A monitoring protocol for assessing changes in deer relative abundance and deer impacts at Lake Tyers, Victoria, Part 1:

Experimental design and survey methodology rationale

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Report prepared for:

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1.2 ABBREVIATIONS

BACI	before-after/control-impact
EVC	Ecological Vegetation Class
GPS	Global Positioning System
ha	Hectare
SD card	Secure Digital card

1.3 GLOSSARY

Experimental units	An entity that can be assigned a treatment
Vegetation plot	Permanently marked area that both deer and native herbivores can access, in which vegetation surveys are to be conducted
Reference site	Site at which no deer control will occur and is used for comparison with Treatment site
Sampling unit	Random samples replicated within and between experimental sampling units
Spatial sampling	Distribution of sample observations or survey locations in space within a defined area
Temporal sampling	Distribution of surveys over time
Treatment site	Site at which deer control by ground-shooting will be conducted

1.4 DISCLAIMER

This report has been prepared on behalf of and for the exclusive use of the Clients, and is subject to and issued in connection with the provisions of the agreement with the Clients. The experimental design and field protocol have been developed with the intention of meeting the objectives of the Lake Tyers Deer Management Trial Implementation Plan but their performance may be restricted by the limitations imposed by the program conditions. This document is not for public distribution and must not be reproduced without permission of the Clients. The author accepts no liability or responsibility for, or in respect to, any use of or reliance upon this report, by any third party.

1.5 CITATION

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2 CONTENTS

2.1 INTRODUCTION

The introduction of deer to Australia began in the mid-1800s and six deer species are now established in the wild: sambar (*Rusa unicolor*), red deer (*Cervus elaphus*), rusa deer (*Rusa timorensis*), fallow deer (*Dama dama*), chital (*Axis axis*) and hog deer (*Axis porcinus*) (Bentley 1998). Deer can have substantial deleterious impacts on natural ecosystems (Cote et al. 2003, Dolman and Wäber 2008). Although information in Australia is limited, there is increasing evidence that deer are causing impacts on natural environments (Davis et al. 2016) and in Victoria, reduction in biodiversity of native vegetation by Sambar has been listed as a Potentially Threatening Process for the reduction in biodiversity of native vegetation under the *Flora and Fauna Guarantee Act 1988* (FFG Act).

Deer populations have increased in the Lake Tyers area (c. 39,000 ha), East Gippsland, Victoria, to levels that are causing concern (Gunaikurnai Traditional Owner Land Management Board 2017). The Lake Tyers area contains a diverse mix of Ecological Vegetation Classes (EVC). Of particular importance is the presence of Littoral Rainforest, listed as critically endangered under the commonwealth *Environment Protection and Biodiversity Act 1999* (EPBC Act) known to be threatened by deer (TSSC 2008). The area is comprised of mixed land tenures including public land managed within parks and and game reserves, and private land including farmland and the Lake Tyers Aboriginal Reserve (Gunaikurnai Traditional Owner Land Management Board 2017). Damage to native vegetation in the area is being caused by deer browsing, antler activities, wallowing and trampling (Gunaikurnai Traditional Owner Land Management Board 2017).

Sambar occur at Lake Tyers Park. There is no self-sustaining population of hog deer present at Lake Tyers Park, although hog deer occur in Ewing Morass Wildlife Reserve, and there are low density populations to the west around Lakes Entrance and Boole Poole Peninsula, and to the east at the mouth of the Snowy River, and individuals occasionally pass through the Lake Tyers area (B. Mills, Trust for Nature 2018, pers. comm.). A small number of fallow deer have been observed to the far north of Lake Tyers around Nowa Nowa and also in farmland in the upper Toorloo arm, however, fallow deer do not occur within Lake Tyers Park (B. Mills, Trust for Nature 2018, pers. comm.). Concerns about deer impacts at Lake Tyers are therefore focussed on sambar.

The impacts and ecology of sambar in Australia were reviewed and summarised by Davis *et al*. (2016). Briefly, sambar are a large deer species (110–240 kg; Menkhorst and Knight 2011), which are generally solitary or form small family groups (Bentley 1998, Menkhorst and Knight 2011). Deer impacts occur primarily through the direct effects of herbivory (Davis *et al.* 2016). Sambar are browsers to intermediate mixed feeders (Parker 2009, Forsyth and Davis 2011). They stand 1.1–1.3 m at the shoulder (Bentley 1998, Mason 2006) and browse up to a height of 2.5 m (Peel et al. 2005), although they more commonly browse at a hieght of 0.6–1.2 m (Bennett 2008). Sambar territory and marking behaviours can also impact on vegetation and include antler rubbing of trees, antler thrashing of understorey vegetation, creating scrapes (scent-marked patches of bare soil cleared of vegetation with hooves and antlers) and wallowing (Bentley 1998, Bennett and Coulson 2011, Menkhorst and Knight 2011, Bilney 2013).

To address growing concerns about increasing deer impacts in the Lake Tyers area, the Department of Environment, Land, Water and Planning have partnered with the East Gippsland Rainforest and Gippsland Plains Conservation Management Networks and Trust for Nature to establish the Lake Tyers Deer Management Trial Implementation Plan which aims to reduce the environmental impacts of deer. It is proposed that volunteer hunters be engaged to control deer using ground-shooting at a landscape scale. Stakeholders are interested in determining how effective this management approach is for mitigating the environmental impacts of deer, and monitoring of the size, distribution and impacts of the deer population have been identified as priorities to inform control programs (Gunaikurnai Traditional Owner Land Management Board 2017).

2.1.1 Objective

The objective of this report is to develop an experimental design and field monitoring method to assess the effectiveness of the landscape-scale Deer Control Program in reducing the impacts of sambar on native vegetation in the Lake Tyers area. To do this, the monitoring methods will aim to:

1. Collect baseline data on the relative abundance and distribution of deer in the area, and enable subsequent monitoring to detect spatial and temporal changes in the relative abundance of sambar at a landscape scale in response to the Lake Tyers Deer Management Trial Implementation Plan, either due to reductions in population density or altered patterns of movement and habitat use.
2. Assess the impacts of sambar on diverse vegetation communities, and detect changes in these impacts over time, using a standardised, repeatable technique.

The monitoring method must be simple enough that it can be easily implemented by volunteers with some botanical knowledge, without the need for sophisticated training. The experimental design must be able to be implemented in two phases: a pilot study, followed by implementation of the full experimental design. The pilot study will test the survey methods and enable collection of preliminary data to guide decisions regarding the ongoing sampling effort required to detect changes with a desired level of precision (Cohen 1988) when the project is scaled up.

The output for this project consists of three documents:

1. **Part 1** of the **monitoring protocol** contains an outline of the background to the project and the rationale behind the experimental design and survey methods;
2. **Part 2** of the **monitoring protocol** contains a step-by-step field manual and Appendices containing maps and coordinates for sample point locations; and
3. **Part 3** is a customised **database** for data collation.

2.2 Experimental design

Two distinct monitoring approaches will be used to evaluate the effectiveness of the Lake Tyers Deer Management Trial Implementation Plan: (1) camera traps will be used to monitor spatial and temporal changes in the relative abundance of deer in response to deer management, and (2) measurement of vegetation structure and density, regeneration and growth in open plots will be used to assess the impacts of deer. The methodology for these two approaches is described in detail under second section of this document (Section 2.4). This first section outlines the experimental framework in which this monitoring will be implemented.

2.2.1 Considerations

The selection of an appropriate experimental approach to assess the effectiveness of the Lake Tyers Deer Management Trial Implementation Plan in reducing deer impacts is influenced by the following considerations:

1. There are limited areas in the region with similar vegetation types and condition, landscape attributes and levels of disturbance that are spatially independent and can be used as a reference site.
2. Ground-shooting is anticipated to commence in autumn 2019, subject to funding approval.
3. Resources available for monitoring are limited.

2.2.2 Management contrasts: before-after/control-impact design

The effectiveness of management interventions can be assessed when different interventions are compared (Lindenmayer and Likens 2010). Given Consideration 3, a manipulative experimental approach cannot be implemented. Therefore, a comparative approach is suggested: Comparison of vegetation between a location where deer are controlled using ground-shooting (Treatment site) to a location where deer are not controlled (Reference site) (Figure 1). This provides a framework to test for deer impacts on vegetation and assess the potential for ground-shooting to reduce impacts and allow vegetation recovery, either through a reduction in the deer population or behavioural responses to shooting (Bennett et al. 2015b).

Given considerations 1 and 3, there will be only one Treatment and one Reference site. Two experimental sampling units will be nested within each of these sites (Figures 1 and 2). This design will provide within-site spatial replication. Restriction of the spatial spread of sampling to the experimental sampling units, rather than across each site, will also ensure that the project is logistically feasible. In particular, nested sampling units will allow phased implementation of the project subject to funding availability (Section 2.3).

Given Consideration 2, it will be possible to collect data ‘before’ deer control. Therefore, a before–after/control-impact (BACI) design (Green 1979, Stewart-Oaten et al. 1986) can be used. This design will quantify treatment effects of deer control through comparison of response variables measured at a ‘Treatment site’, with measurements from an unmanipulated ‘Reference site, before and after commencement of deer control (Morrison et al. 2008).

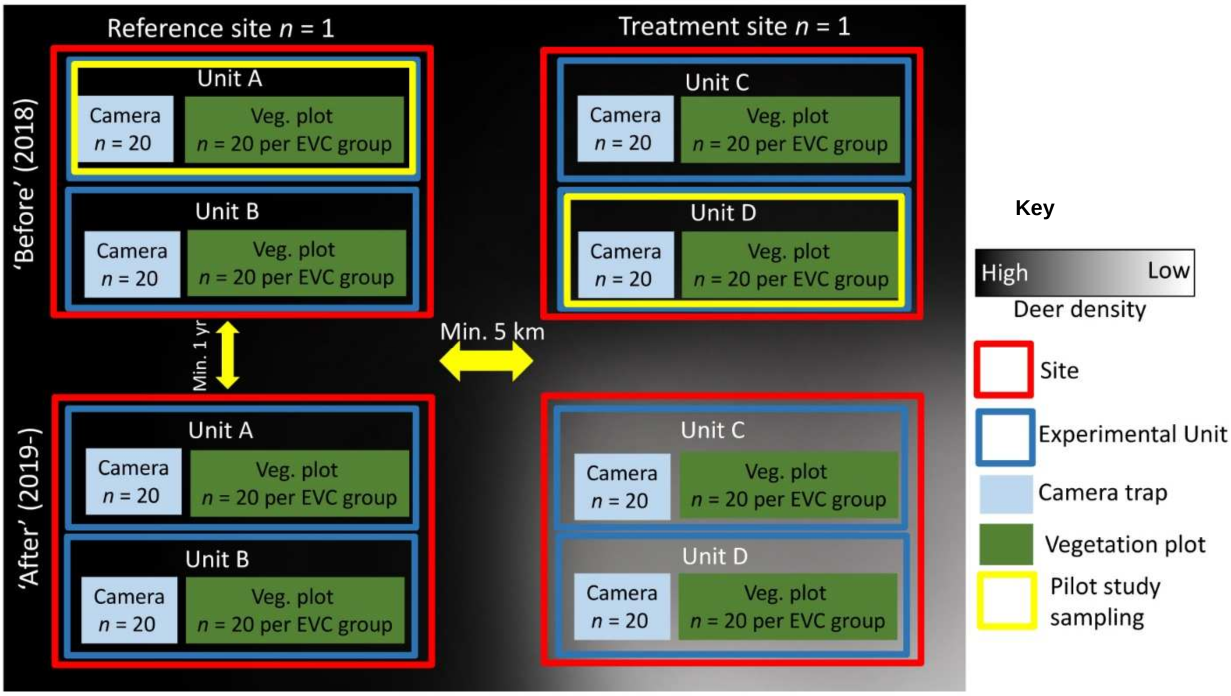


Figure 1. Diagram of experimental design for the Lake Tyers Deer Management Trial Implementation Plan, showing comparisons of data collected using vegetation plots and camera traps between sites subjected to contrasting management interventions (one Treatment site with ground-shooting for deer control, and one Reference site with no ground-shooting for deer control) that are predicted to result in a contrast in the relative abundance of deer. Sampling will be conducted in two replicate experimental sampling units at each site: camera sampling will be spatially randomised across each experimental sampling units; vegetation sampling will be stratified by four EVC groups. Sampling will be repeated annually for ≥ 5 years. A pilot study will be undertaken during the first year of data collection (2018) prior to implementation of the management intervention, and will constitute 'before' sampling The pilot study will use a reduced experimental design consisting of sampling in two experimental sampling units only, and in EVC group 3 (rainforest) only. Subsequent years of monitoring will constitute 'after' sampling.

The study will be implemented using an Adaptive Experimental Management approach. Hence the type and number of deer control treatments, and the associated replication of sites and experimental sampling units, may be modified over time if necessary.

2.2.3 Site selection and allocation of treatments

The lack of replication at the site scale will reduce confidence in separating treatment effects from confounding factors that are unrelated to deer control (Morrison et al. 2008). Therefore, it is important to match the Reference site to the Treatment site to reduce variability between the sites (Morrison et al. 2008). Specifically, it is recommended that the Reference and Treatment sites that will constitute the study area are:

1. of similar size;
2. have similar vegetation (types and condition) and landscape features (in particular, hydrology, given relatively high deer densities observed in riparian zones in the study area);
3. have similar deer densities and levels of deer impact; and

4. have similar land tenure and level of disturbance:
    a. Recreational hunting occurs throughout the Lake Tyers area where permissible. Ewing Morass Wildlife Reserve is a game reserve and will be excluded from the study area because hunting patterns will differ from the surrounding areas; hog deer hunting levels are relatively great, whereas sambar hunting is not allowed. Recreational hunting levels are elevated in the 'Beehive' area compared to other parts of the Lake Tyers area, hence this area will be excluded from the Treatment and Reference sites.
    b. There is little difference in the management of the State Park and Conservation Recreation Zones, so both can be considered for sampling.
    c. The tenure of surrounding land is also important. Similar representation of farmland with respect to area and perimeter is important, as deer have been observed to be in relatively high densities on farmland-forest ecotones, and farmers in the area may not allow recreational hunting but do undertake deer control on their properties.
    d. Fire history in the area is variable and needs to be considered. In particular, a wildfire covering ~1000 ha occurred at Tosteree in 2014.

It is also important that the Treatment and Reference sites are spatially independent, so that the abundance and impacts of deer at the Reference are not influenced by ground-shooting at the Treatment site. Information on sambar deer home range sizes in Australia is lacking, but information for other deer species in other locations suggests that home ranges are generally >50 ha, and commonly hundreds of hectares in size (Nugent 1994, Moriarty 2004, Spaggiari and de Garine-Wichatitsky 2006, Odden and Wegge 2007, Amos et al. 2014). Observations by local managers suggest that deer in the Lake Tyers area move many kilometres when food is scarce (B. Mills, Trust for Nature 2018, pers. comm.). In the Alpine National Park deer control trial, experimental sampling units are separated by only 2 km in places, although deer movement may be hindered by steep terrain in that environment (Davis et al. 2015). Ideally, experimental sampling units would be separated by much greater distances (e.g., >10 km; Simard et al. 2013), although there is potentially a trade-off between achieving spatial independence and ensuring site attributes are similar.

Two landscape-scale (>10,000 ha) sites have been selected in the vicinity of Lake Tyers. One is the broader area surrounding Lake Tyers (c. 34,918 ha) and the other is to the east of Lake Tyers (c. 20,231 ha) (Figure 2). These two sites are separated by a minimum of 5.2 km. These two sites are considered to be the best match available with respect to criteria 1–4. Both contain large areas of public land in which sampling can be focussed and have similar landscape and vegetation attributes. Although the site to the east is smaller and has considerably less rainforest, both sites are large enough to examine landscape-scale changes and impacts within rainforest and other vegetation communities. The deer density at the Treatment and Reference sites is unknown, but is thought to be similar. The broader area surrounding Lake Tyers has been subjectively allocated as the Treatment site because it contains a high proportion of Littoral Rainforest, the focal asset. Further, it is under joint management with the Gunaikurnai

Land and Water Aboriginal Corporation, an important stakeholder. The site to the east of Lake Tyers was therefore subjectively allocated to be the Reference site.

Nested within each of the two landscape-scale sites are two experimental sampling units (i.e., four 'Units' in total; Figure 2). At the Treatment site, Unit A (1,246 ha) is located in the West of the site, and Unit B (1,256 ha) is located in the South East of the site (Figure 2). Unit C (1,237 ha) is in the North East of the Reference site and Unit D (1,251 ha) is in the South of the Reference site. Experimental sampling units have been selected that contain rainforest (an important vegetation type for sampling: Section 2.2.5), while representing the best available matches with respect to criteria 1–4. Experimental sampling units each fall within public land, have similar landscape and vegetation attributes, are of similar size and are thought to have similar deer densities. All Units are in close proximity (<1 km) of private land and dominated by Lowland Forest. While the EVC composition varies slightly among experimental sampling units, the EVCs present are thought to be functionally equivalent. For example, Unit C does not contain Limestone Box forest, but contains significant areas of Shrubby Dry Forest, which is absent from the other units but thought to be functionally equivalent to Lowland Forest in terms of deer utilisation. Unit A has a relatively greater area of Damp Forest compared to Unit B, which has a relatively greater area of Limestone Box Forest. A notable difference between the experimental sampling units is that those in the Reference area are in close proximity to a major source of fresh water (the Snowy River in Unit C and Wombat creek, c. 1 km from Unit D) whereas those in the Treatment area are in close proximity to an estuarine system. All four experimental sampling units have similar road networks throughout (dirt roads only) but Unit A is within 1 km of Princess Highway at its closest point.

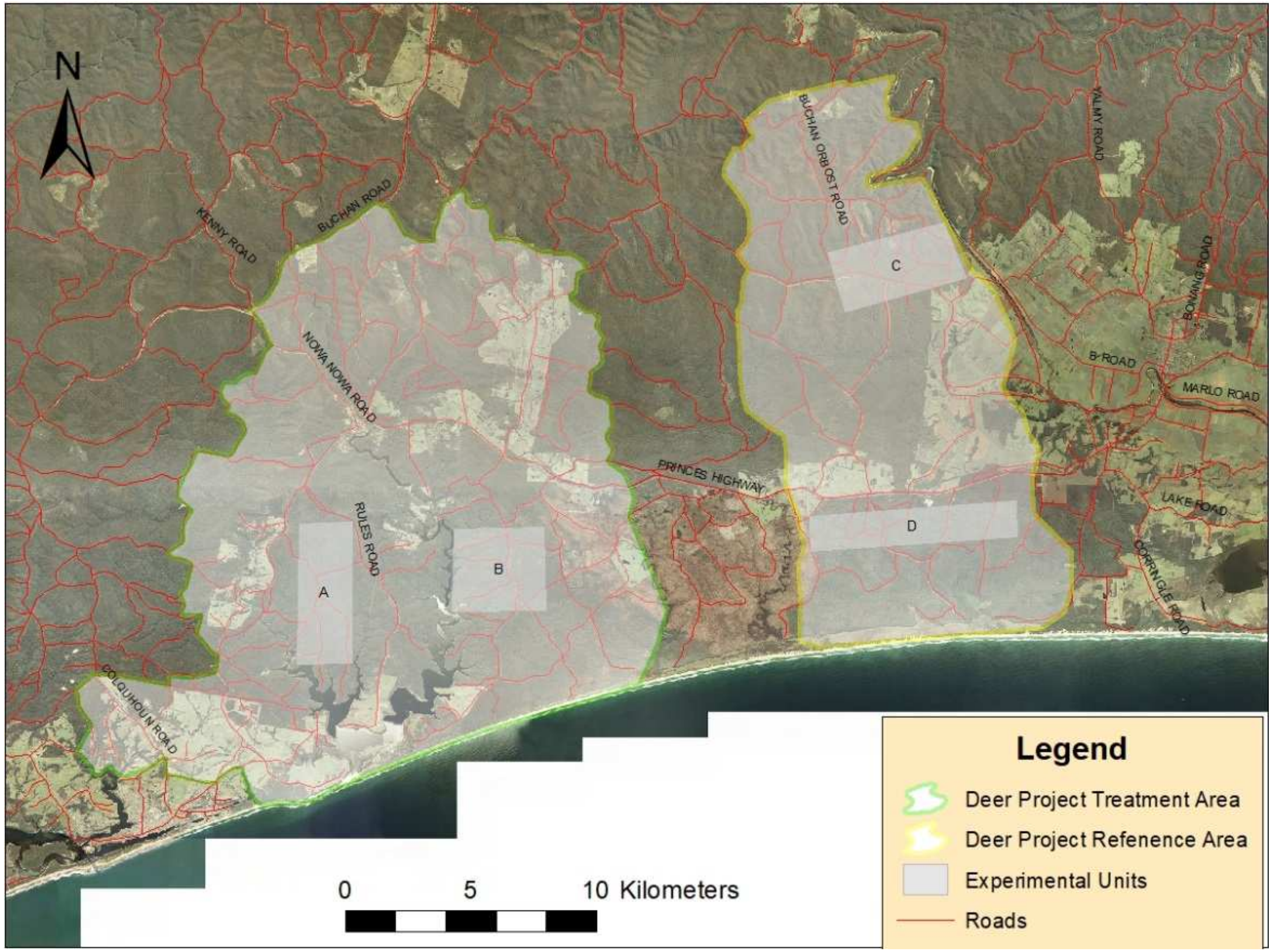


Figure 2. Two landscape-scale sites, each with two nested experimental sampling units, Lake Tyers Region, Victoria. The Treatment site consists of the broader area surrounding Lake Tyers (c. 35,953) and contains Unit A (1,246 ha) in the West of the site, and Unit B (1,256 ha) in the South East of the site. The Reference site is located to the East of Lake Tyers (c. 20,030 ha) and contains Unit C (1,237 ha) in the North East of the site and Unit D (1,251 ha) in the South of the site.

2.2.4 Shooting treatment

The Treatment and Reference sites will be subject to contrasting deer management interventions. Ground-shooting aimed at reducing deer abundance will be conducted across the entire Treatment site as part of the Lake Tyers Deer Management Trial Implementation Plan, but not at the Reference site. Ground-shooting will be conducted by volunteer recreational hunters as part of the Lake Tyers Deer Management Trial Implementation Plan, facilitated by Trust for Nature, in partnership with the Australian Deer Association. The methods used to conduct ground-shooting will be managed by incident controller(s) from PV and other staff and are not described here. The frequency, intensity, commencement and duration of shooting are yet to be confirmed. Shooting will likely be undertaken throughout winter, commencing after the Easter long weekend and continuing through until mid-November.

Recreational hunting occurs at both the Treatment and Reference sites at similar levels and therefore should not compromise the experimental design. To implement a management contrast, the Lake Tyers Deer Management Trial Implementation Plan will aim to reduce deer abundance substantially more than recreational hunting does. This approach will be similar to that employed by Simard et al. (2013) who in their test of the efficacy of hunting as a tool to reduce overabundant white-tailed deer (*Odocoileus virginianus*) in Canada, aimed to harvest 30-50% of antlerless deer at treatment sites, compared to control (reference) sites where harvest rate was 5-7%.

2.2.5 Spatial sampling distribution

2.2.5.1 Randomised sampling: deer distribution and relative abundance

This study aims to document landscape-scale responses of deer to management intervention. There is good reason to presuppose that sambar are more or less likely to occupy particular habitat types within the study area (Gormley et al. 2011). When this is the case, the precision of occupancy estimates can be improved by using a stratified random sampling approach that ensures adequate representation of all vegetation types in the sample (Nelson and Scroggie 2009). However, resources available are inadequate to robustly quantify deer occupancy or relative abundance within different vegetation types. Therefore, a randomised sampling approach (i.e., without stratification) across each experimental sampling unit will be used to document landscape-scale responses of deer to management intervention, modified from Davis et al. (2015):

1. The Treatment and Reference sites will be divided into grid cells (Jacobson et al. 1997, Allen et al. 2015), within which no more than one camera will be placed, to ensure the sampling is spread across the landscape. Camera traps will be placed randomly within each grid cell. Tobler and Powell (2013) recommend that the survey area covered by camera traps should be at least the area of one home range, with camera traps spaced less than one home range radius apart. Home range sizes of sambar in the Lake Tyers area are unknown. Based on the size of the smallest experimental unit (Unit C: Table 1), a cell size of 61. 89 ha (786.71 m × 786. 71 m) will be used to fit a sample of 20 cameras. International estimates of female sambar deer home ranges (c. 2 km^2; Lewis et al. 1990,

Fraser and Nugent 2005) suggest that these grid cells will ensure appropriate sampling coverage and camera separation.

2. Sampling locations will be separated by ≥ 100 m, so that the likelihood of detecting the same animal at adjacent sites is negligible, or at least that detecting the target species at one site has no effect on the likelihood that it will be detected at other sites (Nelson and Scroggie 2009).
3. There is an extensive track network in the area (Figure 3), providing good access (generally < 400 m walk) to all areas of vegetation in the experimental sampling units. Nonetheless, samples will be restricted to < 400 m from a road or track to ensure that sampling is efficient. In addition, cameras will be positioned > 10 m from a road or track to reduce the risk of theft.

2.2.5.2 Stratified sampling: impacts on vegetation

This project aims to capture information on deer impacts at a landscape scale, as well as at small scales in vegetation communities where deer densities and impacts are thought to be relatively high. It is therefore essential to monitor across the dominant vegetation communities, as well as within vegetation communities that are heavily impacted but occupy relatively small areas. Assessment of impacts within specific vegetation communities requires a stratified sampling approach to ensure that each vegetation type of interest is adequately sampled.

Vegetation sampling will be stratified by Ecological Vegetation Classes (DELWP 2017). A large number of EVCs occur in the Lake Tyers area (Figure 3). These have been divided into four groups for sampling (Table 2)**Error! Reference source not found.**:

EVC group 1: Lowland Forest, which comprises the majority of the study area (Table 1) but appears not to be used extensively by deer.

EVC group 2: Limestone Box Forest and Damp Forest. These two forest types also comprise substantial proportions of the study area (Table 1). They have been grouped because they differ from Lowland Forest in that they are wetter and contain species that appear to be more palatable to deer.

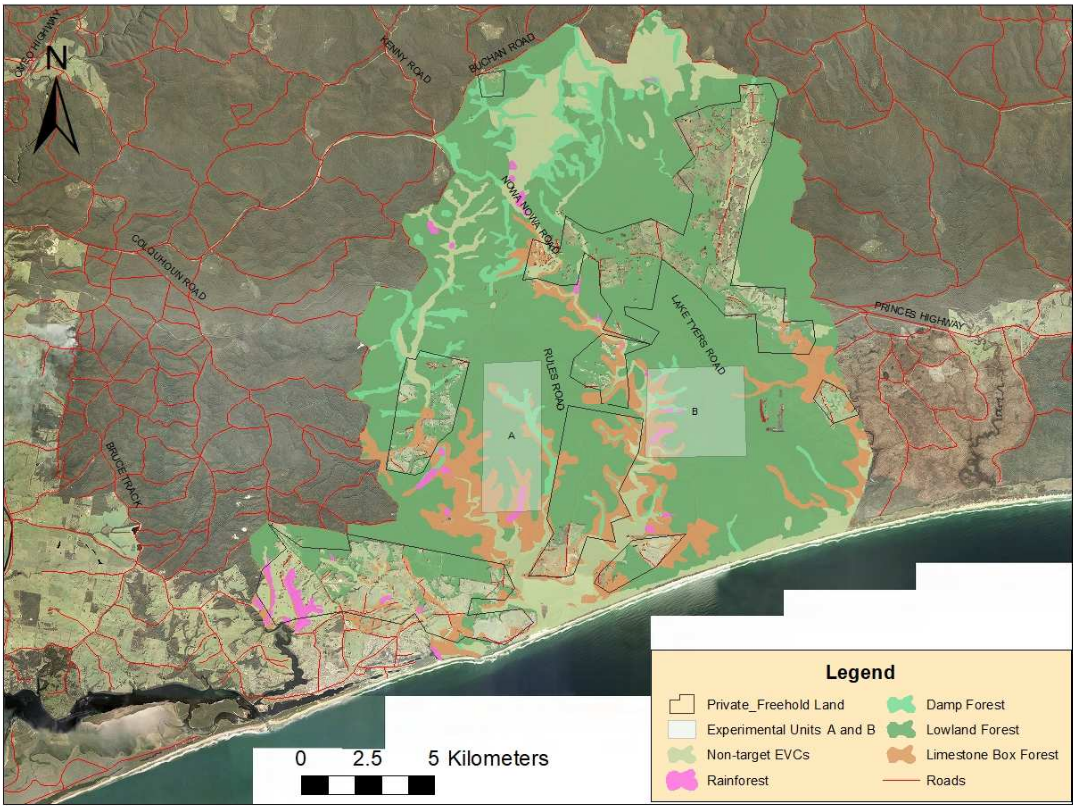
EVC group 3: Rainforest: Warm Temperate Rainforest, Dry Rainforest and Littoral Rainforest. Monitoring in these rainforest communities is important because Littoral Rainforest, given listing of this community as critically endangered under the commonwealth *Environment Protection and Biodiversity Act 1999* (EPBC Act), and listing of the other rainforest communities listed the *Flora and Fauna Guarantee Act 1988*. However, these three rainforest communities have been grouped because they each comprise a relatively small proportion of the study area, occurring in restricted patches (a given experimental sampling unit may not contain all of them; Table 1), yet they are considered to be functionally similar, and they all appear to be used by deer at similarly high levels relative to other EVCs.

EVC group 4: All other EVCs that occur in the study sites[1] have been grouped because they comprise a small proportion of the study area (Table 1) and observational evidence suggests they are not used disproportionately by deer. The exception to this is riparian communities in which elevated deer impacts have been observed, but which have not been grouped with Rainforest due to differences in vegetation composition, and have not been separated into their own EVC group for sampling purposes due to resource constraints.

[1] The EVCs contained within EVC group 4 in the experimental units are: Unit A – Water body (estuary); Unit B – Blackthorn Scrub and Estuarine Wetland; Unit C – Blackthorn Scrub and Shrubby Dry Forest; Unit D – not present.

Table 1. Areas of each EVC group within the study Treatment and Reference site experimental sampling units. Provided are the total areas of each EVC group across each experimental unit, and within the restricted sampling area < 400 m from tracks/roads.

EVC group	**Area (ha)**							
	Treatment site				**Reference site**			
	Across experimental unit		**Within sampling area (<400 m from tracks/roads)**		**Across experimental unit**		**Within sampling area (<400 m from tracks/roads)**	
	Unit A	Unit B	Unit A	Unit B	Unit C	Unit D	Unit C	Unit D
EVC group 1: Lowland Forest	862.69	954.89	523.92	720.22	732.22	970.27	623.18	532.73
EVC group 2: Limestone Box Forest and Damp Forest	377.56	257.22	241.08	225.11	219.87	279.76	153.73	103.73
EVC group 3: Rainforest	5.79	15.35	2.95	14.67	11.69	1.27	9.93	1.27
EVC group 4: All other EVCs	0.07	28.78	0.07	27.64	274.05	0	236.14	0
Total	1246.12	1256.29	768.01	987.64	1237.84	1251.36	1022.98	637.72



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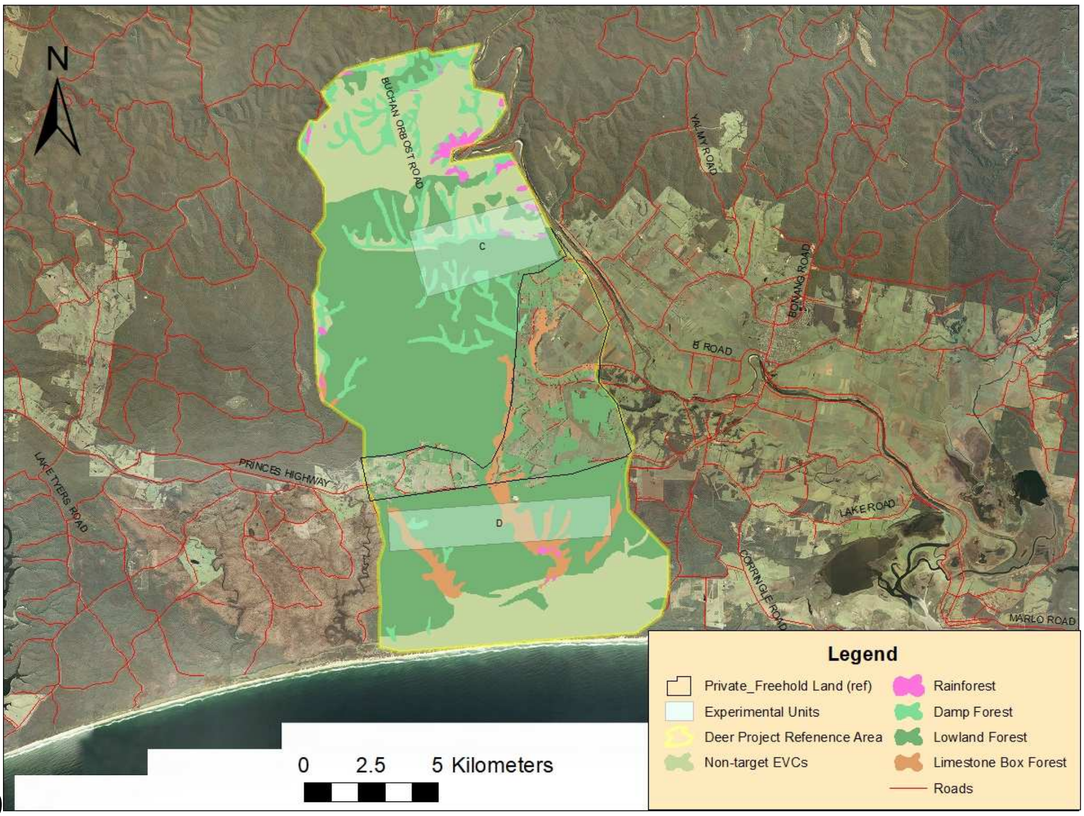

(ii)

Figure 3. Experimental sampling Units A and B at the Treatment site **(i)**, and Units C and D at the Reference site **(ii)**, Lake Tyers Region, Victoria. Shown are target Ecological Vegetation Classes, private/cleared land and roads.

Table 2. Broad descriptions of the main Ecological Vegetation Class (EVCs) (DELWP 2017) (and Littoral Rainforest: Department of the Environment 2015) in which sampling will occur. Detailed descriptions and benchmark species lists for these EVCs may be obtained from http://www.depi.vic.gov.au/environment-and-wildlife/biodiversity/evc-benchmarks#hsf

EVC description
Lowland Forest (EVC 16)
Eucalypt forest to 25 m tall on relatively fertile, moderately well-drained soils in areas of relatively high rainfall. Characterised by the diversity of life forms and species in the understorey including a range of shrubs, grasses and herbs.
Limestone Box Forest (EVC 15)
Occurs on generally well developed terra rossa soils derived from Tertiary limestones that outcrop around coastal streams, gullies and lakes. Open eucalypt forest to 20 m tall with a tall shrub layer understorey and a grass and herb-rich ground layer on sheltered aspects but may be almost completely bare on drier aspects.
Damp Forest (EVC 29)
Grows on a wide range of geologies on well-developed generally colluvial soils on a variety of aspects. Dominated by a tall eucalypt tree layer to 30 m tall over a medium to large dense shrub layer of broad-leaved species typical of wet forest mixed with elements from dry forest types. The ground layer includes herbs and grasses as well as a variety of moisture-dependent ferns.
Dry Rainforest (EVC 34)
Open to closed non-eucalypt low forest or tall shrubland to 15 m tall. Occurs in areas of relatively low rainfall (750-900 mm per annum) and primarily confined to topographic fire refuges most often associated with extensive cliffs, gorges and rock screes on northern and western aspects. Characterised by canopy species that are rain-green (lose leaves during drought).
Warm Temperate Rainforest (EVC 32)
Closed forest to 25 m tall occurring along gullies and river flats. Dominated by a range of non-eucalypt canopy species above an understorey of smaller trees and shrubs and usually visually dominated by ferns and climbers.
Littoral Rainforest (not a described EVC, but closely related to EVC 4)
The Littoral Rainforest and Coastal Vine Thickets of Eastern Australia ecological community is a complex of rainforest and coastal vine thickets on the east coast of Australia influenced by its proximity to the sea. The canopy, which protects less tolerant species and propagules in the understorey from salt laden winds, can range from patchy to closed and may include emergent trees as well as dead trees due to ongoing natural disturbance. The vegetation height depends on the degree of exposure and can range from dwarf to medium.

Vegetation sampling units (plots) within each experimental sampling unit[2] will be spatially randomised using GIS, with the following restrictions:

1. Sampling units will be restricted to the four EVC groups outlined (Section 2.2.5.2).
2. Although the relative area of the four EVC groups is variable (Table 1), sampling units will be allocated evenly among the four EVC groups (*n* = 20 per EVC group per experimental sampling units). An exception is EVC group 4, which will not be sampled in Units A and D because it does not occur in these Units[3]. Samples have been evenly allocated to EVC groups rather than in proportion to the relative area of EVC groups because 20 samples is considered the minimum required to robustly quantify deer relative abundance and impacts at the scale of EVC groups and proportional allocation would not result in a sample size of 20 in EVC groups that occupy a relatively small area.
3. There is an extensive track network in the area (Figure 3), providing good access (generally < 400 m walk) to all areas of vegetation in the experimental sampling units. Nonetheless, samples will be restricted to < 400 m from a road or track to ensure that sampling is efficient. Note that no minimum distance will be applied to reduce edge effects (e.g., from roads or EVC boundaries) because it is assumed that random sampling will sample these in proportion to their occurrence.
4. Samples will be separated by ≥ 20 m. This small separation distance has been selected to allow multiple plots to be placed in small rainforest patches.

2.2.6 Temporal sampling distribution

2.2.6.1 Commencement and duration

It is recommended that sampling commence as soon as possible (mid-autumn 2018) because there is limited time available to collect pre-treatment data prior to the anticipated commencement of ground-shooting in June 2019. Given limited pre-treatment sampling and the lack of spatial replication in the experimental design at the site scale, quantification of treatment effects for this study will be strengthened by incorporating temporal replication (i.e. repeat surveys over a number of years; Morrison et al. 2008). The collection of multiple comparable measurements of the variables of interest at monitoring sites over time will allow

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[2] Modelled EVC mapping available for EVC group 3 (rainforest) was determined to be inaccurate and was improved in GIS using aerial imagery and gully locations prior to determination of random vegetation plot locations for this EVC group. If ground-truthing shows that the area of rainforest within an experimental unit is too small to fit 20 plots separated by ≥ 20 m, those rainforest plots that don't fit within the experimental unit will be randomly placed in the nearest rainforest stands to the experimental sampling unit.

[3] Note that the lack of sampling of EVC group 4 in experimental units A and D is considered acceptable because it is not a target EVC group. However, this will result in an uneven sampling design. If managers want to extend sampling to include EVC group 4 in these two units, vegetation plots can be randomly placed in the nearest areas of EVC group 4 to the experimental unit.

evaluation of changes associated with deer management, through examination of differences in pattern and magnitude of temporal changes between the Treatment and Reference sites (Morrison et al. 2008). It is recommended that the monitoring program is continued for at least five years to enable quantification of short- and medium-term impacts of deer on vegetation.

2.2.6.2 Frequency and timing

Annual sampling is recommended to ensure that monitoring is achievable with the available resources. The timing of ongoing sampling will be dictated by the commencement of sampling in the project establishment year (2018), which is anticipated to be mid-autumn (Section 2.2.6.1). It is important to sample at the same time in each subsequent year at both the Treatment and Reference sites. Ongoing monitoring should be conducted annually, ±30 days of the middle of the first survey in 2018. This timing will avoid the summer and winter holidays, which should decrease the risk of camera theft. No data is available on seasonal changes in use of the study area by deer, however, even if deer densities change seasonally, annual monitoring during the same season each year should detect broad changes in the relative abundance of deer over time. Vegetation and camera sampling will be conducted during the same period to help link data from the two approaches, and to maximise survey efficiency. Because vegetation identification and measurement are not taking place during spring/summer when detection and identification accuracy is generally maximised (Department of Environment Land Water and Planning 2017), vegetation plots may need to be revisited outside the primary sampling period when plants are in flower to aid plant identification.

2.2.6.3 Camera deployment

Nelson and Scroggie (2009) recommend that deploying cameras for a length of time close to the expected endurance of the batteries and/or digital storage medium is generally the best strategy, because the cost of leaving a deployed camera out for a few extra days is effectively zero and maximising the length of the deployments provides a longer sequence of detection/non-detection events at each site, which leads to more precise estimation of the probability of detection and the rate of occupancy. Each camera trap will therefore be deployed for two months, just under the anticipated limit on the function of the cameras due to battery life and memory card capacity. Camera trapping in the area has indicated that deer are commonly detected using a deployment time of two months (B. Mills, Trust for Nature 2018, pers. comm.). If funding becomes available to expand the project in the future, information on seasonal use of the study area by deer could be obtained by repeating the suggested sampling each season.

2.2.7 Sampling effort

The proposed annual sampling effort for the full experimental design is 20 camera traps per experimental sampling unit, and 20 vegetation plots per EVC group (four EVC groups) per experimental sampling unit. That is, 80 camera samples and 280 vegetation plots in total (Table 3). Note that EVC group 4 does not occur in experimental sampling units A and D, and will only be sampled in experimental sampling units B and C, resulting in the total vegetation plot sample size of 280, not 320.

Although general guidelines exist for allocation of survey effort when designing occupancy studies (MacKenzie and Royle 2005), it is often not possible to determine the necessary number of sites to survey using remote cameras prior to conducting the survey (Nelson and Scroggie 2009). There is limited information available on detection probabilities and occupancy rates to guide the sampling effort required to detect changes in deer population abundance in Australia. A study of hog deer in coastal grassy woodland by Davis et al. (in prep) showed that to detect a medium effect size with ~80% power, 40-60 cameras would be required per treatment (depending on the probability of occurrence) but that 20 cameras per treatment was adequate to detect large effect sizes. The sampling effort proposed for the Lake Tyers project is therefore anticipated to be adequate to detect large effect sizes. The proposed effort is in line with that being applied in other deer monitoring programs. For example, the larger deer control trial being conducted in the Alpine National Park involves deployment of 200 cameras across two sites (25 camera traps in each of four experimental sampling units per site) (Davis et al. 2015) and the smaller trial being implemented in the Manningham municipality involves deployment of 20 cameras at each of two sites (Bennett and Davis 2017). Allen et al. (2015) investigated deer habitat use in New Zealand using only 25 cameras, and Gormley et al. (2011) sampled at 40 locations using cameras to examine habitat suitability for sambar in Victoria. The sampling strategy suggested for the Lake Tyers project aims to maximise the precision of estimates of the rate of site occupancy, by maximising the number of sites that are surveyed and the number of days of camera sampling, although with limited resources there is a trade-off between the number of days surveyed at each site and the total number of sites which can be surveyed (Nelson and Scroggie 2009).

Fewer open vegetation plots have been proposed to monitor deer impacts in the Illawarra region (Davis 2017) and Manningham municipality (Bennett and Davis 2017), however, these studies have the advantage of using differential exclosures to strengthen the experimental design. Given the lack of differential exclosures, and the coarseness of some of the the vegetation parameters being measured, a larger sample size is recommended for the Lake Tyers project. The proposed sample size is in line with the minimum sample size of 25 plots per experimental sampling unit recommended for rapid vegetation assessment plots being used to monitor deer impacts across highly variable environments within each experimental sampling unit.in the Alpine National Park (Bennett et al. 2015a).

Given restriction of sampling to within 400 m of the road/track network, it is expected that camera deployment will take approximately one hour per camera, including travel time and will require one person. The vegetation monitoring methods have been designed to be rapid. Implementation times for a similar vegetation survey methodology (Bennett and Davis 2017) have been approximated at one hour per plot for two people when vegetation was sparse and access was easy, but longer when the vegetation was dense, diverse and included extensive regeneration, and/or where access was difficult. It is therefore anticipated that on average, five vegetation plots could be completed by two people in a day. Based on these estimates, total estimates of field time required annually for the full study design are as follows:

- 80 cameras = 10 days[4] (× 1 person)
- 280 vegetation plots = 56 days (× 2 people, i.e. 112 person days)

[4] Note that this does not include processing of camera images.

The estimate of total field time for camera trapping and vegetation plots required to implement the full survey is therefore 122 person days. Funding during the establishment year is not adequate to implement this full design. Instead, a pilot study (Section 2.3) using a reduced version of the full experimental design will be implemented during the first year.

2.3 PILOT STUDY

During year 1 of the study, a pilot study will be undertaken to inform implementation of the full Lake Tyers Deer Management Trial experimental design (Section 2.2). The data obtained will be used to examine variability among samples in the various parameters measured, and power analysis undertaken to guide decisions regarding the ongoing sampling effort required to detect changes with a desired level of precision (Cohen 1988). The pilot study will test the ease, efficiency and resource requirements of implementation of the survey methods.

2.3.1 Reduced experimental design

For the pilot study, the full experimental design (Section 2.2) will be reduced by removing one of the nested experimental sampling units per site (the experimental sampling units randomly selected for use during the pilot study are A and D). That is, sampling will be conducted at each the Treatment and Reference site in one of the nested experimental sampling units only. Further, vegetation impact monitoring will be conducted only in EVC group of most concern – EVC group 3: rainforest. Apart from these two modifications, the experimental design for the pilot study otherwise remains consistent with the full experimental design. This approach allows ‘before’ data collected from the pilot study at both the Treatment and Reference site to form an integral part of the full study, and ensures that the pilot can be easily scaled up as funds become available.

The primary compromise in the reduced sampling design during the first year of the study as that ‘before’ data on vegetation condition will be not be collected for EVC groups 1, 2 and 4. Detecting changes in these vegetation communities associated with deer control will therefore rely on comparison of the Treatment and Reference sites after deer control has commenced. For this reason, managers should consider commencing deer control after autumn in year 2, or in year 3, so that ‘before’ data can be collected in rainforest during year 1, and in the other three EVC groups during year 2.

2.3.2 Reduced sampling effort

Given limited funding available during the project establishment year, reduced sampling effort is key to successful implementation of the pilot study. Land managers currently have funding for 40 cameras for use in this project. This pool of 40 cameras will be split in half to sample both the Treatment and Reference site at the same time. Monitoring vegetation in rainforest will require 20 plots per site annually, substantially reducing sampling effort (Table 3). Total estimates of field time required annually for the pilot study are as follows:

- 40 cameras = 5 days[5] (× 1 person)
- 40 vegetation plots = 8 days (× 2 people, i.e., 16 person days)

[5] Note that this does not include processing of camera images.

The estimate of total field time for camera trapping and vegetation plots required to implement the full survey is therefore 21 person days.

2.3.3 Optional test of vegetation impact methodology

The pilot study will enable the vegetation survey methods to be assessed with respect to ease and efficiency of implementation. The survey methodology can be refined before scaling the project up during subsequent years if for example the vegetation survey methods are too labour intensive, sample sizes are inadequate (or excessive) for detecting change in the vegetation, or some components of the survey are more effective than others at detecting vegetation change. In particular, if the proposed stem counts (Section 2.4.2.2) are too labour intensive, stems counts may be reduced to focus on a sub-set of plant species or sub-sampled from within the 10 m × 10 m plots.

In addition, a test of whether the vegetation survey methods are suitable for detecting changes in vegetation structure and density, regeneration and growth could be undertaken using the fenced exclosures that have been erected in Littoral Rainforest (Bennett 2016). The vegetation survey method could be undertaken in plots within exclosures and the data compared to data collected from open vegetation plots assessed in Littoral Rainforest. A significant difference in the parameters of interest should be detectable, given that the exclosure fences have now been in place for approximately 18 months, although anecdotal evidence suggests that vegetation recovery may take longer than this.

Table 3. Sample sizes for each sampling unit and EVC group at the Treatment site and the Reference site using the full study design, and the reduced pilot design.

	Entire study		**Treatment site**									**Reference site**								
				Unit A				Unit B					Unit C				Unit D			
Sampling unit	Total No. annual samples	Total No. samples over 5 years	Total No. annual samples	EVC group 1	EVC group 2	EVC group 3	EVC group 4	EVC group 1	EVC group 2	EVC group 3	EVC group 4	Total No. annual samples	EVC group 1	EVC group 2	EVC group 3	EVC group 4	EVC group 1	EVC group 2	EVC group 3	EVC group 4
Full design																				
Vegetation plots	280	1400	140	20	20	20	NA	20	20	20	20	140	20	20	20	20	20	20	20	NA
Camera traps	80	400	20	NA	NA	NA	NA	NA	NA	NA	NA	20	NA	NA	NA	NA	NA	NA	NA	NA
Pilot study																				
Vegetation plots	40	NA	20	0	0	20	0	NA	NA	NA	NA	20	NA	NA	NA	NA	0	0	20	0
Camera traps	40	NA	20	NA	NA	NA	NA	NA	NA	NA	NA	20	NA	NA	NA	NA	NA	NA	NA	NA

2.4 MONITORING METHODOLOGY

There are two aspects of monitoring to assess the effectiveness of the Lake Tyers Deer Management Trial Implementation Plan: (1) monitoring spatial and temporal changes in the relative abundance of deer in response to deer management, and (2) assessment and monitoring of the impacts of deer on vegetation. These two aspects of the monitoring require distinct methodological approaches that will be trialled during the pilot study (Section 2.3) prior to implementation of the full Lake Tyers Deer Management Trial experimental design (Section 2.2).

2.4.1 Deer distribution and relative abundance

Camera traps will be used to index changes in the relative abundance of deer over time (temporal changes) between the Treatment and Reference sites (spatial changes). Camera traps are cameras that contain a sensor that is remotely triggered by heat and motion of animals to take a photograph when an animal is present. The number of animals captured or ‘trapped’ by the photographs may be counted as an ‘image of occurrence’ (O'Çonnell et al. 2011). Camera traps have a wide range of applications, including investigation of animal population dynamics over space and time (Nichols et al. 2011), and have been used for this purpose in Australia for monitoring other populations of deer (Davis et al. 2015). Camera traps have been selected to monitor deer during this project because they are relatively easy to use, can sample animals remotely with no requirement for humans to manually operate the equipment (Bridges and Noss 2011), and are likely to cause less disturbance to animals than direct observations (Bridges et al. 2004).

Faecal pellet counts are an alternative indirect measure that can be used to index deer abundance (Parks Victoria 2005). However, herbivore scats can be difficult to identify (Triggs 2003, Liddicoat 2008) and species identification is generally more accurate using camera images (Davis 2014). A number of other mammalian herbivore species occur in the Lake Tyers area including native swamp wallaby (*Wallabia bicolor*) and common wombat (*Vombatus ursinus*), and feral pig (*Sus scrofa*) and goat (*Caprua hircus*) (Gunaikurnai Traditional Owner Land Management Board 2017). Because resources are not adequate to employ differential exclosure plots to allow the effects of deer to be partitioned from those of other herbivore species (Bennett and Coulson 2008), it is essential that the relative abundance of deer and other herbivores is quantified at both the treatment and reference site. Further, deer species other than sambar occasionally occur in the Lake Tyers area, for which impacts and management tools may differ (Davis et al. 2016). For these reasons, cameras are the recommended sampling tool for this project. Camera traps also have the advantage of being able to identify deer sex and age class, and collect data on time and date of visitation that may be of secondary interest to this project.

2.4.1.1 Camera placement and settings

Current camera trapping for deer being conducted in the study area uses subjective placement of cameras to target social wallows. This method has allowed experienced staff to conduct absolute counts of deer at small scales based on identification of individual deer. This method could be scaled up to sample across the landscape using a drone to locate wallows. However, this method will not be employed given its reliance of observer expertise, and potential confounding of wallow use by variables such as climatic conditions. Moreover, to obtain unbiased estimates of occupancy, it is preferable that sampling sites are not selected based on pre-existing knowledge about their potential occupancy (MacKenzie and Royle 2005), hence Nelson and Scroggie (2009) warn against placing cameras at locations known to support the species of interest, or targeting locations with habitat that is considered to be highly suitable. Restricted random locations will therefore be used for cameras (Section 2.2.5).

To increase detection rates, once random camera trap sampling coordinates have been navigated to, camera traps will be subjectively placed to maximise visitation (i.e., located at the nearest trail to the random sampling coordinates; Jacobson et al. 1997). Camera traps be sited to face along trails so that they are triggered by animals moving toward or away from the camera trap, rather than across the field of view, to minimize the effect of a delay between motion being detected and an image being recorded (Bengsen et al. 2011b). A passive survey approach (i.e., no bait lure) will be used so that deer behaviour and habitat use are not altered by camera trap monitoring (Meek et al. 2012).

An important aspect of a camera trap monitoring is to minimise variation in the probability of animal detection among camera traps. This will be done by using only one camera make/model for all sampling (Meek et al. 2014) and using standardised mounting procedures. All camera traps are to be mounted on a tree or stake at 1.3–1.5 m (Nelson and Scroggie 2009, Bennett 2013) facing south, which will help to avoid direct sun light triggering camera traps (Meek et al. 2012). Camera traps will be mounted horizontally to the ground, angled slightly downwards so that the passive infrared sensor is aimed 6 m away at the height of the core body zone of an adult sambar deer. Standardised settings should also be used to maximise detection and accurate identification of animals as follows: 'high sensitivity', 'three images per movement' and 'rapidfire' with no delay (Bennett 2012, 2013, Davis 2014).

To further standardise camera trap detections, the detection zone of each camera trap should be delineated. Following the approach taken by Davis (2014) and Bengsen et al. (2011b), a marker will be used to delineate the detection zone for each camera trap. A 6 m detection zone has been used in coastal (Davis 2014) and alpine (Davis et al. 2015) vegetation in Victoria. This has been successful for determining whether hog deer (Davis 2014) and sambar (E. Thomas, Parks Victoria 2018, pers. comm.) in images occur inside or outside the standardised detection zone. For consistency with these other deer monitoring programs, a 6 m detection zone is recommended for the Lake Tyers project. However, if during the pilot study it is determined that the vegetation at Lake Tyers is too dense to successfully implement a 6 m detection zone, a stake will be instead by placed 5 m in front of each camera trap to delimit a standardized 22.5-m^2 visual field of view, within which the infrared sensor had an effective 2.5-m^2 field of view (Bengsen et al. 2011a). Vegetation within the detection zone will be modified to maximise and standardise detection probabilities and reduce false triggers (Davis 2014, Meek et al. 2014). Vegetation should be removed from within the conical detection zone 6 m in front of each camera trap and to 40° either side of that central line of sight, if it is deemed that the vegetation has the potential to prevent detection or cause false triggers. This was achieved in alpine vegetation with an acceptable level of effort, although relatively open sites were selected in which to place cameras (E. Thomas, Parks Victoria 2018, pers. comm.). To further standardise the detection zone, camera traps will be placed on the flattest section of trail available (Davis 2014).

2.4.1.2 Camera data

The image browser application ExifPro 2.1 should be used to manage and count images to occupancy and frequency data. This free program can help to display, describe and sort large collections of photographs. It can present information embedded in photographs (EXIF metadata) and also allows the user to append descriptions. As much as possible, we recommend the use of established rules for recording camera trap data. However, given that individuals are unlikely to be readily identifiable enough to use methods such as those of Bengsen et al. (2011b), the rules used by Allen et al. (2015) to classify deer sex and age classes and to define independent records of animals may be more appropriate. The presence of other herbivore species should also be recorded.

2.4.1.3 Other herbivore species

Although aiming sensors lower that this is recommended by Meek et al. (2012) for detection of other smaller taxa, mounting specifications have been selected to optimise detection of deer because detection of other taxa is secondary. However, it is anticipated that cameras will detect other medium-large mammal taxa. The image sorting process will quantify the relative abundance of these other species to help determine whether documented impacts on vegetation are caused by deer.

2.4.2 Impacts on vegetation

Protocols have been developed to monitor the impacts of deer on vegetation in Australia, however, these are either too complex to be implemented by volunteers without extensive botanical knowledge (e.g., Davis 2017), or are habitat-specific, for example, the method developed by Bennett (2016) to assess impacts in Littoral Rainforest and the method developed by Bennett et al. (2015a) to monitor deer impacts in alpine peatlands. Managers implementing the Lake Tyers Deer Management Trial want to use a standardised, repeatable technique to monitor changes in vegetation, that is simple and rapid to implement, and can be applied in a wide variety of vegetation communities, ranging from wetland to forest. A new method for monitoring deer impacts on vegetation has therefore been developed to meet these requirements. Although it has been designed to be simple to implement, it does assume that volunteers implementing the methods will have a moderate level of botanical knowledge, or will be able to access expert assistance to identify plants. This is important because the impacts of novel herbivores will be related to relative consumption levels of different plant species (Nuttle et al. 2014).

There is evidence to indicate that sambar browsing and trampling can reduce ground layer and understorey vegetation density, prevent tree recruitment and facilitate weed establishment, leading to overall impacts on species diversity and structural integrity with potential flow-on impacts on canopy cover and tree species diversity (TSSC 2008, Bennett 2016). Three key vegetation attributes have been selected for monitoring because they are known to be impacted on by sambar and other deer species in vegetation communities elsewhere (Forsyth et al. 2015, Davis et al. 2016) and they can be quantified without sophisticated training and equipment: 1. Structure, composition and diversity; 2. regeneration and survival; and 3. vertical growth.

2.4.2.1 Structure, composition and diversity

Vegetation structure will be assessed using three methods. First, changes in the structure of vegetation < 2 m in height will be assessed using a structure pole. Second, changes in the structure of the ground and canopy layers will be assessed using visual estimates of cover. Third, standardised photo points will be used to provide a qualitative visual record of temporal changes in vegetation structure of all vegetation layers. The first and second method will also assess changes in composition and diversity.

Changes in the structure of the ground and shrub layers will be assessed using a structure pole to record foliage density at different height increments. Although more time consuming than the coverboard method (Nudds 1977), a structure pole will remove observer error associated with estimating percentage foliage cover. It will also allow species- and lifeform-specific changes to be detected even if overall vegetation cover remains constant, which would be difficult to assess using a coverboard, which is generally used to estimate the density of all vegetation types combined. Species-level information can also be used to look at the relative occurrence of weeds, which may be more prevalent in areas disturbed by deer. To correspond with vegetation layers and the range of deer browsing heights, the structure pole estimates will be made at five height increments: 0–20, 21–50, 51–100, 101–150, 151–200 cm (modified from Bennett and Davis 2017). Anecdotally, a lack of recovery of some species from deer browse damage, once deer are no longer using an area, has been observed, particularly when a browse line has been established. The structure pole assessment will quantify impacts, as well as recovery or lack thereof, if changes in vertical structure of the vegetation occur.

Changes in the structure of the ground layer will be assessed using visual estimates of cover of ground layer categories. The cover assessment will be undertaken within replicate 1-m^2 quadrats. Ground layer cover estimates (percentage cover; continuous scale 0–100%) will be made for material ≤ 50 cm in height for 14 categories: bare ground, litter, rock, coarse woody debris (dead wood ≥10 cm diameter), forb, grass, sedge/rush, lily, orchid, fern, shrub/tree, climber, moss, and other cryptogams. Bare ground is included as an index of disturbance, which can facilitate weed invasion and soil erosion. In addition, cover estimates (percentage cover; continuous scale 0–100%) will be recorded for the canopy (emergent and upper-canopy combined; Figure 3) and mid-stratum (M1 and M2 combined; Figure 3). These canopy and mid-stratum estimates will capture changes in vertical structure at a height greater than the 0–2 m estimates recorded by the structure pole.

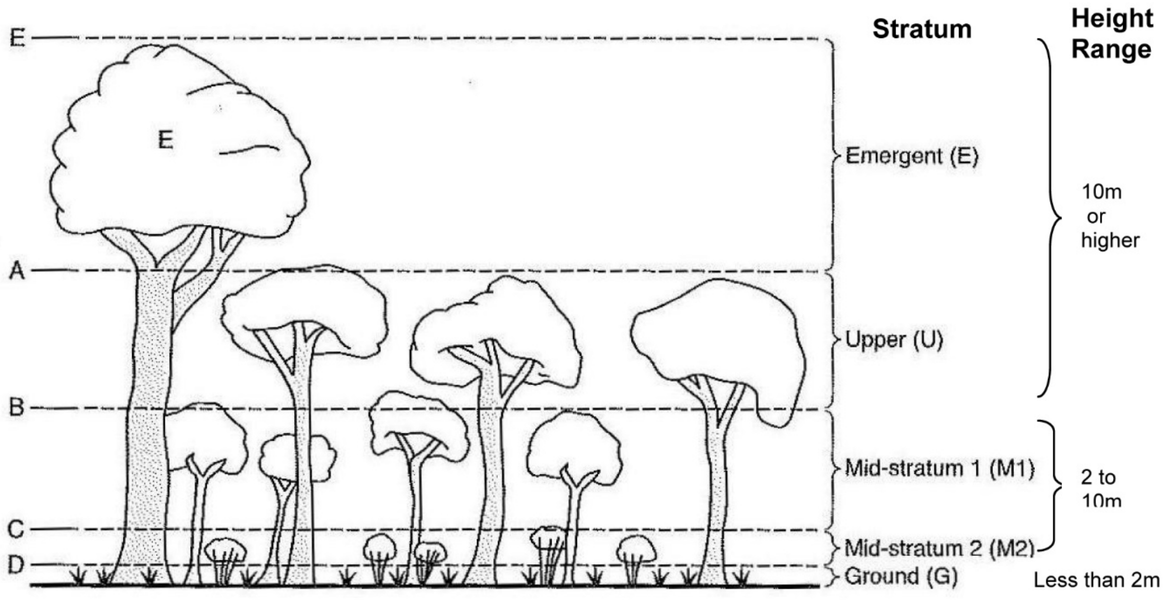

Figure 3. Diagram of vegetation structural layers (sourced from Hnatiuk et al. 2009).

Standardised photographs (using a consistent camera make and model; no zoom; camera at a height of 1.5 m) will be taken of the vegetation from the centre of the plot in the four cardinal directions (North, South, East and West) with the camera angled horizontally, and directly up into the canopy. This will provide a qualitative visual record of temporal changes in vegetation structure.

2.4.2.2 Regeneration and survival

Quantification of tree age structure by species provides an indication of whether regeneration is occurring (Bennett 2016). To assess deer impacts on regeneration, one should consider seedling establishment, survival, and size-class recruitment (Bressette and Beck 2013). To quantify recruitment of woody plants of different genera into different size-classes (and in turn estimate establishment and survival), stem counts will be made for all live and all dead individuals in vegetation plots of each woody plant genera. This method will be expanded from Davis (2017) to include shrubs and woody lianes, but with simplified DBH size-classes (Bennett and Davis 2017). Initially it was proposed that counts be made within three DBH (cm) size-classes: < 2.5, 2.5–10, >10. However, preliminary field trials showed that this approach was no feasible due to extremely high seedling densities in some rainforest plots. Stem counts have therefore been reduced to the larger two DBH size-classes only: 2.5–10, >10. The use of two size-classes will reduce the time required to conduct vegetation surveys, and it will also reduce the influence of large masting events in rainforest with high levels of natural mortality of seedlings. It is anticipated that impacts of deer on survival and size-class recruitment will still be detected using counts of stems that reach the larger size classes. Species-level information can be important when some plant species are preferentially browsed or rubbed, or less tolerant of browsing, rubbing or trampling damage, and these are replaced by other species that are avoided and/or tolerant of these impacts. However, stems counts will be conducted for genera rather than species, because identification of juvenile plants is likely to be difficult and relative preferences of deer, and plant tolerances to deer, are likely to be broadly similar among species within a genus. For multi-stemmed trees, only the largest living stem will be included in the count. Collection of genus-level data will also add to the assessment of changes in composition and diversity (Section 2.4.2.1).

2.4.2.3 Growth and survival

Vertical growth of select understorey and mid-storey shrub and tree species that appear to be targeted by deer will be undertaken. Tagging individual plants is a useful method to assess the impacts of deer on plant growth (Husheer and Robertson 2005). This method is labour intensive when detailed measurements are taken to assess the growth of individual branches (e.g., Bennett 2008), however, simple measurements of plant height can be used to demonstrate deer impacts on growth (Davis and Coulson 2010) and are rapid to implement. The method must be simple enough to be implemented without extensive botanical skills, however, measuring vertical growth on all plants present without identifying species is not advisable. This is because deer preferentially browse some plant species while avoiding others (Nuttle et al. 2014), hence the occurrence of both negative and positive effects on growth rates may mask impacts. Growth will therefore be assessed over time on tagged individuals of five plant species that are easily recognisable, widespread across many vegetation communities in the Lake Tyers area, and for which anecdotal impacts of deer have been observed: *Exocarpos cupressiformis*, *Pittosporum undulatum*, *Pomaderris aspera*, *Indigofera australis* and *Coprosma quadrifida*. Each of these species is heavily browsed by deer, as evidenced by a distinct browse line on these species in the Lake Tyers area. Each of these species is common across the landscape in the Lake Tyers area, *although P. undulatum* tends to be a pioneer species and is not as widely distributed. *Acronychia oblongifolia* is also heavily impacted on by deer (Peel et al. 2005) but has not been included as a target species for tagging because it only occurs in rainforest. As well as providing information on growth, this component of the monitoring will provide data on survival of tagged plants.

2.4.2.4 Deer damage

Documentation of physical damage caused by deer such as antler rubbing does not necessarily provide evidence of consequences for plant fitness or survival (Davis et al. 2016). The current project aims to move beyond quantification of this type of damage to examining its consequences for vegetation communities. However, quantification of physical damage by deer can help to partition the impacts of deer from those of other species (Davis et al. 2016). This is importation for this project, given that deer impacts at Lake Tyers will not be partitioned from those of other herbivores using experimental exclosures.

Deer browsing causes damage such as apical bud removal, bark striping and fraying, antler rubbing, flower removal and stem breakage (Barrett and Schmitz 2013). It has been suggested that deer browsing can be differentiated by that other species due to differences in bite characteristics (e.g., rough-torn edges for deer; Strole and Anderson 1992), however, Stockwell (2003) found that this was not possible in the field. Therefore, this study will broadly quantify the degree of browse damage in combination with an assessment of antler damage (rubbing and thrashing) and bark stripping. Data from browse damage scores can be analysed to compare levels of damage between groups of plants (Marks and Moore 1998). Physical evidence of deer damage will be categorised for all tagged plants. The following deer damage ranking, derived from Richardson (2015) and Bulinksi and McArthur (2003), with the addition of categories for antler damage and bark stripping, will be used (more than one category may be selected): (1) no visible browse damage (0 % biomass removed); (2) light browse damage (browsing of exterior tips; 1-29% biomass removed); (3) moderate browse damage (30-49% biomass removed); (4) heavy browse damage (70-89% biomass removed); (5) terminal browsing (sapling either dead or not expected to recover; ≥90 % biomass removed); (6) antler damage (rubbing and/or thrashing damage such as stem breakage) and/or bark stripping present.

It is possible to measure changes in damage caused by deer trampling and track creation, and wallowing, however, the methods are quite involved (Bennett et al. 2015a). The 'bare ground' cover estimate (Section 2.4.2.1) will provide a simple index of disturbance caused by deer wallowing, trampling, rutting and fighting, that may lead to impacts such as weed invasion and erosion.

2.4.2.5 Vegetation sampling units

Vegetation will be surveyed in permanent 10 m × 10 m plots, which will be compared between Treatment and Reference sites. Vegetation variables will be estimated using quadrats rather than transects because many of the EVCs being sampled occur in small patches and quadrats will avoid the possibility of sampling units extending beyond the study area of interest, which could happen with transects.

Permanently marked quadrats of 10 m × 10 m have been selected because they have been used to quantify rusa deer impacts in littoral rainforest, sandstone gully forest and sandstone heath (Moriarty 2004). While larger quadrats have been selected to quantify deer impacts for some studies of highly heterogeneous environments or rainforest characterised by features such as large trees and canopy gaps that may dominate small plots (e.g., 20 m × 20 m; Davis 2017), 10 m × 10 m plots are expected to be large enough to assess the parameters of interest in this study, and small enough to allow rapid assessments to be undertaken.

2.5 REFERENCES

Allen, R. B., D. M. Forsyth, R. K. J. Allen, K. Affeld, and D. I. MacKenzie. 2015. Solar radiation determines site occupancy of coexisting tropical and temperate deer species introduced to New Zealand forests. PLoS ONE **10**:e0128924.

Amos, M., G. Baxter, N. Finch, and P. Murray. 2014. At home in a new range: wild red deer in south-eastern Queensland. Wildlife Research **41**:258-265.

Barrett, K. J., and O. J. Schmitz. 2013. Effects of deer settling stimulus and deer density on regeneration in a harvested southern New England forest. International Journal of Forestry Research **2013**:11 pages.

Bengsen, A. J., L. K.-P. Leung, S. J. Lapidge, and I. J. Gordon. 2011a. Using a general index approach to analyze camera-trap abundance indices. The Journal of Wildlife Management **75**:1222-1227.

Bengsen, A. J., L. K.-P. Leung, S. J. Lapidge, and I. J. Gordon. 2011b. Using a general index approach to analyze camera-trap abundance indices. The Journal of Wildlife Management **75**:1222-1227.

Bennett, A. 2008. The impacts of sambar (*Cervus unicolor*) in the Yarra Ranges National Park The University of Melbourne, Parkville, Victoria.

Bennett, A. 2012. Sambar presence at Lake Mountain Yarra Ranges National Park and Mount Bullfight Nature Conservation Reserve alpine bogs. Unpublished Client Report for Parks Victoria, University of Melbourne, Parkville, Victoria.

Bennett, A. 2013. Deer in Yellingbo Nature Conservation Reserve, Victoria. Unpublished Client Report for Melbourne Water, University of Melbourne, Parkville, Victoria.

Bennett, A. 2016. Littoral Rainforest and Coastal Vine Thicket of Eastern Australia: A method for monitoring deer abundance and impacts. Unpublished report prepared for Department of Environment, Land, Water and Planning. North Melbourne, Victoria.

Bennett, A., and G. Coulson. 2008. Evaluation of an exclusion plot design for determining the impacts of native and exotic herbivores on forest understoreys. Australian Mammalogy **30**:83-87.

Bennett, A., and G. Coulson. 2011. The impacts of Sambar *Cervus unicolor* on the threatened Shiny Nematolepis *Nematolepis wilsonii*. Pacific Conservation Biology **16**:251-260.

Bennett, A., and N. E. Davis. 2017. A field monitoring protocol for investigating deer impacts in the Jumping Creek Catchment, Victoria. A report prepared for Manningham City Council and the Jumping Creek Catchment Landcare Group. Melbourne, Victoria.

Bennett, A., N. E. Davis, and D. M. Forsyth. 2015a. Monitoring change in deer impact associated with the Parks Victoria deer control trial in the Alpine National Park: survey design and rationale. Unpublished report prepared for Parks Victoria.

Bennett, A., S. Haydon, M. Stevens, and G. Coulson. 2015b. Culling reduces fecal pellet deposition by introduced sambar (*Rusa unicolor*) in a protected water catchment. Wildlife Society Bulletin **39**:268–275.

Bentley, A. 1998. An Introduction to the Deer of Australia with Special Reference to Victoria. 3rd edition. Australian Deer Research Foundation Ltd., Croydon, Victoria.

Bilney, R. J. 2013. Antler rubbing of yellow-wood by sambar in East Gippsland, Victoria. The Victorian Naturalist **130**:68-74.

Bressette, J. W., and H. Beck. 2013. The effects of high deer density on forest regeneration and carbon sequestration.*in* J. A. Daniels, editor. Advances in Environmental Research. Nova Science Publishers.

Bridges, A. S., and A. S. Noss. 2011. Behaviour and activity patterns.*in* A. F. OÇonnell, J. D. Nichols, and K. U. Karanth, editors. Camera traps in animal ecology: Methods and analyses. Springer, London.

Bridges, A. S., M. R. Vaughan, and S. Klenzendorf. 2004. Seasonal variation in American black bear *Ursus americanus* activity patterns: quantification via remote photography. Wildlife Biology **10**:277-284.

Bulinski, J., and C. McArthur. 2003. Identifying factors related to the severity of mammalian browsing damage in eucalypt plantations. Forest Ecology & Management **183**:239-247.

Cohen, J. 1988. Statistical power analysis for the behavioral sciences. 2nd edition edition. Erlbaum, Hillsdale, NJ.

Cote, M., J. Ferron, and R. Gagnon. 2003. Impact of seed and seedling predation by small rodents on early regeneration establishment of black spruce. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere **33**:2362-2371.

Davis, N. E. 2014. Monitoring herbivore relative abundance as part of the Yanakie Isthmus Restoration Project. Unpublished report prepared for Parks Victoria.

Davis, N. E. 2017. Assessing the impacts of wild deer on remnant Threatened Ecological Communities in the Illawarra: Survey design and rationale. Unpublished report prepared for the Office of Environment and Heritage.

Davis, N. E., A. Bennett, and D. M. Forsyth. 2015. Monitoring changes in deer abundance and habitat use associated with the Parks Victoria deer control trial in the Alpine National Park: Survey design and rationale Unpublished report prepared for Parks Victoria.

Davis, N. E., A. Bennett, D. M. Forsyth, D. M. J. S. Bowman, E. C. Lefroy, S. W. Wood, A. P. Woolnough, P. West, J. O. Hampton, and C. N. Johnson. 2016. A systematic review of the impacts and management of introduced deer (Family: Cervidae) in Australia. Wildlife Research **43**:515-532.

Davis, N. E., and G. Coulson. 2010. Mammalian browse damage to revegetation plantings in a national park. Ecological Management and Restoration **11**:72-74.

Davis, N. E., H. Sitters, J. Di Stefano, J. Whelan, J. Wright, L. Ludewigs, and G. Coulson. in prep. Efficiency of frequency and occupancy indices derived from faecal pellet counts and camera trapping to monitor wildlife activity.

DELWP. 2017. Ecological Vegetation Class (EVC) Benchmarks for each Bioregion. http://www.depi.vic.gov.au/environment-and-wildlife/biodiversity/evc-benchmarks Accessed February 2018. Department of Environment, Land, Water and Planning,, Melbourne, Victoria.

Department of Environment Land Water and Planning. 2017. Standard Operating Procedures Field Guide (v 1.1.0) Ecosystem Resilience Monitoring Program. Melbourne.

Department of the Environment. 2015. Approved Conservation Advice for the Littoral Rainforest and Coastal Vine Thickets of Eastern Australia ecological community. Canberra: Department of the Environment. Available from: http://www.environment.gov.au/biodiversity/threatened/communities/pubs/76-conservation-advice-12112015.pdf. In effect under the EPBC Act from 12-Nov-2015.

Dolman, P. M., and K. Wäber. 2008. Ecosystem and competition impacts of introduced deer Wildlife Research **35**:202-214.

Forsyth, D. M., and N. E. Davis. 2011. Diets of non-native sambar deer in Australia estimated by macroscopic versus microhistological rumen analysis Journal of Wildlife Management **75**:1488–1497.

Forsyth, D. M., D. J. Wilson, T. A. Easdale, G. Kunstler, C. D. Canham, W. A. Ruscoe, E. F. Wright, L. Murphy, A. M. Gormley, A. Gaxiola, and D. A. Coomes. 2015. Century-scale effects of invasive deer and rodents on the dynamics of forests growing on soils of contrasting fertility. Ecological Monographs **85**:157–180.

Fraser, K. W., and G. N. Nugent. 2005. Sambar Deer. The Handbook of New Zealand Mammals. Pages 436–442 *in* C. M. King, editor. Oxford University Press, Melbourne.

Gormley, A. M., D. M. Forsyth, P. Griffioen, M. Lindeman, D. S. L. Ramsey, M. P. Scroggie, and L. Woodford. 2011. Using presence-only and presence–absence data to estimate the current and potential distributions of established invasive species. Journal of Applied Ecology **48**:25–34.

Green, R. H. 1979. Sampling Design and Statistical Methods for Environmental Biologists. Wiley, Chichester.

Gunaikurnai Traditional Owner Land Management Board. 2017. Draft Gunaikurnai and Victorian Government joint management plan part 2(c) Lake Tyers State Park – Krauatungalung Country.

Hnatiuk, R. J., R. Thackway, and J. Walker. 2009. Vegetation. Pages 73-125 Australian Soil and Land Survey Field Handbook. CSIRO Publishing, Collingwood, Vic.

Husheer, S. W., and A. W. Robertson. 2005. High-intensity deer culling increases growth of mountain beech seedlings in New Zealand. Wildlife Research **32**:273–280.

Jacobson, H. A., J. C. Kroll, R. W. Browning, B. H. Koerth, and M. H. Conway. 1997. Infrared-triggered cameras for censusing white-tailed deer. Wildlife Society Bulletin **25**:547–556.

Lewis, J. C., L. B. Flynn, R. L. Marchinton, S. M. Shea, and E. M. Marchinton. 1990. Biology of sambar deer on St. Vincent National Wildlife Refuge, Florida. Tall Timbers Research Station Bulletin **25**:1–107.

Liddicoat, E. K. H.-R. 2008. Population density and habitat use of two deer species in a Victorian water catchment. Department of Zoology, The University of Melbourne, Melbourne.

Lindenmayer, D. B., and G. E. Likens. 2010. Effective Ecological Monitoring. CSIRO Publishing, Collingwood.

MacKenzie, D. I., and J. A. Royle. 2005. Designing occupancy studies: general advice and allocating survey effort. Journal of Applied Ecology **42**:1105–1114.

Marks, C. A., and S. J. Moore. 1998. Nursery practices influence comparative damage to juvenile blue gum by wallabies (*Wallabia bicolor*) and European rabbits (*Oryctolagus cuniculus*). Forest Ecology & Management **112**:1-8.

Mason, E. 2006. Secrets of the Sambar. Shikari Press, Bairnsdale, Victoria.

Meek, P. D., G. Ballard, A. Claridge, R. Kays, K. Moseby, T. O'Brien, A. O'Connell, J. Sanderson, D. E. Swann, M. Tobler, and S. Townsend. 2014. Recommended guiding principles for reporting on camera trapping research. Biodiversity Conservation **23**:2321–2343.

Meek, P. D., G. Ballard, and P. Fleming. 2012. An Introduction to Camera Trapping for Wildlife Surveys in Australia. PestSmart Toolkit publication, Invasive Animals Cooperative Research Centre, Canberra, Australia.

Menkhorst, P., and F. Knight. 2011. A field guide to the mammals of Australia. 3rd edition. Oxford University Press, South Melbourne, Victoria.

Moriarty, A. 2004. The ecology and environmental impact of rusa deer (*Cervus timorensis*) in the Royal National Park. Ph D. University of Western Sydney, Sydney, Australia.

Morrison, M. L., W. M. Block, M. D. Strickland, B. A. Collier, and M. J. Peterson. 2008. Wildlife Study Design. Second Edition edition. Springer.

Nelson, J. L., and M. P. Scroggie. 2009. Remote cameras as a mammal survey tool—survey design and practical considerations. Arthur Rylah institute for environmental research unpublished report no. 2009/36. Department of Sustainability and Environment, Heidelberg.

Nichols, J. D., K. U. Karanth, and A. F. OÇonnell. 2011. Science, Conservation, and Camera Traps.*in* A. F. OÇonnell, J. D. Nichols, and K. U. Karanth, editors. Camera traps in animal ecology: Methods and analyses. Springer, London.

Nudds, T. D. 1977. Quantifying the vegetative structure of wildlife cover. Wildlife Society Bulletin **5**:113–117.

Nugent, G. 1994. Home-range size and its development for fallow deer in the Blue Mountains, New Zealand. Acta Theriologica **39**:159-175.

Nuttle, T., T. E. Ristau, and A. A. Royo. 2014. Long-term biological legacies of herbivore density in a landscape-scale experiment: forest understoreys reflect past deer density treatments for at least 20 years. Journal of Ecology **102**:221–228.

O'Çonnell, A. F., J. D. Nichols, and K. U. Karanth. 2011. Preface.*in* A. F. OÇonnell, J. D. Nichols, and K. U. Karanth, editors. Camera traps in animal ecology: Methods and analyses. Springer, London.

Odden, M., and P. Wegge. 2007. Predicting spacing behaviour and mating systems of solitary cervids: A study of hog deer and Indian muntjac. Zoology **110**:261-270.

Parker, B. D. 2009. Feeding biology of sympatric sambar (*Cervus unicolor*) and fallow deer (*Dama dama*) in NE Victoria. The University of Melbourne.

Parks Victoria. 2005. Threat Monitoring Protocol: Deer (Family: Cervidae). Parks Victoria, Melbourne, Australia.

Peel, B., R. J. Bilney, and R. J. Bilney. 2005. Observations of the ecological impacts of Sambar *Cervus unicolor* in East Gippsland, Victoria, with reference to destruction of rainforest communities. The Victorian Naturalist **122**:189-200.

Richardson, J. 2015. Kurrajong survey. Unpublished report. (Federation Training: Lakes Entrance.).

Simard, M. A., C. Dussault, J. Huot, and S. D. Côté. 2013. Is hunting an effective tool to control overabundant deer? A test using an experimental approach. The Journal of Wildlife Management **77**:254–269

Spaggiari, J., and M. de Garine-Wichatitsky. 2006. Home range and habitat use of introduced rusa deer (*Cervus timorensis russa*) in a mosaic of savannah and native sclerophyll forest of New Caledonia. New Zealand Journal of Zoology **33**:175–183.

Stewart-Oaten, A., W. W. Murdoch, and K. R. Parker. 1986. Environmental impact assessment: "Pseudoreplication" in time? Ecology **67**:929–940.

Stockwell, M. 2003. Assessing the Levels and Potential Impacts of Browsing by Sambar Deer (*Cervus unicolor*) in the Upper Yarra Catchment, Victoria. Honours. Monash University, Melbourne.

Strole, T. A., and R. C. Anderson. 1992. White-tailed deer browsing: species preferences and implications for central Illinois forests. Natural Areas Journal **12**:139-144.

Tobler, M. W., and G. V. N. Powell. 2013. Estimating jaguar densities with camera traps: problems with current designs and recommendations for future studies. Biological Conservation **159**:109–118.

Triggs, B. 2003. Tracks, Scats and Other Traces; a field guide to Australian mammals. Oxford University Press, Melbourne.

TSSC. 2008. Listing advice to the Minister for the Environment, Water, Heritage and the Arts from the Threatened Species Scientific Committee (the Committee) on Amendments to the List of Ecological Communities under the Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act). Threatened Species Scientific Committee (TSSC), Department of the Environment, Australian Government: Canberra, Australian Capital Territory.