become a major component of the novel vegetation community in the area. When the abandoned mine was purchased privately and reassigned to a residential property site, the property owners embarked on a spatially extensive control program for the invasive shrub *L. camara*, using a combination of chemical and mechanical strategies, such as slashing, and in situ mulching of above ground biomass (J Reay and J Hudson personal communications). The property owners, in collaboration with local ecological consultants, identified alien plant invasion (particularly by *L. camara* followed by *Ageratina adenophora*) as the primary threatening process inhibiting regeneration of the endangered subtropical rainforest community (R Scarborough personal communications). In addition, alien plant control was deemed a priority restoration action before the introduction of nursery grown native rainforest seedlings to reinstate the vegetation community. *L. camara* is a perennial woody shrub native to south America that has been introduced to numerous parts of the world (Taylor, Kumar et al. 2012) and is well known for dominating the understorey of disturbed forest communities, including in India and eastern Australia, and resulting in declines in native vegetation diversity and seedling establishment (Sharma and Raghubanshi 2007, Gooden, French et al. 2009). Examples of vegetation condition before and after *L. camara* was cleared by can be observed in Figure 3.
Since 2012, Landcare Illawarra, CVA and LLS have provided support through its network of volunteers and by supplying large quantities of native rainforest tree seedlings and guards to protect the seedlings from herbivore attack. Since November 2012, the property owners in concert with Landcare Illawarra and CVA have removed invasive plants from more than 30,336 m², planted more than 3,000 shrub, tree and grass seedlings, and collected and dispersed the seeds of native plants across the abandoned slopes of the mine that were once dominated by *L. camara* (R Scarborough personal communications).

Although Landcare Illawarra and CVA have worked on site for approximately 100 days in total, the property owners frequently visit the revegetation site and perform restoration management activities to maintain the health of seedlings and prevent secondary plant invasion. Maintenance activities include alien plant control (principally hand-pulling seedlings or cut-and-painting basal stems with glyphosate herbicide), mowing long grass, repairing damaged fences and guards, and anecdotally monitoring the activity of vertebrate wildlife across the revegetation zone.

The property owners observed that many of the regenerating native seedlings (both naturally regenerating ones as well as those introduced as nursery-grown tube stock) were failing to establish due to high levels of stem and leaf damage from vertebrate herbivore
attack. Direct visual encounters and indirect signs (e.g. scats and prints) suggested that *C. timorensis* was the principal herbivore threat to the establishing seedlings. Therefore, to facilitate seedling establishment, three different protective treatments were emplaced across the revegetation zones to protect the seedlings from attack. These included corflute guards, individual wire guards and feral fence exclosure plots (Figure 4). Corflute guards are plastic, fluorescent-pink and typically < 50 cm high that surround individual seedlings. They enclose plants in a triangular prism format and are open at the top. The individual wire guards are made of steel mesh with holes that range in diameter from 4 to 10 cm and are up to 1.5 m in height. The top of the guard is open, and are secured to the ground using wooden stakes. The feral fenced plots are constructed with feral fencing wire with holes that have a diameter ranging from 10 – 15 cm, and a height of between 1 and 1.5 m. The wire was used to enclose multiple plant seedlings simultaneously in herbivore-excluded plots. There are a total of 12 fenced plots located on the study site and they are located on two of the four prominent tiers (referred to as benches) that the property consists of. Four fenced plots occur on bench three, which range in size of 20.8 m² and 55.4 m². Bench four contains eight fenced plots, which range in size from 7.5 m² to 80.5 m² (Figure 4).

It should be noted that on several occasions the feral fencing wire slouched due to trampling by free-roaming cattle and falling branches, which enabled access by *W. bicolor* and *C. timorensis* until repaired. During the honours research, a *W. bicolor* was observed on two occasions inside a feral fence plot. Therefore, feral fenced plots were not considered to be complete herbivore-exclusion zones but rather a form of herbivore deterrent.

![Figure 4](image_url). Schematic of the three types of guards used to protect plants on site. A = wire guard, Al = a modified wire guard, B = corflute guard and C = feral fence. Note the difference in size of guards and the number of plants that can be protected at a time.
Alien plant management and revegetation occurred sequentially across the property rather than simultaneously (Figure 5). As such, there is complex temporal variation in L. camara removal from each bench, revegetation and installation of protective guards (Figure 5, R Scarborough personal communications). In general, seedlings were planted primarily in feral fenced plots during 2013 and 2014 – about 3 to 4 years prior to the commencement of this honours project. Seedlings began to be protected by individual corflute guards in mid-2014, followed by wire guards in 2015. The current honours research aimed to assess variation in herbivore damage to seedlings and vegetation communities between the three different protective treatments (with reference also to non-protected control seedlings) that were planted on the property prior to 2016. Seedlings planted in 2017 were not evaluated. In some cases corflute guards for seedlings planted in 2014 and 2015 were replaced with wire cages in 2015 because the seedlings had grown too tall and were experiencing increased attack by vertebrate herbivores (R Scarborough personal communication).

It is important to note that this honours project evaluated a complex site-intensive and long-term restoration program to rehabilitate an abandoned mine that had first been invaded by a suite of invasive plants and then putatively adversely affected by introduced C. timorensis. Given this, there were several important but unavoidable artefacts introduced to this study that may influence the interpretations of results (see section 3). For example, many plants located in feral fenced plots were about one year older on average than those seedlings protected by corflute or single wire guards. It was possible that variation in plant size among the three protective treatments were confounded by plant age. However, this was deemed unlikely for two main reasons: (1) non-protected seedlings that were used as controls to test for herbivore impacts were chosen at random and probably represented a spectrum of ages from before 2013 to the present, and (2) corflute and wire guards were installed at similar times.
2.2 Data collection

2.2.1 Aim 1: Quantification of tree and shrub seedling damage by vertebrate herbivores

Surveys for Aim 1 were carried out between April and May 2017. A total of 900 plants were surveyed, however seven were excluded due to an inability to determine their species (Table 1). Plants were allocated to one of three protective treatments. Unprotected ‘control’ plants were sampled in an interspersed fashion from within the zone of revegetation, and were of an equivalent size (as assessed by height) to the revegetated plants. These ‘unprotected’ control plants were naturally regenerating on the property since planted seedlings were almost always surrounded by a guard (R Scarborough personal communications).

Although I attempted to sample the largest number of replicate seedlings per species per protective treatment to ensure a balanced and robust experimental design, the haphazard and species-rich nature of the revegetation scheme used by restoration practitioners made this impossible. As such, many rainforest species were represented by only a few replicate seedlings, and most species were not evenly represented in each of the protective treatments. The approach taken to deal with these artefacts are explored in data analysis (Section 2.3.1). However, it must be noted that, to the best of my knowledge, this study represents the most comprehensive and intensive assessment of seedling establishment and growth responses to vertebrate damage in a restoration context ever undertaken in Australia, and the first to assess the differential effectiveness of multiple protective treatments.

Several variables for each plant were recorded including: species identity (to species level for most, but Ficus species except for F. coronata were grouped by genus due to

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**Figure 5.** Presents a timeline of the approximate months that different groups of plants were planted. B1 to B4 represents the four tiers (benches) on the study site from 1 to 4. FF = feral fences (each are numbered separately), WG = wire guard, CG = corflute guard.
uncertainties with identifying seedlings), the type of protective treatment surrounding each seedling, seedling vertical height (to nearest 10 cm from soil surface to tallest apical meristem), percentage of branches damaged by herbivore browsing and the presence or absence of bark stripping by deer. The detailed assessment of percentage herbivore damage was adapted from Zamora, Gómez et al. (2001), Palmer and Truscott (2003). Up to 30 lateral branches arising from the central trunk were randomly selected per seedling, and the proportion of their apical tips that had been clipped by vertebrate herbivores was calculated as a measure of browsing damage. If a leaf showed signs of herbivore activity but the stem upon which it was growing had not been clipped at the meristematic growing tip, vertebrate herbivore damage was not recorded. This is because leaf damage could have arisen from invertebrate herbivory, and deer are known to browse on apical growing regions in woody plants (Forsyth, Wilmshurst et al. 2010).

In addition to these quantitative variables, a qualitative assessment of herbivore damage was made for each seedling using a four-point scale. The rapid four-point scale of damage is used by local restoration practitioners to rapidly (but qualitatively) assess the level of browsing damage of vertebrate herbivores on seedlings. The four-point scale is exemplified in Figure 6: “1” = no visual damage; “2” = mild browsing damage apparent; “3” = damage is prominent but the plant appears generally well and is surviving; “4” = high levels of browsing damage and defoliation, such that the plant appears unlikely to establish and survive. These rapid assessment data were generated through personal communication with Richard Scarborough. The aim of this honours project was to provide a detailed assessment of level of browsing damage to validate the rapid four-point damage score used by restoration practitioners.

Figure 6. Depicts plants suffering one of the four levels of browsing used to measure the degree of browsing as a visual estimate. B1 is the lowest, and B4 is the highest.
The presence or absence of bark stripping was also documented per plant. On one rare occasion, considerable damage to bark and tree trunks were caused by Yellow-tailed black cockatoos (*Calyptorhynchus funereus*). This damage was only limited to 2 mature *Solanum aviculare* on site and so were not recorded. Damage by deer produced long and often deep vertical lacerations in tree trunks in comparison to hollows and chunks of wood missing that would be caused by *C. funereus* (see figure 7).

![Figure 7](image_url)

**Figure 7.** Depicts the type of bark stripping that is typical of *C. timorensis* (a), (b) and (c), compared to the damage inflicted by *C. funereus* (d) and (e).

### 2.2.2 Aim 2: Assessing vegetation community responses to deer exclusion using feral fence plots.

Surveys occurred from May to June 2017. A total of 12 feral fenced plots occurred across the property, and all plots were surveyed as part of this honours project. The aim of this part of the project was to assess the effects of feral fence installation on the composition of the regenerating vegetation, including non-target plants that had spontaneously regenerated on site. This was achieved by comparing the vegetation within the set of 12 feral fenced plots with adjacent non-fenced control plots of similar size that deer and other vertebrate herbivores had free access to. Non-fenced control plots were interspersed among the existing feral fenced plots to avoid spatial biases in vegetation differences. It was thus assumed that differences in vegetation composition (particularly for herbaceous, grass and vine species that have regenerated at the site spontaneously) were a function of vertebrate exclusion with feral fencing.
Several variables were recorded per plot including: identity of all species and their abundance using a Braun-Blanquet scale (see below), the total percentage canopy cover directly above 2 m at each plot, the percentage cover of four different vegetation layers, the percentage cover of bare soil, percentage cover of litter, as well as the depth (cm) and volume of litter ($cm^3$ per $m^2$ of soil).

The percentage cover of bare soil and litter were determined using a visual estimate to the nearest 5% across each plot (Gooden, French et al. 2009). The litter depth was estimated by finding the average of five random points across each plot. Litter depth was measured as the vertical distance (cm) from soil surface to the top of the litter layer (Gooden and French 2014).

Percentage foliage cover was estimated for four layers of vegetation within each plot: ground cover extends from 0 – 0.5 m; shrub layer ranges from 0.5 – 2 m; sub-canopy mesophyllous tree layer from 2 – 10 m; upper tree canopy from > 10 m.

The identity and abundance of each vascular plant species (native and alien) rooted within each plot was recorded using a modified Braun-Blanquette cover abundance scale following Gooden, French et al. (2009): (1) <5% cover and one or few individuals, (2) <5% cover and uncommon, (3) <5% and common, (4) <5% and very abundant, (5) 5 – 20% cover, (6) 21 – 50% cover, (7) 51 – 75% cover, (8) 76 – 100% cover.

In order to account for visual biases in canopy cover estimated, a single cover value per plot that encompassed all the vegetation layers was derived using a tested and proven digital photographic technique (Goodenough and Goodenough 2011). Five photographs were taken with a Sony Xperia Z5 digital camera at approximately 1.5 m height in the middle of the shrub layer across each plot. Photograph spectra were converted to two tones (red and
blue) in the program CanopyDigi, with each colour representing sky and foliage respectively. A percentage canopy cover was derived from the proportion of blue pixels per image, and a single canopy estimate was calculated as an average of the five photos per plot (Figure 9). Five photos of the canopy for each fenced and control were taken on the 23rd, 24th, 29th, and 30th of May 2017 between 11am and 3pm as the weather had clear skies and therefore similar levels of brightness.

![Image](image_url)

**Figure 9.** A schematic of the procedure (A → B → C → D) used to calculate the total canopy cover for each feral fence and control plot. Programs used included Corel PaintShop Pro X9 and CanopyDigi

2.2.3 **Aim 3: Monitoring activity of vertebrate herbivores within different habitat treatments with camera traps**

*C. timorensis* and *W. bicolor* are two vertebrate herbivores reported as highly active on site based on anecdotal evidence from the landowners and the wider literature (Hamilton 1982, Wollongong City Council 2017). The aim of this third part of my honours research was to explore how the activity of *C. timorensis* and other vertebrates vary across the property in order to evaluate relative contributions to browsing damage on revegetation.

Herbivore activity was assessed within three different common habitat treatments using camera traps including (1) established patches of mature rainforest, (2) cleared grassy areas that had been cleared of woody vegetation but not revegetated with native seedlings and (3) areas cleared of woody vegetation (namely *L. camara*) and revegetated with native rainforest tree and shrub species. The rainforest habitat treatment had a dense canopy of rainforest species with sparse groundcover consisting of ferns and thick layer of leaf litter. These patches of established rainforest were located at least 100 m from the centre of the property where revegetation occurred. Differences in the treatments can be visualised in
Herbivore activity across these three microhabitats was deemed necessary because it is currently assumed (but not known) that areas undergoing active revegetation with seedlings are more vulnerable to herbivores than established vegetation. It is also assumed but not known that open grassy areas attract *C. timorensis* and may act as magnets for deer activity that spills over into adjacent areas undergoing revegetation (Kuijper, Cromsigt et al. 2009). This study represents a first step in evaluating the activity of vertebrate herbivores at fine scales across microhabitats in the context of ecological restoration.

**Figure 10.** Depicts the three different habitat treatments used to measure differences in the activity of large herbivores including *C. timorensis* and *W. bicolor*. A = forest, B = grass, and C = revegetation.

A total of six camera trap stations were set up in each of the three habitat treatments using Little Acorn Ltl-5210A motion sensing camera, equipped with an infrared flash. Cameras were deployed for a total period of 28 days over two 14-day cycles. Three cameras were placed in three different locations per habitat treatment on July 1st for a total of two weeks. At the end of this period the cameras were redeployed to a different set of three sites per habitat treatment for another two weeks. In total six different camera stations were used per habitat treatment. Given the size of the property and the smaller size of the study site, cameras were at times placed within 30m of each other.

Cameras were set to high sensitivity, positioned at approximately 1.5 –2m by strapping to tree trunks and were then directed towards the ground at approximately 35° to maximise detection of herbivores (Gray and Phan 2011). Cameras were programmed to take three consecutive photos at a time upon detection of an animal. This ensured that an animal
was not simply captured when it triggered the side sensors, and ultimately allowed the camera to capture the animal as it passed through its field of view. This resulted in clear captures of the animal. This study does not intend to identify individual animals and is only focussed on recording visitation rates; therefore, a time delay of 10 seconds was used which allowed animals to be tracked, and a time of occupancy per animal to be measured.

2.3 Data analyses

2.3.1 Aim 1: Quantification of tree and shrub seedling damage by vertebrate herbivores

All data in aim 1 was analysed following this step-wise process.

1. A one-way Analysis of Variance (ANOVA) found that seedlings grown in feral fenced plots were significantly taller compared with seedlings protected by corflute guards and individual wire cages ($F_{2,624} = 39.74, P < 0.0001$). However, it is not possible to attribute this increase in height to the exclusion of vertebrate herbivores by feral fencing because plants in feral fenced plots tended to be about one year older than those protective by corflute or wire guards, and thus height may be a function of age instead of the treatment type (Figure 5). Therefore, due to confounding effects of plant age by treatment, this honours project only evaluated the effects of protective treatment on herbivore damage as a measure of impact.

2. A one-way ANOVA was used to examine variation in proportion of stems attacked by herbivores among the four-point qualitative damage scores, in order to validate the score’s use by restoration practitioners to rapidly assess herbivore damage to seedlings. Where significant differences were found, post hoc comparison of categories was undertaken using Tukey’s HSD test.

3. The variation in proportion of branches browsed by herbivores across all plant species was examined among the four protective treatments using a one-way ANOVA (followed by pairwise Tukeys HSD tests). In this test the unit of replication was the single plant, and plant species identity was not considered ($n = 364$ in feral fenced plots, $n = 124$ in corflute guards, $n = 185$ in wire guards and $n = 220$ for unguarded plants).

4. The final set of analyses aimed to examine if the effects of protective treatment on herbivore damage varied among different plant species. Separate ANOVAs were performed for species with sufficient replication ($n > 5$) in three or more protective treatment categories.
Species present in two protective treatment categories only were examined using $t$-tests (see Table 1 results for detailed description of replication level per species and representation in each treatment category). Species responses had to be examined using multiple separate tests because most species were planted together into single feral fenced plots and were thus not spatially or temporally independent from one another. The parametric tests outlined in steps 1, 2 and 3 were based upon normal distribution of residuals and homogeneous variances. Normality was tested by inspecting plots of residuals and normal quantile plots, and data were square root transformed as necessary to improve normality and homogeneity of variance. In many cases, due to the retrospective focus of this honours project on an existing revegetation project, it was not possible to use balanced, orthogonal designs when conducting ANOVAs. However, calculation of sum of squares are robust to small deviations from orthogonality for one way ANOVAs, although this increases the likelihood of a Type I error where data are non-normal or small sample sizes (Shaw and Mitchell-Olds 1993). Such results must therefore be interpreted cautiously in cases of unbalanced designs with small sample sizes (see results in section 3).

All analyses were performed in JMP 11.0.0 (SAS Institute 2013).

5. Chi-Square tests were initially attempted to assess variation in occurrence of bark stripping among protective treatments and plant species. However, this was not possible because bark stripping frequencies were extremely low and < 5 in all treatment combinations. Whilst observations are summarised in results section 3.1.4, hypothesis testing was not viable for bark stripping.

2.3.2 Aim 2: Assessing vegetation community responses to deer exclusion using feral fence plots.

Differences in percentage cover of bare soil abundance, percentage of litter abundance, average litter depth (cm), and litter volume (cm$^3$/m$^2$) and percentage covers of the five different canopy strata between feral fenced and non-fenced control plots were assessed using $t$-tests.

Differences in total, native and alien species richness (number of different species were plot) between feral fenced and control plots were also assessed with $t$-tests. All Conyza species and Oplismenus species were grouped into taxa due to difficulties with identifying species from vegetative material in the field. Four species could not be identified and were
excluded from subsequent analyses. The richness of native plants within different growth form categories (i.e. herbs, graminoids, climbers, shrubs and trees) were also compared between fenced and control plots using t-tests in order to determine functional plant responses to feral fence installation.

Prior to analysing species composition, species abundance data were adjusted from Braun-Blanquet scores to median percentage covers following Mason and French (2007). Braun-Blanquet scores of 1 to 4 were given values of 1 to 4% each. Scores of 5, 6, 7 and 8 were allocated a percent abundance of 15, 38, 63 and 88, respectively. A matrix of Bray-Curtis similarity percentage was first developed which ranked sites by compositional similarity to one another. Once this was developed, non-metric multidimensional scaling ordination was then used to plot feral fence and unfenced control sites based on their compositional similarities to one another. Finally, permutational multivariate analyses of variance (PERMANOVAs) were used to determine whether plant compositions varied significantly between feral fenced and unfenced control plots (Gooden and French 2014). These compositional differences in plant communities between feral fenced and un-fenced control plots were assessed for all species combined as well as native and alien plants separately using PERMANOVAs. Where significant differences occurred, then similarity percentage (SIMPER) analyses was performed to determine the plant species contributing most strongly to compositional changes.

All compositional analyses were performed using the statistical program PRIMER 6 (Clarke and Gorley 2006) and PERMANOVA + B (Anderson, Gorley et al. 2008).

2.3.3 Aim 3: Monitoring activity of vertebrate herbivores within different habitat treatments with camera traps

Three response variables of vertebrate activity were recorded in this study. The first variable was the activity of each species of vertebrate animal per habitat treatment. Activity was measured as the number of separate animal detections observed over the 28 days of camera deployment per site. I defined separate detections as events with a time delay that exceeded 10 seconds. Where camera captures contained more than one animal, then the number of animals was also recorded as separate events. First, the variation in the probability of a camera detecting an animal (i.e animal detected ‘1’ or not detected ‘0’) among the three habitat treatments was evaluated using a Chi-Square test following a binomial distribution.
Given the limitations in replication (i.e. \( n = 3 \) cameras per habitat treatment per deployment event), it was not possible to examine variation in animal activity through time. Therefore, data were pooled between the two deployment events which resulted in \( n = 6 \) cameras per habitat treatment. Differences in activity of the two most common vertebrate herbivores (\( W. \) bicolor and \( C. \) timorensis) was then evaluated among the three habitat treatments using a Chi-Square test. For all analyses, post hoc comparison of differences in probability were assessed using the Wilcoxon-U test where significant differences occured (Zar 1999).

The second variable was the duration of time that each animal (\( W. \) bicolor or \( C. \) timorensis) spent in each habitat treatment per event (Seconds). This was measured by recording the time between an animal first being observed and when it departed as was indicated by the photos which were time and date stamped. Where several animals occurred in a single detection, then the time that each animal spent in the series of photos being taken consecutively was recorded. Due to an extremely non-normal distribution of residuals, parametric tests such as ANOVA were deemed to be inappropriate analyses for these data. Therefore, non-parametric comparisons of time spent by these animals among treatments were used, which assessed differences by rank with a Kruskal Wallis test following a Chi-Square distribution. Where significant affects were detected, comparisons were made using the pairwise Wilcoxon-U test (Zar 1999).

The third variable measured for each \( W. \) bicolor and \( C. \) timorensis recorded was the time of day that they were observed. These were recorded as during the day (06:30 – 17:00), night (18:30 – 05:00) or crepuscular (05:00- 06:30 and 17:00 – 18:30). Variation in the probability of detection between these time periods was analysed using a Chi-Square test, followed by a non-parametric pairwise comparison to determine when each species was most active.

All analyses were performed on JMP Pro 11.0.0 (SAS Institute Inc 2013)
3. Results

3.1 Aim 1: Effects of ‘protection’ treatments on herbivore damage to native tree and shrub seedlings.

3.1.1. Use of a four-point impact score to rapidly assess herbivore damage to native tree and shrub seedlings.

A total of 900 individual tree and shrub seedlings were sampled across the study site (Table 1). Seven plants were excluded from all subsequent analyses as they could not be identified. There was a strong positive relationship between the four-point damage score that was used by restoration practitioners to rapidly assess herbivore damage in the field and the proportion of stems displaying signs of herbivore damage (Table 2, Figure 11). There was a high degree of confidence that plants given the lowest damage score of ‘1’ by restoration practitioners (i.e. deemed as ‘healthy’ and unbrowsed) did indeed have the very lowest levels of browsing of <2% (Figure 11). Likewise, plants deemed to have a damage score of ‘4’ had significantly greater levels of branch and leaf damage (i.e. > 80%) compared with intermediate damage scores of ‘2’ and ‘3’ (Figure 11). This result indicates that the four-point score can be used with a high degree of confidence by restoration practitioners when rapidly assessing the health and herbivore damage to tree and shrub seedlings during vegetation restoration programmes.

Table 1. Number of plants per species in each of the four ‘protective’ treatments that were sampled for Aim 1.

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<th>Species</th>
<th>Feral fence</th>
<th>Wire guard</th>
<th>Corflute guard</th>
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<th>Total</th>
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Table 2. Result of an ANOVA for variation in the proportion of branches browsed among four-point damage scores that were used by restoration practitioners to rapidly assess herbivore damage in the field.

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</tr>
</thead>
<tbody>
<tr>
<td>Damage score</td>
<td>3</td>
<td>645999.5</td>
<td>215333</td>
<td>1019.616</td>
<td>&lt;0.0001</td>
<td>0.76</td>
</tr>
<tr>
<td>Error</td>
<td>888</td>
<td>187537.2</td>
<td>211</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Figure 11. Average (± SE) percentage of branches browsed across each level of an ordinal four-point damage score used by restoration practitioners to rapidly assess the effects of browsing damage to native tree and shrub seedlings. ‘1’ (n = 567), ‘2’ (n = 183), ‘3’ (n = 83), ‘4’ (n = 60). Browse damage categories that do not share the same letter are significantly different based on Tukey’s HSD test.

3.1.2. Effects of protective treatments on herbivore damage to native seedlings.

The 893 plants included in analyses represented at least 54 species of plants (Table 1). However, given that all *Ficus* species except for *F. coronata* were pooled into a single taxon for analyses, the number of species surveyed was most likely 58, as *F. rubiginosa*, *F. obliqua*, *F. superba* and *F. macrophylla* were observed and sampled on site.

Overall, there was a significant reduction in the proportion of stems damaged by herbivores in response to each of the three forms of protective guards compared with unguarded control plants (Table 3, Figure 12). On average, unguarded plants suffered approximately 40% greater levels of browsing damage compared with plants surrounded by corflute guards, but more than two-times greater damage than plants surrounded by individual wire guards or feral fencing. The proportion of stem damage was significantly higher for plants surrounded by corflute guards than either wire guards or feral fencing. Plants protected by wire guards and feral fencing experienced similarly low levels of browsing of between 10 and 15% of stems on average (Figure 12).

Table 3. Results of an Analyses of Variance for the effect of guard type on the proportion of branches browsed by herbivores (%).

<table>
<thead>
<tr>
<th>ANOVA</th>
<th>DF</th>
<th>SS</th>
<th>MS</th>
<th>F</th>
<th>P</th>
<th>( r^2 )</th>
</tr>
</thead>
<tbody>
<tr>
<td>Guard types</td>
<td>3</td>
<td>61375.16</td>
<td>20458.4</td>
<td>23.546</td>
<td>&lt;0.0001</td>
<td>0.073</td>
</tr>
<tr>
<td>Error</td>
<td>889</td>
<td>772425.36</td>
<td>568.9</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Figure 12. Average (+ SE) percentage of branches browsed in relation to different type of guards used to protect plants: N = no guard (n = 220), C = corflute guard (n = 124), W = wire guard (n = 185) and F = feral fenced plot (n = 364). Guard types which do not share the same letters have significantly different averages.

3.1.3 Variation in effects of protective treatment types on browsing damage among different plant species.

Of the approximately 54 species sampled, only 20 could be examined statistically for the effects of different protective treatments on browsing damage of plants, given limited levels of replication and lost degrees of freedom (e.g. n < 4 seedlings for some protective treatments for most species), and violations of homogeneity of variance (Zar 1999) (Table 1). Furthermore, most species were not evenly represented amongst the four protective treatments (e.g. Homalanthus populifolius seedlings were never protected by corflute guards but were well replicated across the other three protective treatments, Table 1). This meant that browsing damage for most species could only be compared between two or three of the four protective treatments at any one time (see details in Table 1). For an interpretation of the effect of the different protective treatments for each species of plant, see Table 4.

Nine of the 20 species showed significant negative effects of one or more protective treatments on the level of browsing damage (Table 4, Figure 13), which included Acacia maidenii, Cryptocarya micruneura, Ehretia acuminata, Ficus coronata, Glochidion ferdinandi, Hymenanthera dentata, Pittosporum undulatum, Rapanea howittiana and Solanum aviculare. For most species, such as C. micruneura, E. acuminata, F. coronata, P. undulatum and R. howittiana, unprotected control seedlings had between two to four-times
greater levels of browsing damage than seedlings located in either feral fenced plots or surrounded by individual wire cages. For these species, the level of browsing damage on unprotected plants was usually greater than 60% of leaves and stems, but in some species (e.g. C. microneura and F. coronata) the level of damage was as high as 80%. Two species (H. populifolius and S. stelligerum), also showed non-significant trends toward being less browsed when protected by a feral fence, as opposed to being unguarded (P = 0.054 and P = 0.08 respectively) (Figure 13).

There was no significant difference in the proportion of browsing damage among the different protective guards for 10 species of plants, although they all generally showed higher levels of browsing when they were unprotected (Figure 13). These included Acmena smithii, Elaodendron australie, Ficus spp., Guioa semiglauca, H. populifolius, Indigofera australis, P. revolutum, Synoum glandulosum, S. stelligerum and Toona ciliata. Two species (Melia azederach and E. australie) had overall very low levels of browsing damage that did not differ significantly among treatments, although browsing damage was never detected on seedlings protected by feral fenced plots. Furthermore, D. cunninghamii and T. ciliata were not found to be browsed at all.

Surprisingly, two species (G. ferdinandi and P. revolutum) had higher levels of browsing on corflute protected plants compared to those within wire guards, feral fenced plots and unguarded control plants. However, this trend was only statistically significant for G. ferdinandi. Similarly, E. acuminata seedlings protected by corflute guards experienced significantly higher levels of browsing compared to plants which were protected by wire guards and feral fencing (Table 4, Figure 13).
Table 4. Variation in browsing damage for the 20 species with various protective treatments. ANOVAs were deployed for comparisons of three or more treatment categories; t-tests were deployed to compare two treatment categories only. P-values in bold denote significant effects. The column labelled “treatment comparisons (n)” indicates which treatments were compared in each analysis as well as the number of replicates per treatment in parentheses (F = feral fenced plot, C = corflute guard, W = wire guard, N = unprotected plant).

<table>
<thead>
<tr>
<th>Species</th>
<th>Effect</th>
<th>DF</th>
<th>SS</th>
<th>F</th>
<th>t</th>
<th>P</th>
<th>Treatment comparisons (n)</th>
<th>Notes of effects</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acacia maidenii</td>
<td></td>
<td>24</td>
<td>2.16</td>
<td>0.0407</td>
<td></td>
<td></td>
<td>F (18), N (8)</td>
<td>Unprotected control seedlings suffered about 10 times greater browsing damage than seedlings in feral fenced plots.</td>
</tr>
<tr>
<td>Acmena smithii</td>
<td>Model</td>
<td>2</td>
<td>4495.6</td>
<td>2.03</td>
<td>0.16</td>
<td></td>
<td>W (7), C (7), N (8)</td>
<td>Null effect of protective treatments on browsing damage.</td>
</tr>
<tr>
<td>Cryptocarya microneura</td>
<td></td>
<td>14</td>
<td>2.83</td>
<td>0.013</td>
<td></td>
<td></td>
<td>F (8), N (8)</td>
<td>Unprotected control seedlings suffered more than 3-times greater browsing damage than seedlings in feral fenced plots.</td>
</tr>
<tr>
<td>Diploglottis cunninghamii</td>
<td>n/a</td>
<td>n/a</td>
<td>n/a</td>
<td>F (5), C (5)</td>
<td></td>
<td>Statistical comparisons were unnecessary because browsing damage was not evident on any seedling in any treatment, including unguarded control plants.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ehretia acuminata</td>
<td>Model</td>
<td>2</td>
<td>9590.4</td>
<td>9.84</td>
<td>0.0016</td>
<td></td>
<td>F (6), W (8), C (5)</td>
<td>Seedlings protected by corflute guards suffered more than 4-times greater browsing damage on average than seedlings in feral fenced plots and wire guards.</td>
</tr>
<tr>
<td>Elaodendron australi</td>
<td></td>
<td>n/a</td>
<td>n/a</td>
<td>F (7), W (4), C (7), N (7)</td>
<td></td>
<td>Statistical comparisons were unnecessary because browsing damage only occurred on unprotected control plants.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ficus coronata</td>
<td>Model</td>
<td>3</td>
<td>19627.4</td>
<td>7.92</td>
<td>0.0002</td>
<td></td>
<td>F (10), W (15), C (10), N (15)</td>
<td>Unprotected control seedlings suffered between 2 and 3-times greater browsing damage than seedlings in each of the three protective treatments.</td>
</tr>
<tr>
<td>Ficus species</td>
<td>Model</td>
<td>3</td>
<td>5337.60</td>
<td>1.87</td>
<td>0.14</td>
<td></td>
<td>F (6), W (26), C (10), N (7)</td>
<td>Null effect of protective treatments on browsing damage.</td>
</tr>
<tr>
<td>Hymenanthera dentata</td>
<td></td>
<td>46</td>
<td>3.07</td>
<td>0.0036</td>
<td></td>
<td></td>
<td>F (37), N (11)</td>
<td>Unprotected control seedlings suffered more than 2-times greater browsing damage than seedlings in feral fenced plots.</td>
</tr>
<tr>
<td>Glochidion ferdinandi</td>
<td>Model</td>
<td>3</td>
<td>11877.30</td>
<td>9.88</td>
<td>0.0002</td>
<td></td>
<td>F (6), W (26), C (10), N (7)</td>
<td>Seedlings protected by corflute guards had more than 6-times greater browsing damage than either unprotected control plants or those protected by feral fenced plots or wire guards.</td>
</tr>
<tr>
<td>Species</td>
<td>Model Error</td>
<td>R²</td>
<td>Error</td>
<td>F (df), W (df), N (df)</td>
<td>Summary</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>-------------------------</td>
<td>-------------</td>
<td>----</td>
<td>-------</td>
<td>------------------------</td>
<td>------------------------------------------------------------------------</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Guioa semiglauca</em></td>
<td>2 / 27</td>
<td>0.60</td>
<td>0.55</td>
<td>F (6), W (15), N (9)</td>
<td>Null effect of protective treatments on browsing damage.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Homalanthus populifolius</em></td>
<td>2 / 21</td>
<td></td>
<td></td>
<td>F (11), W (8), N (5)</td>
<td>Non-significant trend ($P &lt; 0.1$) towards higher browsing damage on unprotected and wire guarded plants than seedlings in feral fenced plots.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Melia azederach</em></td>
<td>n/a / n/a</td>
<td></td>
<td></td>
<td>W (8), C (8), N (7)</td>
<td>Statistical comparisons were unnecessary because browsing damage was not evident on any seedling in the wire guard and were extremely low in corflute guarded or unguarded control treatments.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Pittosporum revolutum</em></td>
<td>3 / 24</td>
<td>1.89</td>
<td>0.15</td>
<td>F (9), W (6), C (7), N (6)</td>
<td>Null effect of protective treatments on browsing damage.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Pittosporum undulatum</em></td>
<td>3 / 92</td>
<td>6.65</td>
<td>0.0004</td>
<td>F (60), C (7), W (7), N (22)</td>
<td>Unprotected control seedlings suffered about 3-times greater browsing damage on average than seedlings in each of the three protective treatments.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Rapanea howittiana</em></td>
<td>3 / 27</td>
<td>7.34</td>
<td>0.0009</td>
<td>F (60), C (7), W (7), N (22)</td>
<td>Unprotected control seedlings suffered between 3 and 4-times greater browsing damage on average than seedlings in each of the three protective treatments.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Solanum aviculare</em></td>
<td>21 / 14</td>
<td>4.90</td>
<td>&lt;0.0001</td>
<td>F (14), N (9)</td>
<td>Unprotected control seedlings suffered up to 4-times greater browsing damage than seedlings in feral fenced plots.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Solanum stelligerum</em></td>
<td>26 / 17</td>
<td>1.81</td>
<td>0.08</td>
<td>F (17), N (11)</td>
<td>Non-significant trend ($P &lt; 0.1$) towards higher browsing damage on unprotected and wire guarded plants than seedlings in feral fenced plots.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Synoum glandulosum</em></td>
<td>3 / 27</td>
<td>0.57</td>
<td>0.64</td>
<td>F (7), W (5), C (12), N (7)</td>
<td>Null effect of protective treatments on browsing damage.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Toona ciliata</em></td>
<td>47 / n/a</td>
<td>n/a</td>
<td>n/a</td>
<td>F (33), N (16)</td>
<td>Statistical comparisons were unnecessary because browsing damage was not evident on any seedling in either the feral fenced or unguarded control treatments.</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
3.1.4. Association between bark stripping by rusa deer and type of protective treatment.

In general, the occurrence of bark stripping was rare across the study site. In total, 38 plants showed signs of bark stripping which represented just over 4% of the entire cohort of seedlings. Of these, 24 were unprotected, nine were located in feral fenced plots, three in wire guards and two in corflute guards. Statistical comparison using Chi Square tests were not possible due to insufficient data as n <5 for all protective treatments, except for unguarded plants (Zar 1999). Generally, 11% of all unguarded plants showed visible signs of bark stripping, which was considerably higher than those which were protected. In comparison, less than 3% of plants which were protected by either a feral fence, wire guard, or corflute guard showed any sign of bark stripping (Figure 14).

The 38 plants that showed signs of bark stripping represented 16 out of the total of 54 species surveyed (Figure 15). The incidence of bark stripping on plant species irrespective of the guard type, could not be analysed further as the rate of incidence of bark stripping was n <5 for most species except *M. azederach* and *A. maidenii*. While *E. quadrangulata* and *Breynia oblongifolia* appeared to have the highest rates of bark stripping, a total of three and one plant were sampled respectively, and therefore such patterns cannot be considered.
reliable. In contrast A. maidenii and M. azederach suffered higher rates of bark stripping that ranged between 25 – 30% and these were sampled in higher numbers (n = 26 and n = 25 each). Other species with a reasonable amount of replicates and which suffered rates of bark stripping above, or close to 10% of all plants sampled included Brachychiton acerifolius, Claoxylon australe, H. populifolius, S. aviculare, and T. ciliata. Most species however typically had lower rates of bark stripping, with less than 5 in every 100 plants being bark stripped by deer.

![Figure 14. Presents the proportion of plants within each type of treatment that show signs of bark stripping as present or absent](image-url)
Figure 15. Presents the proportion of plants, by species which show signs of bark stripping in some form. Numbers at the top and bottom of each column represent the number of replicates of bark stripping that was present or absent respectively. ACMA = Acacia maidenii, ACME = Acacia melanoxylon, ACOB = Acrochyna oblongifolia, ALSU = Alectryon subsinereus, BRAC = Brachychiton acerifolius, BROB = Breynia oblongifolia, CLAU = Claoxylon austral, EUQU = Eucalyptus quadrangulate, FOCO = Ficus coronata, FIUS = Ficus spp, GLFE = Glochidion ferdinandi, HOPO = Homalanthus populifolius, MEAZ = Melia azederach, PIUN = Pittosporum undulatum, RAHO = Rapanea howittiana, SOAV = Solanum aviculare, TOCI = Toona ciliata.

3.2 Aim 2: Vegetation community responses to vertebrate herbivore exclusion using feral fence plots.

3.2.1 Effects of feral fencing on abiotic conditions and vegetation structure

The percentage cover of bare soil was significantly affected by fence installation, being approximately five times lower in fenced than adjacent unfenced control plots (Table 5, Figure 16). Fenced plots also had significantly higher percentage cover of leaf litter, although only by approximately 10 % (Table 5, Figure 16). Litter depth and volume, however, did not differ significantly between fenced and unfenced plots (Table 5, Figure 16).

There was no significant difference in average total canopy cover (%) between the fenced and unfenced plots (Table 5, Figure 17). However, the effects of fence installation varied amongst different strata within the canopy. The abundance of groundcover vegetation was significantly more than two-times higher in fenced than unfenced plots (Table 5, Figure 17), and shrub-layer vegetation cover was about 20% higher in fenced than unfenced plots (Table 5, Figure 17). Indeed, across both the fenced and unfenced treatments, the vegetation was dominated by shrub-layer vegetation, ranging from approximately 75% cover in unfenced plots to 90 % in fenced plots. However, the mesophyll tree cover and upper canopy
vegetation cover (which ranged between 30 and 60 % on average) did not significantly differ between fenced and unfenced plots (Table 5, Figure 17), although there was a trend (P = 0.09) towards higher cover of mesophyllous trees in the midstory in fenced compared with unfenced pots.

**Table 5.** Results of two-tailed t-tests for differences in abiotic and canopy structure variables between fenced and unfenced plots (n = 12). Units of measurement for each variable are presented in parentheses. Bold P-values denote significant differences.

<table>
<thead>
<tr>
<th>Response variable</th>
<th>t</th>
<th>DF</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Abiotic condition</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Bare soil cover (%)</td>
<td>3.670</td>
<td>22</td>
<td><strong>0.001</strong></td>
</tr>
<tr>
<td>Litter cover (%)</td>
<td>2.457</td>
<td>22</td>
<td><strong>0.020</strong></td>
</tr>
<tr>
<td>Litter depth (cm)</td>
<td>1.076</td>
<td>22</td>
<td>0.290</td>
</tr>
<tr>
<td>Litter volume (cm³/m²)</td>
<td>1.550</td>
<td>22</td>
<td>0.130</td>
</tr>
<tr>
<td><strong>Canopy structure</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total canopy cover (%)</td>
<td>0.051</td>
<td>22</td>
<td>0.960</td>
</tr>
<tr>
<td>Groundcover (0 – 50 cm stratum height, %)</td>
<td>2.975</td>
<td>22</td>
<td><strong>0.007</strong></td>
</tr>
<tr>
<td>Shrub cover (0.5 – 2 m stratum height, %)</td>
<td>4.449</td>
<td>22</td>
<td>&lt; <strong>0.001</strong></td>
</tr>
<tr>
<td>Mesophyll tree cover (2 – 10 m stratum height, %)</td>
<td>1.782</td>
<td>22</td>
<td>0.089</td>
</tr>
<tr>
<td>Upper canopy cover (&gt; 10 m stratum height, %)</td>
<td>1.582</td>
<td>22</td>
<td>0.128</td>
</tr>
</tbody>
</table>
Figure 16. Average (+ SE) responses of the abiotic conditions including: (a) bare soil cover, (b) litter abundance, (c) litter depth, and (d) litter volume in unfenced (n = 12) and fenced (n = 12) plots. Note the differences in y-axis units and ranges amongst the four plots. Asterisks denote significantly different means based on t-tests.
Figure 17. Average (+ SE) percentage cover of different canopy layers including: (a) total canopy cover (b) ground cover, (c) shrub cover, (d) mesophyll tree cover and (e) upper canopy cover in unfenced (n = 12) and fenced (n = 12) plots. Asterisks denote significantly different means based on t-tests.
3.2.2. Effects of feral fencing on vegetation diversity and composition.

In total, 132 vascular plant species were recorded from across the 24 plots, comprising 35 alien and 93 native plant species (plus four species that could not be identified and were subsequently excluded from analyses). These species represented 41 herbs, 11, graminoids, 17 climbers, five ferns, 10 shrubs and 44 tree species.

There was no significant difference in the total (i.e. native and alien species combined) richness of vascular plants between fenced and unfenced plots (Table 6, Figure 18a). However, native plant species richness was significantly higher in fenced than unfenced plots. On average, a fenced plot contained 25% more native plant species than corresponding unfenced plots (Table 6, Figure 18b). Conversely, alien plant species richness was not significantly affected by fence installation although there was a trend (P = 0.071) towards a higher number of alien species occurring in unfenced compared with fenced plots (Table 6, Figure 18c). There was no effect of fence installation on species richness for either annual or perennial plants. Fence installation also had no effect on species richness of native herbs, graminoids or climbers Figure (18 d, e, f, g, h). In contrast, species richness for both native shrubs and trees was significantly higher in fenced than unfenced plots. On average, fenced plots contained more than twice the number of native shrub species, and about 80% more native tree species, than unfenced plots (Table 6, Figure 18 i, j).

Table 6. Results of two-tailed t-tests for the differences in the species diversity of vascular plants residing within fenced (n = 12) and unfenced plots (n = 12). Bold P – values denote significant differences.

<table>
<thead>
<tr>
<th>Response variable</th>
<th>t</th>
<th>DF</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total species richness</td>
<td>0.17</td>
<td>22</td>
<td>0.860</td>
</tr>
<tr>
<td>Native species richness</td>
<td>2.65</td>
<td>22</td>
<td>0.015</td>
</tr>
<tr>
<td>Alien species richness</td>
<td>1.89</td>
<td>22</td>
<td>0.071</td>
</tr>
<tr>
<td>Life -history</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Annual species richness</td>
<td>1.45</td>
<td>22</td>
<td>0.160</td>
</tr>
<tr>
<td>Perennial species richness</td>
<td>1.09</td>
<td>22</td>
<td>0.287</td>
</tr>
<tr>
<td>Native species growth forms</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Native herb species</td>
<td>1.05</td>
<td>22</td>
<td>0.310</td>
</tr>
<tr>
<td>Native graminoid species</td>
<td>0.37</td>
<td>22</td>
<td>0.710</td>
</tr>
<tr>
<td>Native climber species</td>
<td>0.19</td>
<td>22</td>
<td>0.850</td>
</tr>
<tr>
<td>Native shrub species</td>
<td>2.88</td>
<td>22</td>
<td>0.009</td>
</tr>
<tr>
<td>Native tree species</td>
<td>4.96</td>
<td>22</td>
<td>&lt;0.0001</td>
</tr>
</tbody>
</table>
Figure 18. Average (+ SE) differences in species diversity for: (a) total species diversity including native and alien species (b) native species, (c) alien species, (d) annual species, (e) perennial species, (f) native herb species, (g) native graminoid species, (h) native climber species, (i) native shrub species, and (j) native tree species, between fenced (n = 12) and unfenced (n = 12) sites. Asterisks denote significantly different means based on t-tests.
Overall, the vegetation community composition (i.e. native and alien plant species combined) differed significantly between fenced and unfenced plots (Table 7, Figure 19). Based on the presence/absence of species within a plot, the vegetation composition differed between fenced and unfenced plots by approximately 45% (Table 7, Figure 19 b), yet this degree of compositional dissimilarity rose to greater than 70% when the relative percentage foliage cover of species were considered within the PERMANOVA (Table 7, Figure 19 a). This indicates that fence installation modified vegetation composition primarily by altering the identity of plant species found within fenced areas. These compositional differences between fenced and unfenced sites based on the presence/absence and percentage cover of vascular plants can be visualised in Figure 19 nMDS (b, d, e), in which there is a clear separation between plots based on fencing treatment.

Separate PERMANOVA analyses for native and alien plant species revealed that fence installation significantly affected native vegetation composition but not the composition of alien plant species (Table 7, Figure 19). The dissimilarity in native vegetation composition between fenced and unfenced plots rose to approximately 83% from 63% when the percentage cover of species was considered. Therefore, this result indicates that most of the variation in compositional differences in native vegetation between fenced and unfenced plots was explained by the different types of species occupying the plots. Alien plant composition (based on species presence/absence but not percentage cover abundance) differed significantly between fenced and unfenced plots (Table 7, Figure 19). These compositional differences for alien vegetation were not as strong as those observed for native plants, as dissimilarities between fenced and unfenced sites ranged from about 53% to 60%.
Table 7. Results of the PERMANOVA analyses for the abundance and occupancy of vascular plant species between fenced (n = 12) and unfenced (n = 12) plots. Abundance is measured as percentage foliage cover per species and occupancy was measured as the presence or absence of a species within a plot. Values in bold indicate a significant result.

<table>
<thead>
<tr>
<th>Response variable</th>
<th>Data</th>
<th>Source</th>
<th>DF</th>
<th>MS</th>
<th>Pseudo-F</th>
<th>P</th>
<th>Dissimilarity index (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total species occupancy</td>
<td>Presence or absence of native and alien species combined</td>
<td>Fence treatment</td>
<td>1</td>
<td>4748.8</td>
<td>2.81</td>
<td><strong>0.001</strong></td>
<td>61.38</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Residual</td>
<td>22</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Total</td>
<td>23</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Native species occupancy</td>
<td>Presence or absence of native species only</td>
<td>Fence treatment</td>
<td>1</td>
<td>4738.5</td>
<td>2.59</td>
<td><strong>0.003</strong></td>
<td>63.16</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Residual</td>
<td>22</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Total</td>
<td>23</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Alien species occupancy</td>
<td>Presence or absence of alien species only</td>
<td>Fence treatment</td>
<td>1</td>
<td>3883.5</td>
<td>2.3</td>
<td><strong>0.047</strong></td>
<td>59.19</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Residual</td>
<td>22</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Total</td>
<td>23</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total species abundance</td>
<td>Percentage cover of native and alien species combined</td>
<td>Fence treatment</td>
<td>1</td>
<td>9949.9</td>
<td>4.86</td>
<td><strong>0.001</strong></td>
<td>72.27</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Residual</td>
<td>22</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Total</td>
<td>23</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Native species abundance</td>
<td>Percentage cover of native species only</td>
<td>Fence treatment</td>
<td>1</td>
<td>12425</td>
<td>4.62</td>
<td><strong>0.001</strong></td>
<td>82.89</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Residual</td>
<td>22</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Total</td>
<td>23</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Alien species abundance</td>
<td>Percentage cover of alien species only</td>
<td>Fence treatment</td>
<td>1</td>
<td>3977.8</td>
<td>2.62</td>
<td>0.054</td>
<td>53.93</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Residual</td>
<td>22</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Total</td>
<td>23</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
SIMPER analyses revealed that only 17 of the 93 native species sampled, contributed approximately 70% to the compositional dissimilarity between fenced and unfenced plots (Table 8). Most of these 17 species were more abundant within fenced than unfenced plots. The herbaceous groundcover *Commelina cyanea*, for example, was approximately five times
more abundant within fenced plots than unfenced plots. Three species of native plant (woody shrubs *H. populifolius*, *S. aviculare* and tree *G. ferdinandi*) were never detected within unfenced plots but were relatively abundant and common within fenced plots (possessing average foliage covers of 23%, 18% and 8%, respectively).

Five species did not differ considerably in their abundance between unfenced and fenced plots, including the shrub *H. dentata*, tree *P. undulatum*, vine *Stephania japonica*, graminoid *Lomandra longifolia* and herb *Sigesbeckia orientalis*. The tree *M. azederach* was the only native plant species totally absent from within fenced plots and only occurred within the unfenced plots.

Table 8. Summary of SIMPER analyses, showing the cumulative contributions (up to 70%) of native species to the average compositional dissimilarity between fenced and unfenced plots. Data listed under ‘fence treatment’ and included in the SIMPER analysis are average percent foliage cover abundances for each species (n = 12).

<table>
<thead>
<tr>
<th>Data source</th>
<th>Growth Form</th>
<th>Fence Treatment</th>
<th>Av. Dissimilarity</th>
<th>Diss/SD</th>
<th>Contribution (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Species</td>
<td></td>
<td>Fenced (n = 12)</td>
<td>Unfenced (n = 12)</td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Commelina cyanea</em></td>
<td>Herb</td>
<td>31.42</td>
<td>5.83</td>
<td>7.78</td>
<td>1.09</td>
</tr>
<tr>
<td><em>Homolanthus populifolius</em></td>
<td>Tree</td>
<td>23.33</td>
<td>0.00</td>
<td>6.54</td>
<td>0.83</td>
</tr>
<tr>
<td><em>Solanum aviculare</em></td>
<td>Shrub</td>
<td>18.33</td>
<td>0.00</td>
<td>5.42</td>
<td>0.7</td>
</tr>
<tr>
<td><em>Acacia maidenii</em></td>
<td>Tree</td>
<td>13.25</td>
<td>1.42</td>
<td>4.37</td>
<td>0.68</td>
</tr>
<tr>
<td><em>Oplismenus aemulus</em></td>
<td>Graminoid</td>
<td>8.08</td>
<td>5.33</td>
<td>4.16</td>
<td>1.07</td>
</tr>
<tr>
<td><em>Hymenanthera dentata</em></td>
<td>Shrub</td>
<td>12.08</td>
<td>14.67</td>
<td>3.33</td>
<td>1.29</td>
</tr>
<tr>
<td><em>Toona ciliata</em></td>
<td>Tree</td>
<td>8.33</td>
<td>0.17</td>
<td>3.18</td>
<td>1.07</td>
</tr>
<tr>
<td><em>Lomandra longifolia</em></td>
<td>Graminoid</td>
<td>11.92</td>
<td>10.42</td>
<td>3.04</td>
<td>1.18</td>
</tr>
<tr>
<td><em>Cynodon dactylon</em></td>
<td>Graminoid</td>
<td>9.00</td>
<td>3.08</td>
<td>2.92</td>
<td>0.54</td>
</tr>
<tr>
<td><em>Solanum steligeranum</em></td>
<td>Shrub</td>
<td>6.17</td>
<td>1.25</td>
<td>2.78</td>
<td>0.61</td>
</tr>
<tr>
<td><em>Sigesbeckia orientalis</em></td>
<td>Herb</td>
<td>4.33</td>
<td>4.17</td>
<td>2.68</td>
<td>1.2</td>
</tr>
<tr>
<td><em>Stephania japonica</em></td>
<td>Climber</td>
<td>7.25</td>
<td>11.25</td>
<td>2.65</td>
<td>0.95</td>
</tr>
<tr>
<td><em>Pittosporum undulatum</em></td>
<td>Tree</td>
<td>7.83</td>
<td>9.08</td>
<td>2.58</td>
<td>0.99</td>
</tr>
<tr>
<td><em>Glochidion ferdinandi</em></td>
<td>Tree</td>
<td>8.25</td>
<td>0.00</td>
<td>2.34</td>
<td>0.69</td>
</tr>
<tr>
<td><em>Pandorea pandorana</em></td>
<td>Climber</td>
<td>7.00</td>
<td>4.00</td>
<td>2.06</td>
<td>1.10</td>
</tr>
<tr>
<td><em>Ficus coronata</em></td>
<td>Tree</td>
<td>5.83</td>
<td>1.58</td>
<td>1.91</td>
<td>0.56</td>
</tr>
<tr>
<td><em>Melia azederach</em></td>
<td>Tree</td>
<td>0.00</td>
<td>7.33</td>
<td>1.68</td>
<td>0.45</td>
</tr>
</tbody>
</table>

SIMPER analyses for alien plant presence/absences indicated that 16 species contributed up to 70% to the compositional differences between fenced and unfenced plots (Table 9). In all but two cases, alien species tended to occur more frequently in unfenced compared with fenced plots. However, the grass *Bromus catharticus* was found exclusively in fenced plots, whilst *Oxalis* species occurred in an equal proportion of fenced and unfenced plots. Most of these alien species were disturbance adapted herbs and graminoids. No alien trees were detected. One notable species – the woody, thicket-forming shrub *L. camara,*
which was the focus of primary weed control prior to revegetation at this abandoned mine site – was about three times more common across unfenced compared with fenced plots.

Table 9. Summary of SIMPER analyses, showing the cumulative contributions (up to 70%) of alien species to the average compositional dissimilarity between the fenced and unfenced plots. Data listed under ‘fence treatment’ and included in the SIMPER analyses were proportion of plots occupied by each species (n = 12).

<table>
<thead>
<tr>
<th>Species</th>
<th>Growth form</th>
<th>Fence treatment</th>
<th>Average dissimilarity</th>
<th>Diss/SD</th>
<th>Contribution (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Stellaria media</td>
<td>Herb</td>
<td>0.17</td>
<td>0.75</td>
<td>3.82</td>
<td>6.45</td>
</tr>
<tr>
<td>Euphorbia peplus</td>
<td>Herb</td>
<td>0.42</td>
<td>0.92</td>
<td>3.58</td>
<td>6.04</td>
</tr>
<tr>
<td>Solanum nigrum</td>
<td>Herb</td>
<td>0.50</td>
<td>0.83</td>
<td>2.96</td>
<td>5.00</td>
</tr>
<tr>
<td>Conyza spp.</td>
<td>Herb</td>
<td>0.50</td>
<td>0.75</td>
<td>2.94</td>
<td>4.97</td>
</tr>
<tr>
<td>Delairea odorata</td>
<td>Climber</td>
<td>0.50</td>
<td>0.67</td>
<td>2.91</td>
<td>4.88</td>
</tr>
<tr>
<td>Solanum chenopodioides</td>
<td>Herb</td>
<td>0.08</td>
<td>0.58</td>
<td>2.89</td>
<td>4.79</td>
</tr>
<tr>
<td>Sida rhombifolia</td>
<td>Herb</td>
<td>0.33</td>
<td>0.50</td>
<td>2.72</td>
<td>4.59</td>
</tr>
<tr>
<td>Ageratina adenophora</td>
<td>Herb</td>
<td>0.25</td>
<td>0.42</td>
<td>2.63</td>
<td>4.44</td>
</tr>
<tr>
<td>Oxalis spp.</td>
<td>Herb</td>
<td>0.33</td>
<td>0.33</td>
<td>2.61</td>
<td>4.41</td>
</tr>
<tr>
<td>Famaria muralis</td>
<td>Herb</td>
<td>0.67</td>
<td>0.83</td>
<td>2.54</td>
<td>4.30</td>
</tr>
<tr>
<td>Suchys arvensis</td>
<td>Herb</td>
<td>0.25</td>
<td>0.50</td>
<td>2.49</td>
<td>4.21</td>
</tr>
<tr>
<td>Lantana camara</td>
<td>Shrub</td>
<td>0.17</td>
<td>0.50</td>
<td>2.43</td>
<td>4.10</td>
</tr>
<tr>
<td>Sonchus olearceus</td>
<td>Herb</td>
<td>0.33</td>
<td>0.33</td>
<td>2.23</td>
<td>3.77</td>
</tr>
<tr>
<td>Bromus catharticus</td>
<td>Graminoid</td>
<td>0.42</td>
<td>0.00</td>
<td>2.04</td>
<td>3.45</td>
</tr>
<tr>
<td>Araujia sericifera</td>
<td>Climber</td>
<td>0.17</td>
<td>0.33</td>
<td>2.02</td>
<td>3.41</td>
</tr>
<tr>
<td>Galinsoga parviflora</td>
<td>Herb</td>
<td>0.17</td>
<td>0.33</td>
<td>1.96</td>
<td>3.31</td>
</tr>
</tbody>
</table>

3.3 Aim 3: Diversity and activity of vertebrate herbivores across different habitat treatments at the revegetation site.

3.3.1: Differences in activity of W. bicolor and C. timorensis between treatment types.

Six species of vertebrate animal were observed from 136 capture events over the 28 days of camera deployment across all three habitat treatments, including: 84 W. bicolor, 39 C. timorensis, nine crimson rosellas (Platycercus elegans), one red fox (Vulpes vulpes), two kookaburras (Dacelo novaeguineae) and one Australian raven (Corvus coronoides). See appendix 1 for examples of photographs of W. bicolor and C. timorensis. The probability of detecting an animal did not vary among habitat treatments ($\chi^2 = 0.181$, DF = 2, P = 0.9). For the treatments of forest, grass and revegetation, an average of nine, ten and nine animals were observed each fortnight. This finding suggests that the sites in which cameras were deployed experienced relatively similar levels of activity of all species of animal.

A second analysis was performed that combined W. bicolor and C. timorensis only (excluding birds and red fox), as these were considered the principal herbivores affecting seedling establishment and growth across the regeneration area. Of these two herbivore species, the probability of detecting an animal did not differ significantly between habitat
treatments (Table 10). Within the revegetation, grassy and forest habitats, a total of 50, 37 and 36 large herbivores were reported. When *W. bicolor* and *C. timorensis* were analysed separately for their activity, the number of animals captured did not differ significantly between habitat treatments. Of the 84 *W. bicolor* individuals recorded over the 28 days of camera deployment, 43 occurred within revegetation sites, 16 in the grassy areas and 25 occurred in the rainforest. Likewise, of the 39 *C. timorensis* recorded, seven sightings occurred in revegetation areas, 21 in grassy open areas, and 11 in the rainforest thicket (Figure 20).

**Table 10.** Summarises the results from a Wilcoxin test for a significant difference of all or individual species of herbivore (*W. bicolor* and *C. timorensis*) reported within each treatment type (Gras, revegetation or forest; *n* = 18 locations).

<table>
<thead>
<tr>
<th>Species</th>
<th>DF</th>
<th>Chi Square</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total</td>
<td>2</td>
<td>0.18</td>
<td>0.91</td>
</tr>
<tr>
<td><em>W. bicolor</em></td>
<td>2</td>
<td>0.11</td>
<td>0.94</td>
</tr>
<tr>
<td><em>C. timorensis</em></td>
<td>2</td>
<td>2.52</td>
<td>0.28</td>
</tr>
</tbody>
</table>

**Figure 20.** Presents the average (±SE) number of each animal (wallaby symbol = *W. bicolor* and deer = *C. timorensis*) being recorded per fortnight within each treatment type (Grass = lawn symbol, revegetation = fenced seedling symbol and forest = tree symbol; *n* = 18 locations).
3.3.2: Differences in the time spent by *W. bicolor* and *C. timorensis*

The time spent by *W. bicolor* per animal per site differed significantly among the three habitat treatments ($\chi^2 = 6.22$, DF = 2, $P = 0.045$), but not for *C. timorensis* ($\chi^2 = 0.96$, DF = 2, $P = 0.62$) (Table 11, Figure 21). *W. bicolor* spent up to five times longer within revegetation areas than the established rainforest sites. However, the time spent by *W. bicolor* in revegetated areas was not significantly different to grassy areas (Table 11). Although there was no difference in duration of occupancy among habitats for *C. timorensis*, it is important to note that in more than 30% of detection events the deer spent between 1 and 10 minutes foraging in grassy treatment but only a maximum of 1.5 minutes in either the rainforest or revegetated sites. Therefore, it is likely that deer spend greater amounts of time in grassy habitats, but limits to spatial replication of camera traps and high temporal variation in vertebrate activity meant that I could not detect those differences with statistical significance.

**Table 11.** Summarises the results from the pairwise Wilcoxin method, to determine if each species of herbivore spent a different amount of time in each treatment. $P$ – values denoted in bold indicate significant differences between a pair of treatments.

<table>
<thead>
<tr>
<th>Species</th>
<th>Level – Level</th>
<th>Std error difference</th>
<th>Z</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Wallabia bicolor</em></td>
<td>Reveg – Grass</td>
<td>4.15</td>
<td>-0.77</td>
<td>0.44</td>
</tr>
<tr>
<td></td>
<td>Reveg – Forest</td>
<td>4.32</td>
<td>1.99</td>
<td><strong>0.046</strong></td>
</tr>
<tr>
<td></td>
<td>Grass – Forest</td>
<td>3.36</td>
<td>2.17</td>
<td><strong>0.029</strong></td>
</tr>
<tr>
<td><em>Cervus timorensis</em></td>
<td>Reveg – Grass</td>
<td>3.03</td>
<td>0.20</td>
<td>0.84</td>
</tr>
<tr>
<td></td>
<td>Reveg – Forest</td>
<td>2.45</td>
<td>1.05</td>
<td>0.29</td>
</tr>
<tr>
<td></td>
<td>Grass – Forest</td>
<td>3.14</td>
<td>0.62</td>
<td>0.53</td>
</tr>
</tbody>
</table>
3.3.3: Patterns of *W. bicolor* and *C. timorensis* activity throughout a 24 hour day.

Both *W. bicolor* and *C. timorensis* were most often detected before sunrise and after sunset. *C. timorensis* was never detected on any of the 18 cameras between 0900 and 1800 hours, whilst *W. bicolor* was detected at relatively similar levels throughout daylight hours (Table 12, Figure 22). Likelihood of detection for *C. timorensis* varied significantly across the major time zones of day, night and crepuscular ($\chi^2 = 1.96$, DF = 2, $P < 0.0001$). In contrast, *W. bicolor* detection did not vary significantly across the day although there was a trend toward higher detection pre-dawn and post-dusk ($\chi^2 = 4.92$, DF = 2, $P = 0.085$). *W. bicolor* was detected 5 times more often during the night time compared to the daylight hours. *C. timorensis* was detected 14 times more often at night, compared to daytime hours, and four times more often compared to either dawn or dusk periods.

Table 12. Summarises the results from the pairwise Wilcoxin method, to determine if each species of herbivore was active during a particular time of day. $P$ – values denoted in bold indicate significant differences between a pair of treatments.

<table>
<thead>
<tr>
<th>Species</th>
<th>Level – Level</th>
<th>Std error difference</th>
<th>Z</th>
<th>$p$</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Wallabia bicolor</em></td>
<td>Night – Day</td>
<td>3.19</td>
<td>2.04</td>
<td><strong>0.042</strong></td>
</tr>
<tr>
<td></td>
<td>Night – Crepuscular</td>
<td>3.27</td>
<td>1.56</td>
<td>0.12</td>
</tr>
<tr>
<td></td>
<td>Day – Crepuscular</td>
<td>2.94</td>
<td>-0.59</td>
<td>0.55</td>
</tr>
<tr>
<td><em>Cervus timorensis</em></td>
<td>Night – Day</td>
<td>3.01</td>
<td>3.63</td>
<td><strong>0.0003</strong></td>
</tr>
<tr>
<td></td>
<td>Night – Crepuscular</td>
<td>3.13</td>
<td>3.11</td>
<td><strong>0.0019</strong></td>
</tr>
<tr>
<td></td>
<td>Day – Crepuscular</td>
<td>1.92</td>
<td>-0.96</td>
<td>0.34</td>
</tr>
</tbody>
</table>
Figure 22 Presents the total number of camera detections per species (C. timorensis and W. bicolor) depending on the time of day. Daytime was measured between 06:30 and 17:00h (10.5 hours), Night time was between 18:30 and 05:00h (10.5 hours), and crepuscular hours were from 05:00 to 06:30h and 17:00 to 18:30h combined (3 hours).
4. Discussion

This honours study aimed to determine the effectiveness of different restoration strategies for restoring an endangered rainforest community in the presence of both a native and invasive vertebrate herbivore. The study used a combination of intensive site surveys of individual seedlings and community vegetation to evaluate the effectiveness of different protective treatments in deterring browsers. The study also deployed camera traps to observe the activity of vertebrate herbivores on site and therefore ascertain this to the observed patterns of browse damage.

This study broadly found that the establishment and growth of native seedlings used to revegetate disturbed landscapes can be severely limited by herbivore pressure. However, this study is one of the first to clearly show that herbivore effects can be ameliorated by a combination of different protective treatments (see section 3.1) and that the differential effectiveness of those treatments vary among different plant species. It was also found that protecting multiple species simultaneously using herbivore exclusion plots has the emergent benefit of enhancing the overall diversity of the vegetation community through passive regeneration of native herbs, grasses, vines, ferns and shrubs (see section 3.2). The third major finding of this study was that native vertebrates (*W. bicolor*) likely play as significant a role as feral vertebrate herbivores (*C. timorensis*) in inhibiting the regeneration of native vegetation in revegetated forests (see section 3.3). As will be discussed below, a consideration of the differential effectiveness of protective treatments, variation in impacts among planted seedlings, and exclusion of vertebrate herbivores will ensure that restoration can proceed in a cost-effective manner to achieve biodiversity gain in endangered rainforest communities.

4.1 Effectiveness of the four-point scale for browse damage

The four-point score was used by local restoration practitioners to rapidly assess the condition and health of plants occurring on the restoration site. As with many rapid assessment tools for vegetation condition (e.g. Cook, Wardell-Johnson et al. 2010), the four-point score was a visual, low-resolution estimate of browsing damage. However, I found that the rapid visual score very strongly correlated with the measured proportion of branches browsed, which was a more accurate but time-consuming measure of browse damage. Specifically, each damage score (1 – 4) was correlated with a proportion of branches
browsed, which was significantly different from each other browsing category. This finding indicates that, where the condition of plants must be evaluated quickly and accurately such as in future restoration sites, the use of the four-point damage score can be used with a high level of confidence. However, ensuring the accuracy of the four-point score requires strict adherence to the defining qualities of each category of damage in addition to sufficient training of restoration practitioners before they deploy the score in the field to assess impacts of herbivory. Should the four-point score of damage be used, a significant amount of time can be saved, which will prove extremely useful in circumstances where large quantities of plants must be surveyed such as in restoration projects much like this. Further, this reduction in time will reduce costs which managers could expend on other tasks such as weed management, purchasing more tube-stock or more effective guards.

4.2 Evaluating the effect of guard type in deterring herbivory

The type of guard used in protecting plants was found to have a significant negative effect on the level of browsing damage. Plants which were unprotected suffered more than two times higher browsing damage compared to plants which were surrounded by wire guards or fenced plots. Also, plants in corflute guards suffered approximately 10% less browsing damage than unprotected plants, emphasising the idea that some form of guard, even a short plastic corflute guard, does significantly reduce browse damage. Corflute guards, however, were not as effective as individual wire guards or fenced plots. These results correlate well with the wider literature. While numerous studies have used exclosure experiments to evaluate the impact of browsers such as deer, rabbits and W. bicolor on forest restoration (Montague 1993, Cabin, Weller et al. 2000, Cabin, Weller et al. 2000, Zamora, Gómez et al. 2001, Bellingham and Allan 2003), fewer evaluate the relative effectiveness of different types of individual plant guards in deterring browsing (Mayhead and Jenkins 1992, Montague 1993, Ward and Stephens 1995), and far fewer have evaluated the effectiveness of individual guards compared to exclosures (Kittredge, Kelty et al. 1992). Specifically, the presence of a guard significantly reduces or eliminates browsing damage on plants, such that plant height (the measure used to evaluate response to browse exclusion) is considerably greater compared to unguarded plants (Baer 1980, Mayhead and Jenkins 1992, Montague 1993, Rooney and Waller 2003, Lai and Wong 2005). Much like the results in this thesis, shorter guards such as the corflute guards (<50cm) proved far less useful in protecting plants from browsers, in comparison to taller guards as was found by Bendfeldt, Feldhake et al.
In these studies, plants protected by wire, mesh or plastic guards that were more than 1m in height, grew taller and suffered lower rates of browsing compared to shorter guards (wire/plastic mesh) which were typically less than 60cm in height. Therefore guards must be of sufficient height to deter herbivore browsing (i.e 2 m for *C. timorensis*). Kittredge, Kelty et al. (1992) also found that plants in exclosures were often shorter than their individually protected counterparts, and attributed this to the competition with understory plant species that plants in exclosures often faced. Disappointingly Kittredge, Ketly et al. (1992) did not report browsing damage, and only focussed on growth over a period of years and therefore does not address the impact of browsing as opposed to competition with other plants.

Overall, I consider that fenced plots are far superior at achieving goals of restoration and are far more economical in protecting plants than either wire guards or corflute guards. As was found in section 3.2, fenced plots enclose higher densities of plants, enhance vegetation structural complexity and foliage cover, and allow the unassisted regeneration of herbs shrubs, graminoids, and climbers, which neither corflute nor wire guards provide. Furthermore, fenced plots are highly economical and cost effective to construct as they use far less guard material (stakes or wire) per plant in comparison to individually protecting each plant. Although corflute guards and wire guards do significantly reduce browsing compared to unguarded plants, they often require adjustments (i.e raising the guards, or replacing corflute guards with wire guards) as plants continue to grow taller or produce lateral branches which often become exposed to browsing (J Reay, J Hudson and R Scarborough personal communications). This increases costs associated with purchasing new guards, or investing labour time that could be invested into restoration activities elsewhere. I believe, therefore, that future restoration should focus on establishing fences which are at least 2m in height and properly secured using wooden stakes around the entire fence perimeter which would prevent entry by vertebrate herbivores rather than installing individual wire guards.

It should be noted that most studies which have been compared to this thesis measured plant height before and after applying a protection treatment, as a proxy for browse damage (Kittredge, Kelty et al. 1992, Ret and Kathleen L. Shea 1998, Conner, Inabinette et al. 2000, Ward, Gent et al. 2000, Bendfeldt, Feldhake et al. 2001, McCreary and Tecklin 2001, Sweeney, Czapka et al. 2002, Ponder Jr 2003, Davis and Graeme 2010). However a study similar to mine performed by Keith and Pellow (2005) examined the impact of *C. timorensis* on native vegetation. It qualitatively determined signs of damage (browsing or...
bark stripping) by *C. timorensis* on several species of shrub and tree species and only quantitatively assessed browsing damage (as a percentage of defoliation or branch tip removal) of one species of plant (*Syzygium paniculatum*). This honours study builds upon the work by Keith and Pellow (2005) in examining how the proportion of branches browsed by herbivores varies among multiple species using more than one protective treatment. However, future research will be needed to determine what the longer term consequences of such browsing damage are for plant growth beyond the seedling stage.

4.2.1 *The role of plant identity in influencing browse susceptibility*

It was found that the effects of the different protective treatments on herbivore damage vary among different plant species. Some species showed conflicting findings, with higher levels of browsing observed for protected plants compared with unprotected plants. Unguarded specimens of *G. semiglauca, G. ferdinandi, P. revolutum* and *Ficus spp.* suffered levels of browsing which were either equal to, or lower than plants which were protected by wire guards or feral fences. Unguarded specimens of most species were found on the revegetation area, however unguarded specimens of the above species had to be located from outside the study site and within the rainforest thicket. Browsers including *C. timorensis* and *W. bicolor* are thought to preferentially feed in forest openings similar to the study site due to the abundance of grass that often occurs (Blymyer and Mosby 1977, Davis, Coulson et al. 2008, Kuijper, Cromsigt et al. 2009). This, therefore, may explain why it was not possible to find unguarded specimens on the study site as browsers would have prevented the unassisted regeneration of these specimens (Blymyer and Mosby 1977, Casabon and Pothier 2007, Kuijper, Cromsigt et al. 2009) and also why the unguarded specimens found in the rainforest thicket had lower rates of browsing.

Browsers (*C. timorensis* and *W. bicolor*) seemed to show preference for specific species of plants and this was discerned when examining rates of consumption for unguarded plants. For example, eight species when unguarded suffered browsing damage that exceeded 75%, in contrast to five other species which were browsed at less than 10% across all protective treatments. These preferences of browsers to feed on specific species of plant could be attributed to the three common plant properties which determine the attractiveness of the foliage to herbivores. These include nutrients (i.e nitrogen, and phosphorus for growth of muscle and bone tissue) (Robbins 1993), fibre (including cellulose, hemicellulose and
lignin) which is ingestible to ungulates and reduces the proportion of digestible fibre in forage (Robbins 1993) and secondary metabolites, (tannins, monoterpenes etc) which broadly reduce palatability by interfering with digestive enzyme activity (Robbins, Hanley et al. 1987, Robbins 1993). In addition to the species of plants, a plants palatability also varies spatially according to the different parts of the plant (stems, bark) or the relative age of these tissues, as they influence the chemical composition (Blair, Short et al. 1977), relative digestibility and therefore palatability of the plant to browsers (Hjeljord, Sundst et al. 1982).

Nutritionally, studies do not agree on the role of carbohydrate or protein content of forage in browser preference. While most studies report that deer (roe/mule/black-tailed) preferentially select forage that has a higher concentration of protein and carbohydrate (McArthur, Robbins et al. 1993, Tixier, Duncan et al. 1997, Dostaler, Ouellet et al. 2011), others found no correlation with between forage selection and protein content (Semiadil, Barry et al. 1995, Tixier, Duncan et al. 1997, Forsyth, Coomes et al. 2002, Forsyth, Richardson et al. 2005). Forage selection was however, often strongly determined by the presence and concentration of other chemicals including fibre and secondary metabolites. In all circumstances, an increase in the concentration of fibre consistently decreased the preference for forage as it decreased palatability (Forsyth, Coomes et al. 2002, Forsyth, Richardson et al. 2005, Sauvé and Côté 2007, Dostaler, Ouellet et al. 2011). Likewise, the presence of secondary metabolites can cause browsers to select forage that is nutritionally poor but contains less fibre and secondary metabolites (Semiadil, Barry et al. 1995). Overall, an increase in the concentration of tannins, terpenes, phenolic compounds and monoterpenes consistently reduced the palatability of browse to browsers (Duncan, Hartley et al. 1994, Semiadil, Barry et al. 1995, Alm, Birgersson et al. 2002, Vourc'h, De Garine-Wichatitsky et al. 2002). The interrelation between plant chemistry and palatability is well known at fine scales (i.e. plant-animal bodies) but there is no knowledge of the importance of plant secondary defences or constitutive chemical structures on vegetation response to suites of herbivores at the community scales, especially not in the context of vegetation restoration ecology.

Although *W. bicolor* have shown preference for specific species of tree or shrub over others (Montague 1994), the underlying cause for these observed differences remain unknown. Although the results from camera traps (Section 3.3) suggest that *W. bicolor* and *C. timorensis* actively use the revegetated site, we cannot definitively determine if all the damage was by them. Therefore, further research should focus on first better understanding
the differential role of *W. bicolor* and *C. timorensis* browsing and how they forage for plants within the IESR community, and also the chemical composition of the plants of the Illawarra Subtropical rainforest, to explain the observed difference in browsing intensity among plant species by either herbivore species.

### 4.2.2 Magnitude of bark stripping by *C. timorensis*

Many plants sampled in this study were seedlings which were less than 1.5m in height and lacked a prominent main trunk which largely made these plants unavailable for antler rubbing or bark stripping by deer (Bilney 2013). Although the rate of bark stripping in this study was very low, it could pose a threat to the broader forest ecosystem if the population of deer remains uncontrolled as large trees could be damaged and killed (Verheyden, Ballon et al. 2006). The focus of restoration in the presence of ungulates should focus on the protection of seedlings from herbivory using fences first to ensure juveniles can survive and mature, after which they should be protected from bark stripping.

### 4.3 Effects of feral fence exclosures on vegetation structure and community composition

Unfenced control plots contained significantly higher amounts of bare soil, and lower amounts of litter than fenced plots. I predict that the exclusion of herbivores such as native *W. bicolor* and introduced *C. timorensis* are responsible for these observed differences. Herbivory by introduced herbivores is well documented to increase the cover of bare soil, by the removal of groundcover through feeding, as well as through trampling of soil (Alverson, Waller et al. 1988, Keith and Pellow 2005, Mohr, Cohnstaedt et al. 2005, Knight, Dunn et al. 2009, DiTommaso, Morris et al. 2014, Pedersen, Andreassen et al. 2014, Relva, iacute et al. 2014). The higher amounts of leaf litter in fenced plots is likely to due to the significantly higher amounts of ground cover (e.g. dead grass leaves) and shrub vegetation in the absence of vertebrate herbivory, resulting in a higher rate of litter-fall to the soil surface.

The installation of fences also had a significant effect on the cover of the different vegetation structures. Ground cover and shrub cover were significantly higher in the fenced plots compared to the unfenced plots, however the already-established mesophyll tree cover, upper canopy or total canopy cover did not differ among plots. The differences in the two vegetation structures as well as bare soil abundance among fenced and unfenced plots could be attributed to (i) the planting strategy used on site, (ii) weeding processes, and (iii)
herbivory pressure. Firstly, the idea that more seedlings were planted inside fences and therefore explains this discrepancy in the vegetation cover of shrub or ground cover does not hold, as a data base held by Landcare Illawarra indicates that more than 3000 seedlings, including shrub, trees and grasses were planted in the study site, however just 364 were surveyed in the 12 fenced plots. Evidently, most of the plants have been planted outside of the feral fenced plots in corflute or wire guards which indicates that these plants were present, but have not grown as much. Secondly, manual and chemical weeding occurs both inside and outside fenced plots equally (J Reay, J Hudson and R Scarborough personal communications). The sustained removal of weeds in fenced plots in the absence of browsers could have worked to benefit the planted seedlings and allowed the recruitment of other species, which were then able to better occupy and fill the shrub and ground cover vegetation layers (Randall 1996, Claeson and Bisson 2013). Browsing by C. timorensis and W. bicolor within their browse tiers of approximately 2m and 1m respectively more strongly support the current findings. This pattern of significantly reduced cover of groundcover and understory vegetation in the presence of introduced deer is highly consistent with numerous studies (Wardle, Barker et al. 2001, Vázquez 2002, Allison and Vitousek 2004, Suzuki, Miyashita et al. 2008, Forsyth and Davis 2011, Chollet, Bergman et al. 2015, Crowther, Ortac et al. 2016). Bare soil may have become reduced in fenced plots as a function of the elimination of trampling by herbivores as well as an increase in groundcover growth and litter abundance that covered the soil surface. Previous studies have shown that exclusion of herbivores and increased plant growth may also improve soil stability, nutrients and water retention capacity (Chen, Huang et al. 2007), or providing more suitable opportunities for the recruitment of native vegetation through habitat amelioration processes (Cabin, Weller et al. 2000, Gómez-Aparicio, Zamora et al. 2008).

Total plant species diversity between fenced and unfenced plots did not differ significantly, however fenced plots contained a significantly higher diversity of native plants, a greater diversity of perennial species, and lower diversity of annual species compared to unfenced plots. The diversity of native plants according to their growth form was also found to differ between fence treatments, with target species of restoration (trees and shrubs) being significantly more diverse in fenced than unfenced plots, while graminoid, herb and climber species did not differ significantly. The observed patterns in the abundance and diversity of native species including their growth form, can help predict changes to community assemblages in the presence of a vertebrate herbivores.
When considering the growth form of native plants, deer browsing is restricted to a browse tier of approximately 2m (Bilney 2013). Plants which occur in this tier will be targeted more heavily and explains broadly why shrub and tree seedlings species declined rapidly when they were unfenced (Smale, Hall et al. 1995, Husheer, Coomes et al. 2003, Barrios-garcia, Relva et al. 2012). It is likely that the browsers did not favour ground layer vegetation, resulting in similar levels of diversity between the two treatments. This result was unexpected because previous studies have shown that unfenced areas with high activity of vertebrate herbivores tend to have a higher diversity and abundance of grasses and ground layer herbs because the decline in shrub and tree abundance allows the penetration of more light and soil disturbance to the forest floor (Gill and Beardall 2001, Rooney and Waller 2003, Rooney 2009, Takatsuki 2009). Germination of seeds from grasses carried in the dung of deer is often higher than for shrub or tree species, which also increases the abundance of graminoids relative to shrub or trees in unfenced areas (Rooney and Waller 2003, Takatsuki 2009, Roberts 2012).

All native species except for Hymenanthera dentata, Stephania japonica, Pittosporum undulatum and Melia azederach were more abundant in fenced compared to unfenced plots. These findings indicate that the majority of native plant species are not tolerant of browsing by W. bicolor or C. timorensis. While most native species were more abundant in fenced than unfenced sites, some such as Homalanthus populifolius and Glochidion ferdinandi were totally absent in the unfenced site, suggesting that they are not tolerant to browsing or trampling disturbance at any level. In contrast, M. azederach was totally absent in fenced plots. The increase in the abundance of a few native species outside the fenced plots suggests that they are more disturbance adapted. For example, P. undulatum and S. japonica were never planted as tube stock and were more abundant in unfenced plots. While M. azederach were planted as tube stock, the majority recorded in the unfenced plots were seedlings that had emerged unassisted (<10cm height) and showed no evidence of browse damage. Although P. undulatum suffered high amounts of browsing, the species appears to be tolerant and capable of regenerating in the presence of browsers. The increase in the amount of bare soil caused by the herbivores feeding/treading, coupled with the increased amount of light provided with the decrease in vegetative cover of shrub and ground cover, may provide optimal conditions for the germination of seedlings of these species (Nomiya, Suzuki et al. 2003). Further, M. azederach and S. japonica may possess additional defences against herbivory such as a low nutritive content, high fibre content or contain secondary
metabolites, making them unfavourable forage (Hjeljord, Sundst et al. 1982, Bryant, Provenza et al. 1991, Forsyth, Richardson et al. 2005). Future study of the chemical composition of these tolerant species would help determine the reasons for their increased survival in the presence of herbivores.

Alien plant species appeared highly tolerant and even seemed to thrive in the presence of herbivores. All species of invasive plant species except Bromus catharticus occurred in greater frequency within unfenced as opposed to fenced plots. Most of these species were herbaceous. Likewise, there was a trend towards a higher diversity of alien plant species in unfenced as opposed to fenced plots. Herbs (particularly annual herb species) tend to proliferate in areas of high disturbance such as in the presence of overabundant herbivores, and is often due to their quick growth rate and short life histories (Iravani, Schütz et al. 2011). Similarly, several species of deer are known dispersers of the seed of alien plant species in their native or foreign home range, and this could explain why there are a greater occurrence of several alien plant species in unfenced compared to fenced sites (Malo and Suárez 1998, Bartuszevige and Endress 2008, Williams, Ward et al. 2008). This study shows that in the absence of protective fences, regenerating native vegetation may be limited by multiple interacting threatening processes – namely alien plant invasion and herbivore activity. Further research should examine whether secondary alien plant invasion is facilitated by vertebrate herbivory and, in turn, whether the presence of such secondary weeds boosts the activity of herbivores, resulting in synergistic negative feedbacks to resident native seedlings.

4.4 Differential activity and occupancy of C. timorensis and W. bicolor across the study site

Both species appeared to be nocturnal in their behaviour, choosing to remain inactive for most of the daytime hours. W. bicolor, unlike C. timorensis was reported at consistently low frequencies throughout the daytime hours, which suggests this species still forages throughout the day, in contrast to C. timorensis which was strictly nocturnal. Cervids are well known to be most active at nightfall (Jackson, White et al. 1972, Kuijper, Cromsigt et al. 2009). Both species are herbivores and prey animals. Their elusive behaviours may be an adaptation to avoid possible predation which could occur during the daylight hours.

Higher number of detections of W. bicolor occurred in the revegetation treatment, less in the forest treatment, and least in the grassy treatment. In contrast, there was double the detections of C. timorensis in the grassy treatment, compared to the forest and revegetation
sites. While statistical analyses were unable to detect any significant difference in the activity of each herbivore within each habitat type, these observations suggest that the invasive herbivore (*C. timorensis*) occurred least often in the revegetation treatment, while native *W. bicolor* were highly active where revegetation was occurring. This result suggests that *W. bicolor* are a greater threat to restoration compared with other common vertebrate browsers, particularly because I found that they also foraged for longer durations in revegetation areas compared to other habitat types. This result was unexpected because informal conversations with local land managers, property owners and local residents (as well as presence of hoof prints and scat) indicated that deer were the most active resident vertebrate that caused the greatest damage to vegetation across the region. However, *C. timorensis* is larger, capable of eroding soil, ranges over larger spatial scales throughout the year and damage matured woody vegetation through bark stripping in addition to consuming larger quantities of vegetation. I also found that when detected the deer spend longer foraging for food than the native herbivores. This study was also performed over fine temporal scales (hours for several weeks) and may not represent seasonal variation in deer activity. It is therefore uncertain what the differential importance of these two herbivore species are on vegetation restoration. Exclosures which allow native herbivores such as *W. bicolor* passage, but exclude large ungulate herbivores such as *C. timorensis* (Bennett 2009) would be particularly useful to determine the role of native or introduced herbivores in forest restoration in the Illawarra subtropical rainforest community.

Despite the limited temporal scale of this honours project, previous research has shown that clearings in forests (such as grassy openings and revegetated areas examined in this honours study) are often attractive to herbivores including deer, elk and caribou (Blymyer and Mosby 1977, Lyon and Jensen 1980, Moriarty 2004, Casabon and Pothier 2007) as well as some macropods including *W. bicolor* and *Macropus rufogriseus* (Johnson 1987, Di Stefano, Anson et al. 2007). Upon visiting these clearings, herbivores tend to spend longer foraging there (e.g. deer, Kuijper, Cromsigt et al. (2009)) and at higher densities (e.g. wallabies, Di Stefano, Ansen et al. 2007). In the context of restoration and forest rehabilitation, it is thus more important to install protective treatments (e.g. feral fenced plots) in cleared areas with juvenile seedlings, although it may be necessary to set up trunk protectors in established forests to protect mature shrubs and trees from bark stripping.

No other study has yet compared the relative activity or time spent by *W. bicolor* or *C. timorensis* in either forest openings or across a restoration site. The findings from this study
highlight firstly the threat that *C. timorensis* has on the restoration activity, and also the unexpected negative impacts that a native resident (*W. bicolor*) may pose on regenerating plants. If the activity of either vertebrate herbivore remains unmanaged, the impact they may have on forest restoration could be far greater than what was observed in this study into the future. This poses a serious cost burden and risk to viability of such restoration projects when management resources and funding are scarce. Long term exclosures such as feral fenced plots may provide a long-term option for vertebrate management (i.e. exclusion) to enable vegetation establishment.

5. Management implication and future recommendations

To maximise the effectiveness of a restoration project in the presence of vertebrate herbivores, the selection and pairing of a specific guard type with a species of plant must be carefully selected. Individual plant species were found to demonstrate different susceptibilities to browsers. Scarce management resources may be used more efficiently by not installing a protective guard around plant species that are not browsed heavily by vertebrate herbivores (e.g. *D. cunninghamii, M. azederach*), whilst reallocating more of those guards to protecting the most vulnerable species (*H. populifolius, R. howittiana* and *G. ferdinandi*). Although corflute and wire guards were effective at deterring browsing, neither of these guards are advised to protect plants from vertebrate herbivores in future restoration efforts, especially corflute guards. Both corflute and individual wire guards are only capable of protecting one plant at a time, which is extremely cost-inefficient when restoration sites contain numerous seedlings. As plants grow in height and width, these individual guards often become inadequate in deterring browsing as they need to be raised or widened to prevent browsing of apical or lateral shoots which is unlike exclosure fences. These adjustments were commonplace on this study site and represent a significant labour burden to attendant restoration practitioners, which could be allocated to other tasks, such as weed control. In contrast, exclosure fences are more economical as they enclose numerous plants simultaneously, as well as facilitate the unassisted regeneration of other native plant species, which in turn boosts overall vegetation diversity.

This study addressed in detail the challenges of restoration in the presence of an introduced vertebrate herbivore (*C. timorensis*). Whilst the specific role that *C. timorensis* play in limiting restoration at this site could not be determined, it is nonetheless clear that
herbivory (including by native animals) significantly hinders seedling and vegetation community reestablishment and thus ecosystem recovery following landscape disturbance. In the short term, management protocols must consider secondary disturbance by vertebrate herbivores as a key process limiting restoration success. However, longer term monitoring will be required to determine what effects herbivore pressure has on the trajectory of community recovery and ecosystem rehabilitation.

Key questions for future research should include: what mechanisms (e.g. secondary plant metabolites) underpin differential browsing pressure on regenerating seedlings, what are the synergistic roles of herbivores and secondary weed invasion on native vegetation restoration and whether (in the long term) restored vegetation approaches similar diversity and function to the desired ecosystem.
6. References


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Appendix 1