Monitoring Feral Deer in the Illawarra Escarpment State Conservation Area

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Abstract
Since the beginning of the Anthropocene, human induced impacts have led to reduction in ecosystem functioning and a reduction in biodiversity, the like of which has not been seen for 60 million years. One of the most common impacts faced by scientists and land managers alike is the threat of ecological invasions and effects that they are able to impose on regional environments. To allow for the creation of well informed and successful management procedures, effective and reliable methods of monitoring and surveying populations of invasive species are needed.

Locally, feral deer populations are responsible for a reduction in faunal and floristic biodiversity, the destruction and clearing of large volumes of biomass, the degradation of soil quality, increased rates of soil erosion, and cost residents in excess of $500 000 a year in damage to property. Worldwide, various methods have been developed to monitor trends in deer abundance over time. From a management standpoint relative measures are often favoured as they are generally cheaper and more easily implementable. To investigate the most suitable practice for monitoring deer populations of the area, relative abundance estimates garnered from three different methodologies, camera trapping, modified distance sampling methods and faecal accumulation plots, were compared in terms of their reliability and ability to monitor changes in deer abundance over time.

No two methods provided significantly similar estimates of relative abundance at all sampled sites. At two of the three sites, data from camera traps and spotlight assisted distance sampling procedures were significantly correlated. Faecal plots and daytime visual observations were not significantly similar to any other measures of abundance. Data from faecal accumulation methods were highly variable and may not be suitable for use in scientific or management procedures. Distance Sampling of feral deer indicate that there are 0.4 deer per hectare within the IESCA, which is equivalent to approximately 1090 deer in the conservation area.

It is suggested that long term monitoring procedures focus on either camera trap based methodologies or spotlight assisted distance sampling to monitor deer populations. This information will hopefully allow for the creation and implementation of more effective monitoring and management procedures in the future. Future studies should investigate the effect of increasing periods of deployment on the reliability of estimates garnered from camera traps and faecal accumulation plots. Additionally, these methods should be tested against a population if a known size to test if garnered estimates are representative true abundance patterns.

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Monitoring Feral Deer in the Illawarra Escarpment State Conservation Area

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ABSTRACT

Invasive species currently cause significant environmental and economic losses worldwide, impacting ecosystem functioning and contributing to global losses in biodiversity. With a pronounced upsurge in the occurrences and extent of biological invasions in the past 50 years, the need to formulate and implement effective management strategies has become paramount. To ensure proposed management techniques are effective, correct and efficient methods of surveying and monitoring the abundance of target organisms are needed.

Of the numerous invasive species that inhabit Australia, feral deer, despite being relatively unstudied, are known to have severe and varied consequences for the ecosystems that they inhabit. Of the 200 000 deer currently inhabiting Australia, several populations of concern exist along the south coast of New South Wales. To inform current and future monitoring and management programs, several commonly used methodologies were employed and subsequently compared in their ability to survey deer abundance within the IESCA. Relative abundance indices were created from data collected from areas surrounding Mt Kiera and Mt Kembla by infrared triggered camera traps, faecal accumulation based methods and transect sampling performed during the day and evening over a seven-week period during the deer’s mating season.

Of the four methods tested, camera trapping and spotlight based transect sampling both returned estimates of abundance that were well correlated and consistently recorded the greatest number of deer. Faecal accumulation based methods and daytime transect sampling did not provide estimates of abundance that were correlated with any other method and occasionally were not able to detect any deer in the area. Distance sampling undertaken during spotlight procedures indicate that local deer populations exist at densities of approximately 0.4 individuals per hectare.

This study illustrates that not all commonly employed methods of abundance estimation applied to deer populations will provide similar measures of abundance and provides some of the first empirical evidence that estimates produced by transect sampling undertaken during different times of the day will differ significantly. While ideally these methods could be validated in areas with a known population, results suggest that for future monitoring programs in this area, methods focused on camera trapping or spotlighting would be most effective.
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LIST OF ABBREVIATIONS

FAR – Faecal Accumulation Rate

FPI – Faecal Pellet Index

FSC – Faecal Standing Crop

IESCA – Illawarra Escarpment State Conservation Area

ITC – Infrared Triggered Camera Trap

KAI – Kilometric Abundance Index

KAI-I – Individual Kilometric Abundance Index

KAI-G – Group Based Kilometric Abundance Index

NPWS – National Parks and Wildlife Service

RNP – Royal National Park
1 INTRODUCTION

1.1 INVASIVE SPECIES AND ECOLOGICAL INVASIONS

Since the beginning of the Anthropocene, rates of species extinction have dramatically increased on both global and local scales (Dirzo et al., 2014; Pimm et al., 2014). Anthropogenic impacts such as habitat destruction and fragmentation, climate change, hunting and the introduction of invasive species have had noticeable effects on ecosystems worldwide, reducing biodiversity and ecosystem functioning and contributing to this extinction crisis (Fahrig, 2003; Worm et al., 2006). Of these numerous impacts, one of the most prominent issues commonly faced by scientists and environmental managers alike is the threat of ecological invasions and the effects that invasive species are able to impose on ecosystems. While many view all species that are non-indigenous to an area as invasive, exactly which organisms that should, and are able to be, definitively classified as invasive species have been subject to much debate since the coining of the term with various opposing views arguing for classification on the basis of the organisms’ ‘natural’ range, effects on environments and economies and the degree to which humans have played a role in their dispersal (Colautti & MacIsaac, 2004).

Historically, the geographic distribution of species has been primarily determined by a combination of relatively long term processes such as selection, drift, geographic isolation and shifts in climatic conditions (Davis & Shaw, 2001; Krebbs, 2014). These patterns of distribution have changed in more recent times, and are likely to continue to change with increases in anthropogenic activity leading to a pronounced upsurge in the number of occurrences and extent of biological invasions across the globe (Mack et al., 2000; Pyšek & Richardson, 2010). Though the vast majority of immigrating species do not survive initial dispersal and acclimation periods, the small fraction of surviving constituents often thrive in their new environments due to the lack of natural predators or competitors, human interference and habitat modifications created by the processes of previously invading species (Fig. 1) (Mack et al., 2000; Simberloff & Von Holle, 1999). Currently the most accepted and widely employed definition of invasive species used by scientists and managing bodies alike, including all facets of the Australian Government and its enterprises, is that of a species that has been
subject to an, or a series of, anthropogenically mediated dispersal event(s) allowing for the
inhabitation of a novel environment to the detriment of native species (Colautti & MacIsaac,
2004). The magnitude of the effects that invasive species have on their environment will
depend on the extent of their naturalisation and inhabitation of their novel range (Fig. 1). To
determine whether or not a species may fit these criteria and is in need of a targeted
management response, the effects that the species impose on their environment and the
characteristics of the invasive population should be closely monitored, especially when these
effects may be discrete or density dependant.

Fig. 1: Stages of a biological invasion. Each ‘filter’ removes approximately 90% of potentially invasive species. Organisms at different stages of invasion will exhibit effects of differing intensity and require varying approaches to manage effectively (adapted from Colautti & MacIsaac, 2004; Krebbs, 2014; Lockwood et al., 2005).
This great concern and need to monitor the influx of invasive species is caused by the potential that these organisms have to cause major damage to regional environments, economies, ecosystems and cultures (Clavero & Garcia-Berthou, 2005; Pimentel et al., 2005; Thomsen et al., 2011). This holds a particular importance to Australian environments as its unique biosphere has evolved in relative isolation for millions of years and contains some of the highest levels of biological endemism in the world. High levels of endemism and isolation from distinct biological environments have been thought to be linked to an increased susceptibility to invasion events and disproportionately high environmental impacts from invasive species (Thresher, 1999). These pronounced impacts have already been observed in Australia with invasive species having numerous observable effects on native environments including, but not limited to; the reduction in regional biodiversity (Bax et al., 2003; Butchart et al., 2010), the alteration of fire regimes (Rossiter et al., 2003), the modification of vegetation communities and structure (Meissner & Facelli, 1999; Moriarty, 2004a), the reduction in abundance of native species through direct predation or direct competition with sympatric species (Fogarty & Facelli, 1999; Moriarty, 2004a) and the inducement of morphological changes in consumers (Phillips & Shine, 2004). The majority of these impacts are density dependant with varying population densities and distributions causing impacts of differing intensity.

In addition to these ecological impacts, invasive species currently cost the Australian economy over $5 billion dollars (AUD) a year though their detrimental effects on the agricultural industry and the implementation of necessary research and management programs (Gong et al., 2009; Sinden et al., 2004). While the effects of common pest species such as the feral cat and Cane Toad are relatively well studied and quantified, the distribution and abundance of novel species such as feral deer and camels are less well documented. With the lack of information concerning these species, the true impact of these organisms cannot be quantified and the criticality of the situation truly assessed. To formulate effective management procedures, monitor and predict possible implications that these novel invasive species may bestow on certain environments, effective methods for surveying changes in distribution and abundance are required. Without information on the occurrences and characteristics of these species, the implementation and effectiveness of any required or applied management procedures are likely to be significantly affected.
1.2 FERAL DEER IN THE SOUTH COAST AND ILLAWARRA ESCARPMENT STATE CONSERVATION AREA

A widespread, ecologically plastic clade of ungulates, invasive and overabundant deer have quickly emerged as a significant threat to numerous environments around the world due to their generalist diet, hoofed feet, large ranges and the social and economic impacts that they bring with them (Côté et al., 2004; Tuckwell, 2003). Invasive and overabundant deer species are often some of the most influential components of ecosystems and with the effects of deer causing cascading effects on trophic structures, species distributions and abundances and even landscape morphology (Salk et al., 2011). As these impacts have become more pronounced and studied, land managers within Australia have begun to take the threat of environmental degradation by deer more seriously and have put in place procedures to manage feral deer populations in areas of high ecological value.

Within Australia, deer have been present since the early 19th century after their deliberate introduction by European settlers (Moriarty, 2004b). Present day distributions are a product of the actions of early acclimation societies, releases and escapes from farms and deliberate translocations by hunting groups (Moriarty, 2004a, 2004b). Currently there are an approximate 200 000 feral deer distributed across Australia with majority of these populations inhabiting areas of the south eastern coast. Several of these populations have been identified as having great potential to significantly damage and affect the surrounding environment which has led to the implementation of various management programs in southern New South Wales and northern Victoria (NLWRA, 2008). One of these populations of concern is currently the target of a management program within the Illawarra Escarpment State Conservation Area (IESCA). The IESCA is a protected area situated within the greater Illawarra Region, located along the south coast of New South Wales, consisting of a series of fragmented segments of land dispersed throughout areas of rainforest and sclerophyllous forests. Deer have been present within the IESCA and surrounding areas since the 1980’s with populations likely stemming from original populations of Rusa Deer (*Cervus timorensis*) first released in the neighbouring Royal National Park (RNP) in 1907. With bolstered feral deer populations after bushfires in the mid 1990’s within the RNP saw deer disperse into surrounding areas and the IESCA’s identification as a hotspot for potential for illegal translocations for hunting purposes, deer in the area quickly became a pest species of significant concern, causing ecological problems and damage to local agricultural land and property (Wollongong City Council, 2013).
1.3 THE IESCA DEER POPULATION

Despite the implementation of management procedures, there has been little effort made to study the abundance or demographic characteristics of deer populations in the area. Anecdotal evidence and hunting records indicate that the majority of deer within the IESCA and in the Greater Illawarra area are Rusa Deer with smaller numbers of Fallow (*Dama dama*), Red (*Cervus elaphus*) and Sambar Deer (*Cervus unicolor*) also present (Dawson, 2012; NPWS, 2011). Additionally, while sex ratios of Rusa Deer in the RNP are approximately 1:1 (Moriarty, 2004a), hunters across the Illawarra have harvested varying proportions of fawns, males and females, suggesting that varying population structures are exhibited over the larger area (Dawson, 2012). However, this data is likely to be influenced by different hunting groups’ preferences as many ‘trophy hunters’ will preferentially target large males with well-developed antlers, meaning data is not likely to be truly indicative of the demography of the local deer population.

No records exist on the absolute abundance of deer within the area. The relative abundance of local deer populations was mapped by Dawson (2012) using Faecal Pellet Indices along ecotonal boundaries of the escarpment itself. Faecal data indicates that deer activity is highest in areas surrounding Mt Kiera and Mt Kembla. When compared to results from studies performed overseas, faecal pellet data indicated that densities of deer are likely to be approximately 116 to 49 deer per km$^2$ in areas north of Mt Kembla with densities closer to 24 deer per km$^2$ in the area’s southern reaches (Dawson, 2012). However, data from relative abundance measures, especially from methods that vary with climatic and other regional factors, cannot be expected to provide accurate representations of true population numbers when applied to other environments (Forsyth et al., 2007).

Current management procedures consist primarily of ground shooting, with options such as trapping, fencing and habitat corridor manipulation being investigated to control the abundance and combat the effects that deer impose on the environment (NPWS, 2015; Wollongong City Council, 2013). Research into the characteristics of the IESCA deer population are needed to ensure that implemented management programs are well informed and effective.
1.3.1 Effects of Invasive Deer

Invasive and overabundant deer populations have numerous impacts on their environment. These include the reduction of native vegetation cover, increases in levels of soil erosion, the facilitation of invasions by exotic plant species, decrease in leaf litter macrofaunal biodiversity and abundance and the exclusion of sympatric species. One of the greatest concerns raised by the presence of deer within an invaded environment is the fact that deer are ungulates. All terrestrial ungulates possess thick keratin coating over the edge of the toes that bear most of the animal’s weight, commonly known as hooves. These hooves are able to disturb soil structure and uproot vegetation through normal organism movement (Côté et al., 2004). In environments that have evolved with a distinct lack of ungulate presence, such as Australia, the movement of these animals cause a distinctive decrease in the abundance and diversity of macrofauna that inhabit leaf litter (Forsyth et al., 2010) and significantly reduce the ground covering and biomass of native plants (Relva et al., 2010; Vavra et al., 2007). This destruction of biomass and the lessening of macrofaunal diversity opens up further opportunities for invasions by non-indigenous plant species (Simberloff & Von Holle, 1999). The trampling of ungulates, in combination with their grazing patterns, has also been shown to increase rates of soil erosion (Evans, 1997) and significantly affect soil quality (Mohr et al., 2005). This may prove to be a major problem for Australian landscapes as soils are generally poor in quality and may significantly affect agricultural operations and the recruitment and success of plant growth in revegetated areas. Farmed ungulates in Australia have already been shown to have significant effects on the recovery of soils, however the extent of deer related damage is unknown (Drewry, 2006). One of the major issues with these trampling related effects is the difficulty to monitor the effects of soil disturbance in areas of terrain where regular monitoring in impracticable, such as mountainous regions and heavily forested areas. As trampling is likely to be density dependant, monitoring the changes in density and abundance of regional deer populations is likely to give indications of the extent of these effects and allow for the implementation of relevant management procedures.

The effective management of ungulate populations requires a detailed understanding of target species’ feeding habits and diet composition (Bookhout, 1996). This knowledge is of particular importance when dealing with non-indigenous ungulates in relatively novel ecosystems and plant communities due to the likelihood that local vegetation communities have not evolved adaptations to sufficiently combat this evolutionary pressure (Forsyth & Davis, 2011). The
unspecialised, ruminant digestive system possessed by the majority of deer species allows for a great deal of dietary plasticity, giving rise to the rather generalist herbivore diet they exhibit. This generalist biology allows deer to alter their diet to less palatable food sources when more nutritious options are unavailable and have made deer a highly successful opportunistic feeder and invader (Forsyth et al., 2002; Nugent et al., 2001).

Rumen analyses have shown that while deer consistently avoid or prefer some plant species (Forsyth & Davis, 2011), deer diet is often dictated by their environment with larger volumes of grasses being consumed in cleared environments, shrubs in woodland and heath regions and trees in within rainforest environments (Moriarty, 2004a). Various studies have shown that significant dietary overlaps exist with native Australian mammals, primarily a number of Macropod species, and deer inhabiting Australia including Hog Deer (Davis et al., 2008), Rusa Deer (Moriarty, 2004a) and Fallow Deer (Boyle, 1995; Duncan, 1987). Due to the sheer volume of vegetation consumed by Rusa Deer and the existing dietary overlap between Rusa Deer and the Swamp Wallaby (Wallabia bicolor), it is suggested that the competition pressure created by a singular Rusa Deer in the RNP is equal to that of four Swamp Wallabies (Moriarty, 2004a). A negative correlation between Rusa Deer and Swamp Wallaby abundance combined with below average abundances of Swamp Wallaby within the RNP is thought to be due to this relatively high competition between the two species for resources, especially during the winter when resources are more sparse (Moriarty, 2004a). The effects of this overlap and subsequent competition for resources are likely to have led to the competitive exclusion of various animals that may have otherwise occupied relevant niches and is likely to be in effect in numerous other environments where significant populations of feral deer exist.

While possessing unspecialised digestive tracts, seasonal variations in the diets of some of the more well studied species of deer present in New South Wales have been noted. Forsyth and Davis (2011) showed that Sambar Deer, while primarily thought of as browsers, more closely resembled grazers during the summer months with the majority of their diet consisting of grasses, further supporting Moriarty’s (2004a) observations that deer diets and dietary overlaps with native mammals vary with season. While this pattern is present within some Australian environments, more research is needed to confirm whether this pattern is present throughout more environments as this is not observed in all settings. Deer are often seen as permanent to semi-permanent grazers in temperate locations and browsers in tropical settings. These seasonal differences often come at a time when plants are beginning to shoot, leading to low
densities of woody plants and a low success rate of recruitment in plant species in these environments (Keith & Pellow, 2005). These issues are particularly prominent in areas with threatened or endangered vegetation communities, such as the IESCA. As this effect has been demonstrated to be density dependant, like the effects of trampling, the monitoring of deer abundance is needed to quantify the effect that deer may have on vegetation communities.

The direct social and economic impacts that deer have on areas in which they are introduced to and abundant within are also well recognised, mainly through their direct damage to the agricultural industry and private property. The ability of feral deer to spread diseases to livestock is well documented around the world, with many of the deer species present within Australia, including Rusa, Red and Fallow Deer, contributing to these epizootics (e.g. FrÖLich et al., 2002; Mackintosh et al., 2004; Reid et al., 1999). While strict biosecurity procedures in present day Australia have seen the incidences of deer-spread disease remain low, diseases such as Deer Haemorrhaging Disease and Blue Tongue Virus, have been identified in populations of deer and nearby farmed ungulate communities, suggesting deer may assist in the spreading of these diseases (Parsonson & Snowdon, 1985; Weir et al., 1997). Though the relatively fragmented state of deer populations in Australia may mean that the threat of disease transmission by deer and the associated economic impacts are negligible for the moment, the spread of disease by deer in places such the United States of America and Europe costs relevant economies millions of dollars a year in lost revenue and healthcare costs (Bishop, 2004; Gortázar et al., 2006; Šumilo et al., 2008).

In addition to disease risks and agricultural effects, damage to property and human wildlife interactions such as deer-automobile collisions have significant social and economic impacts. In an Australian setting, invasive deer species have had a notable effect on the amenity of communities. Reports from the South Coast of New South Wales indicate that damage caused to property by wild deer populations exceeds $1 million (AUD) per year with further economic damage and humans causalities seen in deer-vehicle collisions (NPWS, 2015; Wollongong City Council, 2013). Greater protections and reserve formation are only likely to increase human-wildlife interactions. To combat these collisions and damage to property, the monitoring and alteration of deer movements is needed to reduce the amount of these negative human-wildlife interactions.
Wild deer populations have a number of ecological, economic and social impacts within Australia and around the world. As the majority of these effects are density dependant, effective survey methods are required to inform correct and efficacious land management practices to minimise the damage caused by these environments.

### 1.3.2 Aims and Significance

With an abundance of threatened and endangered ecological communities within the IESCA and its surrounding areas, the threat that deer herbivory and trampling has on the existence of these communities is an issue of great concern. Furthermore, damage that feral deer cause to property costs residents in the area approximately $500 000 a year with additional expenses also spent on management procedures carried out by both the local council and National Parks and Wildlife Service (Wollongong City Council, 2013). Despite this ecological and economic damage, little information exists on the extent and/or demography of the deer populations in the area. To protect endangered or threatened communities and minimise social and economic impacts, species that are detrimental to ecosystems are often subject to various management programs. The formation of effective management procedures requires a detailed understanding of the distribution, abundance and biology of the target organism. Without a thorough knowledge of deer populations in the area, management procedures are likely to be hindered and their success difficult to monitor. For example, Hone et al. (2010) suggested that to effectively reduce Rusa Deer numbers and combat the relatively high fecundity of the species, 46% of the population would need to be removed yearly. Without known densities or abundances of deer in the area, the success of current culling programs cannot be properly assessed. Information regarding the absolute and relative abundance of deer would prove an effective tool from which future trends and management success can be measured.

Numerous methods exist to estimate of the abundance and distribution of organisms, however, each method will prioritise various factors (e.g. cost-effectiveness, time, accuracy, etc) and have varying levels of effectiveness within different environments. Gathering relevant information concerning the deer populations of the area is often laborious and limited due to economic, social and environmental constraints. The most effective and suitable methods for estimating deer abundance is likely to depend on a number of factors including terrain, habitat type, season and species behaviour (Smart et al., 2004; Staines & Ratcliffe, 1987). Identifying survey methods that provide reliable estimates of abundance can ensure that populations are
managed effectively and encourage regular monitoring practices, while the employment of unsuitable methods is likely to lead to poor management decisions and a spread of misinformation that can have serious consequences for both the target population and the surrounding environment. The comparison of results from various methodologies is also able to illustrate the downfalls of differing methods, allowing for the selection of suitable options and the correction of the apparent bias of some methods.

This thesis aims to provide information on the deer population present in the IESCA and investigate the usefulness of various methods at estimating the relative abundance of deer. More specifically, this thesis will:

1. Provide baseline measures of the relative abundance of deer within the IESCA from which the future trends may be measured to determine the success of current management programs.
2. Compare estimates of relative abundance garnered by camera trapping, visual observations undertaken via transect sampling and faecal data to determine the most effective methods for measuring abundance within the IESCA.
3. Provide additional demographic information and preliminary estimates of absolute abundance on the IESCA deer population that may assist in the implementation and monitoring of current and future management procedures.

1.4 MONITORING TECHNIQUES

Accurate and precise measurements of the abundance, distribution, population characteristics and density of pest species are vitally important for the implementation and development of effective management and conservation strategies. A number of methods have been developed to effectively quantify these features in a number of mammal populations, including camera trapping, transect sampling and faecal based methodologies.

1.4.1 Camera Trapping

First used in 1890’s, camera trapping has proven to be a useful, effective survey technique for the study of large to medium mammals and is more recently being employed in studies of small mammals (Glen et al., 2013; Kucera & Barrett, 2011). With technological advances in the 1960’s leading to an increase in this technique’s popularity as tool for scientific purposes,
camera traps are now widely used to survey and monitor the abundance, movements and behaviour of an array of animals around the world (Meek et al., 2012), ranging from some of the rarest and most inconspicuous animals, such as the Snow Leopard (*Uncia uncia*) (Jackson et al., 2006), to extremely abundant organisms such as White Tailed Deer (*Odocoileus virginianus*) in North America (Roberts et al., 2006) and feral cats (*Felis catus*) in New Zealand (Glen et al., 2013). Other survey methods, such as capture-recapture techniques, strip counts and drive counts are often laborious, time consuming, expensive and restricted in environments with limited visibility (Roberts et al., 2006). It is for these reasons that researchers will often choose to employ camera traps as they have been illustrated to be non-invasive and much more practical than other methods (Foster & Harmsen, 2012). Additionally, despite relatively high capital costs, the deployment of Infrared Triggered Cameras (ITCs) often proves to be the most cost effective practice due to the longevity of the individual units and their ability to provide rapid assessments of faunal communities (Balme et al., 2009; De Bondi et al., 2010; Silveira et al., 2003).

Camera traps generally involve the placement of remotely triggered cameras that are able to capture photographs of target organisms when activated by movement or timers. While these cameras come in a range of designs varying in sensor characteristics (passive infrared, microwave or physical triggers), flash capabilities (no flash, white or infrared) and recording capabilities (video or photography) (Glen et al., 2013), the majority of camera trapping studies of deer and ungulate populations employ ITCs. Cameras with microwave sensors produce false triggers and occasionally fail to trigger when movement is present at rates that are significantly higher than ITCs (Glen et al., 2013). This proves a major issue for methods such as capture recapture models and rate of detection models for describing organism density as these methods are directly reliant on the assumption that all target organisms that pass through the detection zone will be captured. By violating this assumption, the accuracy and reliability of estimates presented by these methods may be compromised. Additionally, while cameras with a white flash yield photographs that are more readily identifiable, the bright, visible flash produced is thought to frighten target animals. While this may be acceptable in studies of presence-absence data or some relative abundance models where only one photograph is required to determine these results (e.g. O’Brien et al., 2003), in studies where multiple captures are required, cameras should be placed in inconspicuous locations that will not significantly affect the capture rates of target organisms. These organisms may flee at the sight of bright flashes and not return,
affecting the rate and likelihood of detection of these animals, significantly affecting the results of density based estimations.

Measuring the abundance of organisms from camera data has been extensively researched and is continually evolving with means of estimating both relative and absolute abundance of organisms taking many forms. Most techniques for estimating abundance from camera trap data come in the form of altered capture-recapture and mark-resight methodologies. These methods generally separate camera data into ‘capture’ or ‘marking’ periods where a subset of the population is either tagged with easily recognisable apparatus such as ear tags or collars (Curtis et al., 2009; Roberts et al., 2006) or, in some species, recorded using recognisable patterns or markings to identify individuals (e.g. Jaguars (*Panthera onca*) (Maffei et al., 2011) and Ocelots (*Leopardus pardalis*) (Trolle & Kéry, 2003)). While cameras locations may be placed to maximise their capture probability on a local scale, altering locations over larger scales is likely cause bias and issues if extrapolating density estimation to areas where habitat may be of different or lesser quality.

One of the main issues with these capture-recapture methodologies is the need to initially mark a subset of the target population as this process can prove expensive and time consuming. While individuals of various species can be identified from natural markings or colourings, individuals of many species studied using camera traps are often not easily identifiable from natural markings alone. Without individual identification, an accurate rate and ratio of ‘marked’ or tagged individuals to ‘unmarked’ individuals captured/recaptured and subsequent estimates of abundance cannot be accurately calculated. Despite this, numerous studies have attempted to employ this capture-recapture methodology to investigate the abundances of these inconspicuous species using markers such as scars, tail length, spots of colour and parasite markings (Kelly et al., 2008; Noss et al., 2012; Watts et al., 2008). While this approach may be acceptable in small populations where all individuals have markers that make them individually identifiable to researchers, the difficulty and problems with the accuracy of these methods increase exponentially as population size increases (Foster & Harmsen, 2012). Issues of observer bias in species with no natural markers have been investigated in populations of both tapirs (*Tapirus terrestris*) (Oliveira-Santos et al., 2010) and pumas (*Puma concolour*) (Kelly et al., 2008). Oliveira-Santos et al. (2010) requested that various researchers count the number of individual tapirs from a data set of camera trap photographs collected using a population of a known size (eight individuals). Observers reported results with population size
varying from four to fourteen. Similarly, Kelly et al. (2008) revealed that, while researchers were in agreement on the identity of specimens in the majority of cases specimens (average 80% agreement between two researchers, 72.9% between three), differences in individual identifications between observers can result in density estimates that vary by over 100%. Additionally, when decisive identifications cannot be made, investigators often assume that the individual in question is the same that was previously captured at that location (e.g. Wallace et al., 2003). However, this assumption is only valid when studying species that have been observed and are well-known to inhabit and defend exclusive territories and can lead to an underestimation of population size if applied in incorrect situations (Foster & Harmsen, 2012). Unless small populations of easily identifiable organisms exist, accurate, reliable measures of abundance from these methods may only be attained through physical capture and/or tagging of a subset of individuals from a population. This process can prove time consuming and be expensive due to the relatively high costs required to capture medium to large mammals and the need for permits to do so in many locations.

To resolve these identification issues, mark-resight methods have been developed. As an alternative to capture-recapture methodologies, mark-resight methods do not require all captured individuals to be marked or captured and therefore allows for the calculation of estimations of abundance from populations where only some individuals have identifying features. This method is generally not used as widely as traditional capture-recapture methods as the need to mark animals with collars or tags defeats the purpose of a non-invasive camera study that is cheaper than live trapping studies (Foster & Harmsen, 2012). However, this mark-resight methodology has previously been employed to study deer without the use of external tagging equipment. Watts et al. (2008) divided data captured by infrared triggered cameras into a marking period where male deer where identified by the appearance of their antlers and a resight period where sightings of these individuals where subsequently recorded. However, in these populations, estimates of population size were rather small (<18 individuals) and had not been quantified via other methods or against a known population to test their accuracy. Therefore, the reliability of this methodology and the power of using antler patterns as a proxy for tagging procedures is unknown.

The need for resightings and recaptures and the problems that these methods entail have led to the development of methods that do not require individual identification to garner estimates of density. Carbone et al. (2001) suggested that photographic rates alone could be
used to determine the density of certain species. This rate of capture has formed the basis for most measures of relative abundance garnered by camera traps (Jenks et al., 2011; Sollmann et al., 2013; Treves et al., 2010). These methods of relative abundance created by the use of remotely triggered cameras are attractive due to the ease of their implementation and their relative cheapness when compared to those estimates of absolute abundance gathered from capture-recapture methods. These estimates, usually presented as animal per camera unit time, have been shown to correlate with absolute estimates garnered from capture-recapture methods and distance sampling methodology (see Section 1.3.2) (Carbone et al., 2001; O'Brien et al., 2003).

The major criticism of these camera based relative abundance indices is that they are overly simplistic (Carbone et al., 2001; Jennelle et al., 2002). It is suggested that for an estimate of relative abundance to provide accurate information on the characteristics of target populations, estimates should demonstrate a functional relationship with the density of individuals and be calibrated with independent estimates of animal density (Jennelle et al., 2002). However, both Carbone et al. (2001) and O'Brien et al. (2003) demonstrated that these estimates of relative abundance are able to give estimates that fall in line with those generated via capture recapture and mark resight methods for large carnivores and suggest that more accurate measures may be gained by employing these methods to measures of ‘prey’ species as they are far more abundant than carnivores and additional captures will minimise the variation observed during sampling, providing more precise measures (Carbone et al., 2002).

Further building on the idea of using photographic capture rates to determine density of individuals, Rowcliffe et al. (2008) developed a method for estimating the absolute density of populations solely from camera properties and photographic rates. Basing this method on the ideal gas model, estimates of population density are derived from the photographic capture rates, movement speed of the target animal and detection area of the camera. In environments that are relatively homogeneous in their terrain and vegetation communities, this methodology has provided estimates of deer abundance that are relatively accurate in populations of a known size. However, this method has been relatively untested in environments that are heterogeneous in terms of their terrain and floral communities, such as the IESCA. Additionally, while this model holds the capability to include group size in its estimates, this model assumes all individuals (or groups) move independently of each other, which is not the case for highly territorial or social species, of which most deer species are. These non-random movements by deer are increased by the exclusion of individuals from ‘rutting territories’ by breeding males.
during deer mating seasons. Additionally, this assumption of random movement is not likely to hold true in mountainous regions such as the IESCA, where various untraversable features exist. This lack of ability to take deer behaviour into account is likely to affect density estimates, making them unsuitable for use in areas with a large variability in deer movement.

While capture-rate based models should not replace precise density measurements, these measures can give rapid and reliable estimates of population size where it would otherwise prove too costly or impracticable (Carbone et al., 2002). The majority of relative density estimates gathered from forested environments with well-maintained maintenance trails, similar to those within the IESCA are performed via transect sampling (see Section 1.3.2), however these methods are labour intensive and time consuming. While capital costs are greater in studies employing camera traps, the longevity of the units mean that it is often the cheaper choice within these environments (Silveira et al., 2003). How estimates of relative abundance garnered from both methods compare to each other is not well known and are thought to vary with different environmental factors such as visibility and terrain heterogeneity.

From a management standpoint, the need for quickly implementable and cost effective methods to estimate deer densities and monitor changes in population trends is paramount. Although capture-recapture models have been seen to be consistently provide the most reliable estimates of deer abundance, the general cost associated with tagging procedures often limit their use. While relative estimates garnered from camera trapping have been shown to provide an agreeable medium between expensive absolute abundance estimates and cheaper needs, how these relative abundance measures compare with those gathered from other methodologies is relatively unknown and unstudied.

1.4.2 Transect Sampling

Transect sampling remains one of the most popular techniques for estimating the density of vertebrates due to the relative speed and ease with which these methods can be performed. Numerous variations of these methods have been employed in studies of the abundance and distribution of deer in environments around the world, allowing for the creation of both relative and absolute measures of abundance (Acevedo et al., 2008; Smart et al., 2004; Vincent et al., 1991). While relative measures are often presented in simple formats, such as deer per unit length or time, and employed to investigate changes in populations over time or distance, a
great deal of research has been performed to enable the acquisition of accurate estimations of absolute abundance from transect data.

The most commonly used technique to estimate deer abundance is that of distance sampling (Buckland et al., 2005; Thomas et al., 2010). Data gathered following this methodology is able to be analysed mathematically using ‘distance sampling theory’, i.e. the probability that an object will be detected decreases in a predictable manner as distance from the observer increases. These methods have three main assumptions, however more recent advancements often allow one or more of these assumptions to be disregarded (Buckland et al., 2005). Assumptions of distance sampling include; objects along the line will be recorded for certain, objects are recorded in their original position and not subsequently re-recorded and that exact measurements of distance are taken (Buckland et al., 2005; Burnham et al., 1980). Several other minor assumptions also exist but their violation is likely to have minimal effects on the resulting estimates of density (Buckland et al., 2005). These methods have been applied in a range of environments for numerous organisms to great success. A number of studies have quantified distance sampling estimates against capture-recapture models and known populations of deer producing estimates similar to those known or given by other methods (Putman et al., 2011; Smart et al., 2004). However, due to the nature of management procedures and the need to conserve and allocate resources, when applied in many professional or management settings many basic features of distance sampling are often violated. This has led to the creation of simple relative abundance measures that do not require as many assumptions and criteria to be met, allowing for the rapid assessment of communities and easily implementable long term monitoring procedures.

Kilometric Abundance Indices (KAIs) are a simply derived method of estimating abundance and comparing variations in abundance over time as well as within groups of competing, mutualistic or predatory organisms (Acevedo et al., 2008; Vincent et al., 1991). These indices are calculated by sampling along standardised routes and calculating the number of target organisms or groups of organisms observed per kilometre sampled. How well these indices, as well as estimates of absolute abundance created from distance sampling methods, represent true patterns of abundance are a cause for debate, especially when transects are performed along roads or worn paths. While regularly used for convenience and repeatability and often the only safe option, the conducting of transect sampling procedures along roads or paths has been suggested to significantly affect density estimates of deer in forested environments (Ward
et al., 2004). Ward et al. (2004) found that deer numbers recorded along roads were relatively low, suggesting deer avoided the marked transects. However, this is likely to vary throughout sites as the frequency of path use and the type of use (i.e., vehicle vs. walking) is likely to affect deer habituation to the use of the path and their actions around the sampled path. An additional concern raised about the conducting of transect sampling to survey deer populations is the fact that there is a long-standing belief that population estimates will vary depending on the time of day the transect sampling is carried out. McCullough (1982) suggested that due to the cryptic nature of deer and their skittish behavior during the day, spotlighting will return more accurate estimates as deer are more active during the night. While this has led to the majority of deer populations being sampled with the aid of spotlights, few direct comparisons between observations performed at night and during the day exist to examine the legitimacy of this claim.

Despite these results and peculiarities, spotlighting and daytime transect sampling methods have been validated against regular distance sampling methodology, helicopter survey techniques and faecal methods for deer and other mammals (e.g., Acevedo et al., 2008; Garel et al., 2010; Garel et al., 2005; Heydon et al., 2000). The main disadvantage to transect sampling methods is the associated labour and time needed to develop reliable estimates of organism abundances and densities. As most animals have the inherent ability to move, great variations in collected data often present themselves within field recordings. This variation means that a far greater number of samples often have to be collected and analysed to achieve results of desired accuracy and precision. This is concerning from a management standpoint due to the resources that may be allocated to a given program. When comparing transect sampling methods and helicopter surveying, a notoriously expensive option, Garel et al. (2005) found that for long-term monitoring of Mouflon (Ovis orientalis orientalis) populations in mountainous environments, helicopter surveying was the most cost-effective option due to the amount of observers that it took to provide results with similar precision to that of the helicopter surveying. Similarly, camera trapping has also regularly been shown to be more cost-effective for long-term management procedures. Various forms of undesirable variation can arise during many forms of transect sampling that may skew results. This observed variation can be caused by numerous factors including the time of day or year that transects are undertaken, the habitat structure of the survey area, weather conditions and seasonal traits of animals (Wilson & Delahay, 2001). To combat this variation, it has been suggested that measurements should be stratified by these various factors, however this is not always possible.
due to limited resources and safety concerns involved in most management settings (Acevedo et al., 2008).

While transect sampling is easily implementable and has the ability to provide accurate estimates of absolute and relative animal abundance, it has several drawbacks. The large number of man-hours required to obtain a precise result and its usefulness in inhospitable terrain draws into question applicability to many situations.

1.4.3 Faecal Indices

Signs and indications of presence left by various species may be used as an indices of organismal abundance (Forsyth, 2005; Silveira et al., 2003). Used by researchers since at least the 1930’s (Bennett et al., 1940; Dice, 1941), one of the most popular approaches for attaining estimates of ungulate abundance/density is to use the amount of faecal matter deposited by target organisms in an area as a proxy for measures of absolute or relative abundance. Numerous models have been developed in an attempt to accurately measure deer and mammal abundance. The two foremost methods for estimating absolute deer abundance are the Faecal Standing Crop (FSC) and the Faecal Accumulation Rate (FAR) methods (Acevedo et al., 2008; Smart et al., 2004). FSC estimates are calculated using the amount of pellet groups in a given area using randomly placed plots (usually stratified by habitat). These measurements are then converted into an estimate of density by applying rates of defecation by deer (often measured in farms) and the rates of decay of pellet groups in that environment. This method differs with the more laborious and time-consuming FAR method which requires the initial clearing of all faecal matter within a plot. These plots are then sampled at a later date to determine the amount of pellets accumulated over a period time. This method eliminates the need for the estimation of dung persistence rates, however still requires that rates of deposition, which can vary between individuals, species, environments and season (Mitchell et al., 1985; Smart et al., 2004), are known.

Smart et al. (2004) tested both the FSC and FAR methods within Atlantic woodlands in areas with populations of deer of both known and unknown quantities. It was found that FSC provided significantly more accurate and precise results than the FAR method. Additionally, the FSC provided more immediate results, making it the more favourable management strategy compared to the FAR technique that often requires long periods of time to accumulate a reasonable amount of faecal matter for a reasonable estimate to be made. Supporting this
observation, Ellwood (2000) also found that FAR methods gave relatively low accuracy. Acevedo et al. (2008) experimented with the FSC method in both open and woody habitats in the Mediterranean and found that this method only provided accurate estimates of population size when only pellet groups of with twenty or more pellets were included in calculations.

While these FSC and FAR methods have proven useful in various locations, it is often difficult and expensive to estimate many of these parameters on site, which often leads to estimates that are based on work from another location (Forsyth, 2005). Using data based on works from other locations, particularly from overseas, can create inaccurate population estimates (Forsyth, 2005). To combat this need for certain parameters to be known, the presence, absence and abundance of faecal matter is often used to produce a relative measure of abundance from which population trends can be measured over time. These relative indices are regularly used in land management as they are generally less expensive and easier to implement than absolute abundance estimates while still providing information that is important to landholders, such as relative population densities, areas of activity and long term changes in population size (Dawson, 2012; Forsyth et al., 2011). Numerous forms of relative abundance measurements exist from these data, taking the form of the total amount of dung along transects or within plots, the accumulation of dung in an area or a combination of the two (Acevedo et al., 2008; Forsyth, 2005; Putman et al., 2011). The most commonly used method is that of the Faecal Pellet Index (FPI) (Forsyth, 2005). This method takes into account total number of pellets and pellet groups located along numerous relatively short transects stratified by habitat characteristics to produce a relative measure of deer activity, allowing for relatively rapid, easily repeatable surveys of deer abundance. This method has been shown to provide reliable long term estimates of relative abundance but is not particularly suited to illustrate short term changes in abundance, regularly requiring two years or a change in population size in excess of 30% before FPIs may show this change (Forsyth et al., 2011). There are currently no generally accepted indirect methods to accurately measure deer abundance trends over the short term.

Accumulation based methods of density estimation are thought to be more representative of short term trends in population sizes due to the inclusion of faeces that was deposited solely within a desired timeframe. This limiting of the amount of time that deer are able to deposit faeces for combats the fact that deer faeces are able to persist within environments for several weeks, months or even years (Acevedo et al, 2008). The main issue with these faecal based
methods of estimating deer abundance is the amount of time that is required to obtain enough faecal material to allow for the calculation of accurate and precise absolute estimates and the varying sizes of plots required to minimise variation. While the length of the accumulation period has a significant effect on the reliability of density estimates of deer, relative estimates are not affected as greatly as those estimating true abundance (Putman et al., 2011). In general, FAR plots are left to accumulate faecal material for approximately 60 days (Smart et al., 2004), however this is not always the case, especially in management situations. While, like many ungulate faeces, faecal pellets deposited by deer are resistant to weathering and may take several months to decay (Forsyth, 2005), when deposited in environments with a high rate of decay of organic material, such as rainforests, pellets may decay significantly faster than if they are deposited in other environments. This loss of faecal material often leads to the need to shorten accumulation periods. To attempt to resolve this shortened accumulation period, Ellwood (2000) investigated deer abundance using the unusually large plots size of 100m$^2$. While average plot sizes for accumulation based methods range from approximately 25m$^2$ to 60m$^2$ (Edwards & Hollis, 1982; Noor et al., 2010; Smart et al., 2004) and the fact that it is generally thought that larger plot sizes will provide more precise and less variable data for analysis, this expanded plot size from Ellwood was found to provide imprecise estimates of deer abundance. Similarly, Noor et al (2010) showed that for dry deciduous forest with populations of Sambar and Chital Deer, circular plots of 5m radii provide the lowest coefficients of variation when compared to both circular and square plots of both smaller and larger sizes. This is likely due the probability that pellet groups will be missed during the sampling of larger areas.

The use of highly variable data to estimate abundance, particularly absolute abundance, is likely to significantly affect given estimates. Despite even the best efforts to gather significantly non variable data, calculated error within these faecal methods is often great, especially when sampling in populations with low to medium densities. In measures of absolute abundance garnered from faecal data, it is common for estimates to have confidence limits of 30-50% of the derived population due to the highly variable nature of the data (Campbell et al., 2004; Putman et al., 2011). While stratifying estimates and measurements by habitat may assist in lessening this variation, the large confidence intervals mean that derived estimates may not be of great utility, even when assessing general population trends (Putman et al., 2011).
Overall, indices derived from faecal data can provide cheap and efficient estimates of deer abundance and have been validated in some environments with high densities of deer. However, the generally large variation in collected data is likely to affect the precision and accuracy of estimates garnered from these methods.

1.4.4 Summary of Methods

Currently there is no generally accepted method for measuring deer abundance across all environments, especially over the short term. All methods of estimating abundance involve the dealing with of certain uncertainties that are exacerbated when subject to different environmental conditions. Various studies have compared differing methodologies with various outcomes (e.g. Roberts et al., 2006; Smart et al., 2004; Wilson & Delahay, 2001). In general, camera trapping is the least laborious if the objectives of the study don require methodologies that require the physical capture or tagging of the animal with transect sampling often proving the most expensive in many areas due to the labour involved. The largely variable data garnered in many faecal pellet studies often hinders their usefulness, however how this variation compares to that garnered by other methods in largely heterogeneous environments, such as the mountainous, rainforested IESCA, is unknown. Different methods will be suitable for application in different environments and the reliability of data garnered by these methods is likely to be affected by a combination of variables. To determine which method is the most suitable for implementation in various situations, methods need to be compared and assessed while taking into account a number of environmental and site specific parameters.
2 METHODS

This chapter provides an overview of the methods and materials used, as well as an outline of the settings in which this study took place. A description of both the regional and temporal settings of this study is presented in Sections 2.1 and 2.2 respectively, while outlines of how data was gathered and analysed to produces relative abundance indices, demographic characteristic and a preliminary measure of absolute abundance is presented in Section 2.3.

2.1 REGIONAL SETTING AND SITE SELECTION

At the request of the Illawarra National Parks and Wildlife Service (NPWS), this study was performed within three areas of the Illawarra Escarpment State Conservation Area where proposed management procedures would take place. All study sites were located within the Illawarra region, approximately 80km south of Sydney. Selected sites were generally representative of the open forest/ rainforest habitat that is common within the IESCA and allowed for easy access and repeatability of methods for the monitoring of future trends in deer abundance. Moriarty (2004a) found that during the rut, Rusa Deer in the neighbouring Royal National Park had home ranges of approximately 5km$^2$ to 6.5km$^2$. As potential sites requested to be used for sampling were spread across approximately 8km of the escarpment, sites were chosen that were separated by approximately 4km to minimise the likelihood that the same deer would be recorded at multiple sites during each round of sampling. The NPWS requested that the exact locations and identifying characteristics of each site not be disclosed. A summary of sites and site specific characteristics can be found in Table 1.

The IESCA sits on the Illawarra Escarpment, an abruptly elevating escarpment that separates the Illawarra Coastal Plains in the east and the Woronara Plateau in the west. Currently, the IESCA is approximately 2635 hectares in size, making up roughly 40% of the escarpment’s total land area and forming almost continuous habitat corridors to three adjacent National Parks and Conservation Areas. These corridors allow the faunal components of these ecosystems to disperse relatively freely which, while a positive for the proliferation of native species, is a major obstacle for controlling the spread of invasive species (NPWS, 2011). The parcels of land that make up the IESCA are located within a matrix of natural bushlands and forests, agricultural areas, coal mining operations and residential zones managed by the Wollongong City Council, the Sydney Catchment Authority as well as private citizens and corporations.
The Park’s fragmented design means that it is likely to suffer exacerbated edge effects from the operations of its surrounding environments and that abundance within the IESCA itself is likely to not be completely indicative of that of which is present throughout the entire escarpment itself.

<table>
<thead>
<tr>
<th>Site</th>
<th>Elevation</th>
<th>Vegetation and Physical Properties</th>
</tr>
</thead>
<tbody>
<tr>
<td>Site 1</td>
<td>340masl</td>
<td>Open forest and rainforest, three residential properties are present within the site. BORDERED by farms with prime grazing areas in the east and rainforest along the escarpment. Minimal grassy areas present themselves within the conservation area itself (approximately 3-4% of land area).</td>
</tr>
<tr>
<td>Site 2</td>
<td>242masl</td>
<td>Mainly open forest and rainforest with a large open area maintained underneath powerlines. Borders privately owned, largely open areas ideal for grazing and relatively undisturbed rainforest managed by the NSW Catchment Authority. Approximately 15% of the area consists of open fields suitable for grazing by deer.</td>
</tr>
<tr>
<td>Site 3</td>
<td>316masl</td>
<td>Rainforest with cleared areas and seasonally used buildings. Surrounded by rainforest, open grassy areas suitable for deer grazing make up 7.5% of the total area.</td>
</tr>
</tbody>
</table>

Table 1: Summary of the attributes of sampled sites. Total area of open fields within sites was calculated from NSW OEH Woody Extent and Foliage Projective Cover Spatial Data (2011)

The greater Illawarra region experiences an oceanic climate with relatively mild temperatures and rainfall spread evenly over all seasons (Peel et al., 2007). Despite these general conditions, the escarpment itself receives much higher levels of rainfall and generates various microclimates over the area due to its steep aspect and the presence of unique geological features. Due to the high amounts of rainfall and protection from harsh climatic conditions offered by the escarpment’s cliff faces, the majority of the escarpment is covered in rainforest.
and wet sclerophyllous forest. While the area’s unique geology and diversity of soil communities have given rise to a diversity of vegetation communities, these communities are often interspersed with the areas of rainforest and open forest and are not thought to significantly affect the connectivity of these environments (NPWS, 2011).

Due to the diversity of vegetation communities and range of habitats present, the Illawarra Escarpment supports high levels of faunal biodiversity. Of the 72 mammalian species found within the IESCA, only five boreal mammals that are not species of deer inhabit the area (NPWS, 2002). The diet of one of the most common mammals within the IESCA, the Swamp Wallaby (*W. bicolor*) significantly overlaps with that of Rusa Deer and this overlap is likely to have a significant influence on the distribution of the Swamp Wallaby (Moriarty, 2004a). While it is unclear whether any major forms of competition occur between feral deer and other sympatric species that inhabit the IESCA, the four remaining mammals, the Common Wombat (*Vombatus ursinus*), Eastern Grey Kangaroo (*Macropus giganteus*), Red Necked Wallaby (*Macropus rufogriseus*) and the Red Necked Pademelon (*Thylogale thetis*), all feed on the same food sources and are likely to experience a similar dietary overlap to that of the Rusa Deer and the Swamp Wallaby. How this overlap may influence the distribution of deer or other species is unknown.

### 2.2 STUDY PERIOD

Sampling was performed from late May to late August to coincide with the rut of Rusa Deer. Sampling was limited to this time frame in the hope that deer numbers at each site would be relatively stable with breeding territories continuously occupied. This time period is also when deer are most actively grazing and roaming around their environments (Lincoln, 1992). Male and female deer are easily distinguishable at this time due to the presence of antlers on the males, allowing for the calculation of sex ratios and proportion of adult females that have dependant calves.

### 2.3 COMPARISON OF RELATIVE ABUNDANCE ESTIMATES

To calculate and compare estimates of the relative abundance of deer, variants of commonly used methods such as camera trapping, transect sampling and faecal pellet indices were employed. As only six cameras were available for this study, cameras and plots were rotated throughout the sites every seven days to ensure enough replicates were available for analysis.
The loss of all cameras used in this study meant that 41 days within the sampling period were unable to be sampled.

2.3.1 Camera Trapping

Camera Traps have been used in an array of research and monitoring situations and have proven to provide reliable information regarding the abundance, distribution and demographic characteristics of various deer species (e.g. Curtis, 2009, Roberts, 2011). To generate a measure of relative abundance, six Reconyx Hyperfire 600 Infrared Triggered Cameras were rotated throughout the three selected sites over the sampling period. ITCs have been shown to produce less false triggers than other motion activated cameras and produce readily identifiable photographs without disturbing the targeted organism (Glen, 2013). When used in studies of mammals, ITCs are reliable and generally only fail to detect an animal when the surrounding area is experiencing high humidity and the ambient temperature is similar to that of the target organism itself (Swann et al, 2011). Over the sampling period, conditions were generally mild with low humidity, minimising the likelihood of a missed capture. Two cameras were placed within each site. Studies on farmed Rusa Deer (Ahmad, 1997) and anecdotal evidence from the local NPWS officers indicate that habitat usage by local populations is likely to vary throughout different stages of the day. Therefore, camera placements were stratified by habitat with one camera placed in areas that are primarily used for grazing (open areas, canopy cover <30%) and another along deer tracks within areas where canopy cover was more dense (>30%). Deer tracks were identified by the presence of deer faecal material and/or hoof marks along paths of disturbed vegetation.

ITCs were fixed to trees at heights of 1-1.4m with locks and elastic roping where necessary to ensure cameras were not easily moved by winds or the brushing past of animals and photographic and detection zones were approximately level with the height of the target animal. Cameras were positioned so that they were relatively parallel to the ground and not obstructed by any residual vegetation to ensure all cameras had similar capture and detection areas. Additionally, cameras were not placed within 100m of the boundary edge to minimise possible edge effects on the data. As preliminary data had shown that photographs of deer produced by these cameras were not easily individually identifiable, separate captures were treated as those that were taken in excess of 30 minutes after the last trigger.
Photographs taken using these camera traps were then analysed to produce a relative abundance measure of the number of deer per camera hour. A second estimate of deer groups per camera hour was also generated to minimise the effect that the aggregation of deer into groups will have on any generated measure.

2.3.2 Transect Sampling

One of the most commonly used methods used to survey deer abundances due to the ease of its implementation and ability to provide a snapshot of local populations is that of transect sampling. Variations of this method have been shown to provide accurate representations of deer populations in an array of environments (e.g. Koganezawa and Li, 2002, Acevedo, 2008). When targeted at deer, transect sampling has typically been performed at night with the use of spotlights as it has long been thought that deer are more cryptic and vigilant during the day and more easily detected in the dark due to their distinctive eyeshine (McCullough, 1982). To test this notion and calculate two separate relative abundance measures, transects ranging from 0.8 to 2km in distance were performed both during the day and night over the sampling period. Transect distance was limited mainly by the fragmented nature of the IESCA as well as the activities of the residents of those sites. Additionally, as with the placement of cameras and faecal plots, to minimise edge effects transects did not extend to within 100m of the site boundary if it was adjacent to urban or agricultural areas or constructs, such as roads or houses. Each transect was walked by two observers four times a week in total, twice during the day and twice at night, over the sampling period. Daytime sampling procedures were performed during the middle of the day while transects performed at night were not started until at least 45 minutes after sunset so estimates may be comparable to other studies. Visual observations made during the night were made with the use of 400 lumen spotlights. Sampling was not performed during rainfall events as, in addition to safety concerns, spotlights during rainfall events has been shown to garner significantly less sightings of deer, leading to underestimations of deer abundance (Garel, 2010). One observer was present during all transects to minimise the observer effect.

For each sighting of deer, the angle at which it was detected, radial distance from the observer, number of individuals within the spotted group, the habitat in which they were present and the group behaviour at and after initial sightings was recorded. The angle of detection between observers and deer groups was measured with a compass. Distances from the path were
estimated based off the distance of deer from previously known landmarks or measured with measuring tape after the conclusion of the transect if no accurate estimation of distance was previously known. When observed deer were present as part of a group, all measurements were taken from the centre of the herd. Deer groups were defined as one or more individuals that were stationary together or moving in the same direction as one another.

From these data, two separate Kilometric Abundance Indices (KAI) were created. Initially a measure of the total amount of deer per kilometre sampled was generated (KAI-I) before a secondary measure of deer groups per kilometre (KAI-G) was created to, as with the cameras, minimise the effects of deer aggregation on the results.

2.3.3 Faecal Data Indices

To generate a measure of relative abundance from faecal data, two plots were set up throughout each site, one in open areas and one along deer trails, to measure the amount of deer faeces deposited over 7 days. To ensure that these estimates were directly comparable to estimates produced by the cameras, plots were placed within the field of view of the cameras themselves. As plots were required to fit in the field of view of cameras with ease, plot size was limited to 5mx5m. Plots this size have also been used in previous studies allowing for comparisons to be easily be made between sites (Ahmad, 1997). As faecal material deposited by deer can persist within the environment for several months and to ensure estimates garnered by this method are directly comparable to those generated by the cameras, plots were cleared of all faecal material and subsequently surveyed after the 7 day period. In closed areas, effort was made to not disturb the existing layers of leaf litter when clearing plots and when subsequent surveying of accumulated material occurred 7 days later. After initial clearing, plots were marked with a singular stake in the north eastern corner of the plot to minimise the amount of equipment left at the site that may influence deer behaviour.

Deer faeces are cylindrical in shape, deposited in distinct ‘pellet’ groups and easily distinguishable from faecal material deposited by other ruminants that inhabit the area and, therefore, pellets recorded in this study are most likely to be solely from deer themselves and not a combination of ungulates as in other several other studies (e.g. Forsyth, 2002). Both the number of faecal pellet groups and the amount of pellets within each group were recorded. Pellets located on the edge of the plot were alternatively included and excluded from results.
From this data a measure of pellet groups per metre squared was created and used as a proxy for relative deer abundance.

2.3.4 Data Analysis

To determine differences between estimates of relative abundance garnered by different methods, data was analysed for variations produced between each method overall as well at site and time specific scales. Both multivariate and univariate analyses were performed to determine whether the three methods produced estimates that were independent of each other using JMP 11 Statistical Software. All data were normally distributed and did not require transforming. To test for significant correlations between methods, a Spearman's Rank Correlation Coefficient was calculated. When data was not monotonically distributed, a Hoeffding’s Independence Test (Hoeffding’s D) was performed to investigate whether or not results from varying methods were correlated. As group and individual measures of deer abundance were created using the same data set for visual observations, I-KAIs and G-KAIs were not tested against each other. Similarly, group and individual measures created via camera trapping were not tested against each other either. After initial multivariate analyses were performed, ANOVAs and subsequent Tukey’s Honestly Significant Difference Tests were performed on both the measures of relative abundance garnered by camera trapping and transect sampling methods to determine whether or not estimates varied significantly between sites or over time. Additionally, a T-Test was used to further investigate the relationship between KAIIs developed through the use of spotlighting and daytime visual observations.

2.3.5 Variation within Methods

Varying procedures that use camera traps and faecal plots to estimate the abundance of deer and other ungulates use a number of varying time periods and plot sizes to do so. To generate accurate estimates of deer abundance, variation in the rates of capture and faecal accumulation should be minimal. Parameters required to accurately measure the abundance through these methods will vary through environments and populations due to a combination of climatic conditions, habitat structure, food availability and the density of the target organism. To determine the period of time that ITCs must be deployed for accurate estimates of abundance to be calculated within the IESCA, six Reconyx Hyperfire 600 Infrared Triggered Cameras were set up across the escarpment for a period of 21 days. Cameras were arranged in a similar fashion to that described in Section 2.3.1. After 21 days these cameras were retrieved and the
amount of individual and groups of deer captured each day were recorded. To investigate how variation in the number of deer captured per day varied with time, the coefficient of variation was plotted over time.

The majority of plots used to estimate the abundance of deer from faecal material are between 25m$^2$ and 100m$^2$ and are allowed to accumulate faecal material for varying amounts of time. While these time periods and sizes are likely to be determined by a combination of deer density and environmental variables such as terrain and weather conditions, basing predictions of deer density on highly variable data can mean that garnered estimates are likely to be unreliable and inaccurate. To determine how faecal accumulation rates vary over time and plot area, three 20 x 20m plots were prepared across the three sites sampled in section 2.3. Each plot was cleared of all faecal material at the start of a 3 week period and routinely surveyed every 7 days. Plots were separated into 64 2.5 x 2.5m plots and sampled in a random order with the number of pellet groups within each plot being recorded. Definitions of pellet groups follow that outlined in section 2.3.3. The coefficient of variation of the mean pellet groups per square metre was then plotted against the total area sampled after 7, 14 and 21 days respectively.

To better inform future management practices, several characteristics of the IESCA deer population, including sex and calving ratios were calculated from data gathered in section 2.3. Additionally, differences in the calculation of mean group size by different methods were analysed. In order to calculate the sex ratio of the IESCA deer population, photographs captured of deer in section 2.3 were analysed to determine the number of adult males and females captured. Adult males were defined as any deer having antlers or any sign of antler growths such as distinct lumps on the forehead while adult females were defined as any deer over approximately 60cm in height that did not show any sign of antler growth. Additionally, female Rusa Deer can sometimes be distinguished from their male counterparts as their necks are significantly slimmer. Similarly, to give an estimate of the ratio of adult females with a dependent calf, the number of females captured by camera traps was divided by the number of calves seen within photographs. Calves were classified as any individual under 60cm in height.

The estimation of a mean group size is particularly important for measures of absolute abundance. Incorrect estimations of group size can significantly skew density estimates leading to severe consequences for management procedures (Rowcliffe, 2008). To investigate the differences in the mean group size calculated by different methods, the mean group size of deer
groups was calculated from data collected during night-time visual observations, visual observations made during the day and data captured by camera traps. As male deer hold ‘rutting territories’ during the mating season and are solitary by nature during this time, a calculation of group size excluding solitary individuals was also calculated to atone for this probable bias within results. These differences between means were then tested through the use of ANOVAs and Tukey’s Honestly Significant Difference Tests.
3 RESULTS

Results of this study are presented in Chapter 3. A broad comparison of the different methodologies and different conditions under which they were applied is presented in Section 3.1. Sections 3.2 and 3.3 give estimates of various demographic characteristics of the IESCA deer population and a preliminary estimate of absolute population size.

3.1 COMPARISON OF METHODS

When using techniques such as automated photography or the sampling of faecal plots to estimate the abundance of deer, it is necessary to ensure that sampling efforts are sufficient enough to return estimates of mean abundance with low coefficients of variation. This can be achieved using a modified rarefaction approach (Collins & Simberloff, 2009). To determine the length of time that cameras should be deployed for to ensure that estimates of abundance are reliable, the variation from the mean amount of deer captured per day was measured across increasing periods of camera deployment. Coefficients of variation decreased relatively quickly over the first 6 to 7 camera days, before decreasing at a much slower rate after this first week of deployment (Fig. 2A). While variation appeared to continue to decline until the end of the sampling period, trends in variation remained relatively unchanged after 15 camera days with coefficients of variation varying from 15 to 18% throughout the remaining sampling period.

Similarly, to investigate the reliability of relative abundance estimates garnered from faecal sampling plots, coefficients of variation were calculated for the mean amount of pellet groups sampled at increasing plot sizes. Measured variation in plots sampled after 7, 14 and 21 days all showed decreasing variation with increasing plots sizes. Coefficients of variation calculated in plots that were allowed to accumulate faecal matter for 21 days reached an asymptote at an area of approximately 25 to 35m² with values stabilising around 4% variation. In plots left for 7 days, variation continued to decrease as plot size increased yet became relatively constant at approximately 3.5-4%. Plots left for 14 days had the lowest levels of variation after coefficients of variation stabilised at approximately 2% at plot sizes of 40m² or greater.
Fig. 2. (A) Variation of the mean amount of deer captured by camera traps with increasing time of camera deployment. (B) Changes in variation from the mean number of pellet groups located in with plots of various sizes as measured after 7, 14 and 21 days.

While it was shown that possibly undesirable variation decreases after cameras are deployed for longer periods of time, the loss of equipment meant that camera traps and faecal sampling plots were limited to deployment periods of 7 days. Additionally, plots were limited to 25m$^2$ in size to ensure that the entirety of the plots would be able to easily fit within the field of view of the ITCs. Observed variation under these conditions is relatively close to minimal observed values with coefficients of variation after the deployment of cameras and 25m$^2$ plots after 7 days equalling 22% and 6% respectively.

Across all three sampled sites, the number of deer captured by cameras located along deer trails in closed environments and those placed within open areas targeting grazing deer did not vary significantly from one another (fig. 3.) Patterns of abundance measured by camera traps are also consistent with anecdotal evidence that deer abundances across sites are subject to
approximately monthly movements with both Sites 1 and 2 showing a significant decrease in deer numbers after 3-4 weeks with peaks at the beginning and end of the sampling period. Site 3 showed a large peak in deer numbers halfway through the sampling period with lulls in an observed deer presence at the beginning and end. Relative deer abundance as calculated from faecal sampling plots in closed and open areas did not correlate with each other at any site (Site 1 $\rho=0.6736$, $p=0.0971$, Site 2 $\rho=0.3000$ $p=0.5133$, Site 3 $\rho=0.3416$, $p=0.4534$) and did not show any significant variations in abundance over time.

Fig. 3. Trends of deer abundance during the rut as measured as individual captured per camera hour (I/CH) across three sites by camera traps located along deer trails within closed canopied environments and in open, grazing areas. Measures across all sites correlate with each other (Site 1 $\rho=0.9214$, $p=0.0032$, $\rho=0.9555$, Site 2 $p=0.0008$ and Site 3 $p=0.9718$, $p=0.0003$).
No two methods presented estimates of relative abundance that were well correlated over time at all three sites. Camera traps recorded the most deer at Sites 1 and 3, while visual observations undertaken during the evening returned the greatest number of deer sightings at Site 2. Group based estimates from spotlight and camera trapping procedures were strongly, positively correlated at two of the sampled sites, with correlations at the third approaching significance (Fig. 4.). Similar patterns of abundance were measured across all sites by group based camera estimates with peaks at the beginning of the sampling period as well as lesser peaks in deer recordings in Weeks 4 and Weeks 6. Individual based estimates of relative abundance from these methods were also relatively well correlated throughout all sites (Site 1 $\rho=0.6514$, $p=0.1130$, Site 2 $\rho=0.6636$, $p=0.1041$ Site 3, $\rho=5610$, $p=1901$)

Fig 4. Relative abundance as estimated by camera traps and spotlighting. Estimates by both methods gave significantly similar results when applied to Site 1 and Site 3 ($\rho=0.7607$, $p=0.0471$ and $\rho=0.8446$, $p=0.168$ respectively), however estimates did not correlate significantly when measured at Site 2 ($\rho=0.5621$, $p=0.1890$).
Similarly, while mean estimates of deer abundance derived from camera traps and faecal plots both indicated that deer abundance is relatively equal at all sites, all KAIIs returned results indicating that deer populations at Site 2 were significantly greater than the other sampled sites (Fig. 5.). KAIIs measured at Site 2 were 200-500% larger than their counterparts estimated at Site 1 and 3.

Fig. 5. Mean estimates of deer abundance as measured by (A) camera traps and faecal sampling plots and (B) KAIIs derived from spotlight assisted and daylight visual observations. Those measures connected by the same letter were not significantly different from each other. All KAIIs measured at Site 2 were significantly larger than those measured at all other sites (KAI-G Night F=13.029, p=0.0003, d.f.=2, KAI-I Night F=14.377, p=0.0002, d.f.=2, KAI-G Day F=10.049, p=0.0012, d.f.=2, KAI-I F=14.077, p=0.0002, d.f.=2). Error bars represent standard error.
3.2 DEMOGRAPHIC DATA

In all trials a total of 366 photographs were captured that were identifiable to species and sex level. The mean number of captures did not vary between sites (Fig. 5A). Overall, approximately 60% of the captured individuals were female while adult males and calves made up the remaining 29% and 11% respectively. A mean sex ratio of 2.07 females to every male was found. Sex ratios did not vary amongst sites or placements of cameras within open and closed environments (F=1.7686, p=0.1442, d.f=41). Similarly, mean ratios of adult females to calves did not vary significantly across sites or placements within closed and open environments (F=0.7337, p=0.6030, d.f.=41). One in every approximately 4.9 females were observed to have a dependant calf.

Mean group size was calculated from both camera traps and transect methods to investigate whether the limited field of view of the automated cameras hindered their ability to accurately estimate group sizes. As adult males will often act and travel in a relatively solitary manner during the rut, estimates of group size were calculated when including and excluding solitary captures. Estimates of group size calculated from cameras placed in closed and open environments were similar, not varying significantly from each other when including or excluding solitary captures (F=0.8287, p=0.5424, d.f.= 28 and F=1.1084, p=0.3734, d.f.=41 respectively).

When including solitary captures, similar estimates of group size were garnered from all methods (F=0.4517, p=0.6379, d.f.=2), however mean group sizes were found to differ significantly between Sites 2 and 3 (F=3.7732, p=0.0264, d.f.=2). Mean group size at Site 2 was significantly larger than that observed at Site 3 (2.6 and 2.1 individuals per group respectively). When excluding solitary captures from group size calculations, an average of 2.6 individuals were found per group. Estimates of group size when excluding solitary captures did not vary significantly amongst differing methods and sites (F=2.4424, p=0.0916, d.f.=2 and F=0.5641, p=0.5705, d.f.=2) (Fig. 6).
Figure 6: Mean group size as calculated by different methods (A) excluding solitary captures and (B) excluding solitary captures. When including solitary captures, group size was found to be significantly larger at Site 2 than those at Site 3 ($F=3.7732$, $p=0.0264$, d.f.=2). When excluding solitary captures, there was found to be no significant difference between the group size calculated across sites ($F=2.4424$, $p=0.0916$, d.f.=2 and $F=0.5641$, $p=0.5705$, d.f.=2). Error bars represent standard error.

### 3.3 ABSOLUTE ABUNDANCE

Data from spotlight procedures and Distance 6.2 distance sampling software indicate that areas of the IESCA are likely to have deer densities of approximately 0.4 deer per hectare (CV=0.175, 95% confidence interval 0.3-0.6 deer per hectare). This equates to a population of approximately 1090 deer throughout the entire IESCA.
4 DISCUSSION

This section provides a contextual analysis of the results. Similarities and differences between different methods are presented in Section 4.1, while an investigation into the overall demographic characteristics of the IESCA deer populations are presented in Section 4.2. It is hoped that results of this study may provide preliminary information to assist in the creation of effective and successful long term monitoring programs.

4.1 COMPARISONS OF METHODS OF RELATIVE ABUNDANCE ESTIMATION

The overarching aim of this study was to compare the effectiveness of various commonly used methodologies in estimating the abundance of deer within the IESCA. To achieve this, two broad objectives were investigated. Firstly, the conditions under which faecal plots and camera traps were able to provide results with minimal variation were determined. Subsequently, a comparison of relative abundance estimates derived from camera traps, faecal accumulation plots and daylight and spotlight transect sampling was performed to assess the suitability of each method for application in the IESCA.

4.1.1 Variation within Camera Trap and Faecal Accumulation Methods

To increase the accuracy, precision and overall reliability of estimates of abundance and other traits of populations, these features should be measured under conditions that ensure variation in produced results is minimal. While variation arising from naturally occurring differences in populations, such as differences in ages or exposure to varying environmental conditions, should not, an often cannot, be removed from obtained results, technical error/variation caused by imperfect measuring systems or human error should be minimised (Altman & Krzywinski, 2015; Tijskens et al., 2015). By minimising this variation arising from technical procedures, the overall reliability and precision of obtained results and the likelihood that sampled populations are representative of broader populations are improved.

As a part of this study, investigations were undertaken to determine how variation in results produced by camera trap and faecal accumulation based methods changes with increasing periods of deployment and increasing plot sizes respectively. As expected, the mean number of deer captured by ITCs per day was less variable after greater periods of time. After a
significant decrease in variation after 6 days, measured coefficients of the variation plateaued at approximately 15-18% after 15 days (Fig. 2). These results concur with the majority of previous research that has generally suggested that ITCs be deployed for periods of at least 14 days to ensure that suitable quantities of deer are captured and variation between series of captures are low (Carbone et al., 2001; Dougherty & Bowman, 2012; Rowcliffe et al., 2008). Additionally, average measures of variation in this study are in line with or less than those presented in other similar investigations (Hernández et al., 2005). Intrinsic variation measured within populations is dependent on numerous characteristics of these groups. The behavioural responses of deer to differing food availabilities, stages of the rut, males’ holding of various mating greens and numerous other factors will ensure that some inherent biological variation is held within these results. Results from camera traps placed along both deer trails and within grazing areas further suggest that seasonal movements of deer may play a role in this observed biological variability. Data from ITCs showed distinct peaks and troughs in deer abundance at sites over different weeks, aligning with anecdotal evidence from residents and NPWS staff that deer herds will travel throughout the escarpment on an almost monthly basis. Additionally, Moriarty (2004a) found that Rusa Deer in the neighbouring RNP had significantly larger home ranges during the winter when male individuals would become more solitary in their behaviour and travel in search of possible mates. These large amounts of movement by deer are likely to be responsible for the high variability with these results. Due to the possibility that these movements are season specific, further research is required to assess the ideal conditions under which cameras may be deployed for during other times of the year.

In a manner similar to that of the ITCs, faecal accumulation plots showed a significant decrease in variation with increased sampling effort (i.e. greater plot size). While it has been generally accepted that larger plot sizes will produce results with less in built variation (Ellwood, 2000; Smart et al., 2004), Noor et al. (2010) found that when using faecal accumulation rates as a proxy for the relative abundance of deer in wetland and forested environments, medium sized plots (approximately 70m²) produced estimates with the lowest coefficients of variation, with largest plots having only slightly lower estimates of variation than smaller plots. Results in this study align with the more general view that larger sampled plots will provide more precise estimates of abundance. Noor et al. (2010) suggested that the higher variation in results drawn from larger plots were caused by the increased likelihood that pellet groups may have been missed during their surveying. In this study, plots were sampled by counting deer faecal material within smaller contiguous 6.25m² plots which were then combined to provide density
estimates of deer faeces over larger areas. This methodology meant that sampling procedures were likely to be much more thorough and accurate than simply sampling large areas as a whole, which may account for the differing patterns in variation observed between the studies. Although more laborious, separating larger areas into smaller plots before sampling may assist in the obtaining of more reliable results from these methods in the future.

4.1.2 Comparison of Relative Abundance Estimates Derived from Camera Trapping, Daylight and Spotlight Transect Sampling and Faecal Accumulation Plot

No two monitoring approaches employed in this study returned estimates of relative abundance that were significantly correlated across all sites. Results from daytime transect sampling and faecal accumulation plots did not align with any other methods at any site. Two of the most commonly used sampling methods, spotlighting and camera trapping using ITCs returned results that were significantly, positively correlated at two out of the three study sites.

4.1.2.1 Faecal Accumulation Plots

Faecal accumulation plots did not provide estimates of abundance that were well correlated with those produced by other methods. These results are generally concordant with those from previous research which has shown that accumulation based faecal methods do not correlate with distance based sampling methodologies (Ellwood, 2000; Smart et al., 2004). No current works have been performed to directly compare indices of abundance derived from camera traps and accumulation based methodologies. Acevedo et al. (2008) showed that to generate accurate estimates of Red Deer abundance, only pellet groups containing 20 or more pellets should be used for analysis, most likely due to the fact that these pellet groups were representative of a ‘full’ defecation event and not residual pellets that have not been removed from or entered the plots in other ways. This observation was further supported as several deer along trails were captured by ITCs defecating while walking, likely leading to the scattering of faecal material across plot boundaries. Due to the limited number of groups that contained 20 or more faecal pellets, pellet groups that were greater than 5 pellets were used in this study. Generally, most species of deer will deposit between 20-30 pellets per defecation event, so this requirement of 20 pellets may also be needed to accurately quantify the abundance of this population. However, very few studies have placed a limit of on the minimal number of pellets needed for a pellet group to be used in later analysis. Therefore, methods employed in this study are comparable to other studies which have shown that accumulation based faecal
methods can produce estimates of deer abundance that are in line with those produced by other methods (Alves et al., 2013; Staines & Ratcliffe, 1987).

In general, male deer will defecate significantly more than their female counterparts which may lead to differences in abundance estimates between areas where sex ratios are different (Lunt et al., 2007). The proportion of calves, males and females in this study did not differ significantly throughout the sites, therefore estimates can be assumed to be generated solely on the basis of abundance and not a variable sex ratio. In environments where these faecal based estimates have been shown to provide accurate estimates of abundance, the age and structure should be investigated to determine if these play a role in the reliability of garnered results.

The main difference between the methodology applied in this study and those that have validated these types of approaches against other methods, is the length of time in which plots were allowed to accumulate faeces. While accumulation periods in this study were limited to 7 days due to the loss of equipment, Alves et al (2013) used an accumulation period of 264 days to calculate mean deer abundance. This extended period of accumulation allows for the deposition of large amounts of faeces but is unable to capture seasonal or smaller time scale patterns of abundance within areas. As this study was solely focused on trends of deer abundance during the rut, this extended accumulation period would prove unsuitable for capturing variation over this time. Those studies that have found accumulation based methods were poor at providing estimates of abundance that are concurrent with other methods have used accumulation periods of 2 months or less to ensure calculated results were representative of certain seasons and estimates were available within a certain timeframe (Ellwood, 2000; Smart et al., 2004). Those who have come to the conclusion that accumulation based methods are unsuitable have also noted the absence of faecal material in several of their plots. This issue was present within several plots in this study as well, contributing to the relatively high variability present with faecal based estimates. This further suggests that these periods of accumulation may not have been sufficient to allow for the deposition of faeces in a manner that is truly representative of land use and spatial distribution, leading to these conclusions that these methods are inaccurate and unsuitable for use in these environments.

Due to the long periods of time that are required for plots to accumulate enough faecal material for accurate estimations of deer abundance to be made, these methods may prove unfavourable in management situations where rapid assessments of density and spatial distribution are often
required. Additionally, the length of time that is required for suitable estimations to be generated means that these methods are unlikely to be able to demonstrate any seasonal or temporal variation in communities, which may make alternative methods the more favourable choice.

### 4.1.2.2 Daytime and Spotlight Transect Sampling

Transect sampling undertaken during the daytime did not provide estimates of abundance that were well correlated with those produced by other methods. Estimates of group size did not vary significantly between methods, suggesting that a significantly different number of deer groups in total were recorded during these methods rather than corresponding methods. This outcome corresponds with previous research showing that daylight sampling underestimates true densities of populations (Gill et al., 1997).

Since the 1980’s, it has been suggested that daylight and spotlight based methodologies will produce significantly different estimates of deer abundance (McCullough, 1982). However, there has been little empirical evidence presented to investigate the patterns of abundance presented by these two methods. McCullough (1982) suggested that most species of deer are more readily spotted during the evening due to their cryptic behaviour and skittishness during the day. With the ability to use spotlights to observe the distinctive eye shine of deer and a generally reduced flight response exhibited during the night, a more reliable number of deer in the area may be achieved through spotlight based sampling. While little research has been performed to directly compare estimates garnered by transects performed during the day and night, spotlighting as a method for estimating both relative and absolute abundance of deer has been seen to provide accurate and reliable results, despite the reduced visibility at night (Acevedo et al., 2008; Koganezawa & Li, 2002). McCullough (1993) also showed that calculated sex and calving ratios were significantly different when calculated from the same population during spotlight and daytime procedures. In this study, significantly more deer and deer groups were recorded during spotlighting procedures, further supporting McCullough’s (1982) suggestion that sampling procedures during different times of day will produce varying estimates of deer abundance. Gill et al. (1997) also found that daytime transect sampling recorded significantly less deer than those recorded at night. However, the study by Gill et al (1997) was performed with the aid of thermal imaging equipment during the night. Thermal imaging equipment is able to detect significantly more deer at greater distances in forested environments than spotlighting, as well as sampling performed during the day (Focardi et al.,
Therefore, while daytime transects detect significantly less deer than night time thermal imaging based approaches, how these thermal imaging based estimates compare with those undertaken by spotlight procedures is relatively unknown.

Deer numbers recorded along transects performed during the day were insufficient to produce an absolute abundance estimate. While extending the length of these transects may eventually be able to generate estimates of absolute abundance, provided sufficient numbers of deer are recorded, this lack of observed deer means that transects undertaken during the day are likely to underestimate actual populations and lead to an inaccurate measures of trends in abundance. Additionally, this need for longer transects and corresponding man hours also adds to the financial cost of monitoring procedures, despite not necessarily providing accurate estimates of abundance (Focardi et al., 2001). The lack of deer recordings along transects performed during daylight means that this method will be less statistically powerful and most likely a poor method of estimating and monitoring deer abundances in the area.

4.1.2.3 Camera Trapping and Spotlighting

At both Sites 1 and 3, group based estimates of relative deer abundance derived from camera traps and spotlight transect methods were significantly, positively correlated with results at Site 2 approaching significance. Individual based estimates were also relatively well correlated between these two methods (p between 0.1901 and 0.1041). While group based results approached significance at Site 2, spotlight data remained constant over the middle of the sampling period, continually capturing more deer than the corresponding camera traps. As with this study, previous research employing camera trapping based relative abundance indices have found that rates at which target animals are captured are positively correlated with results of spotlight based transect methodologies (O'Brien et al., 2003; Rovero & Marshall, 2009). While this pattern did not present itself significantly across all sites, inherent differences between sites were recorded. Though slightly different vegetative communities were present at Site 2 (dominated by sclerophyllous vegetation rather than rainforest), the main difference between the sites was the composition of the sites themselves. Sites 1 and 3 had between 3-7.5% open grazing areas within each of them, with open areas making up approximately 15% of Site 2 (NSW OEH, 2011). While data from this source is five years old, visual observations made in the field and the assessment of more recent satellite imagery appear to agree that this composition has not changed significantly over this time period. With greater amounts of open areas, visibility is increased during spotlighting surveys, increasing the likelihood that deer will
be spotted at distance further from the sampled transect, inflating KAls. Additionally, this relatively large proportion of open areas may lead to behavioural differences in deer within these areas, such as the extended use of open areas for grazing and a reduced flight response due to the deer habituation to people, vehicles or other animals. Due to the unequal usage of open and closed lands, spotlighting may be inappropriate for the surveying of some species of deer (Dougherty & Bowman, 2012).

Concerns have been raised over the appropriateness of conducting transect based methods of estimation along roads or paths due to the hypothesised differences in deer distribution and behaviour near these trails (Garel et al., 2010; Koganezawa & Li, 2002). All transects performed in this study were performed along well maintained trails with most being used daily by a small number of vehicles. Deer have been observed to use areas of habitat around well used roads significantly less than other areas (McShea et al., 2011), potentially impacting spotlight studies, however, how deer respond to roads and paths that are used irregularly is far less studied. While performing visual observations along randomly located transects throughout the environment may provide a more representative sample of the overall regional setting, the noise and disturbances that this may create, especially in thick, forested environments, may lead to the fleeing of deer from the observer. Additionally, when performing spotlighting procedures, the limited visibility introduces an increased element of risk, especially when transects placed randomly as these may require the traversing of difficult or potentially dangerous environments. The need for safety and the introduction of potential disturbances to wild populations often limits the usage of spotlight procedures to pre-existing paths, and did so in this case. Despite this need for safety and the hypothesised differences in behaviour, the relative size of deer and general deer behaviour has been seen to make these methods suitable for measuring both absolute and relative abundance.

Only group based methods from both sites were significantly correlated at Sites 1 and 3. Group based estimates are less affected by the effects of deer aggregation and the inflation of estimates produced due to the increased detectability of large groups and therefore may be favoured when group size may impact measures of deer abundance (Acevedo et al., 2008). The lack of statistically significant correlation from indices derived from the total number of individual deer recorded by these methods may be due in part to the ability of both methods to capture total group size. Whereas observations made in person while undertaking transect sampling procedures generally have the ability to view groups and landscapes as a whole, camera traps
are limited to a distinctive field of view and detection angle, meaning some group members may be not recorded. Despite this difference in the range of detection offered by both methods, mean estimates of group size did not vary between methods, suggesting that the majority of individuals within groups were captured by both methods. This lack of correlation between individual estimates may also be due to the seasonal and behavioural differences in deer at this time. During the rut, males become more solitary while females and calves generally remain within groups. While calculated sex ratios were equal across site, if one method favours the detection of a certain sex than it may explain why group based estimates are more strongly correlated than individual based measures. Repeat or similar investigations may be performed outside of the breeding period to determine if these individual measures may align with each other or alternatively, transects performed with the use of more powerful spotlights or thermal imaging equipment that enable the classification of sex may be useful for determining if a bias is present between methods.

Both camera trapping and spotlight based transect sampling are two of the most commonly used methods to survey and monitor changes in deer abundances over time. While these estimates only significantly correlate at two sites, the limited time period and amount of resources may have had an effect on the reliability of the results of this study. When dealing with camera based relative abundance indices cameras are generally deployed for 14 days or more (Carbone et al., 2001; Dougherty & Bowman, 2012; Rowcliffe et al., 2008). Results of this study suggest that minimal variation within deer captures occur when cameras are deployed for 15 or more days with coefficients of variation varying from 15 to 18% after this period. The loss of ITCs limited the time that cameras were able to be deployed for to 7 days. Though measured variation was still relatively low after this time period (coefficient of variation = 22%), this may have contributed to the lack of correlation between the two methods at Site 2. Future studies should aim to increase in the length of time and the amount of cameras that are deployed to determine if this pattern is truly present at all times.

4.1.2.4 Measuring Abundance Within Open and Closed Environments

Estimates of relative abundance from camera traps did not vary between trails within closed canopy environments and areas used for grazing (fig. 3). This pattern was not observed in the accompanying faecal plots. Various bodies of research have illustrated that the stratification of methods through different habitats can improve the accuracy and precision of obtained results (Forsyth, 2005; Foster & Harmsen, 2012; Marques et al., 2001; Rowcliffe et al., 2008). Results
of this study differ from these works with camera trap data suggesting that there are no significant differences in abundance between the two identified habitat classes. This is may be due to the relatively small proportion of grazing areas within sites. Forsyth (2005) recommended that the study area be stratified only when deer are subject to significantly different pressures across the broader landscape, such as when large areas of different habitat are present within the greater study area or if areas are subject to different hunting protocols. All study sites in this study were mainly forested with open areas making up approximately 3-15% of the total area, meaning variation in habitat structure may be too minimal to elicit a noticeable response from deer. However, the scarcity of these open habitats and the resources that they provide may alternatively be expected to cause an influx of deer into these areas, meaning that more deer should be captured by cameras within these environments. Rusa Deer have the ability to shift their diet and act as both a grazer and a browser (Moriarty, 2004), allowing individuals to use both environments equally, which may explain the similarities observed between both open and closed environments.

This pattern of equal abundance across the two habitat types was not seen in the estimates from faecal plots, perhaps for a number of reasons. Firstly, the ITCs used in this study are only able to record the number of deer that pass through its detection zone, not necessarily the amount of time spent by deer within these areas. In general, more repeat photographs of deer were recorded in open areas, suggesting that these areas were utilised more than their closed counterparts, which largely only recorded deer as they travelled along tracks in one direction. If deer do indeed use these open areas significantly more than they use closed environments, than it is likely that faecal material would be more concentrated in grazing areas, leading to this observed difference between environments. Additionally, deer do not defecate in a completely random manner and may even favour ‘latrine’ areas (Acevedo et al., 2008; Edwards & Hollis, 1982). This overlap of randomly placed plots and semi random defecation behaviour is likely to introduce some level of variation to the results which may lead to the observed differences between the two habitat types. Furthermore, between the two types of habitat, varying abiotic and biotic factors will influence the decomposition and removal of faecal material differently (Forsyth, 2005). Within closed areas of rainforest, the general dampness, the large amount of organic material and macro invertebrates within the soil is likely to increase the rate of decomposition of faecal material. Additionally, leaf litter was not cleared when searching plots for faecal material, leading to the likelihood that some pellet groups may have
been missed in closed environments. These factors may have introduced variation between environments that is responsible for this observed difference between methods.

While, faecal material suggests deer abundances, or at least habitat usage by deer, varies between open and closed environments within the IECSA, plots sizes were limited by the field view of the ITC’s. To further investigate whether or not this pattern is observed throughout the entirety of the IESCA and surrounding areas, plots of larger sizes may be beneficial as their ability to record additional pellet groups may provide results that are more representative of true patterns of distribution (Acevedo et al., 2008; Ellwood, 2000; Smart et al., 2004).

4.1.2.5 Recommendations and Implementation

Overall, camera traps and spotlighting procedures were the only methods to provide estimates of abundance that were similar and well correlated over time. These are also two of the most regularly used methods to estimate deer abundance and have been proven to provide accurate estimates of abundance in environments similar to the IESCA around the world. Ideally, methodologies would have been performed and compared to a population of a known size to determine which methods are able to provide the most accurate and reliable results. As this is not often possible, correlation between two methods is often thought to be a good indicator of their usefulness and reliability (Acevedo et al., 2008). The poor ability of accumulation based methods to record enough faeces for reliable estimates of abundance to be made and the semi random fashion in which deer defecate suggest that these methods are unsuitable for use in the IESCA. Additionally, the general behaviour and cryptic nature of deer raises questions over the usefulness of daylight sampling procedures. From a management standpoint, spotlighting and camera trapping represent two of the most expensive methods of monitoring deer abundance. In terms of camera trapping, the purchasing of ITCs incurs relatively large capital costs to the relevant managing body. However, the longevity of these units and the possibility of use in future studies means that this is likely to be the cheaper option in the long term as these methods require the least number of man-hours to set up and operate. Spotlight and daylight transect sampling were by far the most labour intensive methods, requiring the greatest number of man-hours and, therefore, the greatest expense in personnel. While faecal based methods may have little to no capital costs and require only slightly more man hours to set up than camera traps, the questionable ability of these methods to provide reliable estimates of abundance combined with the extended period of time that it may take to generate this estimate means that this may be of little use to management programs.
Based on the results presented in this study and the relevant cost effectiveness of each method, camera trapping and spotlight based transect sampling appear to be the most suitable methods for surveying relative deer abundance in the IESCA. If proposed management programs are long term in nature, the relatively high capital costs associated with the deployment of ITCs may be small in comparison to the ongoing personnel costs required for spotlighting procedures. Nevertheless, each method holds within it different capabilities to measure characteristics of deer populations. While, photographs captured by camera traps may be analysed to produce measures such as species composition and sex ratios, spotlighting approaches have the inbuilt ability to survey the absolute abundance of deer with obtained data being relatively easy to transform into that required for use in distance sampling methods (Acevedo et al., 2008). Depending on the proposed nature of the monitoring program and its broader aims, either method may be employed to monitor and survey the abundance of deer within the IESCA.

4.2 DEMOGRAPHIC DATA AND ABSOLUTE ABUNDANCE ESTIMATES

4.2.1 Demographic Data

All deer captured by ITC’s with the exception of one group of Fallow Deer (D. dama) were Rusa Deer (C. Timorensis). This observation fits with anecdotal evidence, council and government reports and previous research that the majority of deer in the area are Rusa Deer (Dawson, 2012; NPWS, 2002; Wollongong City Council, 2013). While the presence of Red Deer (C. elaphus) and Sambar Deer (C. unicolor) have been reported in the area, no captures were recorded by camera traps.

Results from camera traps suggest that the IESCA deer population has a sex ratio of 2 females to every male. While farmed populations of Rusa Deer are produced at approximately even sex ratios (Moriarty, 2004; Woodford & Dunning, 1992), sex ratios of wild Rusa Deer populations vary greatly due to the presence of sex specific mortality factors (Clutton-Brock et al., 1984; Oka, 1998). While deer from the RNP, the hypothesised source population of those now inhabiting the IESCA have sex ratios approaching 1:1 (Moriarty, 2004), populations of deer in the IESCA have a number of sex specific factors acting upon them that are likely not felt to the same extent as those in the RNP. The main sex specific pressure acting of IESCA deer populations is that of the unequal targeting of adult males by ‘trophy’ hunters. The IESCA has
been identified as a ‘hotspot’ for illegal hunting by the NPWS with various signs of hunting and reports illustrating the prevalence of hunting in the area (Public Service Association, 2012). Illegal hunters are most often participating in ‘trophy’ hunting practices, targeting males with well-developed antlers. This preferential removal of adult males from the population is likely to influence the sex ratios observed in the IESCA.

Sex ratios were calculated via the identification of identification of sex through photographic images from camera traps. As the sex of young deer is difficult to determine (Putman et al., 2011) this characterisation was determined predominantly by the presence of antlers of the signs of antler growth. Due to this study being entirely performed within the ‘rut,’ all males two years of age or more can be expected to show obvious signs of antler growth (Moriarty, 2004). One possible problem with this method is that all sampling was performed during the rut, during which males are known to hold ‘rutting territories.’ As breeding males hold these breeding grounds, non-breeding or unsuccessful males are likely to be forced out of these areas. If this is indeed the case and non-breeding males are not present is these areas than it is likely that these sex ratios will be an underestimate of male deer in the area. Any male that is less than a year old can be expected to be counted as a calf due to the reliance on a mother when less than 7 months old and the unlikeliness that they will exceed 60cm. Therefore, it is likely that some males that are between 1 to 2 years of age may have been counted as a female, distorting calculated sex ratios. To avoid some of these sources of possible bias in the results, deer populations in the area need to be surveyed at different times of the year to investigate whether these observed sex ratios are truly representative of the deer populations of the area.

4.2.2 Absolute Abundance of the IESCA Deer Population

Through the use of distance sampling, density estimates of approximately 0.4 (95% confidence interval 0.3-0.6) deer per hectare were recorded within the IESCA. This estimate is consistent with levels of density measured in areas of high habitat quality in the RNP where deer have been observed to cause the removal of large amounts of biomass and significantly reduce the success of the establishment and recruitment of native plants (Moriarty, 2004). If this measure of absolute abundance is consistent within the entire IESCA, then approximately 1090 individual deer can be expected to be inhabiting the conservation area. However, all transects were performed within areas that have been identified as high habitat quality and experiencing high levels of deer activity (Dawson, 2012). This is likely to have meant that these measures
are an overestimate of deer abundance across the whole site as density is not likely to be equal across the entire IESCA. This overestimate is likely to be especially noticeable in the northern reaches of the IESCA where deer activity is estimated to be remarkably lower than that recorded at sites sampled in this study. If measures of activity are truly an indicator of abundance, then this density is likely to be significantly lower in these sites. Further transects undertaken at additional sites representative of these areas, would provide a more accurate estimate of absolute abundance across the entire IESCA and the escarpment itself.

To combat the high fecundity of Rusa Deer populations, approximately 46% of the population would have to be removed each breeding cycle to halt population growth (Hone et al., 2010). These high levels of fecundity are a trait present in numerous invasive species and cause major issues for numerous management authorities and programs. Current management programs are targeting areas with a likely population of approximately 650 individuals. These ground shooting based procedures are responsible for the removal of less than 100 individuals per year, suggesting current deer control programs in the area are not sufficient to halt population growth. To effectively limit population growth and reduce population size, a larger control effort is needed that is able to remove greater numbers of deer from the environment.
5 CONCLUSION AND RECOMMENDATIONS

The overarching aim of this thesis was to investigate and determine the most effective and suitable methods to monitor deer abundance in the IESCA and to provide any additional demographic information on the deer populations currently inhabiting the area. In this chapter, a summary of the major findings of this report is presented. Additionally, a number of recommendations and areas of future research, both broadly applicable and specific to the IESCA, are also suggested to improve current and future monitoring programs.

5.1 CONCLUSIONS

The major finding of this study was that not all methodologies currently applied to measure deer abundances in forested environments will provide similar results. Results from this study indicate that spotlight based transect sampling and camera trapping based methodologies will provide similar estimates of abundance with abundance measures relatively well correlated over the three sampled sites.

Since it was suggested in the 1980’s, transect sampling and road count procedures for the monitoring and surveying of deer abundance has taken place predominantly at night. This study provides some of the first empirical evidence that procedures undertaken to survey deer abundance during different times of the day will lead to significantly dissimilar estimates of abundance being recorded. As spotlighting based transect sampling has been validated in environments with known populations or those estimated by capture-recapture methodologies worldwide, daylight transect sampling appears to be unsuitable for use for the monitoring of deer abundances.

Results of this study also indicate that accumulation based faecal methods of abundance estimation are unsuitable for application in the IESCA, perhaps due to the non-random fashion in which deer deposit faecal material or the high rates of decomposition in rainforest environments. However, the relatively short deployment period used in this study is likely to not have truly captured how these indices vary with changes in deer abundance.

Preliminary estimates of absolute abundance presented here also indicate that deer inhabiting areas surrounding Mt Kembla and Mt Kiera are present at densities of approximately 0.4 deer per hectare. If this density is present throughout the IESCA, then a population of approximately
1090 deer can be expected to be residing within the entire conservation area. While habitat quality appears to be consistent throughout the IESCA, previous research has shown that these sampled areas experience some of the highest levels of deer activity in the region. As deer activity in areas north of the study site was measurably lower than areas south of the study region, this estimate is likely to be an overestimate of deer abundance in the area.

### 5.2 RECOMMENDATIONS AND FUTURE RESEARCH

The main objective of this study was to identify the methodologies that are the most suitable for the monitoring of deer abundance within the IESCA. Results indicate that, while no two methods were perfectly correlated across all sampled sites, relative indices derived from camera trapping and spotlighting were the most similar. These methods are also two of the most commonly validated methods in woodland and forested environments. It is recommended that future monitoring programs should employ either of these methods, depending on the goals and length of the project. If proposed programs are going to be long term in nature, then the high capital costs associated with the purchase of relevant cameras may prove a worthwhile investment to combat the high personnel costs associated with repeated spotlight counts. While camera traps also come with the inbuilt ability to easily record various demographic factors such as species composition and sex ratios, spotlighting procedures also have the ability for derived relative abundance indices to be easily transformed into absolute measures with the application of distance sampling theory. While providing correlated results, these methods should be validated in areas with known populations of deer in similar environments if possible to determine if these methodologies do provide reliable and accurate results.

The foremost shortcoming of this study was the limited time that camera traps and faecal plots were able to be deployed for. The loss and limited amount of ITCs available meant that deployment periods were limited to 7 days, despite results of this project indicating that the most precise and reliable estimates by both methods were garnered after periods of at least 14 days. Future studies should remedy this by using longer deployment periods and investigating if faecal accumulation based methods provide similar estimates of abundance after a deployment period of any length. Factors affecting the decomposition rates of faecal material could also be investigated to further improve the accuracy and understanding of these methods, not only with the IESCA but within environments worldwide. Additionally, this study was focused entirely in areas surrounding both Mt Kembla and Mt Kiera, a small section of the total...
IESCA. To determine if patterns of abundance vary throughout the entire IESCA further surveys of abundance are needed, especially in its northern reaches where previous research has suggested that deer activity is significantly less in these areas. This point is particularly important when it comes to producing a more reliable estimate of absolute abundance in the area. Further research could also aim to quantify the effects that performing transect based methods along maintained roads has on density estimates and correlate levels of deer abundance amongst sites with those of sympatric or species that may be affected by the presence and effects of deer, such as the Swamp Wallaby, to determine if there is any level of competitive exclusion occurring in the area.

Overall, spotlighting and camera trapping appear to be the two most suitable methods for monitoring deer abundance within the IESCA. However, further research is needed to determine the reliability of estimates garnered from both methods and the conditions under which faecal based measures may produce reliable and accurate results.
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