

**The Population Ecology of
Wild Horses
in the Australian Alps**

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A thesis submitted in fulfilment of the requirements of the degree of Doctor of
Philosophy in Applied Science at the University of Canberra

December 2002

ACKNOWLEDGEMENTS

I am deeply grateful to my supervisor Jim Hone for his guidance and unfailing support over the duration of this study. His clear thinking and passion for ecology are inspiring.

The Australian Alps Liaison Committee supported the project financially over the years. I would also like to thank Graeme Enders and Brett McNamara for their encouragement. The University of Canberra and Australian Geographic also provided financial support.

NSW National Parks and Wildlife Service and Parks Victoria staff provided logistical support in the field in particular Pam O'Brien, Danny Corcoran, Dieuwer Reynders, Scott Perkins, Kris Rowe and Chris Rose. Technical staff assisted with the production of maps including Phil Zylstra, Gary Koh and Belinda Hille (NRE).

I am glad to have had the opportunity to meet and work with horse-riders from the Alpine Brumby Management Association who taught me things about brumbies and the mountains that I couldn't learn in a university. Craig Edwards, Bronwyn Edwards, Dean Backman, Rob Jenkins, Leisa Caldwell and Gary Caldwell have been generous with their time, knowledge and horses. I could not have written the brumby-running chapter without their help.

There were many people that helped in the field in all kinds of weather, and shared many experiences for which I am grateful. People that gave generously of their time include Dave Little, Barry Aitchison, James Dawson, Janet Hines and Dieuwer Reynders. It was also a pleasure to work with Peter Potroz and Grant Halverson from Helicopter Aerial Surveys during the aerial survey.

I would like to thank the Applied Ecology Research Group for their support over the years. It has been an excellent research environment offering many opportunities to learn about a wide range of topics and get help in my own research. AERG is the source of long-term friendships. Technical and administrative staff facilitated the research, and David Pederson provided statistical support.

Finally I would like to thank friends and family for all their input and discussions around campfires, dinner tables and on the back of horses, and their personal support through a long project.

ABSTRACT

In this thesis I examine the population ecology of wild horses (*Equus caballus*) in the Australian Alps. Wild horses were first introduced into the Alps over 150 years ago. Paradoxically, they are a feral animal impacting on the environment, but are also a cultural icon. Managing wild horse populations is contentious and needs to be founded on knowledge of their population ecology. This is the first study of its kind in the Australian Alps and therefore has a broad focus. Four general areas were addressed: distribution, estimation of abundance and density, population dynamics and the influence of brumby-running. The study was conducted between 1999 and 2002 inclusive in the Australian Alps national parks, which form a contiguous protected area in south-eastern Australia from the Australian Capital Territory (ACT) in the north, through New South Wales (NSW) and into Victoria in the south.

The current distribution of wild horses in the Australian Alps national parks is patchy. There are five major discrete populations in NSW and Victoria with the northern most population in Kosciuszko National Park (NSW) bordering on the ACT. A review of published material and oral history reveals historic influences on distribution. The presence of each population is associated with introductions by people. The distribution of some populations expanded after introductions and many have fluctuated over time. Distributions have been contained or reduced through control by people, natural events such as snow and drought and by geographical barriers. Park managers eliminated a population of wild horses in the ACT in the 1980s. Distributions have expanded in areas without active management (notably northern Kosciuszko National Park) and are likely to continue expanding under a policy of no management.

In February and March 2001, abundance and density of wild horses were estimated by helicopter aerial survey in areas where the mapping showed horses to occur. This was the first time that these parameters have been estimated empirically. I compared three different aerial survey techniques (strip, mark-recapture and line transect) based on relative accuracy and precision and found that line transect analysis gave the highest, most precise estimate.

Given that aerial surveys usually underestimate abundance, this method was also likely to be the most accurate. Mark-recapture over a 50m wide strip gave a similar result but lacked precision. Strip and mark-recapture techniques performed poorly over 200m strips because animals were missed. Numbers observed dropped off dramatically beyond the 50m strip. Line transect analysis (for both observers combined) gave an estimate of 5010 (± 1408 SE) horses while mark recapture over 50 metres gave an estimate of 4915 (± 2733 SE). These estimates correspond to a density of 1.8 horses km^{-2} over the area surveyed (2789km^2). The results suggest that aerial surveys of large mammals using a wide strip width (200m) and mark recapture analysis may seriously underestimate population density.

The population dynamics and demography of wild horses were estimated at three sites, Big Boggy, Cowombat and Currango, every spring and autumn over 3 years. The sites were spread widely across the Alps with the aim of obtaining a broad understanding of population dynamics. The survey used Pollock's robust design and natural markings were used to identify individuals. There was a seasonal spring peak in population size at Big Boggy with no clear seasonal trend at the other two sites. Mean wild horse densities determined at Big Boggy (2.01km^{-2}) and Currango (2.13km^{-2}) were not significantly ($p < 0.5$) different to the density calculated in the aerial survey, whereas density was significantly higher at Cowombat (6.4 km^{-2}). Census techniques were of limited use in estimating annual population growth rate because of low precision.

Demographic analysis showed that none of the populations were increasing at the maximum intrinsic rate ($\lambda = 1.2$), and the Big Boggy and Cowombat populations may be stable ($\lambda = 1.0/\text{yr}$). There was an apparent trend of food limitation across the sites. Body condition was positively related ($p < 0.01$) to pasture biomass. The Currango population was increasing ($\lambda = 1.09$) associated with higher recruitment, body condition and pasture biomass than in the other two populations studied. The Cowombat population had the lowest annual finite rate of increase ($\lambda = 1.03$), and horses at this site were in the poorest condition and pasture biomass was lowest. The Big Boggy population was intermediate between the two.

Annual adult survival was constant in all populations at 0.91. Survival in the first three years of life was more variable with the average at each site ranging from 0.63/yr to 0.76/yr. An average of 0.26 female foals was born per adult female per year. Sensitivity analysis showed that population growth rate is most sensitive to changes in adult survival, followed by fecundity and then survival in the first three years of life. The dynamics observed at each site was representative of the demography of wild horses in other parts of the world and is typical for large mammalian herbivores.

Brumby-running is a form of harvesting that is currently being used to control wild horses in the Alpine National Park (Victoria) and is soon to be trialled in Kosciuszko National Park (NSW). The effectiveness of brumby-running has not been assessed prior to this study. Data collected by the Alpine Brumby Management Association and predictive modelling were used to examine the influence of brumby-running on the wild horse population in Alpine National Park. Brumby-runners remove about 200 horses per year with a preference for young animals and adult females. More horses are caught in autumn (61/yr) and least in summer (30/yr) ($p < 0.05$). Brumby-runners do not appear ($p > 0.05$) to target horses in poor condition. One skilled brumby-runner caught an average of 1.16 horses/day, while his companions caught an average of 0.55 horses/day. Brumby-runners show behaviour analogous to social carnivores. Predictive modelling suggests that brumby-runners could suppress the population of wild horses in Alpine National Park similar to the effects of predators, or human harvesting of other large mammals. Selecting young animals in the harvest reduces the impact of harvesting on the population compared to unselective harvesting, while selecting adult females increases the predicted impact.

There are several management recommendations based on the findings of this thesis that address concerns for both environmental impact and the cultural value of wild horses. The distribution of wild horses should not be allowed to expand further, and the size of the wild horse population should be prevented from increasing further. Other management recommendations that are more complex involve reducing some populations so that the level of environmental impact they are causing is acceptable. This requires a definition of 'acceptable impact.' Finally managers should consider eradicating smaller populations.

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1. GENERAL INTRODUCTION

This thesis is about the population ecology of wild horses (*Equus caballus*) in the Australian Alps. It is the first study of its kind in the Australian Alps, and therefore has a broad focus. The project was established in an applied science setting, and stemmed from a need to improve our understanding of this controversial species. It aims to provide interested parties with information to determine the best approach to management. The study also has broader applications. It explores survey techniques for large mammals in rugged, inaccessible terrain and contributes to our knowledge of the population dynamics of this widely distributed species. Finally, the thesis examines brumby-running for the first time. This is a cultural activity of horse harvesting, which has been associated with wild horses in the Australian Alps since their arrival. Brumby-running is being considered as a control option.

This introductory chapter sets the scene for the thesis. It provides a brief overview of the wild horse: its evolution, ecology and history in Australia followed by a description of the Australian Alps national parks and target areas for research on population ecology of wild horses in the Australian Alps. Finally it defines the aims and scope of the project.

1.1. Wild horse evolution and ecology

Through the process of this research, I have had many interesting conversations about how we should refer to my study animals. I have had people irate at me referring to them as 'brumbies' because it is too romantic a name for a feral pest. Brumby is a common name for wild horses in Australia (McKnight 1976). On another occasion, I had someone in tears after I called them feral and explained that the definition of feral is a domestic animal gone wild (Berger 1986). However the term feral has a different common usage in Australia which is loaded with negativity. Car stickers like 'shoot ferals' and 'if it's feral it's in peril' exemplify this point. I have chosen to call them wild horses. This may not be completely correct terminology, but it is a neutral term.

Horses are from the Order Perissodactyla (odd-toed hoofed mammals), which includes horses, zebras, tapirs and rhinoceroses (Hickman *et al.* 1990). The Perissodactyls dominated the Eocene (39-55 million years ago) (Duncan 1992 and references therein). The modern equids evolved in North America and became widespread, colonising South America and the Old World. Since the Pleistocene (2 million years ago) the once-abundant wild equids have declined. They disappeared from the Americas about 8000 years ago, and declined in the Old World more slowly. The reason for their extinction from the Americas has been ascribed to overkilling by people, disease or climate change (Duncan 1992 and references therein).

There is debate about the date of first domestication of horses (Pennisi 2001). A common theory is that wild horses from the grassland steppes of Eurasia were hunted as food and eventually tamed about 5000 years ago. Thus all domestic stock come from one area. However recent research using mitochondrial DNA suggests that local horses have been domesticated independently all over the world (Pennisi 2001). Horses have played an important role in human history. They have been fundamental to the success of agriculture, wars and transport. This role has declined with industrialisation, however horses still play an important role in industrialised societies through their recreational use.

Our understanding of the population ecology of wild horses comes from studies around the world. In a recent study of wild horse socio-ecology, Linklater (2000) reviewed feral horse populations in North America, South America, Europe and Australasia. Wild horses occupy a broad range of habitats from deserts to high rainfall areas, low to high altitudes and all latitudinal areas except the polar circles. Comparison across different populations showed that horses had remarkably similar social and spatial organisation and that group sizes (range 1 – 28) and home range size (range 0.8 – 303km²) varied as much between as within populations (Linklater 2000). The size of focal study populations ranged from 20 to over 400 horses. The density of populations ranged from 0.1 to 35.4 horses km⁻², and adult sex ratios varied from 0.03 to 1.85 males per female. Variation in densities and sex ratios were often the result of management practices such as removal of animals.

The maximum rate of increase observed for any wild horse population was 30% per year (Duncan 1992), however values of 21% per year are more common (Eberhardt 1987, Garrott *et al.* 1991, Cameron *et al.* 2001). Horses are birth pulse species with generally 80-90% of adult females (over 2 years) producing foals each year (Garrott *et al.* 1991). Survivorship of large mammalian herbivores is high in adults with little annual variation and generally lower and more variable earlier in life (Gaillard *et al.* 1998, 2000) and horses are similar (Garrott & Taylor 1990).

Variability in wild horse population dynamics occurs at different locations due to the fact that different limiting factors are operating on the components of the population. For example mountain lion predation (*Felis concolor*) drastically reduces juvenile survival (Greger & Romney 1999), and food limitation reduces juvenile survival, delays age at first reproduction and reduces fecundity (Duncan 1992, Scorolli 2001). Management practices can potentially limit populations by targeting fecundity (for example Garrott *et al.* 1992, Kirkpatrick *et al.* 1997) or survival (for example Eberhardt *et al.* 1982, Garrott & Taylor 1990, Berman 1991).

Horses were first introduced into Australia in 1788 (Dobbie *et al.* 1993). They adapted well to conditions in eastern Australia, and numbers rose rapidly. Between 1830 and 1850 they increased from an estimated 14 000 to 160 000 largely by natural increase. Horses were first recognised as feral pests in Australia in the 1860's. Many became redundant with the increase of mechanisation, giving rise to large unmanaged herds. Australia has the highest number of wild horses in the world (Dobbie *et al.* 1993). They occur mostly in remote, usually rugged, semi-arid areas. The largest populations are on unfenced pastoral country in the Northern Territory and Queensland. The major concentration of wild horses in New South Wales and Victoria are found in the Australian Alps (Dobbie *et al.* 1993).

1.2. The Australian Alps

The Australian Alps national parks (AAnps) form a contiguous protected area running from Namadgi National Park in the Australian Capital Territory (ACT) through New South Wales (NSW) to the Avon Wilderness in Victoria (Figure 1.1). The area is unique from the rest of the continent, which is characterised by low relief. The AAnps are typically mountainous

with extensive alpine, sub-alpine and montane environments. They also contain a range of other environments extending as low as 300m above sea level such as cypress-pine woodland, box, stringy-bark and peppermint open forest and wet forests. A Memorandum of Understanding (MOU) between the two states, the ACT and the Commonwealth aims to encourage the consistent and cooperative management of the AANps as a biogeographical unit (Australian Alps Liaison Committee 2002).



Figure 1.1: The Australian Alps national parks (Australian Alps Liaison Committee 2002).

Wild horses occur in Kosciuszko National Park in NSW (Figure 1.1), the Alpine National Park in Victoria and used to occur in the Namadgi National Park in the ACT (see Chapter 2 for details). Kosciuszko National Park in New South Wales is large, covering an area of approximately 698000 hectares (Good 1992). It is a UNESCO biosphere reserve nominated because the lands within the Park are an outstanding example of alpine environments, which

contain unique communities, and areas with unusual natural features of exceptional interest (Good 1992). The region offers many recreational opportunities for visitors each year, and supplies good quality reliable flows of water to the Snowy Mountains Hydro-Electric Scheme and adjacent farmlands.

The Alpine National Park in Victoria covers 646200 hectares and is one of the State's outstanding natural regions listed on the Register of the National Estate (Department of Conservation & Environment 1992). The Cobberas/Tingaringy Unit comprises 178400 hectares of public land. The unit forms part of the headwaters of the Murray River, of vital significance for water supply to the region. The fundamental aim of management is to conserve the natural environment, upon which all other activities are dependent (Department of Conservation & Environment 1992).

The Namadgi National Park is an area of 94000 hectares in the southern part of the ACT (ACT Parks & Conservation Service 1986). The Park is considered to have values for nature conservation and as a scientific reference area. It contains aboriginal and historic sites and low-key recreational and tourist activities occur throughout the area. It plays an important role in water supply for the region (ACT Parks & Conservation Service 1986).

1.3. The Australian Alps environment

The Australian Alps run adjacent to the southeast coast of Australia (Figure 1.1). The mountain range is relatively low by world standards with the largest peaks, including Australia's highest mountain around 2200m. The Alps has a cooler and wetter climate than the rest of the continent, with snow persisting at higher elevations in winter months. The climate varies across the Alps depending on location. Rainfall is generally high and temperatures cool to cold on elevated areas such as Kiandra, Thredbo and Hotham Heights (Figure 1.2). For each site respectively, the mean annual rainfall is 1564mm, 1803mm and 1553mm; the mean daily maximum temperature in the warmest month is 21°C, 21°C and 16°C in January or February; the mean daily minimum in the coldest month is -5°C, -4°C and -4°C in July. Rainfall is lower and temperatures higher at lower elevations including Orrol Valley and Dartmouth Reservoir (Figure 1.2). Mean annual rainfall at these sites is 864mm

and 1042mm, mean daily maximum temperatures occur in January or February at 25°C and 29°C and mean daily minimum temperatures are -2°C and 2°C in July for each of the sites respectively. Rainfall is higher on the western side of the range, with a distinct rainshadow on the eastern side (Green & Osborne 1994) (Dartmouth is on the west while Suggan Buggan and Orroral Valley are on the east) (Figure 1.2). Weather data for the eastern side of the Australian Alps from Jindabyne to the Snowy River National Park are scarce. This region is quite different from the rest of the Australian Alps national parks due to its low elevation and low rainfall. At Suggan Buggan (south-east of the Pilot, Figure 1.1) the mean annual rainfall (1957-1977) was 583mm. This is 40 percent less than Thredbo, which is 50km to the north on the mountain range. 16 km to the north of Suggan Buggan at the confluence of Running Waters Creek, mean daily maximum temperature (1985-1988) occurred in January at 29.6°C; mean daily minimum over the same period was in May at 0.7°C (Pulsford 1991).

The Alps consists of extended undulating plateaus running south-southwest created during a period of geological uplift. These plateaus fall away steeply at the edges. Erosion by rivers has cut back into the plateaus creating large, v-shaped valleys (Australian Alps Liaison Committee 2002). In NSW, the plateaus are large and relatively intact, while in Victoria, the higher plateaus are a series of isolated high plains with much of the area being made up of dissected terrain of steep valleys and narrow ridges (Australian Alps Liaison Committee 2002).

The rock-types on the higher elevation areas are remnants of either basalt capping (for example Mt Jim, Bogong High Plains and Round Mountain near Mt Jagungal), a planed off metamorphic complex (for example most of the Bogong High Plains), or an exposed granite batholith (for example Mount Kosciuszko and surrounds). Basins mark these elevated areas where water accumulates forming bogs and fens. In the sub-alpine zone, cold air drains into these basins creating alpine conditions and an inverse treeline (Australian Alps Liaison Committee 2002).

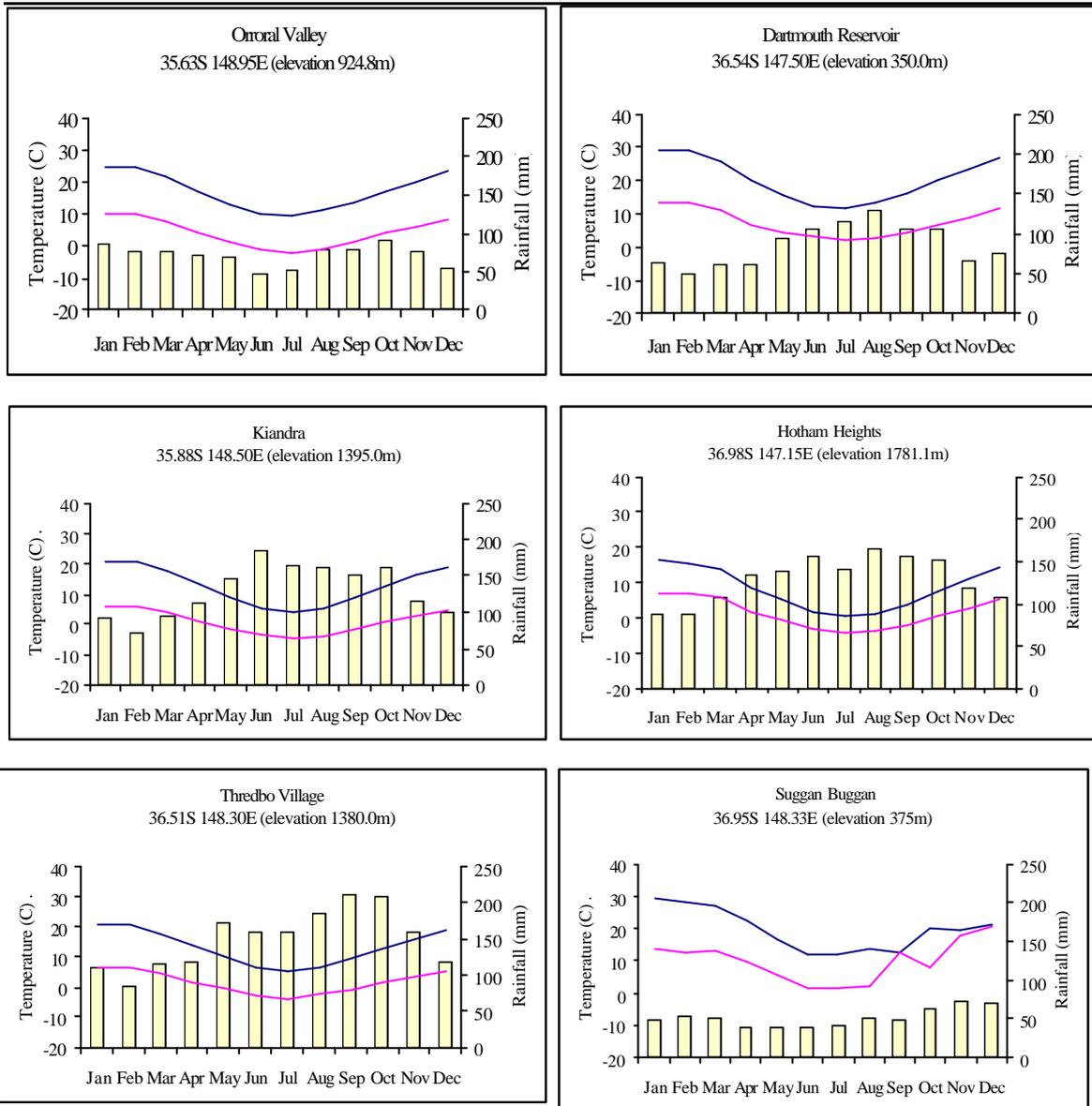


Figure 1.2: Mean daily minimum and maximum temperatures (°C) and rainfall (mm) for six sites in the Australian Alps. All data are from the Bureau of Meteorology (2002) except Suggan Buggan (Pulsford 1991). Temperature data for Suggan Buggan were recorded 16km north at Running Waters Creek.

Slopes and ridges are probably the most extensive landform in the Australian Alps national parks and were formed as rivers cut through the uplifted rock (Australian Alps Liaison Committee 2002). This has created narrow ridgelines with more soil on the slopes. At the base of these slopes, river valleys form with deeper soils. Rugged terrain exists in the form of escarpments, gorges and waterfalls. There are some glacial features, such as morains, at the highest elevations.

Vegetation in the Australian Alps is characterised by a sequence of communities with distinct altitudinal zones. These are broken into lowland, montane, sub-alpine and alpine (Australian Alps Liaison Committee 2002). The bounds of these zones occur at different elevations depending on location. For example the lower limit of the alpine zone (the treeline) lies between 1800m and 1900m in NSW and declines to 1750m in Victoria (Green & Osborne 1994). Lowland communities are widespread and are typically very open Eucalyptus woodland communities with an herbaceous understorey (Australian Alps Liaison Committee 2002). The montane zone contains three main communities: open forest (dry sclerophyll), tall open forest (wet sclerophyll) and alpine ash (Australian Alps Liaison Committee 2002). The sub-alpine zone is a combination of sub-alpine and alpine communities. Exclusive to the sub-alpine zone is sub-alpine woodland and sod tussock grassland. In areas of cold air drainage, alpine communities exist. Alpine communities include alpine shrubby heath, alpine herbfield, alpine bog, short alpine herbfield and feldmark (Australian Alps Liaison Committee 2002).

1.4. Grazing in the Australian Alps

The Australian Alps has a history of cattle (*Bos taurus*) and sheep (*Ovis aries*) grazing since the 1800s. Cattle and sheep grazing in alpine and sub-alpine lands within what is now Kosciuszko National Park commenced on the more accessible areas by the 1830's, and continued until the 1950's (Bryant 1973). Investigations into the impact noted the almost complete destruction or loss of water holding capacity of many bog and fen communities through direct physical damage (Good 1992 and references therein). On the Main Range, grazing caused severe erosion and the destruction of vegetation communities and formations (Good 1992). At a lower altitude, on Nungar Plain in the north of the Park, Bryant (1973) found that sheep and cattle grazing caused the continual loss of cover of all grasslands

studied except the densest; even here deterioration occurred. Grazing of stock is no longer allowed in Kosciuszko National Park.

Grazing has a similar history in the Alpine National Park, however it is still permitted in some areas (Department of Conservation & Environment 1992). Grazing runs originally extended to cover almost all of the alpine and sub-alpine vegetation communities, and the grassy understoreys of the open forests and woodlands. Grazing has been greatly reduced. In 1991, about 1700 head of cattle were grazed in alpine and sub-alpine environments (above 1220 metres elevation) (Department of Conservation & Environment 1992). The same concerns about the impact of stock grazing as in Kosciuszko exists for the Alpine National Park.

Pastoral settlement in what is now Namadgi National Park concentrated on the broader valleys (ACT Parks & Conservation Service 1986). Grazing and associated seasonal burning has had widespread effects mostly in the valley and lower slopes. Gudgenby Station was the last area from which stock were removed in 1987. Namadgi is currently free of the impacts of cattle, sheep and horse grazing.

There is a concern that wild horses will have similar impacts to cattle and sheep (NSW NPWS 2002a). In the Alps, they were seen to be a problem during last century (Helms 1890 in Dyring 1990). Many scientists (Costin 1954, Walters & Hallam 1992, Dobbie *et al.* 1993, Green & Osborne 1994) have expressed concern about the threat of increasing numbers of wild horses to environmental values of the Alps. In the Australian Alps, Dyring (1990) identified track formation, soil compaction, change in vegetation structure and streambank disturbance as impacts of wild horses on sub-alpine and montane environments. In New Zealand, Rogers (1991) showed that wild horses were eliminating certain high elevation tussock communities through preferential grazing, impacting on high nutrient flushes areas by trampling and grazing and they also threaten several vulnerable plant species.

1.5. Culture and wild horses

There is a considerable amount of folklore associated with wild horses in the Australian Alps. Banjo Patterson made them famous with his poem “The Man from Snowy River” which later became the basis for a movie. Banjo Paterson and wild horses feature on the Australian ten dollar note. A local author Elyne Mitchell also wrote children’s fiction about life as a wild horse in the Australian Alps. Many Australians know something about wild horses and see them as having cultural value.

1.6. Target areas for research on the population ecology of wild horses in the Australian Alps

There are several cases where wild horses are successfully managed based on biological research. In central Australia, there has been a study on the population ecology and environmental impact of the resident wild horses (Berman 1991). This has led to a management strategy that maintains the population at a level that prevents extensive degradation of the land and high mortality of horses associated with droughts (Dobbie *et al.* 1993). In New Zealand, ‘Kaimanawa’ horses inhabit alpine and sub-alpine environments, and the Wildlife Act 1953 protects them. Detailed studies on the environmental impact and population ecology of these horses have been running since 1989 (Department of Conservation 1995). The studies show that increasing numbers of horses were having adverse effects on the rare and indigenous qualities of the area. Numbers are now being maintained at ecologically sustainable levels (Department of Conservation 1995).

In 1992, there was a review of wild horses in the Australian Alps national parks which recommended that an action plan for feral horses be developed and implemented at the earliest opportunity (Walters & Hallam 1992). The action plan was to be preceded by two key studies: 1) a study of wild horse demography and their ecology, and 2) a study of wild horse impact on vegetation communities and the environment of the Australian Alps national parks. This thesis addresses the first objective. The second objective is partially being addressed by a long-term vegetation monitoring project in the Alpine National Park, which commenced in 1999 (Theile & Prober 1999).

There are a multitude of aspects of wild horse ecology that could be studied to improve our understanding and hence management of the species and their impacts. As with most research, resources and time are limited, so research effort needs to focus on a few key areas. The key areas of focus are briefly outlined and justified below.

1.6.1. Distribution

The distribution of a species is one of the key elements of its ecology (Krebs 1994). Distribution defines the broad geographical limits inside which the species may be found more or less permanently established (Andrewartha & Birch 1954). A series of distribution maps over time illustrate the rate of spread or contraction of species (Krebs 1994), or the response of distribution to control or reintroduction programs. Dyring (1990) produced a distribution map of wild horses in the Australian Alps. Distribution maps were also important in the current study for reducing the cost of estimating abundance by aerial survey by limiting the area covered in the survey.

The mechanisms that affect the distribution of a species vary. For example in Australia the red fox (*Vulpes vulpes*) distribution has been closely tied to their main prey item, European rabbit (*Oryctolagus cuniculus*) (Saunders *et al.* 1995), while the spread of horses was associated with pastoralism (McKnight 1976). Insights into mechanisms that influence a species abundance can help to focus management activities.

1.6.2. Abundance

The goals of managing wild animal populations are usually expressed in terms of population size (Lancia *et al.* 1996). For endangered species, managers try to increase population size, for pest species they try to reduce population size and for harvested populations they try to maintain a population size to optimise harvest (Caughley 1977a, Lancia *et al.* 1996). Population estimates are particularly useful if they are repeated through time. Population trend data is useful for assessing the success of management actions or response of the population to natural events. Examples are evident for feral goats (*Capra hircus*) (Pople *et al.* 1998) and feral pigs (*Sus scrofa*) (Hone 2002). There have been no attempts to estimate the

size of the wild horse population across the Australian Alps. The only population estimates of wild horse numbers at this scale have been based on anecdotal evidence (Dyring 1990).

Aerial survey is often the only way of surveying large animals such as horses over a large area. Aerial survey techniques have improved over time, addressing the main problem with aerial survey, undercounting (Caughley 1974, Pollock & Kendall 1987). Several techniques have been developed to reduce the problem of undercounting, including strip counts (for example Southwell 1989); mark-recapture (for example Bartmann *et al.* 1987) and line transect (for example Buckland *et al.* 1993). It is not clear what the most suitable method is for counting wild horses in the Australian Alps.

1.6.3. Population dynamics

A basic understanding of population dynamics is essential to the development and implementation of population management goals for whatever objective (conservation, harvesting or control). Population dynamics are the variations in the size and density of a population (Begon *et al.* 1990). Variations reflect a change in demographic parameters including birth rate, survival rate and/or distribution (Caughley 1977a, Caughley & Sinclair 1994) and can be defined by the unifying variable population growth rate (Sibly & Hone 2002). Examples of an understanding of population dynamics aiding management decisions for wildlife populations are illustrated for the northern spotted owl (*Strix occidentalis caurina*) (Lande 1988) and moose (*Alces alces*) (Solberg *et al.* 1999). There was no research into the population dynamics of wild horses in the Australian Alps prior to the current study.

An initial approach to a study on a population whose ecology is poorly understood should be observational and should be assessed in the light of expected values from general experience (Krebs 1988, Eberhardt in press). Such studies should be followed up by more detailed experimentation once the major mechanisms operating on the population are understood. In large mammals, the primary cause of natural population limitation is food supply (Sinclair 1989, 1996). This has been observed in wild equids (Choquenot 1991, Duncan 1992) and may be occurring in the wild horses of the Australian Alps.

A final key area of application for population dynamics to management is sensitivity of population growth rate to change in key demographic parameters. It can be used to help make decisions on what component of the population to target if undertaking control or conservation measures (Lande 1988).

1.6.4. Brumby-running

Wild horse populations are often limited by human management (for example Eberhardt 1987, Department of Conservation 1995, Bryan 2001). The only active management of wild horses in the AAnp is brumby-running in the Victorian Alps. Brumby-running is a form of harvesting where horse-riders rope wild horses and remove them from the population. Harvesting often leads to a reduction in wildlife populations as illustrated for white-tailed deer (*Odocoileus virginianus*) and moose (Solberg *et al.* 1999) (Fryxell *et al.* 1991). Brumby-running is thought to have a limiting effect on the wild horse population in Victoria, but it has never been examined empirically.

1.7. Aims and scope of the thesis

This thesis aims to make a distinct and significant contribution to knowledge and understanding of the population ecology of wild horses in the Australian Alps. The only previous study of wild horses in the Australian Alps was on their impact in sub-alpine and montane environments (Dyring 1990). The current study provides baseline data on the current distribution of wild horses in the Australian Alps and puts it into an historical context to determine key mechanisms affecting their distribution (Chapter 2).

In Chapter 3, I use several aerial survey techniques to estimate abundance of wild horses in the Australian Alps national parks. I compare strip, mark-recapture and line transect techniques. The design is a repeatable method of estimating abundance of wild horses, which will allow ongoing monitoring of the population.

Chapter 4 examines the population dynamics of three wild horse populations within the Australian Alps. The populations were surveyed every spring and autumn for three years to

yield information on the abundance, density and growth rates of the populations as well as estimating key demographic parameters: fecundity and survivorship. The study makes a preliminary assessment of whether it is likely that wild horses in the Australian Alps are food limited like most wild herbivores (Sinclair 1989, 1996). Sensitivity analysis (Lande 1988) was used to assess the relative effect of targeting different life history stages in management.

Chapter 5 examines the demography of wild horses caught and removed by brumby-runners in the Victorian Alps using data collected over four years. This is compared to the demography of the wild horse population in general (Chapter 4). The behaviour of brumby-runners is examined using predator-prey theory. Finally predictive modelling is used to assess the likely effect brumby-running is having on the wild horse population in the Victorian Alps and is discussed in terms of wildlife harvesting and control strategies.

In the final chapter, I outline the major findings of the study relevant to wild horses in particular and ecology and wildlife management in general. I then outline management recommendations and future research directions.

2. DISTRIBUTION OF WILD HORSES IN THE AUSTRALIAN ALPS

2.1. Introduction

In the broadest sense, ecology is the scientific study of the interactions that determine the distribution and abundance of organisms; or more simply, the study of where organisms are found, how many occur there and why (Krebs 1994). Distribution and abundance are linked because the factors that affect the distribution of a species also affect its abundance (Krebs 1994). The distribution of a species is defined by the broad geographical limits inside which the species may be found more or less permanently established (Andrewartha & Birch 1954). Within this distribution there may be favorable areas where abundance is high, and less favorable areas where the numbers are low or animals are temporarily absent (Andrewartha & Birch 1954).

Barriers to dispersal can prevent a species from occupying all areas that are suitable for survival. Translocation experiments are the first way to test whether barriers are limiting the distribution of a species (Krebs 1994). If a translocation is successful, then it shows that the distribution was limited because the area was inaccessible or because the species failed to recognise the area as suitable living space (Krebs 1994). In these cases, the potential range is larger than the actual range. A classic example of large translocation experiments has been the movement of animals around the world by people.

A newly liberated population will disperse, maintain the same distribution or contract. Dispersal may occur only when a critical threshold density is reached in the existing distribution or it may occur by simple diffusion regardless of density (Caughley 1977a). The speed of dispersal varies greatly between species. Rabbits can spread at 64km per year, while sambar deer (*Cervus unicolor*) spread at 0.6km per year (Caughley 1977a). If there are no barriers to dispersal, and a species can survive after its translocation, behaviour can prevent the species from expanding its distribution (Krebs 1994). Behaviour of large mammals that limit distribution is often related to factors such as density. For example the range of the wood bison (*Bison bison*) remained relatively static for periods of time before they dispersed

in a pulse; intraspecific competition for food is the likely stimulus for each range expansion (Larter *et al.* 2000).

The reasons for an unsuccessful translocation (and hence limited distribution) fall into three categories: the presence of other species, physiological ecology (Krebs 1994) and number of individuals translocated (Wolf *et al.* 1996). The classic example of one species limiting the distribution of another is predators limiting their prey (Krebs 1994). Predators can also limit the abundance of a species without excluding them completely. Competitive species, parasites and disease may limit the distribution. Environmental factors such as temperature and water availability affect one or all of the key elements to success of a population: survival, growth and reproduction (Krebs 1994). The physiology of every species limits the environments it can inhabit (Andrewartha & Birch 1954, Krebs 1994). These in turn affect distribution and abundance.

The success of a translocation depends on the number of animals translocated (Wolf *et al.* 1996, Komers & Curman 2000). In a study on translocation of ungulates, Komers & Curman (2000) found that when less than 10 individuals are introduced into suitable habitat, the populations frequently go extinct. However, if more than 20 animals are introduced, the populations have positive growth. Success of translocations (in particular re-introductions) of birds and mammals are generally improved if they are released into high quality habitat (Wolf *et al.* 1996). Translocation success is not correlated with reproductive potential or number and duration of releases (Wolf *et al.* 1996).

The distribution boundary of a species fluctuates with time (Andrewartha & Birch 1954) and this concept is generally referred to as metapopulation dynamics (Hanski & Gilpin 1991). Through time, local populations undergo extinction and re-colonisation (Harrison 1991). Stochastic events (demographic, genetic and environmental) cause the extinction of local populations (Harrison 1991) while environmental stochasticity (such as drought and fires) appears to cause the most extinctions (Harrison 1991).

The movement of horses around the world by humans has been a series of large-scale translocation experiments, and horses have shown themselves to be adaptable. A classic example of the success of horses when transported to a new environment was the release of horses at Buenos Aires in 1537 (Darwin 1845 in Caughley 1977a). They spread south to the Straits of Magellan by 1580. This is an incredible dispersal rate of 48km/year.

Horses first arrived in Australia with the first fleet in 1788 (Dobbie *et al.* 1993). Their consequent spread was slow and erratic and was closely tied to settlement by pastoralists (McKnight 1976). Horses have occupied all but the wettest and driest regions of Australia, and the range and size of populations have fluctuated. These fluctuations have been determined primarily by three factors: first, the amount of control exercised by humans; second, weather conditions- drought or non-drought; and third, additions to the wild populations from domesticated animals (McKnight 1976). The main means by which their range permanently retracts has been through exclusion due to increasingly intensive farming practices (McKnight 1976). McKnight (1976), Dobbie *et al.* (1993) and Strahan (1995) have presented maps for wild horses in Australia, which show that the distribution in the Australian Alps is small and isolated from other populations.

The potential of behaviour to limit the distribution of wild horses is difficult to assess. They are highly social animals (Linklater 2000) and there is a general belief that they are strongly tied to their home range (McKnight 1976). Dispersal of young animals is common in many species (Caughley 1977a), and may be the primary form of dispersal in wild horses. Emigration of young horses (age one to three years) from their natal area occurs frequently (Kaseda *et al.* 1997).

Species that have the potential to limit the distribution of wild horses in Australia are humans, wild dogs and parasites. Australia is unusual in that there is a distinct lack of potential predators of large herbivores excepting humans and the dingo. The role of humans as predators usually only focuses on indigenous people. In other continents, indigenous people have had a strong influence on the survival of introduced horses, however in Australia there are few recorded instances of Aborigines capturing or killing wild horses (McKnight

1976). Control measures by humans in general can also be viewed as a form of predation (see Chapter 5) but is usually considered separately (for example McKnight 1976). Wild horse control programs typically aim to reduce or maintain the density of a wild horse population (Eberhardt *et al.* 1982, Garrott *et al.* 1991a), while other programs also include range reduction (for example Bryan 2001).

Equids form part of the diet of wild dogs (Newsome *et al.* 1983a), but McKnight (1976) and Choquenot (1991) argue that wild dog predation doesn't have a significant effect on wild horse populations. Horses are hosts to internal and external parasites (Duncan 1992). The worst internal parasites of wild horses on the Camargue are nematodes, which can cause internal damage and eventual death (Duncan 1992). Horseflies (Tabanidae) are external parasites that affect horses through blood loss and disease transmission (Duncan 1992). However these parasites did not prevent the Camargue population from increasing in size. Therefore, parasites are unlikely to have a strong effect on distribution.

The most characteristic feature of wild horse habitat is the presence of grasses and forbs or edible shrubs, usually in expansive environments (Berger 1986). The key environmental variable limiting the distribution and abundance of horses in much of Australia is rainfall (McKnight 1976, Dobbie *et al.* 1993). Similarly, droughts cause large reductions in the abundance of kangaroos such as red kangaroos (*Macropus rufus*) (Caughley *et al.* 1987, Cairns & Grigg 1993). Temperature is unlikely to limit the distribution of wild horses in Australia because populations persist at temperatures as low as -40°C in mountainous areas of North America and in environments where temperatures reach over 42°C such as Australian deserts (Berger 1986, Dobbie *et al.* 1993). An additional environmental variable of importance in the Australian Alps is snow. The alpine zone is characterised by a continual snow cover for at least four months of the year, with about 60% of the annual precipitation falling as snow (Green & Osborne 1994). The sub-alpine zone is characterised by a continuous snow cover for one to four months of the year (Green & Osborne 1994). Snow causes the seasonal migration of horses in The Granite Range (Utah), and occasional catastrophic mortality (Berger 1983). Snow also affects the survival of other ungulates such

as bighorn sheep (*Ovis canadensis*), mule deer (*Odocoileus hemionus*) (Picton 1984) and moose (Solberg *et al.* 1999).

Dyring (1990) mapped the distribution of wild horses in the Australian Alps in detail including state forests and private land. The main features of the distribution in Kosciuszko National Park were small isolated populations in northern Kosciuszko at Peppercorn, Wild Horse and Nungar Plains, and in nearby Bago and Maragle State Forests. There was a small population at Botherum Plain. In southern Kosciuszko there was a large continuous population south of Thredbo River extending to the Victorian border. It also extended to the east, but not to the park boundary. In Victoria, Dyring (1990) showed that the most extensive population of wild horses occurred in the Cobberas-Tingaringy National Park (now the Cobberas-Tingaringy Unit of the Alpine National Park) and surrounds. The next largest population occurred on the Bogong High Plains with smaller populations at Mount Willis (west of Lake Dartmouth) and on the Moroka and Caledonia Rivers in Wonnongatta-Moroka National Park (now the Wonnongatta-Moroka Unit of the Alpine National Park).

Dyring (1990) identified humans, drought and fire as important factors that may affect distribution of wild horses and suggested a negative relationship exists between numbers of rabbits and wild horses. Higgins (1994) provided a history of horses in Namadgi National Park (discussed below).

The primary aim of this chapter is to present a revised distribution map of wild horses in the Australian Alps national parks to be used to define the area covered in the aerial survey (Chapter 3). The second aim is to explore ecological theory to explain the current and historical distribution of wild horses in the Australian Alps. This will be used to assess the potential for them to expand their range in the future.

2.2. Methods

The current distribution map was compiled over several years (1999-2001). Initially NSW and Victorian wildlife atlases were searched, and Dyring's (1990) maps were reviewed. Then discussions were held with local managers and park users to better determine boundaries. I

targeted people with a long association with the area and an interest in wild horses. These estimates were checked by personal observation over much of the range between 1999 and 2001. Oral histories relating to past distribution were collected at the same time. The recording of oral history was by no means exhaustive. I have pieced it together here to draw a picture to help us understand patterns of horse distribution in the Alps through time. The methods used in this study are similar (but on a smaller scale) to those used by McKnight (1976) in his survey of feral livestock in Australia, and to Dyring's (1990) study of feral horses in the Australian Alps. Once maps were considered to be fairly accurate, they were digitised. This presented difficulties due to land tenure boundaries across states. I digitised the distribution of wild horses in Kosciuszko National Park on the NSW National Parks and Wildlife Service (NPWS) geographical information system with the help of technical staff. An employee of Natural Resources and Environment, Victoria digitised the map for Victoria based on my hand-drawn map.

The distribution maps were confined to the Australian Alps national parks. This was done for several reasons. Firstly, the study area needed to be defined, and this is a well-established boundary. Secondly, the Australian Alps national parks funded the project. Thirdly, the status of horses, whether domestic or wild, becomes unclear once out of the national parks.

2.3. Current distribution of wild horses in the Australian Alps

Wild horses are found in Kosciuszko National Park, NSW (Figure 2.1) and the Alpine National Park, Victoria (Figure 2.2). Wild horses are only present on the border of the Australian Capital Territory at present though they have occurred there in the past (Higgins 1994).

There are four populations in Kosciuszko National Park: the population with the largest area is in southern Kosciuszko and there are smaller populations in northern Kosciuszko, on the north-east boundary near Talbingo reservoir, and on Snowy Plains north of Lake Jindabyne (including Botherum Plain) (Figure 2.1). All of these populations extend into adjacent private and state forest lands. There are two main populations in the Alpine National Park, Victoria (Figure 2.2). The population with the largest area is in the Cobberas-Tingaringy Unit in the

north of the park. This population is connected to the population in southern Kosciuszko (Figure 2.1). A second population is located in the Bogong High Plains. Several other smaller populations occur which are not mapped because they may only be temporary (see Section 2.4 for details).

2.4. Translocation of wild horses in the Australian Alps

The first record of horses being released in the Australian Alps was by Davey O'Rourke in 1843 (Dyring 1990). He released 70 mares and 2 stallions at Black Mountain in Victoria. Black Mountain is on the eastern watershed of the Great Dividing Range near the NSW border. These horses are thought to be the ancestors of much of the wild horse population in the northern part of the Alpine National Park, and the southern part of Kosciuszko National Park (Dyring 1990). An exact date of release of wild horses into northern Kosciuszko National Park is not known, however, it was probably in the mid-1800s when graziers started using the area (Australian Alps Liaison Committee 2002, Ted Taylor pers. comm.). The population was generally restricted to the northern end of its current range. George Day released horses in the 1960's in the area near Kiandra and Goandra at the southern end of the current northern Kosciuszko population (Ted Taylor, John Rudd pers. comms.). These two populations have joined but animals tend to be different colours- predominantly greys in the south and predominantly chestnuts and bays in the north.

The first wild horses in the Australian Capital Territory came from several sources in the early 1900s (Higgins 1994). In the Brindabella Range, horses escaped from the Franklin family of Brindabella Homestead. In the south (Mt Scabby and adjoining areas), horses escaped from poorly fenced farms at Yaouk. In the Cotter Catchment, it is said that the Oldfield family lost horses in the bush while they lived at Cotter Hut prior to the resumption of this area for the Cotter Catchment in 1913. Wild horses are thought to have arrived at Jumbuck, Kangaroo (now Smokers Gap), Smokers and Emu Flats as the result of fleeing from the brumby-runners (see Chapter 5 for details on brumby-running) around 1906.

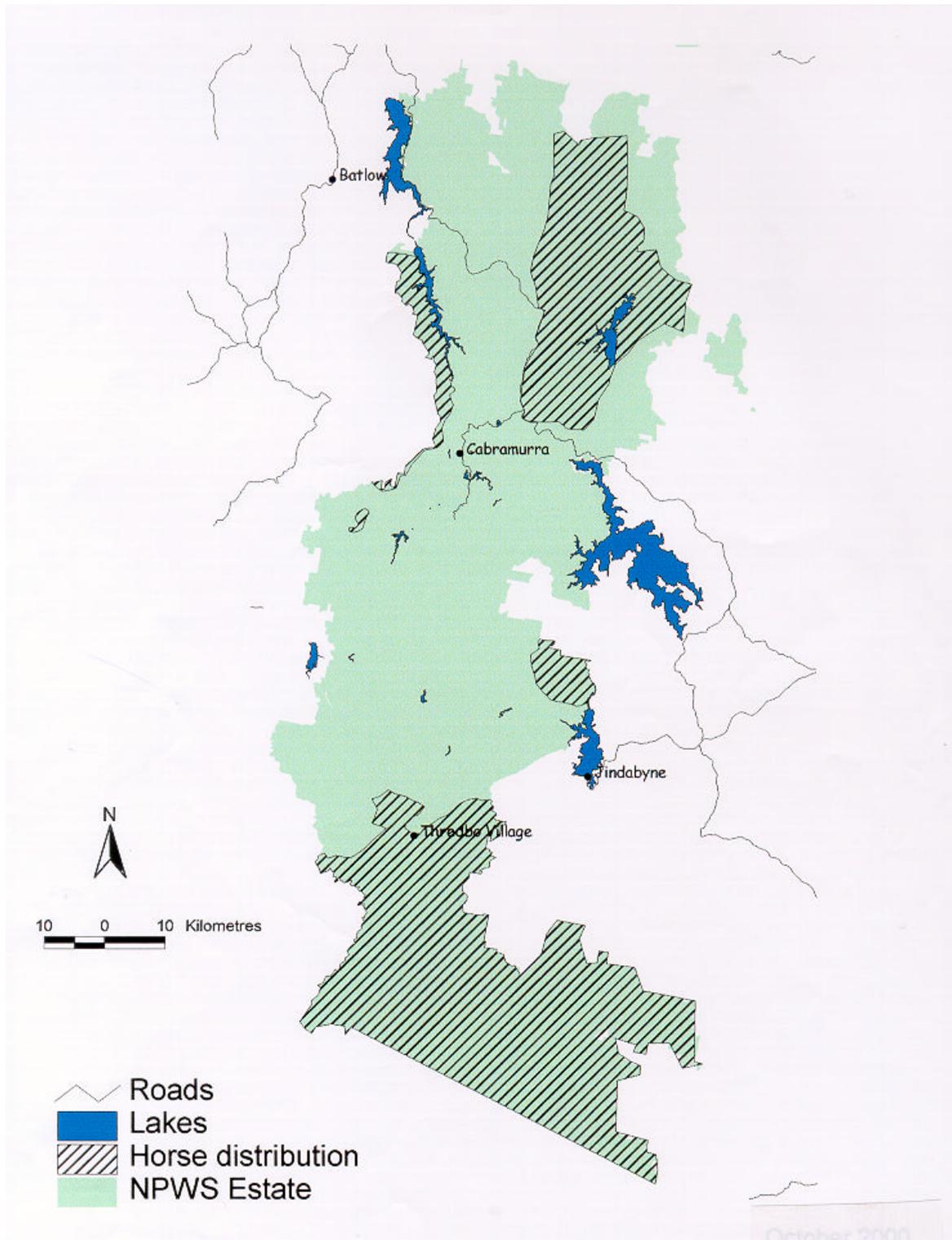


Figure 2.1: Distribution of wild horses (hatched) in Kosciuszko National Park (grey). Map produced courtesy of NSW NPWS.

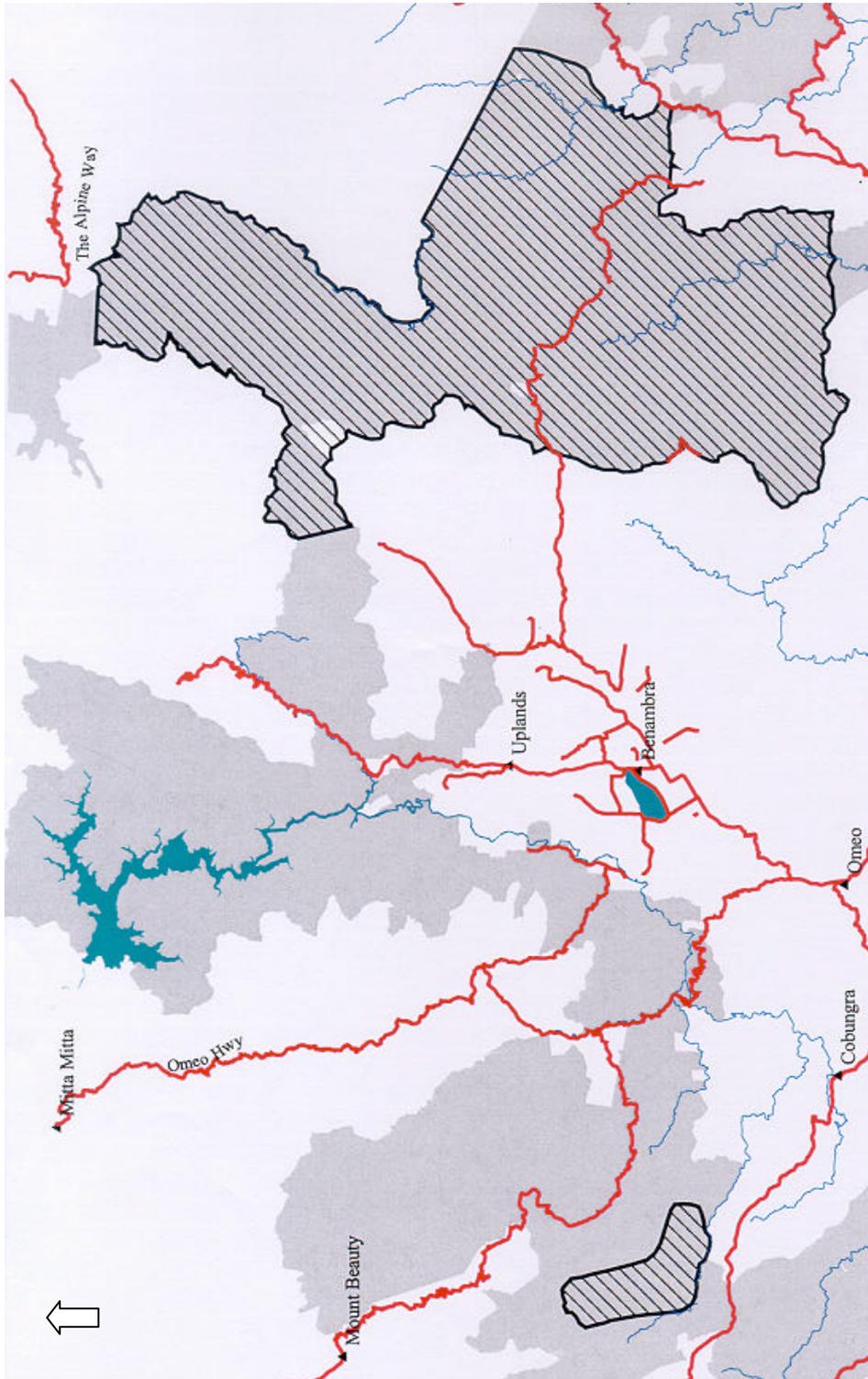


Figure 2.2: Distribution of wild horses in the Alpine National Park, Victoria. The Alpine National Park is grey, horse distribution hatched and roads are dark grey. Map produced courtesy of the Department of Natural Resources and Environment. The scale is 1cm = 5.5km.

The date of release of horses in the area adjacent to Talbingo reservoir is not known, however they have been present in the adjacent state forests for some time (Neil Druce pers. comm.). The population inhabiting Snowy Plain on the eastern boundary of Kosciuszko National Park is a fairly recent addition to the wild horse populations in the Australian Alps. Neville Harvey first released these horses in 1958 (Barry Aitchison, pers. comm.). Wild horses have been present on the Bogong High Plains (Victoria) since before the 1900s (Ron Riley pers. comm.). They were also present on adjacent plateaus historically (Helen Packer pers. comm.).

Small isolated populations of wild horses would have appeared at different locations within the Australian Alps national parks over the past 150 years. The permanency of such populations varies. Examples of small isolated populations from recent years are given below. Two horses were observed in the Jagungal area of Kosciuszko National Park in 2000-2001 and were thought to be of domestic origin (Barry Aitchison pers. comm.). There have been no reports of sightings of these animals in 2002. Wild horses used to be found in the Wonnangatta-Moroka Unit of the Alpine National Park near the Caledona River and on the Moroka River (Dyring 1990). The animals on the Caledona River are no longer present, and the animals on the Moroka River were thought to number 6-20 in 2000 (Anon pers. comm.). There have been recent sightings of wild horses in the Howqua area south of Mt Buller in the Wonnangatta-Moroka Unit of the Alpine National Park (Charlie Pascoe pers. comm.). The first sightings (of at least two mares and two foals) were recorded in August 2001.

A rapid rate of spread was not observed in wild horses in the Australian Alps. Some populations do not appear to be spreading at all, while others appear to spread slowly. The distributions of the two large populations of wild horses in the Alpine National Park, Victoria were similar in 1990 to the present (Dyring 1990). The population around Thredbo in southern Kosciuszko has extended in range north towards Mt Kosciuszko by about 5-10km in 10 years. The distribution on the eastern side of southern Kosciuszko may have spread a little when comparing the current distribution with the distribution mapped by Dyring (1990) but I am not confident on the current distribution because horses are in very low densities in this area and this was not an area of focus. Horses appear to have spread at a faster rate in

northern Kosciuszko. Dyring (1990) indicated only a few isolated populations of wild horses in northern Kosciuszko, while the current distribution is connected throughout the entire area. The maximum rate of spread of wild horses in northern Kosciuszko based on a comparison between the current map (Figure 2.1) and Dyring's map is 4km per year. This trend should be interpreted with caution because Dyring (1990) did not focus on northern Kosciuszko and is not confident on the accuracy of the map in this area (Jenny Dyring pers. comm.).

2.5. What prevents wild horses from occupying all of the Australian Alps?

2.5.1. Humans

Graziers controlled the distribution of wild horses in much of the Australian Alps from the mid-1800s to as late as the 1960s. Different graziers had different approaches to wild horses; some were responsible for their release into new areas, others eradicated them locally. Examples of control of wild horses by graziers occur the length of the Australian Alps. One well known example is the Freebody's who had a lease extending through what is now the Ingegoodbee and Pilot wilderness area in southern Kosciuszko National Park. The Freebody's had a policy in the 1930s and 1940s of shooting all wild horses on their leasehold, and were fairly successful in keeping them out (Dave Pendergast pers. comm.).

Myles Dunphy, a key figure of the conservation movement in Australia, recorded this in his journal (Dunphy 1934) in June 1934:

“All through these wild and cold highlands wild horses roam free in innumerable mobs, matching their horse-sense against their deadly enemies, the Freebody rifles. There is a scheme afoot and started, to wire fence the summit of the Great Divide; so soon the poor harried beasts will be unable to change their grounds to accord with the seasons, and henceforth icy winds, deep snow and that awful barbed fence will spell the doom of the thundering mobs. It ought not be so, it does not seem right that the last of the Monaro brumbies be wiped out because of some sheep and cattle. Surely posterity would care to view their running free in land which had been theirs so long. The ‘Man From Snowy River’ will never die, why then the horses?”

In northern Kosciuszko, in the areas near Coolamine and Currango, fencing excluded wild horses during the grazing era (Ted Taylor pers. comm.), while the Nankervis' kept wild

horses off the Ramshead Range (Dave Pendergast pers. comm.). Horses are present in all of these areas today.

Both humans and snow affected wild horse distribution in the ACT (Higgins 1994). Wild horses were present in what is now Namadgi National Park from the early 1900s until 1987 in varying numbers (Higgins 1994). Their historic distribution can be seen by the presence of wild horse trapyards (Figure 2.3). Their distribution fluctuated through time. In the 1930s, brumbies were regularly chased by brumby-runners in the Smokers Flat area and all were eventually caught (Higgins 1994). Merve Tong reintroduced them, and the population was thought to number about 200 in the 1950s and 1960s despite brumby-running. Brumby-running came largely to a close in Namadgi at the end of the 1960s. Wild horses at higher elevations suffered a severe reduction in numbers in the early 1960s due to heavy snowfalls. However they persisted in lower areas. It is not clear how much effort was made to suppress the wild horse population in Namadgi National Park from the 1960s through until the 1980s. However the last wild horses (33) in Namadgi National Park located in the Cotter catchment were culled in accordance with the Namadgi National Park Management Plan (Higgins 1994).

Management activities (or lack thereof) by Park agencies have lead to wild horse range contractions and expansions. Park managers have successfully excluded wild horses from Snowy River National Park, Victoria (Dave Ingram pers. comm.). Wild horses ranged the rugged rainshadow region of Snowy River National Park several decades ago but they have been controlled and there do not appear to be any wild horses in the area currently (Dave Ingram pers. comm.). Conversely, wild horses have not been actively managed in northern Kosciuszko for decades, and the number and distribution of wild horses in this area have increased (John Sutton, pers. comm.).

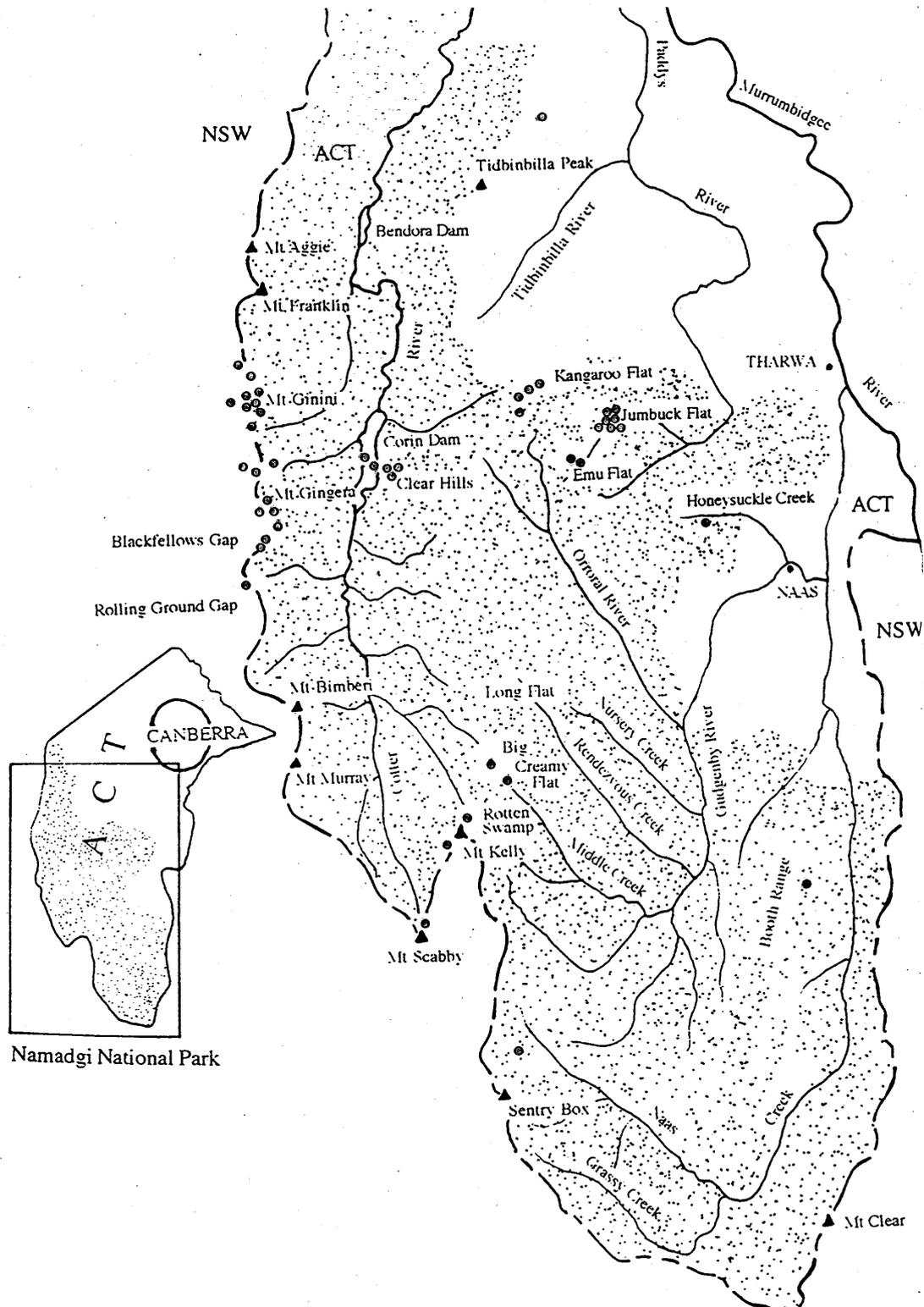


Figure 2.3: Wild horse trapyards known to have existed in the Namadgi area (Higgins 1994). Small circles represent trapyards.

2.5.2. Geographical barriers

The population of wild horses on the Bogong High Plains (Victoria) appears to be contained on the plateau with very few refuges at lower elevations. The Bogong High Plains is an isolated plateau surrounded by steep dissected terrain. The population of wild horses is thought to have been fairly stable for a long time at several hundred animals (Ron Riley pers. comm.). The factors thought to be limiting the size of the population are brumby-running and snow (Ron Riley pers. comm.).

The Snowy Plains (NSW) population appears to be contained on all sides by private property or rugged terrain. The distribution of these horses has not extended beyond the grassy flats and woodland around Snowy Plain after 40 years of occupation (Barry Aitchison pers. comm.). The population is also subject to brumby-running, which has not caused a range extension as suggested around 1906 for wild horses in Namadgi National Park (Higgins 1994).

In recent years wild horses have been resighted in Namadgi National Park. They were sighted at Murrays Gap on the NSW/ACT border in 1999 (Geoff Young pers. comm.) and again during the aerial survey (2001) for the current project (pers. obs.). Murrays Gap and nearby flats are a series of grassy clearings on the border between NSW and the ACT. The topography becomes steep and the hills timbered below these flats in the ACT before reaching the upper Cotter River, where wild horses occurred historically (Higgins 1994). The more rugged terrain appears to be preventing the re-invasion of horses at present.

Wild horses were not present on the Ramshead in 1990 (Dyring 1990). The first sign of wild horses in recent years was about 6 years ago (Ken Green pers. comm.). Horses were present on the Ramshead Range every summer from 1998/99 to 2000/01 (pers. obs.). Access to the Ramshead is limited for wild horses. They must first cross a road and then travel up through sub-alpine woodland for at least one kilometre. This appeared to be a barrier for many years. It is thought that the reason for wild horses moving into this new area was a result of population pressure (Pam O'Brien pers. comm.).

2.5.3. *Environment*

Vegetation communities and hence habitat are strongly defined by elevation in the Australian Alps (see Chapter 1). Wild horses occupy all elevation zones from 300m above sea level to 2200m above sea level, in habitats ranging from grassland to tall open forest (pers. obs.), but they do not occur throughout all of the Australian Alps (Figure 2.1 and 2.2). There may be environments within elevational zones that are unsuitable for horses. Dyring (1990) found that wild horses made use of heaths and grasslands and tended to avoid forests. A broad-scale assessment of habitat use by wild horses in the Australian Alps is beyond the scope of the current study. Such an assessment may provide insights into the role of different habitat types on the distribution of wild horses.

Dyring (1990) reports that wild horse populations generally fluctuate in response to drought years and locally in relation to bush fires. A major event for wild horses in the rainshadow area on the eastern side of the Australian Alps was the 1982-83 drought. This killed off most of the horses in the area (Rob Jenkins, Gary Caldwell & John Rodgers pers. comms.). The kangaroo, wallaby and rabbit populations also apparently suffered massive losses. There is general agreement that the wild horse population in the area is strongly influenced by drought (John Rodgers, Gary Caldwell, Neen Pendergast, and Dave Ingram pers. comms.). There is very little water available in this environment. This is in contrast to most of the Australian Alps where water is freely available.

Large snow events have had a limiting effect on wild horse populations in high elevation areas through history in the Australian Alps. In some cases snow has limited numbers in an area, in other cases entire populations have been exterminated by snow. For example, wild horses were present in the Brindabella Range in large numbers (at least 200 horses) in 1939 (Higgins 1994). However the entire population died during the winter of 1964 (Higgins 1994). Another report of the same event suggests it occurred in 1961 (Resource and Environmental Consultant Group 1973), which coincides with the year of greatest snow depth since 1954 (Bureau of Meteorology 2002). At a similar time, about 60 wild horses died where Thredbo Village is now located (Dave Pendergast pers. comm.). In 1900, an extremely

heavy snow year was believed to have driven the local wild horse population extinct in northern Kosciuszko, dead horses were found in the forks of trees once the snow melted (John Rudd pers. comm.).

2.6. Discussion

Humans have played a fundamental role in determining the distribution of horses in the Australian Alps. The presence of horses in an area can usually be traced back to a translocation event. This is true in general for wild horse populations around the world, and within Australia. Reintroduction programs (Kirsch 2002) support the remaining population of native horses (Przewalski), while other wild populations were introduced and are typically managed by humans (for example Eberhardt *et al.* 1982, Duncan 1992, Department of Conservation 1995). The initial establishment of wild horse in Australia occurred with European settlement (McKnight 1976). The establishment of wild horses in the Australian Alps was no exception. Translocation experiments (Krebs 1994) are analogous to the large-scale movement of horses around the world by people. Horses have shown that they are adaptable to a wide range of environments (Berger 1986, Linklater 2000).

Most translocations of wild horses in the Australian Alps appear to have been successful, except when very few animals were translocated. Unfortunately only a small number of translocations were examined here, and it is likely that there are many unrecorded translocations of populations of various sizes. Nevertheless, small populations of ungulates (less than 10 animals) are less likely to establish after a translocation than larger ones (Komers & Curman 2000). The lack of success of these smaller populations may be due to stochastic events (Harrison 1991) or removal by humans.

The rate of spread of wild horses in the Australian Alps (unassisted by humans) is remarkably low (<4km per year) compared with the rates documented by Darwin (1845) (48km per year). The records of wild horses spreading in Australia all indicate a fairly slow rate of spread typically associated with settlement by pastoralists (McKnight 1976). This low rate of spread is in contrast to other species introduced to Australia such as the European rabbit (64km per year) (Caughley 1977a) and the red fox (associated with rabbits) (Saunders

et al. 1995), which have spread largely unassisted by humans after their initial introduction. The low rate of spread of wild horses in the Australian Alps should make it easier to prevent them spreading further than if they spread at a greater rate.

Wild horses have not spread to occupy all of their potential range within the Australian Alps national parks despite introductions to multiple sites and an ability to disperse. There seems to be several reasons for this, which vary across the Alps. The key factor that appears to have limited their distribution is humans. Other factors include geographical barriers and environmental events while behaviour and other species may also make a contribution. People have had a major effect on the distribution of wild horses across Australia. Of note, their range contracts as farming practices intensify (McKnight 1976), or when there are concerted control efforts (Bryan 2001). Conversely without control wild horse populations naturally increase and spread (McKnight 1976). The same patterns are evident within the Australian Alps national parks.

Barriers of dense vegetation and steep slopes appear to prevent horses from colonising new areas in the short term. The question in these cases is whether horses will eventually overcome these barriers. The effectiveness of a barrier will depend on the form of the barrier and the pressure on it from the wild horse population.

Behaviour, environmental events and other species may have acted in conjunction with humans to restrict the expansion in range of wild horses by keeping them below a threshold density. If wild horses are similar to wood bison in that they are pulse dispersers (Larter *et al.* 2000), they must reach a threshold density before they disperse and colonise a new area. If humans, other species or environmental events keep populations below this threshold density, then they won't expand in range. For example, wild dogs could limit wild horses at low densities. Predators can hold scarce populations at low levels, while at high densities the prey population may escape the limiting effect of predation (Pech *et al.* 1992). This is one possible explanation for the lack of range expansion in some areas where wild dogs are common. Populations in areas subject to drought and snow events may be kept below a

threshold density similarly, populations subject to brumby-running may also be kept below a threshold density.

The effect of heavy snow events in causing mortality of wild horses in the Australian Alps is similar to that reported for wild horses in Utah (Berger 1983, 1986) and other large mammals (Picton 1984, Solberg *et al.* 1999). Such mortality may be extreme and represent local population extinction, and hence a change in the distribution of wild horses. In the longer-term, heavy snow events may become less frequent if global warming raises mean temperatures in the Australian Alps (Whetton 1998). Such changes may facilitate the expansion of the distribution of wild horses in the Australian Alps.

There are clearly several factors that have influenced the dispersal and distribution of wild horses in the Australian Alps since they were first introduced in 1843. I have presented a brief synthesis of these factors in the context of ecological theory. It will be interesting to see the patterns of wild horse distribution in the Australian Alps in the future. It is likely to depend heavily on the approach to management. If they are not actively managed, they may disperse into new areas. Alternatively, management may result in a range reduction.

3. ESTIMATING DENSITY AND ABUNDANCE OF WILD HORSES USING AERIAL SURVEY

3.1. Introduction

Estimation of the size or density of any population of animals is essential to their effective management (Lancia *et al.* 1996). It provides a baseline from which managers can make predictions about the response of the population to a variety of potential scenarios, allowing optimal use of limited resources, and preventing unnecessary suffering of animals through ineffective control. This is particularly important for wild horses as management is closely scrutinised by the public (for example Carruthers 2000). Repeated estimates of abundance through time are imperative for assessing trends in abundance and gauging the success of any control measures (for example Pople *et al.* 1998) or the effect of environmental factors on the population (for example Berman 1991).

Aerial survey is a technique that has been used by biologists for about 60 years (Caughley & Sinclair 1994). It has been reviewed and developed since its inception (for example Caughley 1977b, Pollock & Kendall 1987, Seber 1992). Aerial survey is often the only practical way of surveying populations of large animals ranging over a large area. Despite the advantages of using aerial survey to estimate abundance and density of animals, many shortcomings of the technique have been identified. The challenge has been to improve accuracy, ensure repeatability and increase precision.

Accuracy is a measure of how close a population estimate is to true population size. Undercounting is the rule in aerial survey (Caughley 1974, Pollock & Kendall 1987) and can result in a large negative bias. Eleven studies have compared aerial survey with a different method. From 88% to less than 10% of the animals counted from the alternate method were counted from the air (Le Resche & Rausch 1974, Caughley 1977a, Short & Bayliss 1985, Short & Hone 1988). The causes of bias fall into two categories: animals that are potentially visible to observers but are not seen (perception bias), and those that

are concealed (availability bias) (Marsh & Sinclair 1989). The factors that have been identified as influencing bias in aerial survey include: species being surveyed, vegetation cover, survey specifications (height above ground, speed, strip width), weather conditions and observer experience (for example Caughley *et al.* 1976, Bayliss & Giles 1985, Cairns 1999).

Repeatability is the constancy of bias between samples (Pople 1999). A monitoring technique is repeatable if any proportional difference in population size between two areas or times is reflected in the estimate derived by that technique. Factors that affect repeatability in aerial surveys are survey specifications, environmental variables (for example temperature, cloud cover) and behaviour of the species (often related to environmental factors such as time of day) (Pople 1999).

Precision is the variability in repeated measures of the same population. It can be expressed as the coefficient of variation (CV). CV is an expression of the standard error as a proportion of the estimate. Both the variation in the technique (repeatability) and the variation arising from sampling will determine precision. Estimates are more precise when animals are at higher densities (Southwell 1989) and when the distribution of animals is homogenous relative to sampling (Caughley & Sinclair 1994). Precision can be improved by increasing the area surveyed or by increasing sampling intensity (Caughley *et al.* 1977). There is a wide range of CVs reported in the literature. For example CVs for kangaroos range from as low as 5% (Caughley *et al.* 1977) to greater than 50% (for example Short & Hone 1988). Some of the lower CVs in the literature are inaccurate, underestimating the variability with incorrect equations (Barnes *et al.* 1986, Pople 1999).

Initially aerial surveys attempted total population counts with only limited success (Goddard 1967). Total counts rely on the assumptions that no animals are counted twice and no animals are missed (Krebs 1999 p. 147). Aerial surveys were soon modified as a form of sampling (Jolly 1969, Caughley 1977c). The most common form of sampling is by transects. It has several advantages over the other forms: sampling blocks or quadrats

(Krebs 1999). Transect sampling returns higher estimates and precision than quadrat counts (Teer *et al.* 1985), and is more cost effective because it involves less time flying (Krebs 1999). Transect surveys have been refined over time using a variety of approaches primarily to reduce negative bias but also to improve precision. The methods fall into three categories: strip, mark-recapture and line transect.

Aerial strip surveys have been used extensively in Australia for kangaroo management and have undergone a constant process of review (Southwell 1989, Cairns 1999, Pople 1999). Negative bias has been identified as a major issue with estimates from 64% to less than 10% of ground counts (Short & Bayliss 1985). More accurate results can be achieved using narrower strip widths but with a compromise of precision (Hone 1988). Several studies have developed correction factors to improve accuracy of strip estimates. Correction factors have been established in a variety of ways (for example Caughley *et al.* 1976, Short & Bayliss 1985, DeYoung *et al.* 1989). Critics of the use of correction factors show that estimates are less precise than indicated (Barnes *et al.* 1986). The accuracy of correction factors has also been shown to vary between species (Short & Hone 1988).

Mark-recapture models have been applied to aerial surveys to correct for perception bias which is an issue of animals potentially visible but not seen (Marsh & Sinclair 1989). Bartmann *et al.* (1987) tested the efficacy of mark-recapture models in estimating mule deer populations. They found that, in general, precision is improved and estimates are more accurate than straight counts. However, there is still a high probability that mean population estimates will be low. A mark-recapture model (double-count) was adapted for counting emus (*Dromaeus novaehollandiae*) from the air and found that it increased estimates by 32% (Caughley & Grice 1982). However, there is still a high probability that mean population estimates will be low. Mark-recapture models have subsequently been used for aerial surveys of feral livestock (including horses) (Bayliss & Yeomans 1989a, Graham & Bell 1989, Pople *et al.* 1998, Black 2000). The key assumptions are independence of mark and recapture events and equal catchability (Caughley & Grice

1982). Violation of these assumptions leads to an underestimate of abundance (Caughley & Grice 1982).

Line transect methods also reduce the negative bias associated with aerial surveys (Buckland *et al.* 1993). The first evaluation of aerial line transect surveys, which was on bottlenose dolphins (*Tursiops truncatus*) (Leatherwood *et al.* 1978), was found to be unfeasible. However, subsequent surveys have shown more promise. Hone (1988) found that line transect estimators accurately estimated the density of a known number of feral pig carcasses and had higher precision than strip count estimates. An evaluation of aerial line transects for estimating mule deer densities returned accurate estimates with relatively good precision (White *et al.* 1989). It was also successfully used to estimate pronghorn density (*Antilocapra americana*) in Wyoming (Johnson *et al.* 1991). However line transect estimates of feral water buffalo (*Bubalus bubalis*) in woodland and forest habitat underestimated abundance by about half (Skeat 1990). Line transect estimates are based on the assumptions that objects on the line are always detected, objects are detected at their initial location, animals are counted only once and distances are measured accurately (Buckland *et al.* 1993). Failure of the first assumption leads to negative bias (White *et al.* 1989, Skeat 1990).

Aerial surveys of wild horses have estimated population sizes by total counts, strip counts and mark-recapture methods. In the USA and New Zealand, typically total counts are used (USA: Frei *et al.* 1979, Garrott *et al.* 1991a, New Zealand: Rogers 1991, Linklater & Cameron 2002). In the USA, estimates are considered to be fairly accurate (Garrott *et al.* 1991a) though they have not been rigorously evaluated, and in New Zealand populations are probably overestimated due to counting animals more than once (Linklater & Cameron 2002). In Australia, aerial surveys of horses have been forms of sampling. These include mark-recapture estimates (Graham & Bell 1989, Black 2000), mark-recapture with habitat specific correction factors (Bayliss & Yeomans 1989b) and strip counts with correction factors (Skeat 1990). In New Zealand the average density was 3.6 horses km⁻² (Cameron *et al.* 2001). All of the other horse studies gave population density estimates below 1 horse km⁻².

Expected densities of horse populations can be predicted based on body mass. There is an inverse relationship between the body mass of a mammalian species and its population density (Damuth 1981, Freeland 1990). The expected density of feral horses in the Alps based on these equations assuming a mass between 200 and 550kg (range between a 12h horse in very poor condition to a fat 15h horse (Huntington 1991)) is 1.9-0.8 horses km⁻². However, Freeland (1990) has found that large mammals introduced into new environments consistently occur at higher densities than predicted by the equation of Damuth (1981). Densities in the order of 16 to 40 times the densities recorded in native habitats were observed and may be due to the lack of predators and pathogens in the new environments (Freeland 1990).

The aim of this chapter is to compare strip counts, mark-recapture and line transect techniques for estimating the abundance and population density of wild horses in the Australian Alps national parks.

3.2. Methods

3.2.1. Survey area

The areas selected for survey were based on the distribution maps from Chapter 2. All areas where horses are known to occur within the boundaries of the Australian Alps national parks were surveyed except Talbingo and Byadbo in Kosciuszko National Park. Talbingo and Byadbo were not surveyed because the horses are in very low densities and the terrain is very rugged. The regions surveyed are summarised in Table 3.1.

Table 3.1: Summary of regions surveyed during the wild horse aerial survey of 2001.

Region	Area (km ²)	Number of Transects	Total transect length (km)	Approximate cost (\$)
North Kosciuszko	795	25	412	4000
Snowy Plain	77	6	42	700
South Kosciuszko	758	22	379	6600
North Victoria	1069	30	587	9100
Bogong High Plains	90	8	54	1000
TOTAL	2789	91	1474	21400

3.2.2. Survey design

A pilot survey conducted in April 2000 to test equipment and techniques, resulted in minor survey modifications. The full-scale aerial survey was carried out in late summer and autumn between 16-21 February and 13-14 March 2001 in a Bell Jet Ranger helicopter with the doors removed to improve visibility. This period was chosen because temperatures are generally below 15°C so horses are less likely to be sheltering under trees, but it is not too cold to fly with the doors off. Helicopter are preferable to fixed-wing aircraft for aerial survey because results are considerably more precise and less biased, however the cost is greater by a factor of 4 (Clancy 1999).

We flew parallel east-west transects following every odd gridline on a topographic map of 1:100 000 scale. The start point of the first transect was selected by random and the sampling systematic without replacement. The areas to be surveyed were overlaid on the maps and used for navigation during flight using an onboard computer (AeroNavitraker) linked to a differential GPS. The average transect length was 16km (range 2-33km). Transects were flown 2km apart to optimise sampling intensity and minimise the likelihood that horses were recounted on consecutive transects. In New Zealand,

Linklater and Cameron (2002) found that horses had an average linear flight distance of 1km when flushed by a helicopter. Given that the New Zealand survey was done in more open country and a lower survey height (60m) than the current survey, it is unlikely that we counted the same group more than once.

The aircraft was flown at 100km h^{-1} and 100m above ground level (with the assistance of a radar altimeter). These were held as close to constant as possible given the uneven terrain. One observer was positioned in the front seat (M. Walter) and a second in the rear (J. Hone) with both being on the left-hand side of the aircraft. The pilot sat on the right side. Observations were recorded independently and without collusion (to enable mark-recapture analysis). Observers did not tell each other of sightings or point to any horse groups during transects. A strip 200m wide was surveyed giving a sampling intensity of approximately 10%. This strip was broken down into five intervals (described below) so the data could be analysed using line transect methods.

The area counted was delineated using a bar attached to the underside of the helicopter and stabilised with struts (Figure 3.1). To calibrate the bar and strip width classes, cloth markers $2\text{m}\times 2\text{m}$ were placed on the ground at 50 metre intervals, the helicopter was then flown over the markers at survey height and marks were made on the bar delineating 50, 100, 150 and 200m intervals delineating four intervals. The first interval was further broken down to 0-15 and 15-50m using the skid as a divider delineating 5 intervals. Smaller intervals were not possible because groups are often spread out over about 50m. The general procedure was similar to the methods described by Clarcy (1999).



Figure 3.1: A Bell Jet Ranger helicopter was used to conduct aerial surveys. The doors were removed and a bar stabilised with struts was attached to the helicopter.

The survey took a total of 30 hours over six days and was carried out between 0900 and 1700 hours. Survey breaks were taken during transit, refueling and lunch. Observers also called for breaks if they were tired or cold. At the start of each session, the date, time and weather conditions were recorded. The weather was generally cool (average 11°C), sunny and winds were calm. Animal behaviour can affect the repeatability of some aerial surveys. Kangaroo behaviour varies with time of day and affects the accuracy of counts (Short & Hone 1988), but this is not the case for all animals. Samuel *et al.* (1987) showed that elk (*Cervus elaphus*) behaviour only appears to affect detection because it is correlated with vegetation and group size. Studies of horses suggests that horse behaviour doesn't alter noticeably with time of day (personal observation, Dyring 1990, Black 2000).

Both observers had stopwatches, which were synchronised at the start of each transect. When a group of horses were observed number of animals in the group, time, distance

interval and vegetation cover was recorded in a notebook. The time was later used to assess whether one or both observers saw the group. The groups were considered to be the same if times were within 10 seconds of one another. Vegetation cover was broken into three classes: open, woodland/forest and edge (of woodland/forest and open). The edge class was included because horses were often seen on the woodland/clearing edge during ground based surveys. A group was defined as a set of individuals in close proximity to each other (<25m diameter) and behaving as one unit.

3.2.3. *Analysis*

Mean group size and standard error were calculated from three data sources: observer 1, observer 2 and from ground surveys by observer 1 in autumn (March-May 2001) in the same survey area. Ground survey estimates were assumed to be accurate. The methods for the ground survey are described in Chapter 4. In brief, data were obtained during 18 days of intensive area counts of study sites. In cases where group size counts for the same group differed between observers, it was assumed that the higher estimate was correct because it is likely that one of the observers failed to see all of the animals. Mean group size was calculated for each observer over each distance interval. Difference in group size based on distance interval was tested for each observer and both observers combined using one way analysis of variance (ANOVA).

Estimates of density and number of horses and groups of horses were made using three different techniques and their merits were assessed. A summary of these analyses is shown in Table 3.2.

Table 3.2: Summary of analyses used to estimate density and number of horses and groups. Analyses are broken down based on Observers. Observer 1 - M. Walter, Observer 2 - J. Hone, both = both observers.

ANALYSES	Observer 1		Observer 2		Both	
	Horses	Groups	Horses	Groups	Horses	Groups
Strip Estimate						
0-50m	Y	Y	Y	Y		
50-100m	Y	Y	Y	Y		
100-150m	Y	Y	Y	Y		
150-200m	Y	Y	Y	Y		
0-200m	Y	Y	Y	Y		
Mark-Recapture Estimate						
0-50m					Y	Y
0-200m					Y	Y
Line Transect Estimate						
4 intervals		Y		Y		Y
5 intervals		Y		Y		Y

Estimates of abundance and density of horses and groups for strip counts (uncorrected for any bias) were made using the ratio method because transects were unequal lengths (Caughley & Sinclair 1994, p. 202). Standard errors were calculated assuming sampling without replacement (Caughley & Sinclair 1994, p. 202) and were based on variation between transects. An estimate of the abundance and density of horses was also made by the product of numbers of groups and mean group size. The variance of this estimate was the exact variance of a product (Leatherwood *et al.* 1978). It was expected that estimates from strip counts would be negatively biased, based on results in the literature (for example Caughley 1974, Pollock & Kendall 1987).

Mark-recapture estimates the total number of animals based on the number seen by only one of each of the observers and both observers. The total number of horses and groups were estimated within each transect using the modified Petersen estimator (Chapman 1951 in Caughley & Sinclair 1994, p. 213) for the 0-50m strip width, and separately for the 0-200m strip width. The ratio method was then used to calculate estimates of

abundance and density using the mark-recapture estimates for all transects. Standard errors were calculated assuming sampling without replacement (Caughley & Sinclair 1994, p. 202) and were based on variation between transects. A similar sequence of calculations occurred for horse groups. All calculations for strip and mark-recapture analyses were made in EXCEL. Where observers put the same group in different distance intervals (on 5 occasions), data were designated to the interval recorded by observer 1 and observer 2 alternatively.

Line transect data were analysed using program DISTANCE 3.5 (Buckland *et al.* 1993). Data collected by each observer were analysed independently and then combined as interval data. When observers recorded a group over 2 interval classes, they were entered into the interval class with the most horses. This was done for data separated into four and five interval categories. The estimator adjusted for the presence of observers on only one side of the aircraft. Four detection functions were fitted to the data: half normal, hazard rate, uniform and negative exponential, and cosine, simple polynomial and hermite polynomial series expansions applied. The best functions were selected as those with the lowest value of the Akaike's Information Criteria (AIC) (Buckland *et al.* 1993). AIC provides a quantitative method for model selection with an optimisation rather than hypothesis-testing framework. It attempts to identify a model that fits the data well and does not have too many parameters (principle of parsimony). The expected value of group size was computed in DISTANCE by regression of $\log(s(i))$ on $g(x(i))$ where $s(i)$ is group size of the i th observation and $x(i)$ is distance to the i th observation. The expected value of group size was used in analyses when it was closer to mean group size calculated from ground counts than mean observed group size.

Attempts were made to estimate abundance by combining the mark-recapture and line transect analyses, such as described by Borchers *et al.* (1998), however the attempts were unsuccessful. Such combinations could be a useful future development of the survey method. The analyses used in the present study did not explicitly attempt to correct for the effects of vegetation because of time constraints.

3.2.4. *Optimising sampling intensity and precision*

The relationship between the coefficient of variation and sampling intensity was explored as a means of assessing the optimum trade-off between the two. The relationship between estimated sampling intensity given the results from the current study, is denoted (Caughley *et al.* 1977) by

$$\hat{SI} = SI_{used} \times \frac{s^2}{(\hat{N} \times CV)^2}$$

Where SI = sampling intensity, s^2 = variance, \hat{N} = estimated number of animals for the current study and CV is the desired coefficient of variation.

3.3. Results

3.3.1. *Group size*

Observer 1 saw 34 groups from the air with a mean group size of 4.91+/- 0.61SE, while observer 2 saw 42 groups with mean group size of 3.79+/- 0.57SE. Group sizes observed from the ground were 5.65+/-0.51SE for 34 groups. Therefore observer 1's estimated mean group size was 86.9% of ground estimates and observer 2's was 67.1%. Observer 1's estimates were not significantly different from ground counts ($t = -0.93$, d.f. = 66, $p > 0.05$) but observer 2's estimates were significantly different ($t = -2.35$, d.f. = 74, $p < 0.05$). The difference in estimated group size between observers is a result of observer 1 seeing more horses per group and observer 2 seeing more singletons and pairs missed by observer 1 (Figure 3.2).

Mean group size was higher in the 0-50m strip for both observers than all other distance intervals (Figure 3.3). Mean group size in the 0-50m strip was 6.1 for observer 1 and 4.6 for observer 2. However there was no significant difference in group size between distance intervals for observer 1 (single factor ANOVA, $F=1.41$, d.f. = 3, 30, $p > 0.05$), observer 2 (single factor ANOVA, $F=0.70$, n = 3, 38, $p > 0.05$) or both observers combined ($F=0.98$, d.f. = 3, 48, $p > 0.05$).

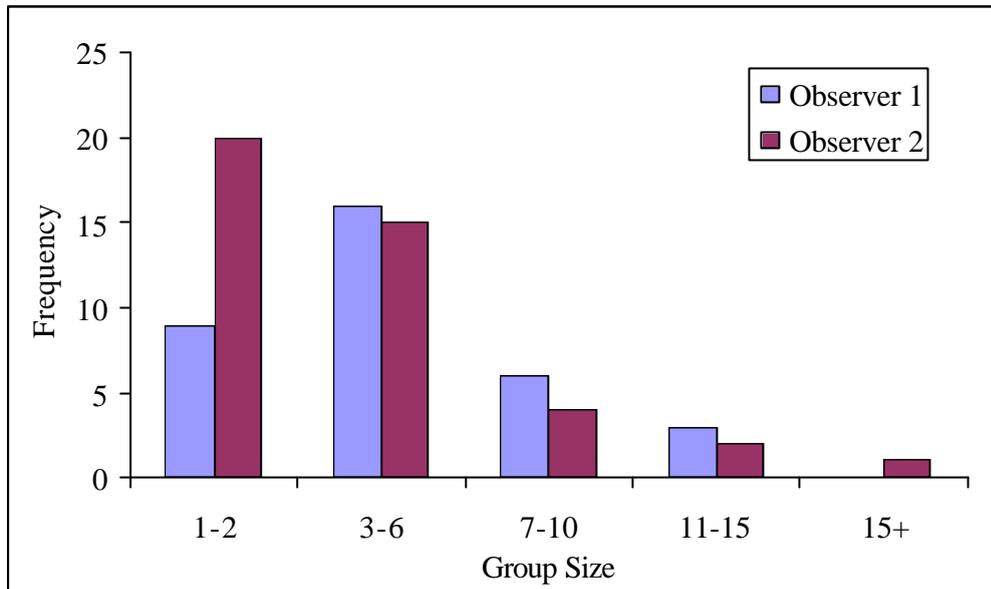


Figure 3.2: Frequency of observations of different sized groups of wild horses for observer 1 and observer 2.

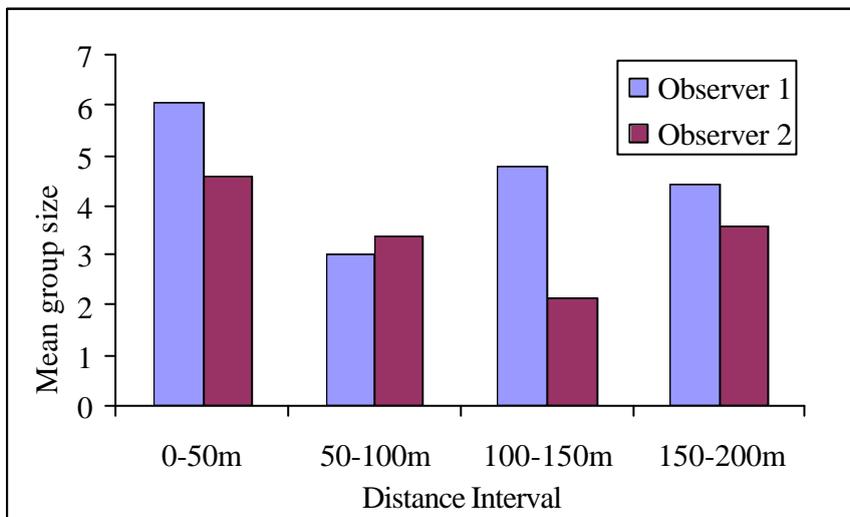


Figure 3.3: Mean group size of wild horses estimated for both observers for each distance interval.

Observed group sizes in the three main survey areas, North Kosciuszko, South Kosciuszko and North Victoria were calculated and showed an interesting result. For North Kosciuszko, mean observed group size was $6.38 \pm 1.23SE$ ($n=16$) while South

Kosciuszko and North Victoria were $3.00 \pm 0.57SE$ ($n=22$) and $3.00 \pm 0.45SE$ ($n=11$) respectively. The northern Kosciuszko area is open with extensive plains, while south Kosciuszko and North Victoria are more heavily timbered with fewer, smaller clearings.

3.3.2. *Vegetation cover*

Most groups were observed in woodland/forest (57%) (Figure 3.4). Observer 2 saw more groups in woodland/forest than observer 1. Similar numbers of groups were seen in edge (22%) and open habitats (21%).

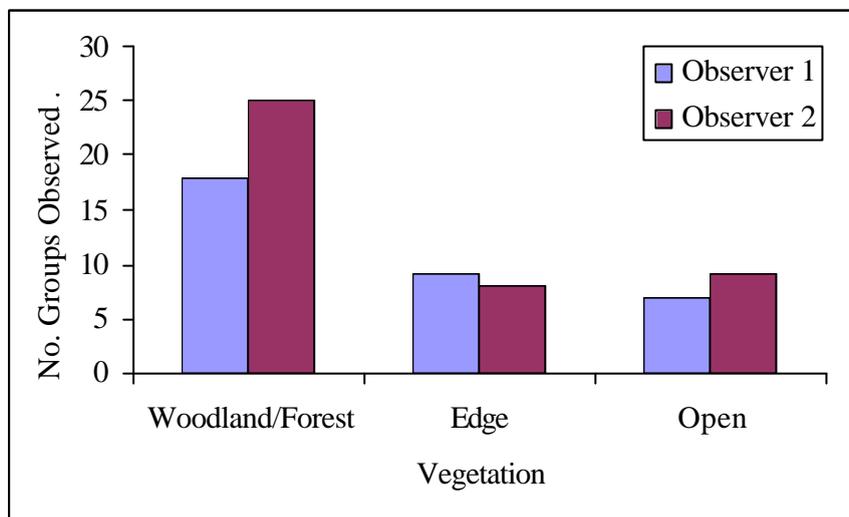


Figure 3.4: Numbers of groups of wild horses observed by each observer in different vegetation classes.

3.3.3. *Strip count estimates*

The accuracy of population estimates based on strip counts was strongly influenced by strip width. Abundance estimates based on individual horses for both observers were around 3500 in the 0-50m strip (Table 3.3). Estimates over the 0-200m strip were strongly negatively biased, being only 43 and 44% of 50m strip estimates for observer 1 and 2 respectively. The effect was even greater for group estimates (Table 3.4). Estimates for groups in the 0-200m strip were only 16 and 23% of the 50m strip for observer 1 and 2 respectively. Mean population estimates for 0-50m strips based on both horses and

group counts ranged from 3419 to 4271 individuals (Table 3.3, 3.4). This equates to densities ranging between 1.2-1.5 horses km² over the area surveyed.

Table 3.3: Estimates of wild horse density (km²) and abundance (N) derived from strip counts of individual horses for both observers. Percent coefficient of variation (CV) and 95% confidence intervals (95% CI) are presented for estimates.

Strip (m)	Density	CV	95% CI	N	95% CI
Observer 1					
0-50	1.32	25.0	0.67-1.96	3670	1871-5469
0-200	0.57	29.0	0.25-0.89	1579	683-2475
Observer 2					
0-50	1.24	22.3	0.70-1.77	3444	1941-4948
0-200	0.54	25.8	0.27-0.81	1503	743-2263

Table 3.4: Estimates of density of groups (km²) of wild horses, density of wild horses and abundance of wild horses derived from strip counts for both observers and mean group size from ground surveys. Percent coefficient of variation (CV) and 95% confidence intervals (95% CI) are presented for estimates.

Strip (m)	Group Density	CV	95% CI	Horse Density	CV	95% CI	No. Horses	95% CI
Observer 1								
0-50	0.22	26.3	0.11-0.33	1.23	57.0	0-2.59	3419	0-7237
0-200	0.12	94.1	0.01-0.23	0.65	95.9	0.03-1.27	1814	84-3542
Observer 2								
0-50	0.27	39.9	0.06-0.48	1.53	62.4	0-3.40	4271	0-9499
0-200	0.14	95.1	0.01-0.27	0.80	95.9	0.03-1.57	2242	84-4379

The estimated numbers of horses per strip dropped off dramatically after 50m (Figure 3.5). Estimates in the 50-100m, 100-150m and 150-200m strips were only 24.7, 24.8 and 22.7% of the 0-50m strip respectively for observer 1. Similarly, observer 2's estimates were 40.6, 14.3 and 19.8% of the 0-50m strip respectively. The reduction in the number of groups observed with distance from the aircraft was more gradual with estimates for the 50-100m, 100-150m and 150-200m strips being 50.2, 31.3 and 31.3% respectively of the 0-50m strip for observer 1 and 55.0, 29.9 and 25.1% respectively for observer 2 (Figure 3.6).

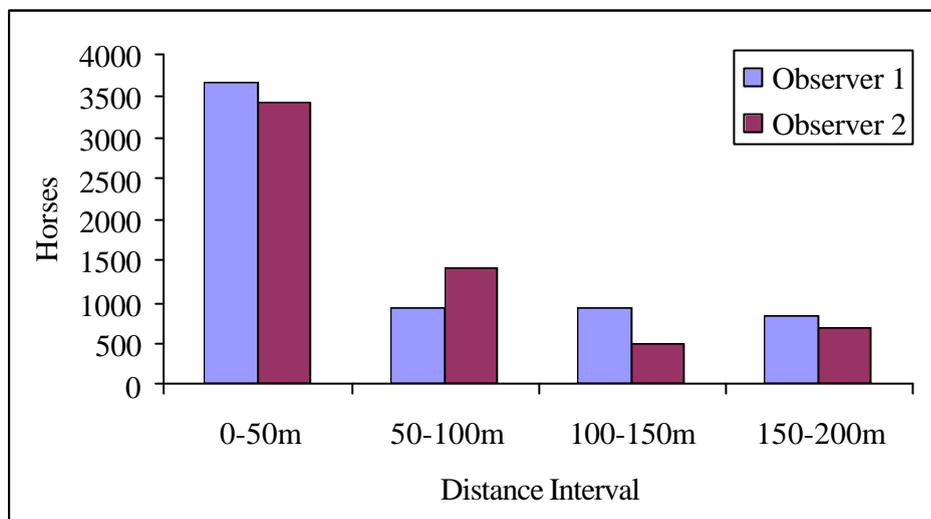


Figure 3.5: Abundance estimates from strip counts of wild horses per distance interval for both observers.

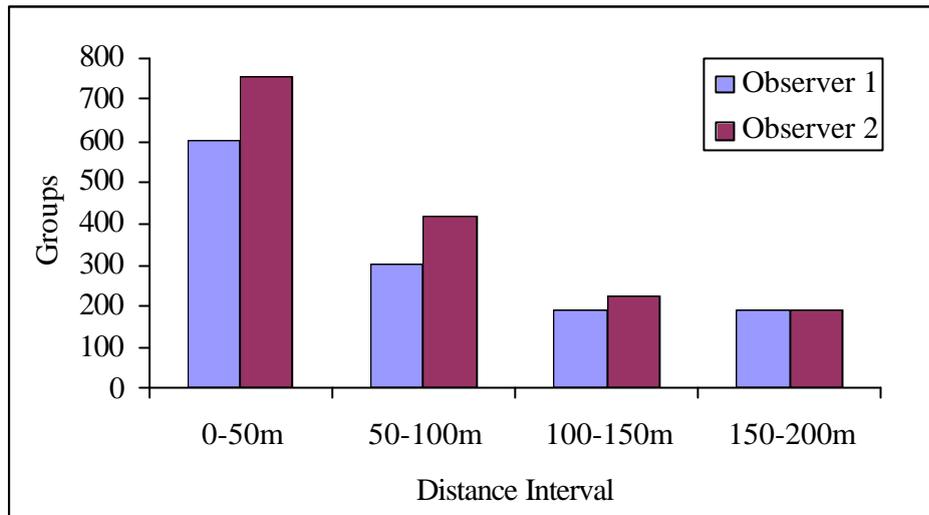


Figure 3.6: Abundance estimates from strip counts of wild horse groups per strip for both observers.

The precision of strip estimates based on individual horses was greater than those based on group counts. All estimates based on horse counts had CVs below 30% (Table 3.3). The CV of estimates of groups over 0-200m was very high at approximately 95% for both observers (Table 3.4). The CV for 0-50m strip estimates of groups was lower at 26.3 and 39.9% for observer 1 and 2 respectively. The low precision for group estimates is compounded when converted to estimates of horse density and abundance due to the variation associated with group size (Table 3.4).

3.3.4. *Mark-recapture estimates*

Density and abundance estimates using the mark-recapture technique (Tables 3.5 & 3.6) were higher than the strip estimates for both 0-50m and 0-200m strip classes (Tables 3.3 & 3.4). The greatest increase in estimates was evident for groups over 0-200m where mark-recapture estimates were 3 to 5 times greater than the corresponding strip estimates for independent observers (Tables 3.3 & 3.4). The effect was not as dramatic over the 0-50m strip for group counts with increases between 15 and 49%. The 0-50m group based estimate (Table 3.6) was the highest (4915) of all the mark-recapture estimates; the lowest estimate (2061) was based on horses observed over 0-200m (Table 3.5). Mark-

recapture estimates based on individuals in the 0-50m strip were 18-26% higher than the corresponding strip estimates for independent observers. In the 0-200m strip the estimates were 31-37% higher.

Table 3.5: Estimates of density (km^{-2}) and abundance (N) of wild horses derived from mark-recapture analyses. Percent coefficient of variation (CV) and 95% confidence intervals (95% CI) are presented for each estimate.

Strip (m)	Density	CV	95% CI	N	95% CI
0-50	1.56	28.3	0.69-2.42	4345	1936-6755
0-200	0.74	17.9	0.48-1.00	2061	1339-2783

Table 3.6: Estimates of wild horse density (km^{-2}) and abundance (N) derived from mark-recapture analyses on individual horses and groups. Percent coefficient of variation (CV) and 95% confidence intervals (95% CI) are presented for each estimate.

Strip (m)	Density Groups	CV	95% CI	Density Horses	CV	95% CI	No. Horses	95% CI
0-50	0.31	21.8	0.18-0.45	1.76	55.6	0-3.68	4915	0-10268
0-200	0.19	18.1	0.12-0.26	1.09	54.7	0-2.26	3045	0-6308

As with the strip counts, the mark-recapture estimates were much higher for the 0-50m strip analysis than the 0-200m analysis (Table 3.5, 3.6). Estimates of individuals in the 0-200m strip were 47.4% of 0-50m estimates, and 61.9% for groups. Estimates of horse numbers were highest when analysis was based on groups and multiplied by group size (Table 3.6) rather than analysis based just on horses (Tables 3.5). In the 0-50m strip, the group based population estimate was 13% higher than the horse based estimates.

Mark-recapture analysis shows that strip estimates for each observer were negatively biased. None of the 95% confidence intervals of the sighting probabilities include 1.00 for individuals or groups. Given the data, the probability of the first observer sighting an individual in the 0-50m strip was 0.79 \pm 0.04SE, and the second observer 0.76 \pm

0.04SE. In the 0-200m strip, the probability of the first observer sighting a horse was $0.74 \pm 0.03\text{SE}$ and the second $0.71 \pm 0.04\text{SE}$. The probability of the first observer sighting a group in the 0-50m strip was $0.58 \pm 0.13\text{SE}$ and the second observer $0.73 \pm 0.10\text{SE}$. In the 0-200m strip the probability of the first observer sighting a group of horses was $0.57 \pm 0.08\text{SE}$ and the second observer $0.71 \pm 0.07\text{SE}$.

Precision was high for estimates in the 0-200m strip at 17.9% CV (Table 3.5). Group density estimates had relatively high precision. However, population estimates based on groups by group size had lower precision with CVs of 55.6% over 0-50m strip and 54.7% over 0-200m (Table 3.6).

3.3.5. *Line transect estimates*

The negative exponential model had the best fit to the data for all analyses (Table 3.7, 3.8 and 3.9). This was because of the spiked distribution of sightings (Figure 3.7). The second best models were not consistent. The hazard rate and uniform models ranked second for different analyses. The uniform models tended to fit the data poorly, but were ranked high because they had high precision. The hazard rate models tended to fit the data, but had lower precision because they used more parameters. The half normal models tended to fit the data poorly and have relatively low precision. In most cases the key functions gave the best fit. The cosine series expansion gave the best fit in several cases for the half normal and uniform functions.

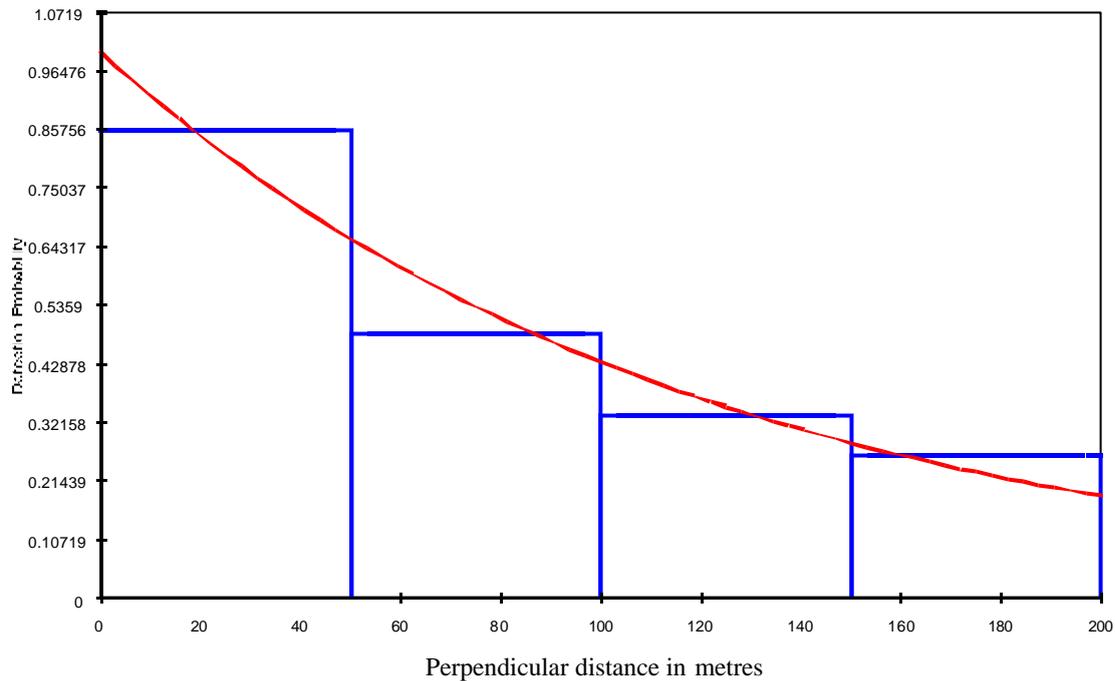


Figure 3.7: Negative exponential model fitted to line transect data of wild horses for both observers combined.

Separating data into 4 and 5 distance intervals did not have a significant effect on the results using the negative exponential model, as evident from the overlap of the 95% confidence intervals. However, the data in 5 intervals gave a slightly better fit for negative exponential models for each observer while the 4 intervals gave a better fit for observers combined. For observer 1, the probability (p) of a greater chi square value (used as a measure of goodness of fit) was 0.66 for 4 intervals and 0.70 for 5 intervals. For observer 2, p was 0.84 for 4 intervals and 0.91 for 5 intervals. For both observers combined, p was 0.86 for 4 intervals and 0.55 for 5 intervals.

Expected mean group size estimates were made in DISTANCE 3.5 by regressing \log (observed mean group size) against the detection function. In all analyses, the estimates were greater than the observed group size. However, the correlation was low (for example 0.15 for 4 interval negative exponential model for both observers combined). Observer 1's observed mean group size was closer to mean group size estimated from ground surveys than expected mean group size for all models except the uniform models.

Therefore observed mean group size was used in all population estimates in all cases except for the uniform models for observer 1. For observer 2 and both observers combined, the expected mean group size was closer to the mean group size from ground estimates in all cases, so it was used in all analyses instead of observed mean group size. The analyses for both observers combined gave the expected group size closest to that estimated from ground counts. Observer 1's estimates were higher and observer 2's were lower than ground estimates. Estimates of group density for the best models range from 0.25-0.38 km⁻² across all line transect analyses. This equates to a horse density range of 1.41-2.15 km⁻² based on mean group size from ground counts. The only strip count estimate that falls within this range is for observer 2's group based estimates over 0-50m (1.53 horses km⁻²) (Table 3.4). The mark-recapture estimates that fall in this range are 0-50m horse and group based estimates (1.56 horses km⁻² based on horses, 1.76 horses km⁻² based on groups) (Tables 3.5 & 3.6).

3.3.6. Reducing the coefficient of variation by increasing sampling intensity

The CVs in the analyses vary from 17.9-96.1%. The analyses which were most accurate and precise were the mark-recapture 0-50m based on individual horses (CV=28.3%), and the line transect analysis using 4 intervals and the negative exponential model for both observers combined (CV=28.1%). The relationship between sampling intensity and the coefficient of variation were assessed for these two analyses.

Figure 3.8 shows that to achieve progressively smaller CVs, sampling intensity increases exponentially. Also, for a given desired CV, mark-recapture analysis requires a lower sampling intensity (Figure 3.7) For the line transect analysis, a CV of <10% is not achievable. For the 0-50m mark-recapture analysis a 10% CV is possible but not feasible. It would require a sampling intensity of 0.20, an eight fold increase on the current sampling intensity of 0.025. That would have line transects spaced 244m apart which is considered too close to avoid double counting of horses. A CV of 20% is more practical but would come at the cost of doubling the sampling intensity of the survey. A CV of 20% could also be expected for line transect analysis if sampling intensity was doubled.

Table 3.7: Estimates of density of groups of horses and estimates of density and abundance of horses (km²) derived from line transect analysis (using program DISTANCE 3.5) for observer 1 using data in 4 and 5 intervals. The percent coefficient of variation (CV) and 95% confidence intervals (95% CI) are presented for estimates. Delta Akaike Information Criteria (Δ AIC) values are shown. The goodness of fit of each detection function (probability of a greater chi square value (P)) is given. Expected group size is presented for each model ($s(i)$ (SE)).

Model	Δ AIC	P	$s(i)$ (SE)	Groups of Horses			Horses				
				Density	CV	95% CI	Density	CV	95% CI	N	95% C.I.
4 Intervals											
Negative Exp.	0.00	0.66	7.34 (1.09)	0.25	30.0	0.14-0.44	1.22	32.5	0.65-2.28	3393	1809-6362
Uniform (cosine)	1.07	0.39	6.03 (0.89)*	0.18	22.7	0.12-0.28	1.10	27.1	0.65-1.86	3073	1817-5199
Half Normal (cosine)	1.24	0.79	7.04 (1.01)	0.24	29.2	0.14-0.43	1.20	31.7	0.65-2.22	3342	1807-6180
Hazard Rate	1.43	0.61	7.37 (1.07)	0.27	80.2	0.06-1.11	1.31	81.2	0.31-5.54	3652	864-15441
5 Intervals											
Negative Exp.	0.00	0.70	7.38 (1.09)	0.25	29.6	0.14-0.45	1.25	32.1	0.67-2.33	3485	1870-6493
Hazard Rate	1.15	0.77	7.59 (1.11)	0.30	48.5	0.12-0.75	1.46	50.0	0.56-3.76	4061	1571-10500
Half Normal (cosine)	1.40	0.67	7.04 (1.01)	0.25	28.9	0.14-0.43	1.21	31.4	0.66-2.23	3388	1843-6229
Uniform (cosine)	1.68	0.34	6.03 (0.89)*	0.18	22.7	0.12-0.29	1.10	27.0	0.65-1.87	3082	1823-5210

*Expected group size used in density and abundance estimates

Table 3.8: Estimates of density of groups of horses and estimates of density and abundance of horses (km²) derived from line transect analysis (using program DISTANCE 3.5) for observer 2 using data in 4 and 5 intervals. See Table 3.7 for explanation of nomenclature.

Model	Δ AIC	<i>P</i>	<i>s(i)</i> (SE)	Groups of Horses			Horses				
				Density	CV	95% CI	Density	CV	95% CI	N	95% C.I.
4 Intervals											
Negative Exp.	0.00	0.84	4.77 (0.76)*	0.33	26.3	0.20-0.56	1.60	30.8	0.88-2.89	4453	2457-8073
Uniform (cosine)	1.22	0.46	4.41 (0.67)*	0.24	19.4	0.16-0.35	1.05	24.7	0.54-1.56	2936	1515-4357
Half Normal	1.46	0.40	4.41 (0.67)*	0.24	21.0	0.16-0.37	1.07	26.0	0.65-1.78	2986	1801-4950
Hazard Rate	1.74	0.77	4.59 (0.72)*	0.32	53.7	0.12-0.88	1.46	56.0	0.51-4.15	4071	1431-11585
5 Intervals											
Negative Exp.	0.00	0.91	4.78 (0.76)*	0.34	26.1	0.20-0.56	1.62	30.5	0.90-2.92	4516	2502-8152
Hazard Rate	1.64	0.93	4.55 (0.73)*	0.36	43.4	0.16-0.83	1.65	46.2	0.69-3.97	4605	1914-11081
Uniform (cosine)	1.76	0.49	4.41 (0.67)*	0.24	19.4	0.16-0.35	1.05	24.7	0.65-1.70	2940	1819-4753
Half Normal (cosine)	1.87	0.82	4.60 (0.44)*	0.32	25.4	0.19-0.52	1.46	29.8	0.82-2.59	4058	2278-7229

*Expected group size used in density and abundance estimates

Table 3.9: Estimates of density of groups of horses and estimates of density and abundance of horses derived from line transect analysis (using program DISTANCE 3.5) for both observers combined using data in 4 and 5 intervals. See Table 3.7 for explanation of nomenclature.

Model	Δ AIC	P	s(i) (SE)	Groups of Horses			Horses				
				density	CV	95% CI	Density	CV	95% CI	N	95% C.I.
4 Intervals											
Negative Exp.	0.00	0.86	4.93 (0.62)*	0.36	24.8	0.22-0.59	1.80	28.1	1.04-3.10	5010	2907-8634
Uniform (cosine)	0.95	0.53	4.53 (0.59)*	0.28	18.8	0.19-0.40	1.26	22.8	0.81-1.97	3525	2259-5500
Half Normal	1.35	0.44	4.51 (0.58)*	0.28	19.9	0.19-0.41	1.25	23.7	0.79-1.98	3477	2191-5516
Hazard Rate	1.70	0.89	4.85 (0.64)*	0.36	55.6	0.13-1.03	1.77	57.2	0.61-5.11	4923	1702-14238
5 Intervals											
Negative Exp.	0.00	0.55	4.95 (0.65)*	0.38	24.4	0.23-0.61	1.87	27.8	1.09-3.20	5216	3043-8938
Hazard Rate	1.32	0.53	4.89 (0.65)*	0.43	38.9	0.20-0.91	2.11	41.1	0.96-4.63	5884	2684-12901
Half Normal (cosine)	2.06	0.35	4.86 (0.64)*	0.36	24.2	0.23-0.58	1.76	27.5	1.03-2.99	4898	2873-8350
Uniform (cosine)	2.09	0.19	4.53 (0.59)*	0.28	18.8	0.19-0.40	1.27	22.8	0.81-1.98	3538	2269-5516
*Expected	group	size	used	in	density	and	abundance	estimates			

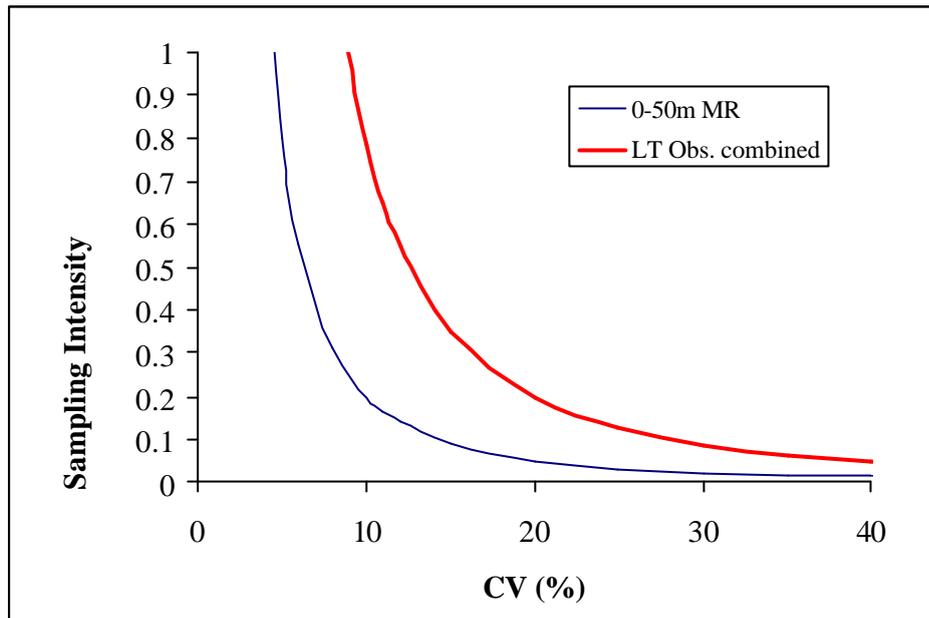


Figure 3.8: The relationship between sampling intensity and the coefficient of variation for 0-50m mark-recapture analysis and line transect analysis for both observers combined 0-200m (4 intervals, negative exponential model).

3.4. Discussion

3.4.1. *Abundance and density of horses in the Australian Alps national parks*

The best estimate of abundance of wild horses in the Australian Alps national parks from the current study is 5010 horses based on the line transect analysis for both observers combined. This corresponds to a density of 1.8 horses km^{-2} in the areas surveyed. The total number of horses in the AAnp is likely to be higher than this because not all areas were surveyed including the Byadbo and Talbingo areas. It is also possible that the estimate is low because availability bias (Marsh & Sinclair 1989) remains unquantified. The density of horses across the entire AAnps is lower than 1.8km^{-2} because the entire area was not surveyed.

Density was not uniform across the areas surveyed. If densities were uniform the estimated standard error would be zero. No horses were seen in the southern section of Northern Victoria (lower Buchan River). But, horses were seen frequently on the high

plateau country on the NSW/Victoria border, and in the open plains in the northern section of North Kosciuszko. Therefore some areas had densities higher than 1.8km^2 , and some areas had lower densities.

In 1990, the population densities of wild horses in the Alpine National Park in the area covered by the current survey was thought to be 1500-1700 individuals based on anecdotal evidence (Dyring 1990). In the current survey, the estimated population size over the same area is 2086 (± 586 SE) horses (1.8 ± 0.5 SE horses/ km^2 in 1159km^2). The current estimate is higher than that of Dyring, but the difference is not statistically significant because Dyring's estimate falls within the 95% confidence intervals of the estimate for the current study.

From the current study, estimates of wild horse population size in Kosciuszko National Park are 2934 (± 824 SE) (1.8 ± 0.5 SE horses/ km^2 in 1630km^2). Dyring (1990) suggested that there were several hundred horses in southern Kosciuszko and approximately 100 in northern Kosciuszko (refer to Chapter 2). It is not known how accurate these estimates are. If they are accurate then the wild horse population in Kosciuszko has increased at least threefold in the last decade.

A density of 1.8 horses km^{-2} falls at the upper end of the expected density of horses based on use of the density-body mass equations of Damuth (1981) and Freeland (1990). The high value may be partially explained by the fact that wild horses are not native to the Australian Alps, and introduced species can occur at higher densities in new environments because they may lack predators and pathogens (Freeland 1990). Refer to Chapter 4 for further discussion of wild horse density.

3.4.2. Assessment of analysis techniques

This study illustrates that analysis techniques influence density and abundance estimates in aerial survey. Strip, mark-recapture and line transect techniques are often used separately for aerial surveys. This survey compared the three techniques and found that

they gave a range of estimates in density and abundance with varied levels of precision. We cannot know how accurate each technique was because true population size is not known. We do know however that undercounting is the rule in aerial survey because not all animals are detected (Caughley 1974, Pollock & Kendall 1987).

The most important factors affecting estimates were strip width, correcting for perception bias and group size estimation. A narrow strip (0-50m) returned higher, and presumably more accurate results for strip (Figures 3.5 & 3.6) and mark-recapture (Tables 3.5 & 3.6) analyses than a wider strip (0-200m) because detection of animals dropped off rapidly in consecutive distance intervals. Adjustments for perception bias in mark-recapture analyses and line transect analyses for two observers combined meant that they gave higher and presumably more accurate estimates than the corresponding strip and line transect analyses for independent observers. Ground based group size estimates were used to improve the accuracy of estimates because observed mean group size from the air was an underestimate. It was also used to assess the accuracy of expected group size calculated in line transect analyses. The ground based data were based on intensive area counts of sites so were more accurate than aerial counts. During ground counts groups were observed for much longer periods on many occasions, than during the aerial survey.

Precision is a function of natural variability and analysis techniques. Variability in group size, low densities of horses and uneven distribution across the landscape limited the level of precision attainable in this survey. These are common issues in population surveys (Southwell 1989, Caughley & Sinclair 1994). There was a large range of precision across analyses, which was linked primarily to the number of parameters used in the estimation. Precision can be improved by increasing sampling intensity (see end of discussion) or increasing the area surveyed (Caughley *et al.* 1977). Another option is stratifying the survey (Krebs 1999). This was done initially in this survey by only sampling where horses were known to occur and excluding areas of very low density. Further stratification in future surveys may lead to an improvement in precision. Post sampling stratification to improve precision was not used.

Precision was highest in strip and mark-recapture analyses based on individual horse estimates rather than the product of group estimates and group size because the variability in the second case was a product of both group estimates and group size. One component of model selection by AIC in line transect analyses is the number of parameters (Buckland *et al.* 1993). The negative exponential model performed better than the similar hazard rate model because it had less parameters. Line transect analyses for individual observers returned very similar results to strip count estimates based on groups over 0-50m but with higher precision. Similarly, line transect analysis for both observers combined returned similar results to mark-recapture analysis based on groups over 0-50m but with higher precision.

Strip width

The number of horses and groups observed dropped off dramatically with distance from the line. This resulted in large negative bias for strip and mark-recapture estimates in strips greater than 0-50m. Similar trends have been observed in other aerial surveys (Caughley *et al.* 1976, Hone 1988). This has serious implications for survey design. Strips 200m wide are a standard width in strip and mark-recapture aerial surveys in Australia (for example see Southwell 1989, Black 2000). This is clearly too wide for aerial survey of horses in the Australian Alps. Even a strip 0-100m wide will lead to an underestimate because numbers of horses sighted dropped off so dramatically after 50m.

The sharp decline in the number of horses observed with distance from the line is probably due to vegetation cover. The area was predominantly woodland and forest that could conceal the horses leading to availability bias. Vegetation cover is a major factor influencing the visibility of large herbivores (Short & Bayliss 1985, Bayliss and Yeomans 1989b, Samuel *et al.* 1987). It is possible to observe animals from directly above through the vegetation layer, but as strip width increases the angle of view increases, the canopy and tree stems become harder to look through. This is not such an issue in more open country where a 200m wide strip can return the same results as line transect estimates (Pojar *et al.* 1995).

Line transect analyses performed better than strip or mark-recapture over wider strips because except near the line, there is no assumption that all animals are detected (Buckland *et al.* 1993). A line transect aerial survey of mule deer in pinyon-juniper woodland with some expanses of sagebrush succeeded in returning accurate abundance estimates (White *et al.* 1989). Line transect analyses are discussed in more detail later.

Correction of perception bias

Mark-recapture analyses and line transect analyses for both observers combined gave higher abundance estimates because they attempted to correct for perception bias. Observers did not see all animals and these analyses attempted to correct for this. The mark-recapture estimates were 15 and 49% higher for groups over 0-50m and up to five fold higher over 0-200m than strip estimates. Similarly, the best line transect models for both observers combined gave estimates 10-44% higher than the best models for each individual observer. Line transect analysis for individual observers are likely to be underestimates because mark-recapture analyses showed that not all animals in the first distance interval were observed by both observers.

Assessment of assumptions of the mark-recapture analyses show that estimates may be negatively biased. The key assumptions in the mark-recapture model that may be violated in aerial survey are 1) that animals/groups have equal likelihood of being sighted, and 2) that animals are correctly identified and recorded during marking and recapture. The first assumption can be difficult to satisfy in aerial survey. Both observers have the same vantage-point and horses that are difficult to see for one observer will be difficult to see for the other, and some animals may be completely concealed from both observers by vegetation. The effect of unequal likelihood of detecting horses will lead to negative bias (Caughley & Grice 1982). But, underestimates are only serious when the mean of detection probabilities for observers is <0.45 (Caughley & Grice 1982). Since detection probabilities are all above 0.56, underestimates are unlikely to be serious. The second assumptions may be a problem when animals occur at high densities (Pollock & Kendall

1987). The low densities of horses and the use of stopwatches to record times of sightings means that it is unlikely that the assumption was violated.

In the present study, the estimated probability of sighting a horse was 0.74 (observer 1) and 0.71 (observer 2) in a strip width of 200m. That equates to a visibility correction factor of $1/0.74 = 1.35$ and $1/0.71 = 1.41$ for observers 1 and 2 respectively. These correction factors are lower than those used in the Northern Territory with a similar strip width of 200m (Bayliss & Yeomans 1989b). The correction factors used in that study were 1.48 for floodplain (equivalent to grassland in the present study) and 1.78 in woodland. Correction factors used by Skeat (1990) in fixed-wing aerial surveys of wild horses in the Northern Territory, using a strip width of 150m, ranged from 1.0 for floodplains to 3.4 for dense woodland. Correction factors used by Black (2000) in fixed-wing aerial surveys of wild horses in the Northern Territory, using a strip width of 200m, ranged from 1.06 to 1.13 over during surveys over a range of habitats from shrubland to woodland. The effect of strip width on the mark-recapture estimates in the present study strongly suggest that many of these correction factors are too low, and hence estimated population densities would be too low (negatively biased).

Estimating group size

The estimates of mean group size from ground counts showed that mean group size was underestimated from the air. As described above, and in the methods, the ground counts used intensive area counts over 18 days so will produce more accurate group size data than the aerial counts. The underestimation of group size from the air is likely to be the result of missing smaller groups and not counting all individuals in a group. In a survey of elk, 100% of groups with 16-30 individuals were seen while only 22% of singletons were seen (Samuel *et al.* 1987). Graham and Bell (1989) showed a similar trend for horses, donkeys (*Equus asinus*) and cattle, and observers frequently estimated group size differently for larger groups. The size of larger donkey groups was underestimated by 12% in an aerial survey (Graham *et al.* 1982). Vegetation cover probably exacerbates this problem.

The use of group size data estimated from ground surveys increased the estimates of horse abundance. Higher estimates were obtained using strip and mark-recapture techniques based on the product of estimated number of groups and mean group size calculated from ground estimates. However precision was generally low because of the variability associated with group size.

The ground counts were also useful for assessing estimated group size from line transect analyses. The program DISTANCE estimated group size in all analyses by regressing \log_e (observed group size) against the detection function. The regression gave better estimates of mean group size than observer 2 and both observers combined. However observer 1's observed mean group size was more accurate than estimated mean group size. The regressions were poorly correlated. This is in contrast to mule deer where mean group size increased significantly with distance (White *et al.* 1989).

Observer experience can affect the accuracy of aerial survey estimates (Le Resche & Rausch 1974, Caughley *et al.* 1976, Frei *et al.* 1979). Estimates from individual observers were generally similar, but there were a few apparent differences. Firstly, the second observer saw more groups (notably smaller groups) than the first observer. This is probably the result of observer 2 having more experience in aerial survey. It was not associated with observer 2 splitting a group into two or more groups. Observer 1 tended to see more individuals per group. This is probably a result of having a lot of experience estimating horse group size from the ground. In a study that included horses, Bayliss & Yeomans (1989b) found that errors were trivial associated with larger groups being easier to detect than smaller groups and underestimating the size of larger groups.

Line transect analysis

Line transect analysis is very useful because it does not have the assumption that all animals in the strip are sighted. This was beneficial in this study as detection of animals clearly declined with distance from the line. However, despite all of the complex

modelling, it returned similar estimates to those made over 0-50m strips. The major advantage of line transect models is the increase in precision. Hone (1988) also observed this improvement.

One disadvantage of line transect analysis is the number of analytical decisions that need to be made (Southwell & Weaver 1993, Anderson & Southwell 1995). These include which model to use, whether data are analysed in interval form (and if so how many), and whether to truncate the data. The latter two points were not really an issue in this study. Data were only collected in four and five intervals. More intervals were not really an option in this survey due to reasons discussed in the methods. This limited opportunities for truncation and variations in number of intervals. Four or five is considered to be the acceptable minimum number of intervals for line transect analysis (Buckland *et al.* 1993). Four or five intervals have been used successfully in other aerial surveys to model detection functions and return accurate estimates (Hone 1988, White *et al.* 1989). One disadvantage of so few intervals is lower precision (Southwell & Weaver 1989, White *et al.* 1989).

The model that fitted the data best in all line transect analyses was the negative exponential key function. Despite the shortcomings of this model (Buckland *et al.* 1993), it appears to be the most appropriate for this survey. One of the criteria for robust estimation in line transect analysis is that the model has a shoulder near the line (Buckland *et al.* 1993). However, no shoulder was evident in the data. The spike was prominent even in the 0-15m distance interval when data was analysed in five intervals. White *et al.* (1989) found that the negative exponential line transect models were not biased in estimating mule deer densities. The spiked negative exponential function tended to correct for missing animals on the line, and hence appeared to have less bias than estimators with a shoulder.

The half normal and uniform models are popular models because they have a shoulder. However, it is the fit of the model near the line that is most important (Buckland *et al.* 1993) and these models consistently performed poorly near the line. They gave detection

probabilities below 1 on the line. Thus they violated the assumption that all animals on the line are seen. In addition, the curves drop off gradually which was not the case with the data.

The hazard rate model fitted the data, as assessed by the p-value, almost as well as the negative exponential model and better in a few instances. The density estimates of groups of horses for the hazard rate model were very similar to those calculated by the negative exponential model. However, the hazard rate model isn't very efficient because it uses more parameters and therefore has low precision. This inefficiency meant that it didn't perform as well as the negative exponential model and hence had a lower ΔAIC ranking.

One of the key assumptions in line transect analysis is that all animals on the line are seen. The combination of both observers' data was more likely to meet this assumption than each observer independently. Skeat (1990) found that aerial line transects underestimated water buffalo abundance by nearly half because not all animals on the survey line were sighted. In the mule deer study by White *et al.* (1989), they had two observers on one side of the aircraft and combined their data for line transect analysis and succeeded in obtaining accurate results with the negative exponential model.

The three other key assumptions in line transect analysis are that: objects are detected prior to any movement in response to the aircraft, objects are correctly counted in the proper distance interval, and animals are not counted twice (Buckland *et al.* 1993). The first of these assumptions may be violated if the animal moves in response to the observer. It is not a serious problem if the movement is random. There is no reason to believe that bias resulted from horses responding to the aircraft in the present study. The sightability curves show no evidence of movement. Some groups did not move at all before their distance was estimated, some bunched into smaller groups and some took flight. Flight was in all directions. Hence the problem of horses flushing about 1km reported by Linklater and Cameron (2002) did not occur in the present study. Violation of the second assumption is only a problem if the measurements are consistently biased. The use of distance intervals marked on a bar ensured that observers could easily classify

groups into distance intervals. There was no evidence of violation of the assumption that no animals are counted twice.

3.4.3. Sampling intensity

Increasing sampling intensity can improve precision (Caughley *et al.* 1977). An examination of the relationship between sampling intensity and precision (CV) for the current study showed that it would be difficult to improve precision (Figure 3.8). Lack of precision was largely a function of heterogeneity in the distribution of animals in particular they form groups of various sizes. This is a common reason for reduced precision in sampling (Caughley & Sinclair 1994). Increasing sampling intensity will also increase the chance of double-counting animals (Linklater & Cameron 2002) and increase costs.

4. POPULATION DYNAMICS

4.1. Introduction

The broad aim of this chapter is to provide a deeper understanding of the population dynamics of wild horses in the Australian Alps to improve management. The options available to managers of any wildlife species are either to make the population increase, to make it decrease, harvest it for a continual yield or leave it alone (Caughley & Sinclair 1994). There is a wide range of views on appropriate management of wild horses in the Australian Alps. I do not propose to judge the most appropriate goal in this chapter. Instead I hope to make a scientific contribution to wild horse management through providing the first description of the population dynamics of wild horses in the Australian Alps. I hope this contribution will improve people's understanding of wild horses in the Australian Alps, and aid in achieving management goals efficiently.

4.1.1. *Population demography and dynamics*

'Population' means different things to different people (Caughley 1977a). A general definition of a population is a group of individuals of one species in an area at a certain time (Begon *et al.* 1990). Population dynamics are the variations in the sizes and densities of populations (Begon *et al.* 1990). The speed of that change is measured as population growth rate (Sibly & Hone 2002). Any such change reflects a change in demographic parameters including birth rate, survival rate and/or distribution (Caughley & Sinclair 1994). To understand the dynamics of a population, we need to estimate these demographic parameters (Caughley 1977a, p.6, Sibly & Hone 2002). Estimation and modelling of these parameters provides information on the state of the population, whether it is likely to be stable, increasing or declining, and possible causes. This basic understanding is essential to the development and implementation of population management goals for whatever objective (conservation, harvesting or control).

Population growth rate is the key unifying variable of population ecology (Sibly & Hone 2002). It plays a central role in forecasting future population trends and defining a

population's ecological niche. Population growth rate is typically estimated using census data over time or from demographic (fecundity and survival) data (Sibly & Hone 2002). The finite rate of increase (λ) is the simplest measure of a population's growth rate; it is the ratio of abundance in two successive years (Caughley 1977a, p.51). Rate of increase can also be expressed as the exponential rate of increase (r); the two parameters are related by the equation $e^r = \lambda$ (Caughley 1977a). Exponential rate of increase is particularly useful because it is centred at zero and a rate of increase has the same value as the equivalent rates of decrease, but with the signs reversed (Caughley 1977a, p.52). The intrinsic rate of increase r_m is the maximum exponential rate at which a population with a stable age distribution can grow. It is dependent on the physiology of the species (for example litter size and gestation period), and the quality of the environment. It occurs when no resource (for example food) is in short supply, predation and parasitism are not limiting and there are no competitors (Caughley & Sinclair 1994).

Demographic models such as the Lotka model are often used to describe population dynamics (for example Caughley 1977a) and are an alternative for estimating population growth rate to census methods. The Lotka model considers age-specific survival and reproduction rates (Caughley 1977a). Eberhardt (1985) and Lande (1988) summarised these rates using a single reproductive parameter and two survival values. Survival rates are broken down into adult survival and survival from birth up to the age of first reproduction. The assumption of constant annual adult survival is an approximation but has been shown (Sibly *et al.* 1997) to be very useful. Fecundity is assumed to be constant between age of first reproduction and senescence. Another approach to demographic modelling is matrix modelling, which is also based on age structured survival and fecundity data (Caswell 1989). The latter approach was not used in the present study.

Sensitivity analysis of rate of increase (λ) to a change in key demographic parameters can be used to gain insight into the relative importance of each component of the population to rate of increase. This can help managers decide what part of the population to target if undertaking control or conservation measures (Lande 1988). Sensitivity is the change in λ for a unit change in a demographic parameter.

4.1.2. *Population limitation*

There has been considerable debate over limitation and regulation of animal populations (see Krebs 1994). The debate is confounded by differing definitions (for example Messier 1991, White 2001, Krebs 2002a). For the purpose of this thesis, limiting factors are defined following Messier (1991), as any processes that affect population growth from year to year. In turn animal abundance is affected. A subset of limiting factors are regulating factors which are density-dependent (Messier 1991). Regulating factors are characterised by negative-feedback mechanisms that depress population growth as animal abundance increases.

Limiting factors act on the demographic components of the population (births, deaths, immigration and emigration) (Eberhardt 1977). Factors that limit population growth are either extrinsic or intrinsic (Krebs 2002a). Extrinsic factors include predation, food supply, disease, parasites, weather and landscape. Population control by humans is also a major extrinsic factor to consider and is discussed in detail in Chapter 5. Intrinsic factors are social, physiological and genetic. Social factors do not usually regulate ungulates (Wolff 1997). In large mammals the overwhelming cause of population limitation is food supply (Sinclair 1989, 1996), however other factors such as predators or disease can operate to keep the population below the limit set by food supply.

Animal populations regulated by food will tend towards equilibrium dependent on food availability. A simple way of viewing the relationship between population dynamics and food supply is the demographic numerical response, which links population growth rate to the biomass of available food (Bayliss & Choquenot 2002). At high biomass, r is positive and approaches r_m , but as biomass is reduced, r approaches zero- the population is stable. At very low levels of biomass r becomes negative.

The demographics of a population of large mammals approaching density-dependent food regulation are predicted to show certain characteristics (Eberhardt 1977, Fowler 1987 in Krebs 2002b). As food becomes limiting, individuals have poorer physical condition, higher incidence of disease and parasitism, females have delayed first breeding, and survival rates in

younger age classes decline. Adult survival and reproductive rates are predicted to decline only at high densities. These processes have been shown to generally occur in food regulated ungulate populations (Sinclair 1996, Gaillard *et al.* 1998, 2000).

4.1.3. *Population dynamics of wild horses*

Dynamics of wild horse populations has been studied to various levels of detail around the world. Some of these populations are contained on islands (for example ~200 horses Goodloe *et al.* 2000), by geographical barriers (for example ~ 140 horses Berger 1986, ~150 horses Garrott & Taylor 1990, ~50 horses Greger & Romney 1999), and by fences (for example ~400 horses Linklater *et al.* 2000).

Several studies have estimated maximum annual rate of increase for horses with consistent results. Eberhardt (1987) estimated a r_m value of 0.191 ($I = 1.21$), which is the same as the mean estimated by Garrott *et al.* (1991a). Similarly in New Zealand, r_m was estimated from demographic models to be 0.196 ($I = 1.217$) (Cameron *et al.* 2001). Duncan (1992) recorded the highest exponential rate of increase for any population of wild horses at 0.24 ($I = 1.26$ but cited as 1.30). This population was liberated from domestic stock and appeared to be under optimal conditions. For the purpose of the current study an r_m value of 0.2 ($I = 1.22$) is assumed. Higher values than this appear to be the exception. Since the Australian Alps is a relatively harsh environment, this exception is unlikely to apply.

A summary of demographic parameters estimated in field studies is presented in Table 4.1. Wild horses are birth pulse species producing one young in the summer months coinciding with a peak in available food (Duncan 1992). In the northern hemisphere, foals are born between March and September, with births peaks varying with location (Keiper & Houpt 1984, Duncan 1992, Turner *et al.* 1992, Greger & Romney 1999). In the southern hemisphere births have been recorded between September and April (Berman 1991, Cameron *et al.* 2001). Timing of median date of birth varies between years (Berman 1991, Duncan 1992), and superabundant food can lead to a less strictly defined season (Duncan 1992). Birth sex ratios can vary from year to year, but overall they do not deviate significantly from one (Keiper & Houpt 1984, Berman 1991, Turner *et al.* 1992, Monard *et al.* 1997).

The age of first reproduction of wild horses is typically 3 years (Table 4.1), however fillies reproduce at the age of 2 when density is low and food is abundant (Berger 1986, Duncan 1992). Fecundity patterns are typical for ungulates (Caughley 1976, p. 186), increasing with age up to the age of 6 (Berger 1986, Keiper & Houpt 1984, Garrott *et al.* 1991b, Duncan 1992, Cameron *et al.* 2001). Reproduction in consecutive years is common (for example Berger 1986, Berman 1991). In general, reproductive rates are high, with 80-90% of the prime aged females foaling (Garrott *et al.* 1991b). Mares continue to have high foaling rates until the onset of senescence at approximately 15 years of age (Garrott & Taylor 1990, Garrott *et al.* 1991b).

Survival rates in horses (Garrott & Taylor 1990) are typical of ungulates (Gaillard *et al.* 1998, 2000) where adults have high survivorship with very low yearly variation, while survivorship is generally lower and more variable early in life. Therefore two stage life history demographic models can be applied to them (for example Eberhardt 1985, Garrott & Taylor 1990). Mean annual adult survivorship was reported to be above 90% for all studies reviewed (Table 4.1). Much lower adult survival rates occur in extreme winters (Garrott & Taylor 1990). Male and female survival rates are typically equal in adults (Cameron *et al.* 2001) and juveniles (Keiper & Houpt 1984, Goodloe *et al.* 2000) but the trend is not universal. Higher female adult survival rates have been observed (Garrott & Taylor 1990). Reported survival rates in 0-1 year old wild horses varied greatly (Table 4.1) as predicted by Gaillard *et al.* (2000). Some of this variation may be an effect of sampling because large proportions of foals die in the first month of life (Goodloe *et al.* 2000, Cameron *et al.* 2001). If studies did not observe foals from birth, estimated survival rates of 0-1 year olds will be an overestimate, while corresponding fecundity estimates will be lower.

Table 4.1: Demographic parameters estimated for wild horses from other studies, r = exponential rate of increase, fecundity is measured as the ratio of half juvenile to adult females, \emptyset = survival/year, a = age of first reproduction, j = juvenile, ad = adult. *ratio of juveniles to adults **female live births per female *** weights represent weight of mare.

Region	Site	r	fecundity	\emptyset (0-2 years)	\emptyset (>2 years)	a (years)	Reference
N. America	Oregon	0.18	0.10-0.20*	-	0.93-0.94 (j & ad)	-	Eberhardt <i>et al.</i> 1982
	Assateague Is.	0.09	0.29	0.88 (1 st yr)	-	3 (23%)	Keiper & Houpt 1984
	Utah	0.19	0.19-0.45	0.92 (1 st yr) 0.97 (2 nd yr)	0.95	2 (37%)	Berger 1986
	Montana	0.17	0.25	0.94	0.98	3 (39%)	Garrott & Taylor 1990
	Oregon	0.12	0.23	0.91-1.00	0.99	-	Ganskopp & Vavra 1986
	Cumberland Is.	0.03- 0.07	0.33	0.60	0.924 (female)	3(17%)	Goodloe <i>et al.</i> 2000
	California/Nevada	0	0.27	0.27 (1 st yr) 0.95 (2 nd yr)	0.96	-	Turner <i>et al.</i> 1992
S. America	Nevada	-0.07	0.18	<0.12	-	-	Greger & Romney 1999
	Argentina		0.41 -low density 0.23 -high density				Scorolli 2001
Australasia	Central Australia	-	0.18**	-	0.80 (j & ad)	2 (1 mare)	Berman 1991
	Kaimanawa, NZ	0.07	0.25	0.83 (1 st yr) 0.90 (2 nd yr)	0.90 (2-4 yrs) 0.95 (>4 yrs)	2 (1 mare)	Cameron <i>et al.</i> 2001
Europe	Camargue	0.24	0.0 (<250kg)** 0.25 (300kg) 0.48 (>400kg)	0.8-0.9 (food limited)	1.0 (>6yrs) 0.98 (1-6 yrs)	2 (food abundant) 3 (food limited)	Duncan 1992

Historically, the study of population ecology of large herbivores was difficult because their long generation time meant that obtaining age-structured information involved long-term studies (Gaillard *et al.* 1998). Simplifying demographic parameters into summarised rates has made it easier to undertake studies on shorter time scales (eg. Eberhardt 1985, Lande 1988, Sibly *et al.* 1997, Gaillard *et al.* 2000). The use of models without inclusion of age-structure has its shortcomings. Models that assume constant age structure may fail to predict changes in population size accurately (Clutton-Brock & Coulson 2002). However, this problem depended on the species. Clutton-Brock and Coulson (2002) found that age-structure variation in red deer (*Cervus elaphus*) (which have a similar life history to wild horses) did not have a pronounced effect on predictive modelling compared to the more fecund soay sheep (*Ovis aries*).

Sensitivity analysis (and the similar elasticity analysis) on species with similar life history traits to horses can be used to predict expected values for horses. In general, population growth rate of large herbivores is highly sensitive to changes in adult survival, that is, adult survival has high elasticity (Gaillard *et al.* 2000). Population growth rate is generally less sensitive to change in juvenile survival (low elasticity) and moderately sensitive (moderate elasticity) to changes in fecundity (Gaillard *et al.* 2000). Similar patterns are observed for the northern spotted owl (Lande 1988), which has a similar life history pattern to horses.

Observed rates of increase varied between studies. These variations reflected different forms of population limitation. Many populations were limited by management by humans as reported by Eberhardt *et al.* (1982), Garrott & Taylor (1990) and Berman (1991). Two studies have identified predation by mountain lions as a key limiting factor where juvenile survival was very low, and adult survival didn't appear to be affected (Turner *et al.* 1992, Greger & Romney 1999). Dingoes have been observed to kill foals in the Australian Alps (Newsome *et al.* 1983b), but it seems to be a rare occurrence.

Density-dependent food regulation has been observed in horses (Duncan 1992, Scorolli 2001) and inferred in donkeys (Choquenot 1991). I am unaware of any published numerical responses for horses. Duncan (1992) and Scorolli (2001) observed populations of wild horses

as they approached food limitation, while Choquenot (1991) undertook a manipulation experiment. Choquenot (1991) and Duncan (1992) found that juvenile survivorship was lowered under food limited conditions and age of first reproduction was delayed. There was no difference in observed fecundity in the donkey populations however fecundity was found to be density-dependent in both horse populations. Duncan (1992) found that fecundity in younger females declined as food availability declined. The trend occurred at relatively low densities in the youngest females (2-3 years) and with lower food availability in 3-7 year olds. Production of foals by mares >7 years remained very high, only showing signs of reducing at peak density. Adult survival of donkeys was marginally reduced at high-density (Choquenot 1991), while horses were removed from the study population observed by Duncan (1992) because they were close to starvation. Body condition was a good indicator of the state of the population in all three studies. Body condition at high densities was poor and at low densities was good. There was a close correlation between condition score in mares and available food (Duncan 1992).

4.1.4. Predictions and aims

The demography and population growth rate of wild horse populations in the Australian Alps was unknown at the commencement of the study. The approach of this study was to establish key demographic parameters at three sites under natural conditions and compare them to expected values from the literature. This approach is advisable when first commencing a study on a population whose ecology is poorly understood and before detailed field experiments are undertaken (Krebs 1988, Eberhardt in press). The predicted effects of different limiting factors on demography and food supply are summarised in Table 4.2.

State	r	Fecundity	Survival 0-2 y.o.	Adult Survival	Body Condition	Food Supply
Not Limited	0.2	High (0.4)	High (0.9)	High (0.95)	Good	High
Food limited	0	Reduced (≤ 0.3)	Reduced (< 0.9)	Possibly Reduced (≤ 0.95)	Poor	Low
Predator Limited	0	High (0.4)	Reduced (0.3 1y.o.)	High (0.95)	Good	High
Harvesting Limited	0	High (0.4)	Reduced (< 0.9)	Reduced (< 0.95)	Good	High

Table 4.2: Predicted key population parameters for wild horses and food supply under different forms of population limitation. Likely values in brackets are estimates taken from the literature (see Table 4.1). Rate of increase (r) is annual and fecundity is the number of females born per adult female per year.

The aims of this chapter are to:

1. Define key demographic parameters and determine population growth rate for three populations of wild horses in the Australian Alps and test for differences between sites,
2. Assess these key demographic parameters and population growth rates in the light of knowledge of wild horse population dynamics from the literature with the aim of improving our understanding of what, if anything is limiting wild horses in the Australian Alps (refer to Table 4.2), and
3. To assess the sensitivity of the rate of increase of the populations to changes in key demographic parameters.

4.2. Methods

4.2.1. Study areas

The population dynamics of wild horses were studied at three sites within the Australian Alps national parks. Big Boggy near Mount Kosciuszko in central Kosciuszko National Park, Cowombat on the border of Kosciuszko National Park and the Alpine National Park near The Pilot and Currango in northern Kosciuszko National Park southwest of Mount Bimberi (Figure 1.1). These sites were chosen for several reasons. Firstly to sample a range of habitats, secondly because they were areas of interest for park managers, thirdly because wild horses could readily be found at all of the sites and fourthly because they are independent of each other (too far apart for horses to move between). There was no management of these populations for the duration of the study. Climatic data are not available for the sites however Thredbo River weather station is adjacent to the Big Boggy study area, and Kiandra is approximately 20km south-west of Currango (see Figure 1.2 for summary weather data from these sites). There are no weather stations near Cowombat. The most representative station is probably Thredbo Village 30km to the north.

Big Boggy

The Big Boggy site (148°15'E 36°35'S) covers an area of approximately 35.5km² at 1500-2228m above sea level (Figure 4.1). It is very close to the village of Thredbo in Kosciuszko National Park, New South Wales. The survey area included alpine habitat south of Mount Kosciuszko, and the upper catchment of the Thredbo River. Vegetation communities include the alpine vegetation complex above about 1800 metres, subalpine snowgum stands on hill sides and cold air drainage shrub and herbfields along the Thredbo River (McRae 1989) (Figure 4.2). Big Boggy is the highest of the three sites so horses are subject to more severe winter conditions in particular snow. Horses were first sighted in the alpine area (north of the Alpine Way) about six years ago (K. Green pers. comm.). This area is very sensitive to environmental damage and managers were particularly interested in this area (NSW National Parks and Wildlife Service 2002a). Horse numbers in the area were low and seasonal (M.

Walter unpublished data), therefore the primary area of habitat was the upper catchment of the Thredbo River (Figure 4.1).

Cowombat

The Cowombat study site (148°10'E 36°47'S) is located at the headwaters of the Murray River and straddles the New South Wales and Victorian border (Figure 4.3). It covers an area of approximately 12.7 km² and a range of elevations of 1000-1540m above sea level. It is in the middle of a large area of declared wilderness, with the nearest public access road 13km away. The vegetation in the study area is predominantly montane and mixed gum open forest, with sections of subalpine snowgum stands at higher elevations, cold air drainage shrub and herbfield and tall gum and alpine ash communities (McRae 1989) (Figure 4.2). Cowombat Flat is at the junction of the Murray River and Pilot Creek in the centre of the study area. Upstream along the waterways the topography is generally gentle with rich soils and grassy flats. Downstream of Cowombat Flat, the topography becomes steep and rocky. There are a few clearings dominated by grassland. To the southwest of Cowombat Flat, it rises steeply towards the Cobberas. This is also typically dry and rocky with a few clearings in wetter areas.

Currango

The Currango site is located at the top of Tantangara Dam in northern Kosciuszko National Park (148°40'E 35°42'S). It covers an area of 41.4km² and has the least topographic relief of the three sites, ranging from 1220 to 1370m above sea level (Figure 4.4). It is an extensive plain dominated by cold air drainage shrub and herbfield vegetation communities, which are periodically inundated up to the 1230m contour (McRae 1989) (Figure 4.2). The edges of the plain are montane mixed gum open forest, sub-alpine snow gum stands and tall gum and alpine-ash forest communities (McRae 1989). Currango Plain is 6km long running north-south, and is about 5km wide. It is gently undulating with higher ground sparsely vegetated with snowgum and lower flat ground is swampy. The study area also extended to the west to encompass Dairyman's Flat and the adjacent open forest areas at the headwaters of Muffler's Creek.

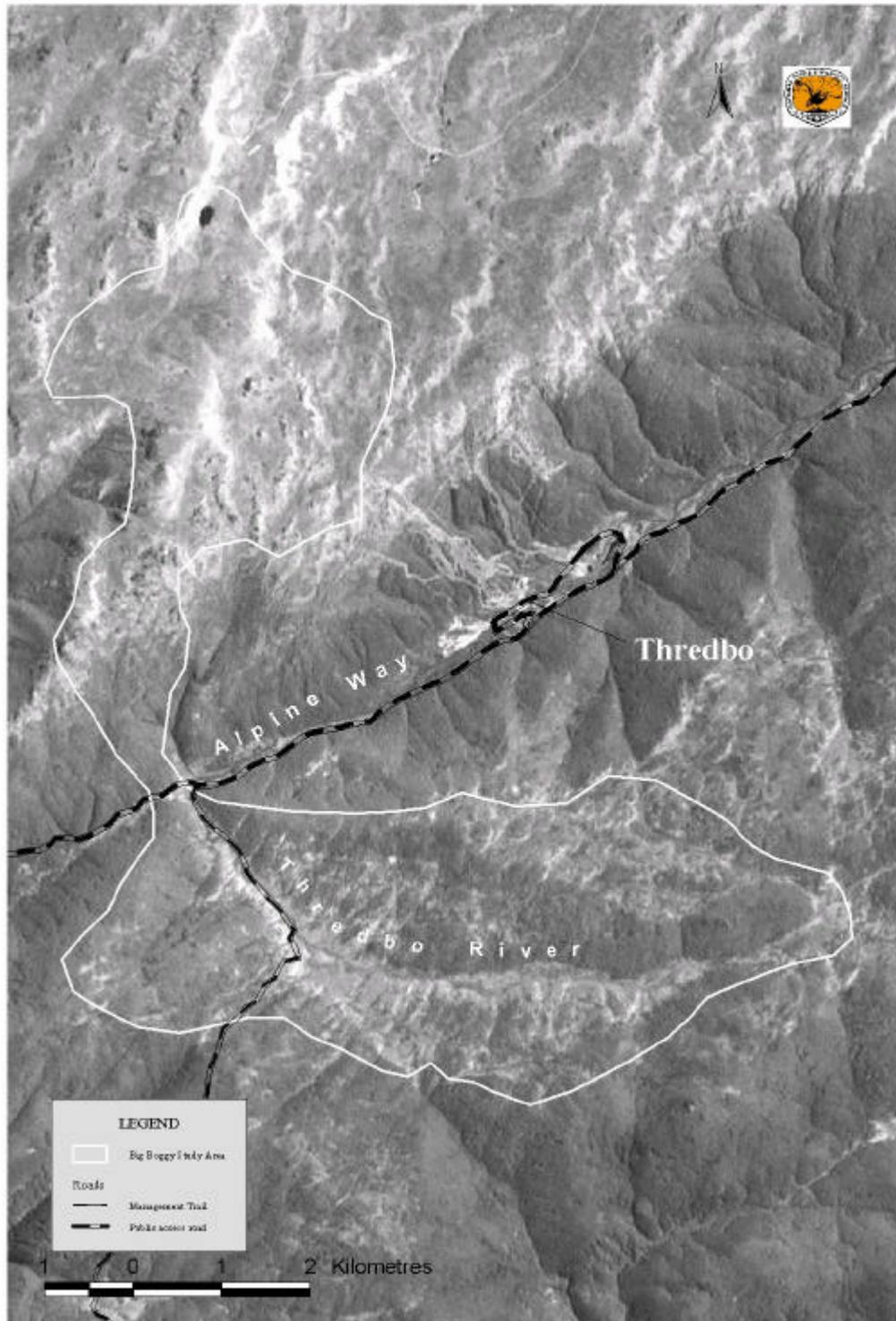


Figure 4.1: Satellite image of the Big Boggy study area. The survey area is outlined in white and roads and management trails are striped lines.



Figure 4.2: Representative habitat with wild horses at each site. Big Boggy (top), Cowombat (middle) and Currango (bottom).

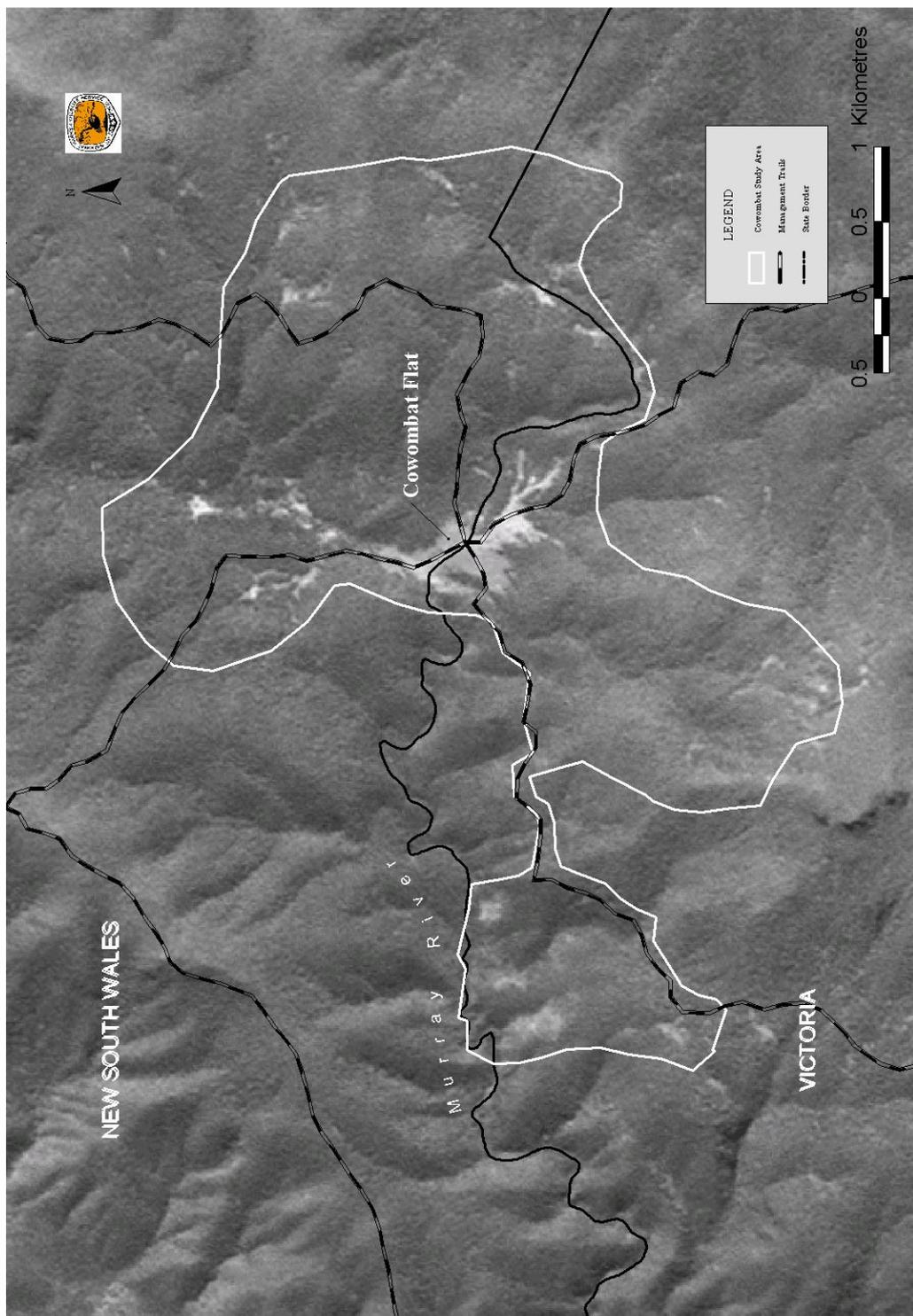


Figure 4.3: Satellite image of the Cowombat study area. The area surveyed is outlined in white, management trails are striped and the New South Wales and Victoria border is a solid black line.

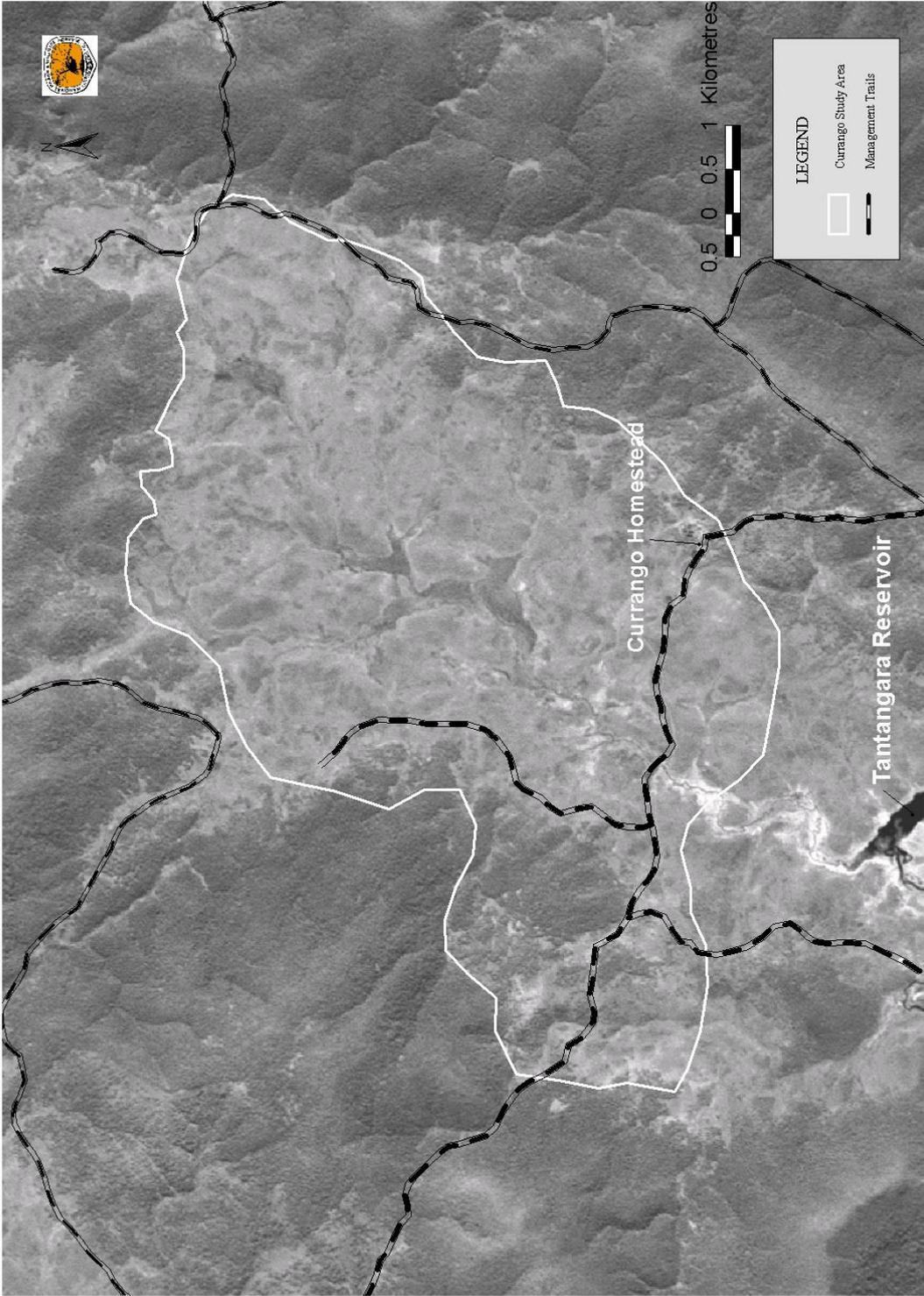


Figure 4.4: Satellite image of the Currango study area. The area surveyed is outlined in white and management trails are striped.

4.2.2. *Survey design*

Population dynamics were studied at the three sites Big Boggy, Cowombat and Currango. The survey design following the structure of Pollock's robust design at each site (Pollock *et al.* 1990) (Figure 4.5). The design is based on both open and closed mark-recapture population models. Closed population models were used to estimate population size from short-term surveys of less than ten days (primary periods). Open population models were used to estimate survival in between primary periods.

Each site was surveyed in spring and autumn from spring 1999 to autumn 2002 giving a total of six primary periods (Figure 4.5). In each primary period, 3 days were spent searching for and identifying horses within the area, which was equivalent to a marking event. Then 3 days were spent covering the same area identifying horses and recording how many were resights and how many were new animals. Survey dates are summarised in Table 4.3. At each primary sampling period, information on sex, age and body condition of all individuals was collected for use in demographic analyses.

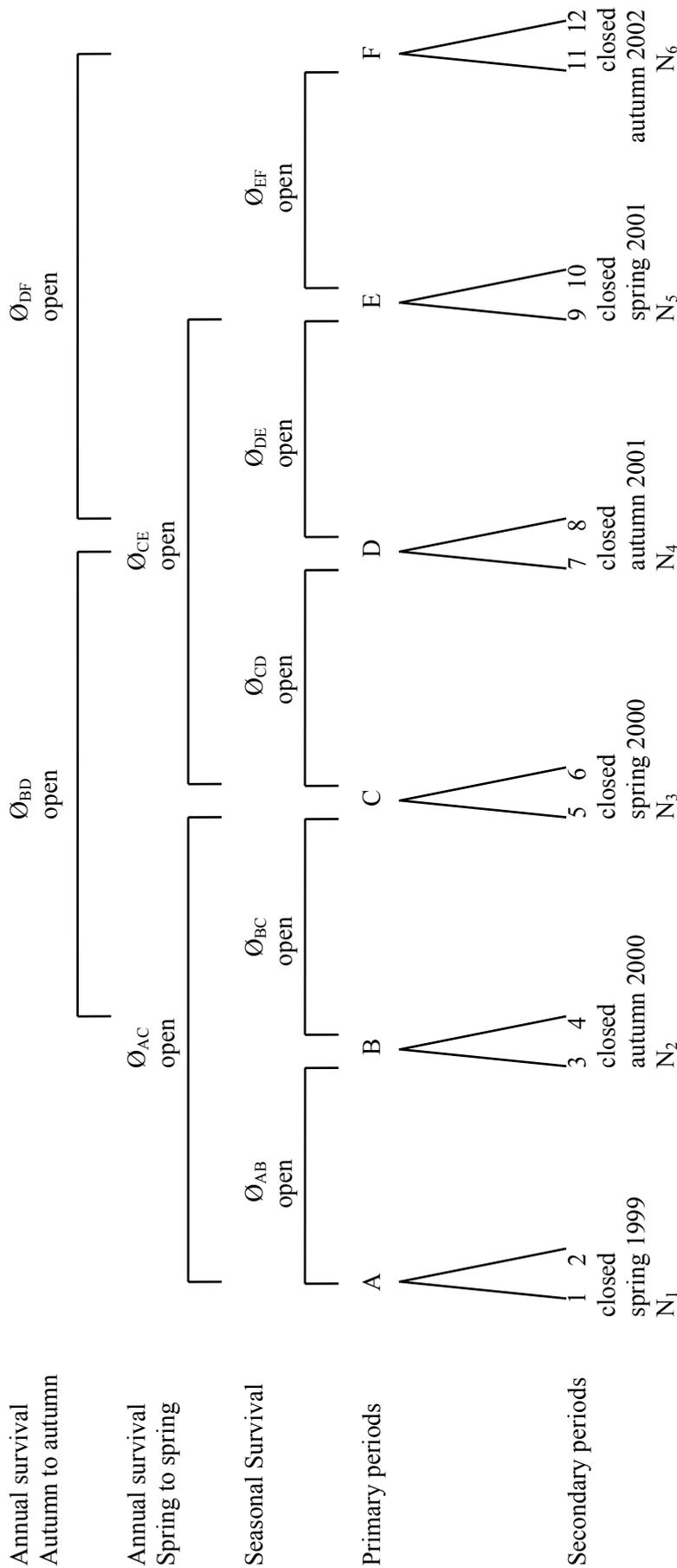


Figure 4.5. Mark-recapture experimental design based on Pollock's robust design used at each study site. The numbers 1, 2 ... 12 represent capture events of 3 days long. Each capture event is paired (e.g. 1, 2 and 3, 4). Each pair is surveyed within a 10-day period and the population is assumed closed over this period. The animals marked in each paired event are pooled and considered as one capture event and the population assumed to be open between primary periods.

Primary Sampling Periods	Big Boggy	Cowombat	Currango
Spring 1999	25/10/99 to 4/11/99	13/11/99 to 19/11/99	23/9/99 to 28/9/99
Autumn 2000	28/3/00 to 5/4/00	16/3/00 to 21/3/00	6/5/00 to 11/5/00
Spring 2000	21/10/00 to 27/10/00	4/11/00 to 10/11/00	22/9/00 to 28/9/00
Autumn 2001	3/3/01 to 8/3/01	23/3/01 to 29/3/01	28/4/01 to 3/5/01
Spring 2001	4/11/01 to 9/11/01	21/10/01 to 1/11/01	8/10/01 to 13/10/01
Autumn 2002	15/3/02 to 20/3/02	13/4/02 to 18/4/02	28/4/02 to 3/5/02

Table 4.3: Dates (day/month/year) of primary surveys conducted at each site.

4.2.3. Individual recognition

Mark-recapture studies typically involve capturing animals and marking them in some way (for example tagging them). In the present study, natural markings were used to identify individuals rather than tags. Natural marks have been used successfully to identify individuals of many mammalian species, such as humpback whales *Megaptera noveangliae* (Stevick *et al.* 2001), and bottlenose dolphins (Wilson *et al.* 1999). Berger (1986) and Turner *et al.* (1992) successfully used natural markings to identify and study wild horses, while Cameron *et al.* (2001) used a combination of natural markings and brands. I used binoculars, a spotting scope and photography to identify individuals based on their natural markings. Records were kept on standard data sheets. Individual distinguishing markings were written down, drawn and photographed in as much detail as possible for each sighting. One person did all surveys and identification to avoid observer bias. A pilot study was made at the Big Boggy in 1999 and the technique was assessed to be viable. During the pilot study, animals were typically easy to approach and the majority of animals could be identified from their natural marks. A portion of animals with indistinct markings could be identified from their membership of a stable group of identifiable animals.

4.2.4. Population size, density and growth rate

An estimate of population size (N_t) was determined twice per year for each study site using a closed population model (Figure 4.5). The model used was a modified version of the Lincoln-Petersen capture-recapture model (Lancia *et al.* 1996). It was possible to be confident on mark-recapture events over the short survey period even when animals were difficult to approach. This is because location and group structure of horses were generally constant over the ten-day observation period. Density was calculated by dividing estimated population size by survey area. Ideally home range information would be used to calculate density with boundary strips (Krebs 1999), however home range estimation was not the focus of this study.

Annual exponential rate of increase (r) was determined at each site in three ways. Firstly the ratio of abundance in two successive years was compared from spring to spring and autumn to autumn to estimate λ . The natural log of the value gave r . The second method estimated r as the slope from the least squares linear regression of $\ln(N)$ versus time (Caughley & Sinclair 1994). The F-statistic was used to test whether the slope was significantly different from zero at the 95% confidence level. The third method used demographic models and is described in section 4.2.6.

4.2.5. Fecundity and survival estimation

Birth rate is an important component of population dynamics contributing directly to population growth rate. Fecundity rate m_x is the number of female live births per female per year (Caughley & Sinclair 1994). Other measures of birth rate are the ratio of juveniles to adults or juveniles to adult females. In ungulates, fecundity rate climbs rapidly from puberty and plateaus through adulthood (Caughley 1976, p.186). For purposes of population analysis, little precision is lost by calculating a single fecundity rate for adults as the weighted mean of the plateau fecundity (Caughley 1976, p.186).

Fecundity was estimated at each site in each autumn sampling period (the end of the foaling season) by determining the ratio of juveniles to adults, and the ratio of half of the juveniles to adult females. These methods were used instead of comparing the ratio of female juveniles to

female adults because it was not possible to determine the sex of 32 percent of foals, and a previous study found that sex ratios of horses at birth are not significantly different from zero (Keiper & Houpt 1984). Two-way analysis of variance was used to test for differences between sites and years at 95% confidence levels.

Survival, like fecundity, is age dependent (Caughley 1977a). The simplest measure of annual survival rate is the number of animals that survive over a year divided by the number alive at the beginning of the year. A more reliable method is to follow individually marked animals through time and estimate survival parameters instead of population size because survival estimators are more robust to the partial failure of assumptions (for example heterogeneity of individuals to capture and recapture) than are estimators of population size (Lebreton *et al.* 1992).

Survival (ϕ) and recapture (p) probabilities were estimated using the Cormack-Jolly-Seber open-population method (Lebreton *et al.* 1992) in program MARK 2.1 (White & Burnham 1999). The initial plan was to use a robust design but varying data quality made this non-viable. Analyses were done in several stages. The first analyses pooled all data within sites and modelled survival and recapture probabilities over time ($\phi(t)p(t)$). This approach was taken instead of starting with age and sex structured models because survival rates in horses are generally high with limited variability and data sets were not large. The input capture histories were classified into 90% and 100% confidence classes. These classes were chosen to distinguish between identifications that I was confident with and those that I was less confident with. Time intervals were set to 0.5 (representing half a year) because surveys were conducted bi-annually. Therefore all survivorship estimates are annual.

Tests for over-dispersion were used to assess whether the time dependent model adequately fitted the data for each site and each confidence level. Over-dispersion is usually a result of important structural variation not explained by the data, or correlation in capture histories between individuals (Anderson *et al.* 1994). The goodness of fit of the global models were evaluated with the bootstrap (500 simulations) approach available in MARK (White *et al.* 2002). The over-dispersion coefficient (\hat{c}) was calculated by dividing the deviance

estimate from the original data set by the average of the simulated deviances and by dividing the observed value of \hat{c} by the average simulated \hat{c} (White *et al.* 2002). Both methods were used because it is not clear which procedure is better (White *et al.* 2002). \hat{c} values were used to assess the structural adequacy of the models. If \hat{c} was less than three the model was assumed to be structurally adequate (White *et al.* 2002) and the \hat{c} value was adjusted in MARK. When \hat{c} was greater than three, the model was considered to have insufficient structure to be used for parameter estimation.

Parameter estimates were obtained by running the global model and models nested within it, then using model averaging based on quasi-Akaike Information Criterion weights (QAICc) (White *et al.* 2002). Program MARK weights models based on Akaike's Information Criterion (AIC_c) (Burnham & Anderson 1998). AIC_c ranks models based on the trade-off between model fit and number of parameters. MARK converts these AIC_c values to quasi- AIC_c values (QAICc) after adjusting for \hat{c} . Models receive weights based on QAICc values, with higher weights for better models.

The next step was to analyse each site independently to test for differences in survival rates between sexes and between two age classes (0-2 and >2 years-old). Data were of varying quality so they could not be examined simultaneously. An example of data of varying quality is that for some juveniles, sex could not be determined. These animals were not included in analyses by sex but were included in age analyses. The same approach for assessment of goodness of fit, \hat{c} adjustment and model averaging was taken as outlined above. Not all global models were fully time dependent because the primary aim of this study was to estimate survival rates for adults and pre-adults, and to look for difference in survival rates between males and females. Contrary to common belief, the global model doesn't have to be the saturated model (Cooch 2002).

4.2.6. *Demographic modelling and sensitivity analysis*

Lande's (1988) modification of the Euler-Lotka equation was used to estimate the annual finite growth rate (λ) of the populations at the three study sites. The equation is,

$$l^a (1 - s/l) = l_a m$$

where a is age (years) at first reproduction, s is annual adult survivorship, l_a is survival from birth to first reproduction and m is fecundity (female offspring/adult female/year). This equation is a simplification of that used by Eberhardt (1985, in press) and Eberhardt *et al.* (1994), as the maximum age is not included. Calculations are not very sensitive to the maximum age used (Lande 1988, Eberhardt *et al.* 1994).

Estimates of rate of increase assumed $a = 3$ years because no 2-year-old females were observed to foal at any of the sites. Adult and pre-adult survival was obtained from estimates based on the age survival model for that site. Survival from birth to first reproduction was calculated as $l_1 l_2 l_3$ where 1, 2 and 3 represent survival to the ages of 1, 2 and 3 respectively. Fecundity estimates were the site-specific estimates obtained in the fecundity section 4.3.2 for the ratio of half juveniles to adult females. Data presented in tables may appear slightly inconsistent due to rounding. All calculations were made with three or more decimal places while data are presented to two decimal places.

Sensitivity of λ at each site to changes in the demographic parameters of fecundity (m), survival to first reproduction (l_a), annual adult survival (s) and age of first reproduction (a) were calculated by implicit differentiation as by Lande (1988),

$$\delta\lambda/\delta x = \lambda/xT$$

$$\delta\lambda/\delta s = \lambda(T-a+1)/(sT)$$

$$\delta\lambda/\delta a = -\lambda[\ln(\lambda/s)]/T$$

$$T = a + s/(\lambda - s)$$

where x is l_a or m , and T is the generation time (years) of the population. Values used for each site were the same as those described above for demographic modelling.

The sensitivity equations were used to calculate the standard error of λ following Lande (1988). The standard error is the square root of the sampling variance. Sampling variance of λ is represented by the following equation,

$$s_I^2 = \sum_p \left(\frac{\partial I}{\partial p} \right)^2 s_p^2$$

where π represents each of the four demographic parameters from the demographic model. Sampling variance of each parameter is represented by the last term in the equation. The estimate for the sampling variance of λ does not include the covariance between sampling statistics. Covariance could arise if for example some of the parameters are estimated from the same individuals (Lande 1988). Exclusion of the covariance between sampling statistics will lead to an underestimation of the sampling variance.

The sampling variances in fecundity estimates were calculated for each site by the mean annual ratio of half of the juveniles to adult females multiplied by the standard deviation, then divided by the number of adult females sampled. The sample variance of annual survival was calculated assuming an underlying binomial distribution as for northern spotted owls (Lande 1988). The sensitivity coefficient of age of first reproduction in horses was very small and contributes a negligible amount to the sampling variance of λ . It is therefore assumed to be zero.

4.2.7. Measuring body condition and biomass

Body condition scores were estimated by visual body fat distribution that was based on a scale from 0-5 (Huntington 1991) when horses were sighted, provided visibility was good. A score of 0 was for a horse in very poor condition and a score of 5 was for a very fat horse. Body scores were summarised for each horse by season by taking the median of the visual body condition scores for all sightings of that individual. Repeated measures ANOVA assuming independence of observations of horses was used in Program SAS (SAS Institute 2000) to determine whether there were differences in body condition between sites, sexes, seasons and years. Not all interactions could be tested because of insufficient data on a few occasions.

Standing pasture vegetation biomass was estimated at each site at the same time as the horse population surveys from autumn 2000 to autumn 2002 using the comparative yield method of Haydock and Shaw (1975). This method is widely accepted and used in grazed pastures (for example Robertson 1987, Tothill *et al.* 1992, Choquenot 1998). It involved pre-selecting a

set of standard reference quadrats in the field. These standards provided a scale over the range of the yields likely to be encountered ranging from 1 to 5. In the present study, photographs of reference plots were taken and were available for reference throughout the sampling. Initially I trained a second observer and myself. We simultaneously rated a series of quadrats until an acceptable degree of agreement was reached. As surveys progressed I ended up being the only person making biomass observations as the second observers changed between surveys and returned less consistent results. A 0.5m*0.5m quadrat was carried out for the second half (3 days) of the population survey. On each survey between 30 and 90 plots were randomly sampled and ranked. At the end of 8 of the 15 surveys, 12 plots were selected covering a range from low to high pasture biomass, then ranked. The typical types of plants in the sample included grasses and forbs. The herbage in these quadrats was harvested and oven dried at 80°C for two days then weighed. A calibration was then established by regression of log of pasture biomass and log of rank. These calibrations were not carried out on all of the 15 survey occasions primarily because of inclement weather.

The standard approach of the comparative yield method is to estimate biomass on each occasion using the calibration regression from the occasion. Since calibration curves were not available on each occasion, analysis of covariance (Sokal & Rohlf 1995) was used to test whether available calibrations by site had the same slope and elevation and could therefore be combined into one regression line to be used for all occasions at that site. Analysis of covariance was done in program SAS (version 8) under the generalised linear model procedure. The tests were done for the Big Boggy (2 regressions), Cowombat (3 regressions) and Currango (3 regressions) separately using type III sums of squares. Four outliers were removed because they had very low values (and hence were heavily weighted on the log scale), the measurements were probably inaccurate and such low values were not characteristic.

First differences between slope were tested. If this was found to be significantly different, then the regression lines were significantly different and could not be combined. If the slopes were not significantly different, then the slope component was removed from analysis and differences in elevations were tested. If they were significantly different, then no further

analysis was made. If the elevations were not significantly different, then all calibration curves for the site were combined to give one calibration equation.

4.2.8. Numerical response

The numerical response of wild horses at the Big Boggy was estimated using Eberhardt's (1998) equation

$$I = (1 + r_m) - \frac{r_m}{a} \left[\frac{H}{V} \right]$$

where λ = annual finite rate of increase

r_m = intrinsic annual rate of increase

a = mean number of horses/biomass (kg/ha), at equilibrium ($\lambda=1$)

H = mean number of horses

V = mean pasture biomass (kg/ha)

Mean pasture biomass was calculated from estimates outlined in Section 4.3.4, and mean population size was estimated as the mean of population estimates for wild horses at the Big Boggy (Section 4.3.1). The equation only applies when λ /year is not different from 1.0. The equation could not be applied to the Currango wild horse population because $\lambda > 1$ (Section 4.3.3). The equation could not be applied to the Cowombat wild horse population due to insufficient biomass data (see Section 4.3.4).

4.3. Results

4.3.1. Population size, density and growth rate

Population estimates through time showed different patterns at each site. The Big Boggy population showed seasonal patterns with higher numbers in autumn than in spring, and the spring estimates were more precise than the autumn estimates (Figure 4.6). Population estimates for the Big Boggy ranged from 57 \pm 1SE to 86 \pm 6SE horses over the duration of the study. Cowombat population estimates also showed seasonal trends but not as clearly as the Big Boggy, and were generally less precise than Big Boggy estimates (Figure 4.7). Population estimates for Cowombat ranged from 64 \pm 6SE to 101 \pm 10SE horses with

seasonal peaks in spring (Figure 4.7). There was no clear seasonal trend in population estimates for Currango, and precision varied between occasions (Figure 4.8). Population estimates for Currango ranged from $69 \pm 4\text{SE}$ to $109 \pm 6\text{SE}$. Precision at the Big Boggy was generally high because recapture rates were generally very high. Recapture rates were also high but variable for surveys at Cowombat and Currango, which is reflected in the variable precision.

The density of wild horse populations varied between sites. The highest densities were recorded at Cowombat (average 6.40 horses/km^2), and the lowest at the Big Boggy (average 2.01 horses/km^2). Average density of wild horses at Currango (2.13 horses/km^2) was slightly higher than at the Big Boggy.

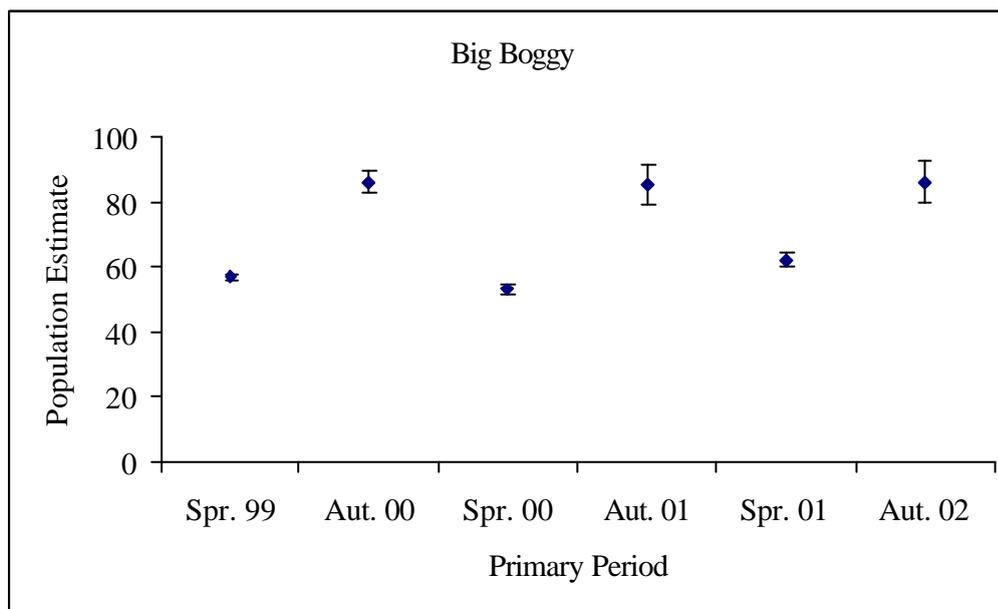


Figure 4.6: Wild horse population estimates (with standard error bars) for the Big Boggy from spring 1999 to autumn 2002 calculated using the modified Lincoln-Petersen estimator.

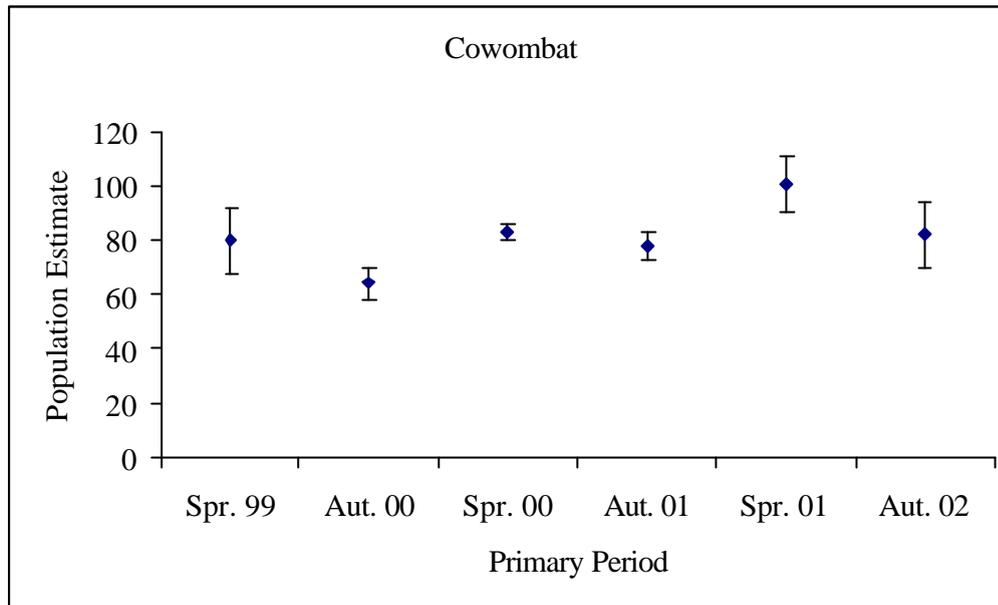


Figure 4.7: Wild horse population estimates (with standard error bars) for Cowombat from spring 1999 to autumn 2002 calculated using the modified Lincoln-Petersen estimator.

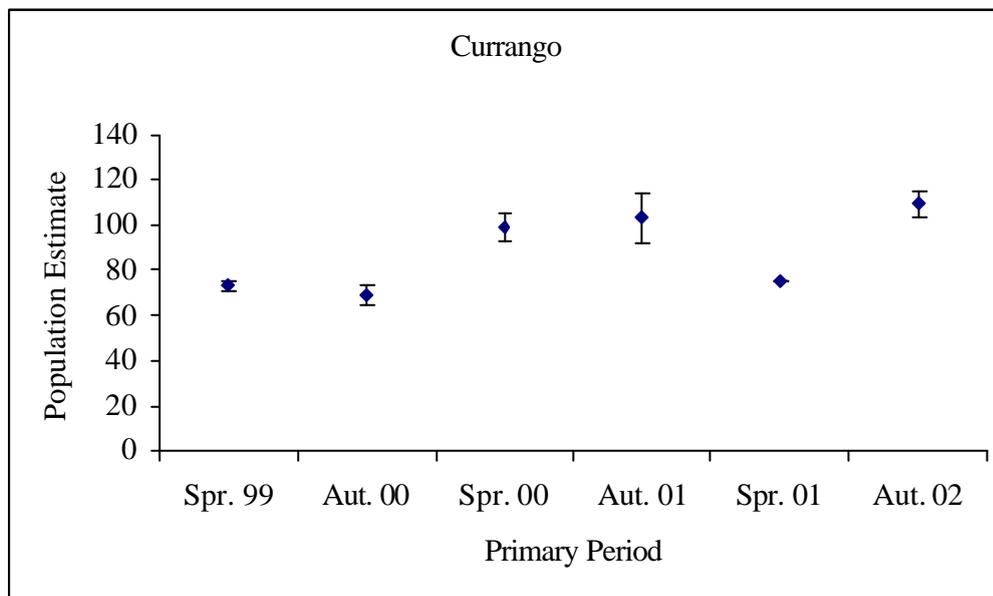


Figure 4.8: Wild horse population estimates (with standard error bars) for Currango from spring 1999 to autumn 2002 calculated using the modified Lincoln-Petersen estimator.

The annual exponential rate of increase (r) of each population is summarised in Table 4.4. Cowombat was the only site that showed a positive rate of increase for each time interval.

Currango showed a decreasing population for the spring 2001-2002 period but increases for all other periods. Estimates of rate of increase for the Big Boggy were close to zero from autumn estimates. Estimates from spring to spring were both positive with the spring 2000-2001 estimate (0.16) being twice that (0.07) of spring 1999-2000.

Table 4.4: Annual exponential rate of increase (r) of the wild horse populations at each site from spring to spring and from autumn to autumn.

	Big Boggy	Cowombat	Currango
Spring 1999-2000	-0.07	0.04	0.30
Spring 2000-2001	0.16	0.20	-0.28
Autumn 2000-2001	-0.01	0.20	0.40
Autumn 2001-2002	0.01	0.05	0.06

When the data were analysed using regression of the natural logarithm of abundance over three years, for each season, there was no significant population change (Table 4.5). The Big Boggy population appeared to be stable, the Cowombat population appeared to be increasing and Currango had discrepancies between spring and autumn.

Table 4.5: Estimated rate of increase (r) of the wild horse populations at each site over three years for spring and autumn abundance data. r = exponential rate of increase and R^2 is the coefficient of determination and ns is not significant.

		r	R^2	Significance
Big Boggy	Spring	0.04	0.29	ns
	Autumn	0.00	0.00	ns
Cowombat	Spring	0.12	0.87	ns
	Autumn	0.12	0.87	ns
Currango	Spring	0.01	0.01	ns
	Autumn	0.23	0.84	ns

4.3.2. *Fecundity and survival*

When birth rate was calculated as the proportion of juveniles to adults at each autumn survey at each site (Table 4.6), Currango had the highest rate (average = 0.28), and Cowombat had the lowest (average = 0.20). Values across all sites ranged from 0.15 to 0.37. None of these values was significantly different as analysed by two-way ANOVA (site: $F = 0.753$, d.f. = 2, 4, ns, year: $F = 0.572$, d.f. = 2, 4, n.s.). The same trend across sites was also evident when fecundity was estimated as the ratio of half of all juveniles to adult females. These differences were not statistically significant either as shown by two-way ANOVA (site: $F = 2.414$, d.f. = 2, 4, ns, year: $F = 0.486$, d.f. = 2, 4, n.s.). The average unweighted estimated juvenile:adult ratio across all sites and all years was $0.23 \pm 0.02SE$. The average unweighted estimated ratio of half juveniles to adult females was $0.26 \pm 0.02SE$. Fecundity estimates from half juveniles to adult females were probably more accurate measures though there was little difference between the two methods (Table 4.6); it was probably more accurate because it did not assume that adults had equal sex ratios. However it may still be biased, underestimating fecundity because foals are likely to have died between birth and the autumn surveys. No 2-year-old females were observed with young. Therefore it is likely that the age of first reproduction was three years.

Table 4.6: Estimated fecundity rates/yr for wild horse populations at each site each autumn calculated as the ratio of juveniles to adults. j = juvenile, ad = adult, half j = half of the juvenile population, ad f = adult females. Values in brackets are the sample size of the adult component of the ratio. Avg = average and SE = standard error.

	Big Boggy		Cowombat		Currango	
	j:ad	half j :ad f	j:ad	half j :ad f	j:ad	half j :ad f
2000A	0.27 (59)	0.25 (32)	0.15 (33)	0.17 (15)	0.37 (43)	0.38 (21)
2001A	0.20 (56)	0.22 (25)	0.21 (43)	0.19 (24)	0.20 (46)	0.28 (16)
2002A	0.19 (53)	0.26 (19)	0.25 (36)	0.28 (16)	0.24 (72)	0.27 (31)
Avg	0.22	0.24	0.20	0.21	0.28	0.31
SE	0.03	0.01	0.03	0.04	0.04	0.03

The number of animals that I was 100% confident in identification was much lower at Cowombat and Currango because horses were harder to approach than at the Big Boggy (Table 4.7). The basic time-dependent survivorship model adequately fitted the Big Boggy 90% and 100% confidence data (Table 4.7). Therefore there is unlikely to be variability in survival rates between different age classes and sexes. The high over-dispersion coefficients for Cowombat analyses at 90% and 100% confidence indicate that the basic time-dependent data lacked sufficient structure to explain the variability in the data (Table 4.7). For Currango, the basic time-dependent survivorship model adequately fitted the 90% confidence data (Table 4.7).

Table 4.7: Summary of basic time-dependent survival analyses for wild horse populations executed in program MARK at each site at 90 and 100% confidence levels. n = number of wild horses included in analysis, and c-hat is the over-dispersion coefficient (refer to methods for two calculations of c-hat). The global model for all analyses was $\emptyset(t)p(t)$ and the number of estimable parameters was 9. C-hat estimates >3 (*) indicate that the model had inadequate fit. There were insufficient animals included in the Currango 100% confidence data set to run analyses (**). QAICc values for the basic model $\emptyset(.)p(.)$ are presented for each analysis.

Site		n	c-hat 1	c-hat 2	QAICc
Big Boggy	90%	81	1.93	2.55	176.62
	100%	56	1.24	1.92	87.66
Cowombat	90%	101	2.35	3.92*	156.99
	100%	33	1.77	5.17*	135.37
Currango	90%	52	1.52	1.68	127.92
	100%	10	**	**	**

A comparison of estimated annual survival rates between 90 and 100% confidence data at the Big Boggy show little difference in estimates (Figure 4.9). There is also little variation in estimated survival rates over time. The range of annual survival estimates for 90% confidence data was 0.89-0.90, and for 100% confidence data was 0.88-0.93. There was substantial overlap in standard error bars for all of these estimates, indicating that there would be no significant difference between the means.

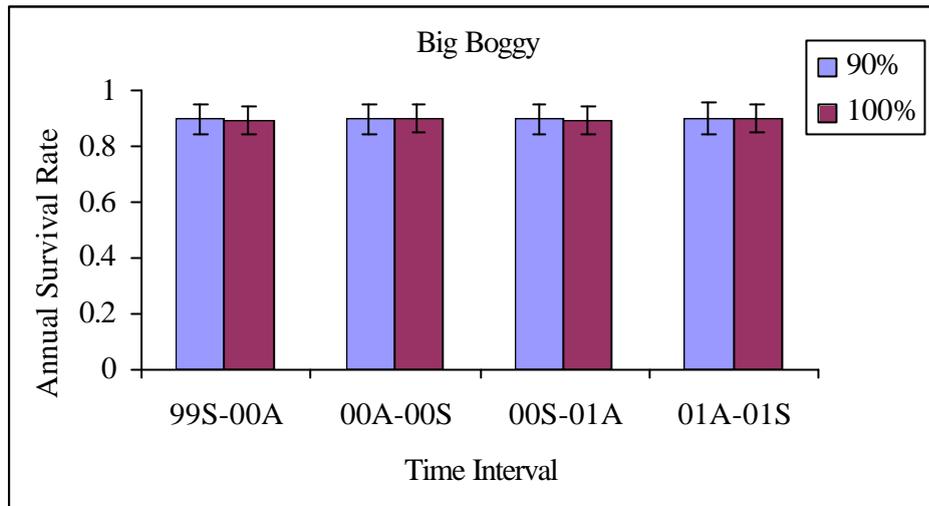


Figure 4.9: Estimated annual survival rates of wild horses at the Big Boggy for 90% and 100% confidence data between primary periods (e.g. 99S-00A is spring 1999 to autumn 2000). Estimates were made using the basic time dependent model. Error bars are standard errors.

Annual survival rates at Currango based on the basic time dependent model were similar to annual survival rates at the Big Boggy ranging from 0.90-0.92 with no significant variation over time (Figure 4.10).

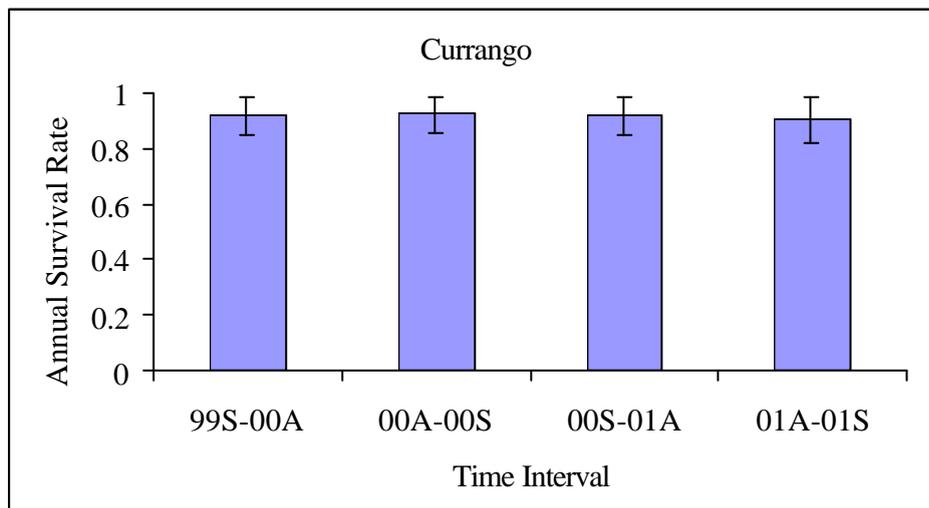


Figure 4.10: Estimated annual survival rates of wild horses at Currango for 90% confidence data at each time period. Estimates were made using the basic time dependent model. Error bars are standard errors.

Grouping data by sex and age gave good model fit for 90% confidence data at all three sites (Table 4.8). Good model fit was also evident for Big Boggy 100% confidence data. The poor model fit for Cowombat 100% confidence data excluded its use in parameter estimation.

Table 4.8: Summary of analyses by sex and age for wild horses executed in program MARK at each site at 90 and 100% confidence levels. n = number of wild horses included in analysis, np = number of estimable parameters in the global model, c-hat is the overdispersion coefficient calculated in two ways. C-hat estimates >3 (*) indicate that the model had inadequate fit.

Site	Group	n	np	c-hat 1	c-hat 2	Global Model	
Big Boggy	90%	Sex	77	12	1.47	1.73	$\emptyset(\text{sex})p(\text{sex.t})$
		Age	78	9	1.47	1.73	$\emptyset(\text{cohort})p(t)$
	100%	Sex	55	12	1.22	1.92	$\emptyset(\text{sex})p(\text{sex.t})$
		Age	52	8	1.19	1.48	$\emptyset(\text{cohort})p(t)$
Cowombat	90%	Sex	74	12	1.50	2.20	$\emptyset(\text{sex})p(\text{sex.t})$
		Age	92	9	1.72	1.44	$\emptyset(\text{cohort})p(t)$
	100%	Sex	27	7	1.47	3.72*	$\emptyset(\text{sex})p(t)$
		Age	32	7	1.55	8.68*	$\emptyset(\text{cohort})p(t)$
Currango	90%	Sex	45	12	1.31	1.67	$\emptyset(\text{sex})p(\text{sex.t})$
		Age	51	9	1.47	1.93	$\emptyset(\text{cohort})p(t)$

There was no consistent trend in differential survivorship between the sexes (Table 4.9). Males had a slightly higher survival rate than females at Big Boggy and Currango, while females had higher survival rates than males at Cowombat. None of these trends are statistically significant as evident by the overlap in 95% confidence intervals (Table 4.9). Some of the variation in estimates is due to model averaging.

Estimated adult survival rate was the same across all sites at 0.91, however variation in estimates was higher at Cowombat and Currango than Big Boggy (Table 4.9). 95% confidence intervals included a survival rate of 1 at Currango but not at Cowombat or Big Boggy. Survival of 0-2 year olds varied across sites. It was high at Big Boggy, but considerably lower at Cowombat and Currango. The lower survival rates were not different

statistically as evident by the overlap in 95% confidence intervals. Large sample sizes that accommodate more complex models are required to explain the high variability.

Table 4.9: Weighted average annual estimated rates of survival of wild horses by age and sex (95% confidence intervals calculated in program MARK) for each site at 90 and 100% confidence. No estimates were made for 100% confidence at Cowombat because data were over-dispersed (Table 4.8) or at Currango because of insufficient data.

		Big Boggy		Cowombat	Currango
		90%	100%	90%	90%
Sex	Female	0.91 (0.76-0.97)	0.90 (0.71-0.97)	0.93 (0.51-0.99)	0.91 (0.61-0.98)
	Male	0.93 (0.80-0.98)	0.92 (0.75-0.97)	0.90 (0.57-0.98)	0.94 (0.49-1.00)
Age	0-2	0.90 (0.73-0.97)	0.92 (0.72-0.98)	0.83 (0.32-0.98)	0.86 (0.41-0.98)
	>2	0.91 (0.73-0.97)	0.91 (0.78-0.96)	0.91 (0.68-0.98)	0.91 (0.63-0.98)

Recapture probabilities were higher at Big Boggy than at Cowombat or Currango (Figure 4.11). Recapture probabilities were high at Big Boggy averaging 0.85 at 90% confidence and 0.93 at 100% confidence. Average recapture probabilities at 90% confidence at Cowombat and Currango were 0.56 and 0.62 respectively.

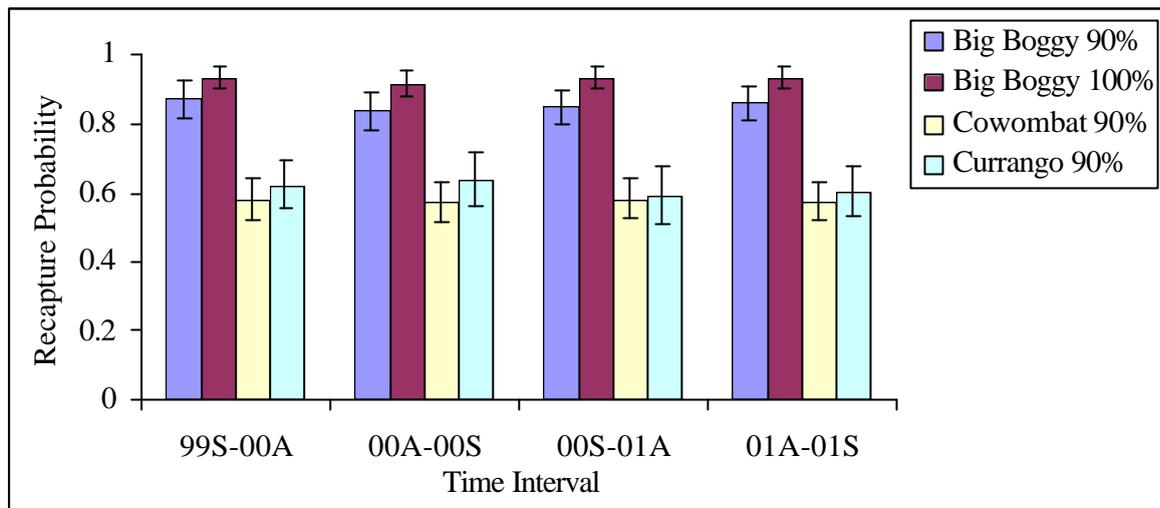


Figure 4.11: Recapture probabilities of wild horses for each site over each time interval. Probabilities are taken from the basic time dependent models for Big Boggy and Currango and from the age model for Cowombat. Error bars are standard errors.

4.3.3. Demographic models and sensitivity analysis

The average estimated rate of increase (I) of the three wild horse populations was 1.06. The Big Boggy and Cowombat populations were stable and the Currango population was increasing but below the maximum intrinsic rate of increase (Table 4.10). Cowombat had the lowest rate of increase and Currango had the highest. The coefficient of variation for estimated population growth was low ranging from 3.4 to 4.5% (Table 4.10). Recruitment ($l_3 m$) was estimated to be lowest at Cowombat and highest at Currango, with Big Boggy intermediate between the two (Table 4.10).

Table 4.10: Summary of estimated annual rates of increase for wild horse populations using Lande's (1988) demographic model based on demographic parameters estimated at the three sites. Data are presented for 90 and 100% confidence levels as described in the methods. Fecundity (m) is taken from Table 4.7. Survival rates (s and l_3) are taken from Table 4.9.

Site		m	s	l_3	$l_3 m$	I	CV (%)	95% CI
Big Boggy	90%	0.24	0.91	0.75	0.18	1.07	3.5	0.98 – 1.14
	100%	0.24	0.91	0.76	0.18	1.07	4.5	0.97 – 1.16
Cowombat	90%	0.21	0.91	0.63	0.13	1.03	3.4	0.96 – 1.10
Currango	90%	0.31	0.91	0.67	0.21	1.09	4.2	1.04 – 1.14

Analysis of the sensitivity of finite rate of increase (λ) to changes in life history parameters found the most influential parameter was adult survival (s) at all sites (Table 4.11). The next most influential parameter was fecundity (m), which was given half the weight of adult survival. Survival for the first three years of life (l_3) has less influence receiving only one fifth of the weighting of adult survival. Age of first reproduction (a) had a negligible effect on population growth rate. Generation time (T) was estimated at between 8 and 11 years (Table 4.11).

Table 4.11: Sensitivity of finite rate of increase of wild horse populations to change in life history parameters. Parameters are as described in the text.

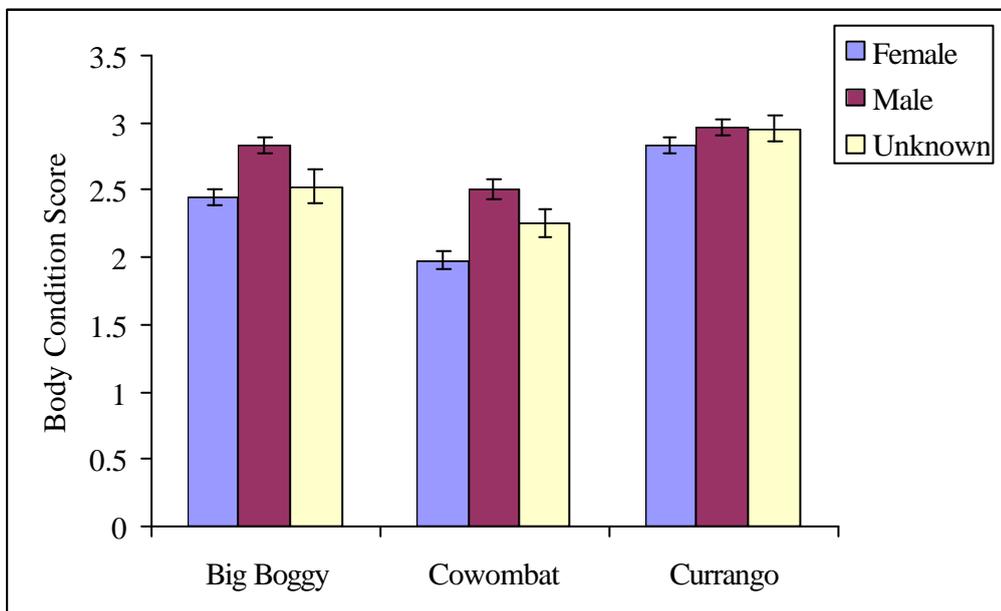
Site		T	l_3	m	s	a
Big Boggy	90%	8.9	0.16	0.50	0.91	-0.02
	100%	8.7	0.16	0.51	0.91	-0.02
Cowombat	90%	10.6	0.15	0.46	0.92	-0.01
Currango	90%	8.2	0.20	0.43	0.90	-0.02

4.3.4. Body condition and pasture biomass

Mean body condition was significantly different between sites, sexes, year and season, and for interactions between two variables for all interactions except year*sex and season*sex (Table 4.12). The mean body condition of wild horses was lowest at Cowombat (2.25+/-0.05SE), and highest at Currango (2.92+/-0.04SE). Mean body condition at the Big Boggy was 2.60 (+/-0.05SE). Females had poorer body condition than males and animals of unknown sex had body condition scores between males and females (Figure 4.12). The interaction between site and sex was significant because Currango showed a different pattern to Big Boggy and Cowombat (Figure 4.12). Female body condition at Currango was high and not different from male body condition (as evident from the overlap of the 95% confidence intervals). The difference in body condition between males and females was greatest at Cowombat.

Table 4.12: Summary of repeated measure analysis of variance results of wild horse body condition.

	d.f.	Sum Squares	Mean square	F	Pr>F
Site	2	33.70	16.85	49.63	<0.0001
Sex	2	16.17	8.09	23.82	<0.0001
Season	1	9.69	9.69	40.09	<0.0001
Year	3	6.64	2.21	9.16	<0.0001
Horse (site*sex)	526	178.56	0.34	1.40	<0.0001
Site*sex	4	4.35	1.09	3.20	0.0129
Site*season	2	4.20	2.10	8.69	0.0002
Site*year	6	8.31	1.38	5.73	<0.0001
Season*sex	2	1.29	0.65	2.67	0.0702
Season*year	1	3.60	3.60	15.37	0.0001
Year*sex	6	2.49	0.42	1.72	0.1147

**Figure 4.12:** Body condition scores for male, female and unknown sexed wild horses at Big Boggy, Cowombat and Currango (with standard error bars).

Horses were in poorer condition in spring than in autumn but the trend was different across sites (Figure 4.13). The difference in body condition of horses between seasons was most marked at the Big Boggy, while it was not significantly different at Currango (as evident by the overlap in the 95% confidence intervals).

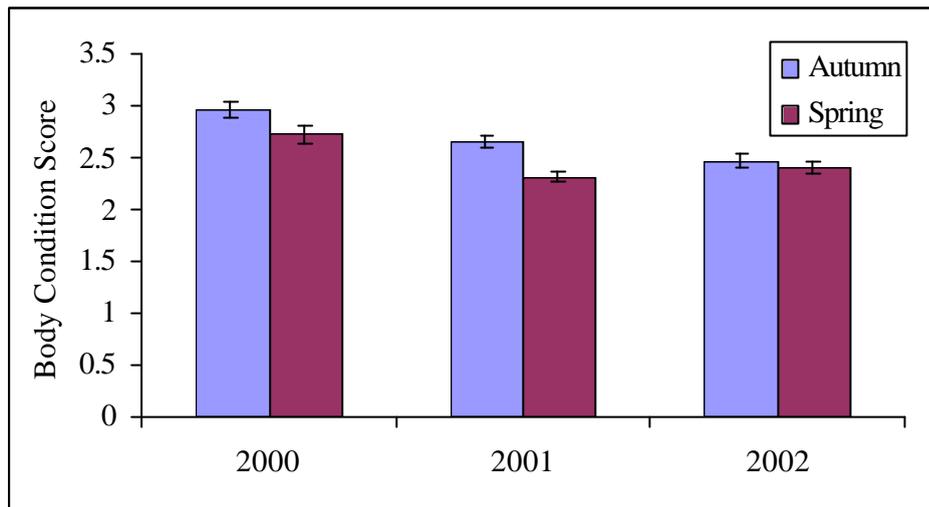


Figure 4.13: Body condition of wild horses at Big Boggy, Cowombat and Currango in spring and autumn (with standard error bars).

Body condition declined over the duration of the study at all sites (Figure 4.14). The trend was not consistent between sites. At the Big Boggy and Currango, body condition was higher in 2001 than in 2000 and the opposite was the case at Cowombat. The greatest difference in mean annual body condition for one site was between 1999 and 2002 for the Big Boggy.

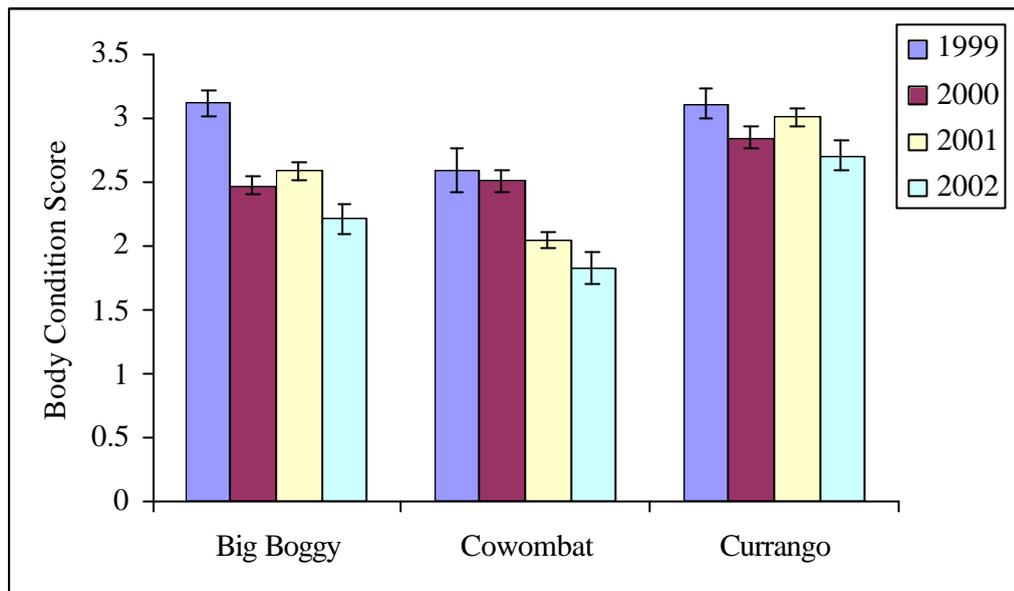


Figure 4.14: Body condition of wild horses at Big Boggy, Cowombat and Currango from 1999 to 2002 (with standard error bars).

The mean rank of pasture biomass for each site and season shows that Cowombat had the lowest biomass and Currango had the highest (Table 4.13). This is the same trend observed in body condition of horses. These differences between sites were significant as shown by 2-way ANOVA ($F = 8.53$, d.f. = 2,8, $p = 0.01$). Tukey-Kramer pairwise comparison among the three sites revealed that there was a significant difference in pasture biomass rank between Currango and Cowombat ($p = 0.01$). However, there was no significant difference between the Big Boggy and the two other sites (Cowombat $p = 0.11$, Currango $p = 0.22$). There was no significant difference in rank for each time period ($F = 0.62$, d.f. = 4,8, $p = 0.66$).

Table 4.13: Mean pasture biomass ranks (+/- standard error) at each site between autumn 2000 and autumn 2002.

	Autumn '00	Spring '00	Autumn '01	Spring '01	Autumn '02
Big Boggy	2.63 (0.11)	2.07 (0.12)	2.64 (0.96)	2.12 (0.14)	1.71 (0.09)
Cowombat	1.78 (0.14)	1.67 (0.09)	1.85 (0.10)	1.82 (0.11)	1.71 (0.12)
Currango	2.28 (0.10)	2.32 (0.08)	2.78 (0.09)	2.58 (0.08)	3.06 (0.10)

The slopes of calibration regression lines were not significantly different at any sites (Table 4.14). The elevation of regression lines were not significantly different at the Big Boggy or at Currango therefore it is justified to pool data across sampling periods and use one calibration curve. The elevations of regression lines were significantly different at Cowombat (Table 4.14), so pooling across regression lines was not advisable. Biomass estimates at Cowombat are presented for Cowombat on three occasions using separate regression lines. Summaries of calibration curves are presented in Table 4.15.

Table 4.14: Summary of results from analysis of covariance of all pasture biomass calibration curves at each site. Tests for differences between slope and elevation for each site are summarised. * indicates significant difference where $\alpha=0.05$.

Site	Slope			Elevation		
	d.f.	F	P	d.f.	F	P
Big Boggy	1	0.02	0.89	1	3.91	0.06
Cowombat	2	0.09	0.92	2	9.17	0.001*
Currango	2	0.53	0.59	2	0.60	0.56

Table 4.15: Summary of calibration curves for pasture biomass estimates used at each site. B = pasture biomass in g/0.25m² quadrat and Rank is the rank given to the quadrat.

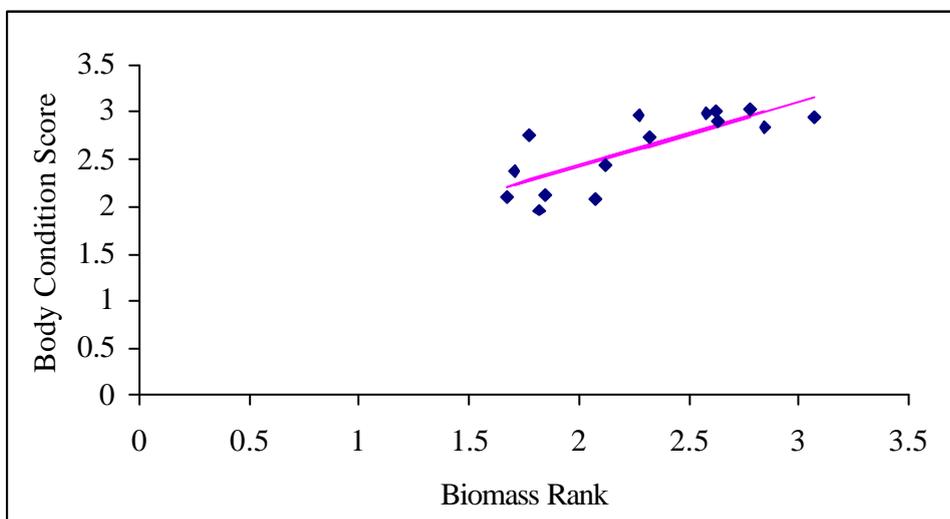
Site	Description	Equation
Big Boggy	Pooled across site	Log B = 1.125 + 2.079*Log Rank
Cowombat	Autumn 2000	Log B = 1.114 + 1.577*Log Rank
	Spring 2000	Log B = 1.223 + 1.577*Log Rank
	Autumn 2001	Log B = 1.375 + 1.577*Log Rank
Currango	Pooled across site	Log B = 1.089 + 2.274*Log Rank

Mean estimates of standing biomass at the Big Boggy ranged from 3474 - 5763 kg/ha (Table 4.16). A similar range was recorded at Currango of 3132 - 5840 kg/ha. The range of estimates at Currango for autumn 2000, spring 2000 and autumn 2001 using individual calibration curves was 1388 - 2760 kg/ha.

Table 4.16: Summary of estimated standing pasture biomass (kg/ha) at the three sites from autumn 2000 to autumn 2002 with standard errors.

	Autumn '00	Spring '00	Autumn '01	Spring '01	Autumn '02	Avg
Big Boggy	4739 (429)	3474 (320)	4972 (430)	3918 (370)	5763 (375)	4573
Cowombat	1388 (172)	1678 (130)	2760 (236)	-	-	1942
Currango	3132 (280)	3441 (249)	4833 (305)	4169 (262)	5840 (370)	4283

There was a significant positive relationship ($r = 0.783$) between pasture biomass rank and body condition score for all sites and seasons pooled ($R^2 = 0.61$, $F = 20.56$, d.f. = 1, 13, $p = 0.001$) (Figure 4.15). The power function had a slightly poorer fit ($R^2 = 0.60$, $F = 19.74$, d.f. = 1, 13, $p = 0.001$). There was also a significant positive correlation ($r = 0.557$) between estimated pasture biomass and body condition for all sites and seasons pooled but the correlation was not as strong, and the slope was not as steep ($R^2 = 0.31$, $F = 5.05$, d.f. = 1, 11, $p = 0.05$) (Figure 4.16). The power function had a poorer fit with a non-significant relationship ($R^2 = 0.25$, $F = 3.68$, d.f. = 1, 11, $p = 0.08$).

**Figure 4.15:** Body condition of wild horses and pasture biomass rank for all sites and all seasons when both condition and biomass were measured.

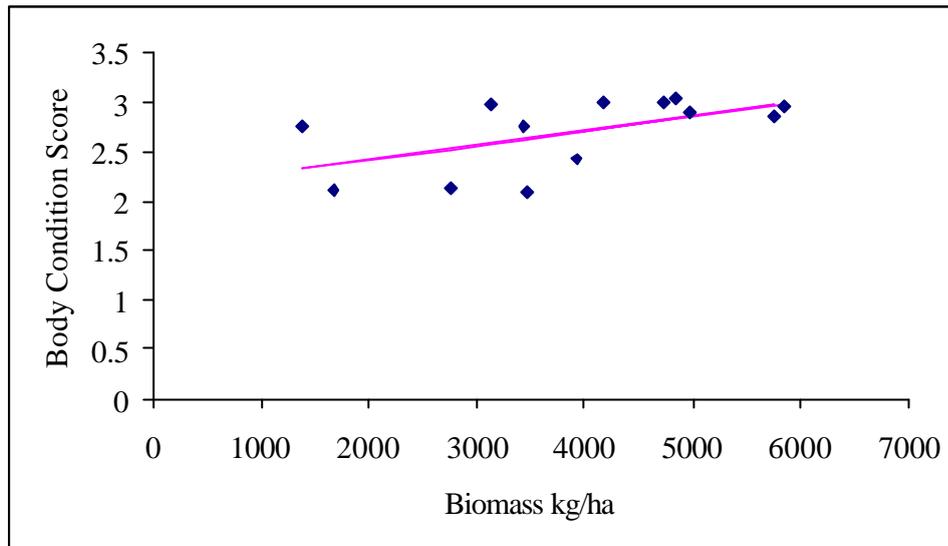


Figure 4.16: Body condition score of wild horses and pasture biomass (kg/ha) for all sites and all seasons when both condition and pasture biomass were measured.

There is a trend across the three sites consistent with the food limitation hypothesis presented in section 4.1.4 (Table 4.17). The Cowombat wild horse population is most likely to be food limited with the lowest rate of increase (estimated from demographic data), horses with the poorest body condition and least available food as measured by pasture biomass. The Currango wild horse population is least likely to be food limited with a positive growth rate, horses in the best condition and the highest pasture biomass rank (Table 4.17). The Big Boggy is intermediate between the two sites.

Table 4.17: Summary of demographic parameters, body condition and pasture biomass rank at the three sites.

Parameter	Cowombat	Big Boggy	Currango
Annual I	1.03	1.07	1.09
Fecundity	0.21	0.24	0.31
Recruitment	0.13	0.18	0.21
Annual adult survival	0.91	0.91	0.91
Mean body condition	2.25	2.60	2.92
Mean biomass rank	1.77	2.23	2.60

4.3.5. Numerical response

As horse abundance at the Big Boggy increases per unit pasture biomass the annual finite rate of increase (I) decreases linearly as estimated by the numerical response (Figure 4.17). It was assumed that $r_m = 1.2$ per year, the mean number of horses was 72, mean pasture biomass was 4573, so $a = 0.01575$. Hence the equation is $I = 1.20 - 12.698(H/V)$. An alternative way of viewing the same relationship shows that as *per capita* food availability increases then I increases but in a curved manner (Figure 4.18), reaching an asymptote at the intrinsic rate of increase. At intermediate levels of food availability the horse population is stable ($I = 1.0$) (Figures 4.17 & 4.18).

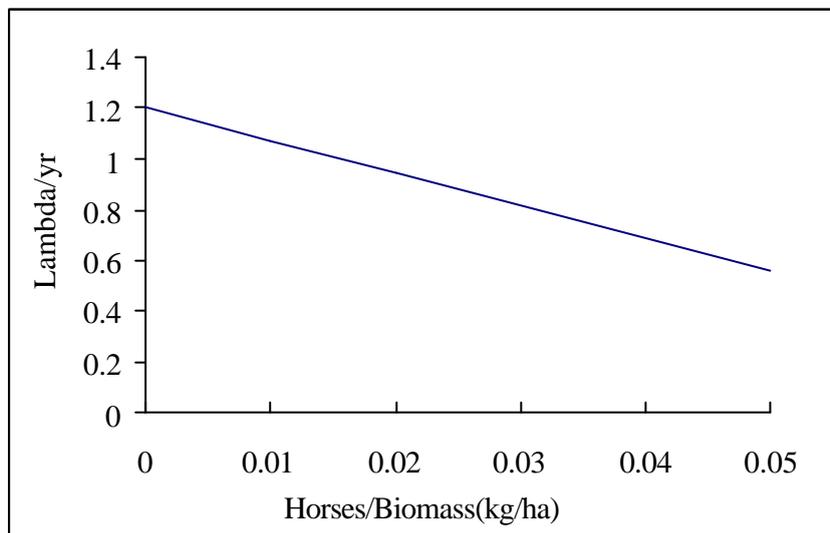


Figure 4.17: Predicted annual rates of increase (I) in the Big Boggy horse population at various levels of wild horses per unit pasture biomass.

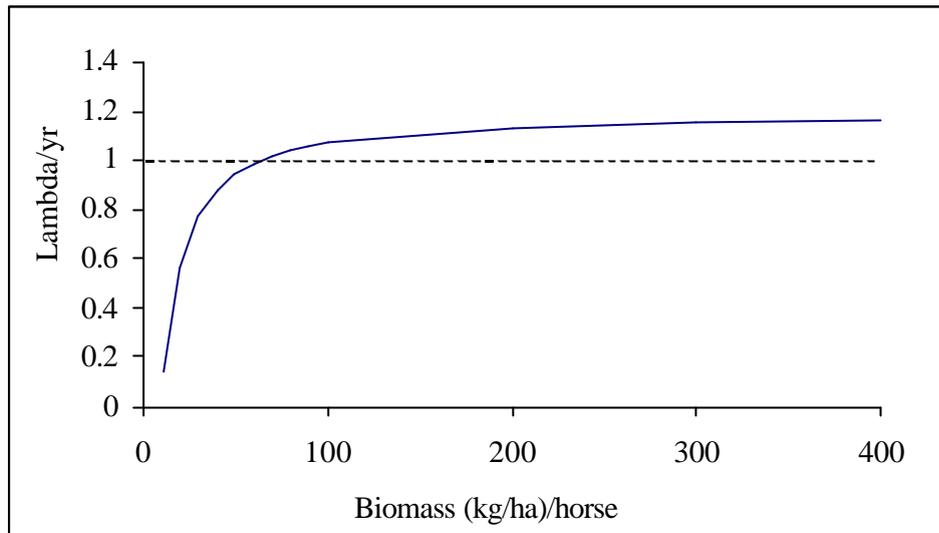


Figure 4.18: The numerical response of wild horses where annual rate of increase (λ) of the wild horse population is a function of pasture biomass (kg/ha) per horse. Calculations are based on the Big Boggy population. $\lambda = 1$ represents a stable population (dashed line).

4.4. Discussion

The population dynamics and demography of large mammalian herbivores typically have the following features- low density, low population growth rates, low annual fecundity, high annual adult survival and lower juvenile survival (Caughley 1976, Gaillard 1998, 2000). Also, populations are usually limited by food (Sinclair 1989, 1996). Wild horse populations in the Australian Alps show dynamics and demography consistent with these patterns.

4.4.1. Abundance and density

The average wild horse density at the Big Boggy and Currango (2.01 and 2.13 km⁻² respectively) fall within the 95% confidence intervals of density estimates for the whole of the Australian Alps national parks based on aerial survey (line transect analysis for both observers combined: 1.04 – 3.10 horses/km²) (see Chapter 3). The average density at Cowombat (6.40 horses/ km²) did not fall within the 95% confidence intervals estimated from the aerial survey. When density estimates are made without boundary strips as occurred here, estimated density is greater than true density, and this problem becomes greater as the area surveyed becomes smaller (Krebs 1999). At 12.7km², the Cowombat site was much smaller than either of the other two sites (35.5km² and 41.4km²).

Estimates of wild horse densities from other studies cover a wide range. Several studies found densities below 1 horse/km² in the USA and Australia (Eberhardt *et al.* 1982, Skeat 1990, Black 2000). Very high densities of horses (>20 km⁻²) have been observed in France (Duncan 1992), the Netherlands (Koene pers. comm.) and Argentina (Scorolli pers. comm.). Densities of wild horses similar to the present studies were observed in New Zealand (2.8 – 5.2 km⁻²) (Cameron *et al.* 2001) and the USA (0.2 – 11.0 km⁻²) (Rubenstein 1981).

The seasonal trends in abundance at Big Boggy and Cowombat and the variable estimates at Currango are consistent with horses moving in and out of the study areas. Seasonal movement was expected at the Big Boggy because the area receives considerable, but variable, snow in winter (see 4.2.1). Berger (1986) observed seasonal movements across an altitudinal range in the response to winter snow. Seasonal movement at Cowombat is

unlikely to be due to snow because it covers a large altitudinal range and does not receive a lot of snow. Seasonal movements are more likely to be due to food availability. Movement was not measured in the current study because of logistical constraints so the movement effect could not be quantified. It is recommended that future research should address this topic. Seasonal trends at Big Boggy are also consistent with seasonal breeding and no movement; peak abundance at the end of the breeding season.

Population estimates at each primary period were based on assumptions of a closed population model. These assumptions are: 1) the population is closed, 2) all animals are equally likely to be captured in each sample, and 3) marks are not lost, gained or overlooked (Lancia *et al.* 1996). The first assumption usually can be met if the interval between samples is short (Lancia *et al.* 1996). There are two components to consider when evaluating the closed population assumption. The first is births and deaths and the second is immigration and emigration. Births (3) and deaths (1) were observed during surveys in the current study. When they were observed the animal was excluded from estimates. Since mortality rates were low (section 4.3.2), and the number of horses born in a population is small during that time span and spread out over time, births and deaths are unlikely to have biased estimates in the ten days within which each survey was carried out.

The assumption of no immigration or emigration is likely to have been violated because horses were observed moving into and out of the survey area on a number of occasions. The movement appeared to be random. Some animals took flight from observers but were typically observed back in the study area before the survey was over. Completely random movement into and out of the study area does not introduce bias, although it decreases precision (Kendall 1999).

The assumption that all animal are equally likely to be captured was unlikely to be met because horses occur in groups of different sizes and capture probability is related to group membership. Heterogeneity in capture probabilities leads to negative bias in population estimates (Lancia *et al.* 1996). This assumption was not tested so bias cannot be measured, but surveys were designed to minimise bias. Surveys were thorough with each marking and

capture event lasting three days. Running surveys for this long resulted in multiple sightings of conspicuous animals, while numbers of new sightings dropped off considerably by the third day. It is believed that if any negative bias resulted from missing inconspicuous animals it would have been small.

The final assumption that marks are not lost, gained or overlooked can be difficult to satisfy when marks are natural. Several strategies were employed to ensure that the assumption was met. Firstly, only one observer made all observations, and most individuals were known for the duration of the entire study based on detailed photographic and descriptive records of markings. Secondly, group structure was generally stable over the duration of the primary surveys. Therefore, if the group or individuals within the group were difficult to identify, the group could be identified based on size and composition. There were occasions where groups were not sighted clearly or group composition appeared to change. On these occasions, 'best guesses' were made rather than excluding them from calculations. Using 'best guesses' prevented negative bias associated with excluding animals from calculations.

4.4.2. Population growth rate

Demographic modelling found that the population growth rate of wild horse populations at all sites was below the maximum intrinsic rate of growth (r_m) observed for wild horses by Eberhardt (1987), Garrott *et al.* (1991a) and Duncan (1992). This can be interpreted in two ways. The first interpretation is that the maximum intrinsic rate of increase for wild horses in the Australian Alps is below that in other areas. This could occur if the habitat was of poorer quality in the Australian Alps. The second and more likely alternative is that limiting factor(s) influence the wild horse populations in the Australian Alps.

The Big Boggy and Cowombat populations appear to be stable because 95% confidence intervals include $I = 1$. This is supported by the census results for the Big Boggy where annual estimates were remarkably constant at the Big Boggy. Wild horses in the region of the Big Boggy and Cowombat are part of a widespread population that were liberated over 150 years ago (Chapter 2) and have not been subject to any active management for decades (NSW National Parks & Wildlife Service 2002b). In contrast, the Currango population had a

positive exponential rate of increase. All wild horses were removed from Currango in 1963 and were in low numbers in surrounding areas (Ted Taylor pers. comm.). High rates of increase were observed in other studies of wild horse populations that were either recently liberated (Duncan 1992) or had large levels of population control prior to the study (for example Garrott *et al.* 1991a). The results for Currango and Cowombat should be viewed with some caution because they are based on 90% confidence in identification of animals.

Demographic models returned estimates of population growth rate that are more representative of the entire population than those estimated by census methods. The census method was clearly affected by movement of animals into and out of study areas. That is, population growth rate estimates from census methods were a result of immigration and emigration rather than just *in situ* recruitment. This was clearly evident at Currango where on two occasions (spring 99-00, and autumn 00-01) observed annual r was above the highest r_m value (0.24) recorded for wild horses (Duncan 1992). The Currango study area was a large plain and only incorporated a small component of the adjacent woodland. Wild horses were often observed on the margins of the study area. The plain probably only constituted part of the home range of many groups. Movement into and out of the Big Boggy and Cowombat sites also occurred but probably not to the same extent because the areas were better defined by landscape boundaries, which restricted movement of horses such as rocky ridges. The population growth rate estimates obtained from demographic modelling are more robust to movement of animals, because they are based on apparent survival estimates. For further discussion see Section 4.4.3.

The preferential use of either demographic modelling and census methods depends on the study and the study animal and how well model assumptions can be met. The best approach initially is to use both methods if possible and compare results. Demographic modelling and census methods can return similar population growth rate estimates as found for the northern spotted owl (Lande 1988, Burnham *et al.* 1996), but this is not always the case. Hoyle *et al.* (2001) showed that demographic modelling gave lower estimates of population growth rate than census methods based on mark recapture; they believed the mark-recapture estimates were superior because demographic modelling underestimated survival. Demographic

models were also found to underestimate population growth rates of snowshoe hares (*Lepus americanus*) for similar reasons (Haydon *et al.* 1999).

The precision of population growth rate estimates based on demographic models is dependent upon the precision of the estimates of demographic parameters. Obtaining precise estimates of population growth rates has been a challenge (Taylor & Gerrodette 1993, Eberhardt *et al.* 1994). Taylor and Gerrodette (1993) have shown that it is difficult to obtain variances of population growth rate estimates with adequate statistical power. Statistical power can be improved by maximising the number of animals marked and working with relatively small populations (Taylor & Gerrodette 1993). It would be difficult to improve statistical power in population growth rate estimates for wild horses in the Australian Alps because capture rates were high and populations were relatively small. Some improvement may be achieved by marking animals for better identification; however, this would be very costly. An alternative would be to use census methods over a larger area because statistical power improves with larger sample sizes for census methods (Taylor & Gerrodette 1993). The estimate of variance of population growth rates may be an under-estimate because it uses only sample size and assumes a binomial distribution in program MARK.

4.4.3. Survival and fecundity

The fecundity estimates for the current study (0.26 \pm 0.02SE) fall within the range of estimates from other wild horse studies summarised in Table 4.1 (range from 0.18 to 0.48 half juveniles/female/year). In a detailed study on reproduction in horses Garrott *et al.* (1991b) found that fecundity was at the upper end of these estimates with 80 – 90 percent of prime aged females foaling. High and relatively stable fecundity is expected for prime aged female ungulates (Gaillard *et al.* 2000). The lower fecundity estimates in the current study are likely to be due to a combination of two factors. Firstly, some foals die in their first month of life (Goodloe *et al.* 2000, Cameron *et al.* 2001). Surveys were done at the end of the foaling season, and foals that died prior to the survey would not have been sampled. Secondly, Duncan (1992) and Scorolli (2001) observed a reduction in fecundity in horses due to food limitation, where fecundity was closely related to body weight (Duncan 1992). In the current study, the trend in fecundity across sites (highest at Currango and lowest at

Cowombat, Table 4.6) corresponded to the trend in body condition of females across sites (Figure 4.13). This suggests that fecundity levels observed in wild horses in the Australian Alps are influenced by the body condition of females, which is related to food availability.

Adult survival in large herbivores is typically high and shows little year to year variation (Sinclair 1996, Gaillard *et al.* 2000). The current study was not an exception with annual adult survival estimates (0.91/year) at the low end of the range reported in other wild horse studies (0.90 – 1.00/year, Table 4.1). Juvenile survival in large herbivores shows strong variation annually compared to adult survival (Gaillard *et al.* 2000). This trend was observed for horses in the Australian Alps where survival estimates for the first two years of life had a greater range than adult survival. The rates estimated in the Australian Alps were similar to most of those from other studies (Table 4.1). The exceptions being the very low rates of survival of 0-1 year-olds (0.27 and <0.12) observed by Turner *et al.* (1992) and Greger and Romney (1999) respectively where mountain lions preyed on young horses. The estimates from the current study are likely to be over-estimates for first year survival because surveys were conducted in autumn, therefore mortality in younger animals born early would have gone undetected.

Studies have found males of some species of birds and mammals have lower survival rates than females (Clutton-Brock *et al.* 1985 and references therein). However no significant differences in survival between the sexes were observed for wild horses in the Australian Alps and there was no consistent trend across the sites (Table 4.10) (adult sex ratios are reported and analysed in Chapter 5). The literature on sex-specific survival of wild horses is mixed. Cameron *et al.* (2001) found that there was no significant difference in survival between adult (>4 years old) male and female wild horses, while female yearlings and sub-adults had significantly higher mortality rates than males. In contrast, Garrott and Taylor (1990) found that male wild horses had significantly lower survival rates than females in all years except extreme winters when survival rates were not significantly different between the sexes. These results suggest that male wild horses do not suffer higher mortality due to winter weather than females as Clutton-Brock *et al.* (1985) proposed for red deer.

Survival analysis was limited in the current study due to small sample sizes. Some of the variability in survival estimated in the current study (Table 4.9) may have been explained by more detailed models. It would be advantageous to explore other effects that may have influenced survival in the future including season. In seasonal environments like the Australian Alps, mortality is likely to be seasonal especially at high densities. Survival rates of juvenile large herbivores are often affected by seasonal limitation in food supply (Gaillard *et al.* 1998).

The first assumption that may have been violated in survival estimation is that survival and capture probabilities within each sampling period are homogenous (Pollock *et al.* 1990). It is unlikely that this assumption was met in the current study for several reasons. Primarily, horses occur in groups, and each group behaved differently to the presence of an observer (some groups took flight, so could not be identified for inclusion in survivorship analysis). However, violation of this assumption produces relatively small bias in survival estimates (Carothers 1973), and becomes relatively unimportant when capture probabilities are high (Pollock *et al.* 1990). Therefore it is unlikely that failure of this assumption caused bias in the current study.

The second assumption that may have been violated is that marks are not lost or missed. If animals lose their marks, the numbers of recaptures will be low resulting in an underestimation of survival (Pollock *et al.* 1990). This was an issue in the current study. It was addressed by only including animals with distinctive natural markings in analyses at two levels: 90 and 100% confidence. At the Big Boggy there were sufficient data to compare results at these two confidence levels. Survival estimates were very similar suggesting that no bias results from using 90% confidence data as expected, only the confidence intervals became wider. Excluding animals from analysis (i.e. using 100% confidence data only) is equivalent to them not being captured. Therefore only a smaller sample of the population was used in survival analysis using 100% confidence data. It is believed that this sample was representative because their inclusion was based on natural markings which are not correlated with survival. Unfortunately there was insufficient data at the 100% confidence level for Currango to make the same comparison, and the models had poor fit for Cowombat

100% data (Table 4.7). The lack of fit for the Cowombat 100% data may have been due to sparse data because \hat{c} values estimated using the second method (as opposed to the first method) are often high with sparse data (White *et al.* 2002).

Movement of animals is unlikely to have caused bias in survival estimates. An implied assumption of the survival model is that emigration from the study area is permanent (Pollock *et al.* 1990). Temporary emigration does not bias results either if it is completely random (Kendall *et al.* 1997). Furthermore if temporary emigration is not random, estimates may be biased with respect to the immediate population, but not with respect to the super-population (animals which are associated with the area sampled) (Kendall *et al.* 1997). Temporary emigration does result in higher variance though (Kendall *et al.* 1997). Movement into and out of the study areas was observed but not measured in the current study. It is not known whether or not it was random. If movement was random, survival estimates would not have been biased but variances would be higher. If movement was not random, then survival estimates are representative of the super-population with increased variance.

The final assumption of the survival model is that all samples are instantaneous relative to the intervals between samples. This assumption was met by surveying bi-annually for ten days (see Figure 4.5 & Table 4.3).

In conclusion, it is possible that some of the survival estimation assumptions were violated in the current study. However, the effect of any of the violations will result in small bias or increased variance in estimates. Therefore, the survival estimates for this study are likely to be accurate despite being at the low end of those reported in the literature.

4.4.4. Sensitivity of finite rate of increase to changes in life history parameters

Finite rate of increase of wild horses was found to be most sensitive to changes in adult survival at all sites in the Australian Alps. This is in keeping with findings for long-lived species with low levels of reproduction (as reviewed in large herbivores by Gaillard *et al.* 1998, Eberhardt in press) and evident in birds (northern spotted owl, Lande 1988). This sensitivity to adult survival is apparent in horses whether the population is increasing, stable

or decreasing (Hone unpublished data). This is in contrast to feral pigs where population growth rate is most sensitive to adult survival only when the population is increasing (Hone unpublished data).

When populations are unmanipulated, the relative importance of a component of a population to population growth rate depends on the temporal variation of that component and its sensitivity (Gaillard *et al.* 2000). For example, in the present study adult survival has high sensitivity and little variability, so may have little impact on population growth rate in an unmanipulated population. In contrast, juvenile survival is more variable with lower sensitivity and may have greater importance in observed population growth rate in an unmanipulated population.

If the aim of management is to control wild horses, then sensitivity analysis shows that targeting adult survival is most important. The next most important area to target is fecundity and then survival of 0-3 year-olds. A manager's ability to manipulate these components will depend on many things such as difficulty in administering fertility control (for example Stafford *et al.* 2001). Compensatory responses in survival, fecundity and age of first reproduction in the population should also be considered in any management program. That is, when culling reduces density, survival and fecundity may increase and age of first reproduction decrease, as reported for feral donkeys (Choquenot 1991).

4.4.5. *Body condition and biomass*

The poorer body condition of females relative to males in the current study has been observed in other wild horse populations, and is associated with the energetic costs of lactation and/or reproduction (Berman 1991, Duncan 1992, Cameron *et al.* 1999). It has also been reported in other species including red kangaroos (Moss & Croft 1999). Female body condition and available pasture biomass were lower at Cowombat. This indicates that density-dependent food limitation is more likely to be affecting the population at Cowombat. The poorer female body condition will in turn affect the success of their young and hence recruitment. The growth and survival of young horses is dependent on the milk they receive from their mothers; larger mothers produce larger foals (Duncan 1992). In times of food

shortage, Choquenot (1991) found that high juvenile mortality of donkeys was associated with poor juvenile body condition and slower growth. Similar patterns occur in other ungulates (eg. wild reindeer Skogland 1985 and roe deer Gaillard *et al.* 1997).

The seasonal variation in body condition of wild horses observed in this study has been reported in other wild horse populations. Duncan (1992) observed that breeding mares were in best condition in autumn and worst in late winter. In temperate, seasonal environments, winter is the season when food limitation often operates (Duncan 1992, Gaillard *et al.* 1993) and this may occur in the Australian Alps. The high range in body condition between seasons at the Big Boggy (Figure 4.13) can be explained by the more severe climate at this site (due to its high elevation). Wild horses at the Big Boggy appeared to be subject to lower food availability in winter. They were frequently observed digging through snow to reach food on opportunistic winter visits. Winters are not as severe at the other two sites because they are at lower elevations.

There is no clear reason for the observed decline in body condition over the study. Declines in body condition over time have been observed in association with a reduction in available food (Duncan 1992). However, there was not a comparable decline in biomass in the current study. One possible explanation for the observed trend is sampling error. It is possible that there was a trend over time where lower body conditions were assigned to horses of the same condition. If this were the case, the relative differences in body condition between the sexes or sites would not be affected.

The correlation found between body condition and pasture biomass (Figures 4.15, 4.16) for wild horses in the Australian Alps have been observed in other studies. Such a relationship is associated with population regulation by food limitation (Choquenot 1991, Duncan 1992). Body condition of animals is a good indicator of the status of the population with respect to carrying capacity: where the condition of the animals declines, the closer they are to carrying capacity.

The summary of results presented in Table 4.17 is consistent with the food limitation hypothesis and suggest that the populations were limited by food at varying levels as predicted in Section 4.1.4 and reproduced below in bold in Table 4.18. There was a trend across sites with Cowombat showing the strongest evidence of food limitation and Currango the least.

State	r	Fecundity	Survival 0-2 y.o.	Adult Survival	Body Condition	Food Supply
Not Limited	0.2	High (0.4)	High (0.9)	High (0.95)	Good	High
Food limited	0	Reduced (≤ 0.3)	Reduced (< 0.9)	Possibly Reduced (≤ 0.95)	Poor	Low
Predator Limited	0	High (0.4)	Reduced (0.3 1y.o.)	High (0.95)	Good	High
Harvesting Limited	0	High (0.4)	Reduced (< 0.9)	Reduced (< 0.95)	Good	High

Table 4.18: Predicted key population parameters for wild horses and food supply under different forms of population limitation. Likely values in brackets are estimates taken from the literature (see Table 4.1). Rate of increase (r) is annual and fecundity is the number of females born per adult female per year.

4.4.6. Numerical Response

The estimated numerical response for the population at the Big Boggy (Figures 4.17, 4.18) is similar in shape to the numerical responses of predators eating voles (*Microtus* spp.) (Turchin & Hanski 1997), wolves (*Canis lupus*) eating deer and moose (Eberhardt & Peterson 1999) and ferrets (*Mustela furo*) eating rabbits (Barlow & Norbury 2001). Each study used the same model of the numerical response though estimated parameters in different ways. The method of estimating parameters used in the present study was the same as used in the vole study, of empirical point estimates of the two parameters, the intrinsic rate of increase and the ratio of food per horse at equilibrium. The estimated numerical response of horses is

preliminary and could be refined with further study. For parameter estimation, a long-term data set was not available and an experimental approach was not feasible. In such situations the method used provides approximate estimates, as discussed in general by McCallum (2000, p. 271). The numerical response is useful as it summarises the horse population responses to variation in *per capita* food availability, to complement the observed demographic responses in body condition and recruitment of the wild horses.

5. BRUMBY-RUNNING AS A FACTOR INFLUENCING WILD HORSE POPULATIONS: THE VICTORIAN ALPS CASE STUDY

5.1. Introduction

Brumby-running is a management option currently employed in the Victorian Alps. It is thought to have a limiting effect on the wild horse population but the effect has never been examined. Management of wild horses in the Australian Alps is not based purely in the context of an introduced species but also as one with cultural value (NSW National Parks and Wildlife Service 2002a). Brumby-running is an activity that began soon after the arrival of horses in the Australian Alps and is closely tied to the culture of wild horses in the area.

5.1.1. Brumby-running in the Australian Alps

Brumby-running is a highly skilled activity and involves horse riders chasing wild horses and roping them. Typically it involves a group of horse-riders working together to track and locate wild horses. Once horses are sighted, the riders make chase. The wild horses typically run along established trails and the riders pursue them until they are on the tail of the mob. A rider comes alongside a wild horse where the terrain and vegetation permit and ropes the horse (Figure 5.1). Once the wild horse is brought under control, a specially designed halter is put on. The horse is trained to lead and taken back to camp. Other riders often accompany brumby-runners but do not catch horses. They may help with the handling of the wild horses.

Brumby-running is closely tied to the cultural significance of the wild horses. Brumby-running was part of the inspiration for Banjo Paterson's poem "The Man from Snowy River" (1898), and remains a strong part of local folklore. In the area south of Jindabyne in NSW for example, Brian Pendergast recites how his father caught his first wild horse when he was 14, in 1895; Bill Mugridge recalls chasing wild horses in his early teens on the Pinch and Ingegongbee Rivers in NSW in the 1920s (pers. comms.). Similar tales are recited in many communities adjacent to the Australian Alps national parks. The culture extends to festivals and street and establishment names, and is a tourism attraction.



Figure 5.1: A wild filly roped by Bronwyn Edwards. Dean Backman is getting a halter ready to put on the filly.

Brumby-running and other activities have influenced wild horse populations in the Australian Alps since they were first introduced (Chapter 2). In the grazing era, there was a lot of interchange between domestic horses and wild horses. Domestic horses escaped and were let go. Free-running horses were removed by trapping, shooting, brumby-running and snaring, then trained as saddle horses and pack horses, used for dog food and sold for profit. These activities have been restricted in the past 50 years as land tenure has changed and access to areas has been reduced to meet conservation objectives.

The role of brumby-running across the Australian Alps national parks has varied because the area is so large and covers two states and a territory. Approaches to brumby-running in NSW and Victoria are summarised below. The ACT does not currently have wild horses, so brumby-running is not a management consideration. Kosciuszko National Park in NSW is in the process of writing a new plan of management that will be available in 2003-2004. Approaches to management of wild horses are being reviewed. Aerial shooting of horses was

recently banned from all NSW national parks and is currently not a management option (NSW National Parks & Wildlife Service 2002a). In southern Kosciuszko National Park, the NPWS introduced a system of licensed brumby-running in about 1970. However there were concerns raised about the activity in 1972 and the licensing was discontinued (NSW National Parks & Wildlife Service 2002a). There is anecdotal evidence that brumby-running was popular from the 1960s to 1980s, with local saleyards at Jindabyne making the activity economical. However with the declaration of the Byadbo and Pilot wilderness areas in 1982, recreational horse riding became illegal under the Wilderness Act in most of the wild horse range in southern Kosciuszko National Park. Some people continued to ride horses and run brumbies in these areas but with decreasing frequency. In 2001, the NSW NPWS began a public consultation process with the aim of addressing the management of wild horses in the alpine and adjacent area of Kosciuszko National Park (NSW National Parks & Wildlife Service 2002a). A draft plan of management was written in May 2002. The plan aims to trial brumby-running (roping) and trapping as methods of control. This draft plan is currently being reviewed and should be put into action by the end of 2002.

Recreational horse riding is still permitted in much of the wild horse range in northern Kosciuszko National Park. However, a permit is required to brumby-run and none are issued. The activity occurs illegally and remains unquantified. In northern Kosciuszko National Park, the impact of horse riding confounds the impacts of wild horses. Managers are currently reviewing recreational horse-riding policy, and the wild horse management policy.

In the Alpine National Park in Victoria, brumby-running was practiced under a permit system from 1992 to 2001. This has undergone a review and now runs under a contract. The Alpine National Park Management Plan (Department of Conservation & Environment 1992) estimated that prior to 1992, up to 100 brumbies per year were removed by brumby-runners in the Cobberas-Tingaringy unit. Brumby-running was regarded as a management tool for control of brumbies, not as a recreational activity and is only permitted in certain areas (Department of Conservation & Environment 1992). The plan aimed to evaluate the effectiveness of brumby-running as a control measure by the following actions: 1) estimate the number of wild horses required to be removed each year to maintain effective population

control, 2) issue target numbers for brumby-runners to remove, and 3) require brumby-runners to keep accurate records of horses removed. The first two actions are yet to be initiated. The Alpine Brumby Management Association (ABMA) has carried out the third action for the past four years. Only ABMA members are permitted to brumby-run in the Alpine National Park.

5.1.2. Brumby-runners and wild horses as predators and prey

Consideration of brumby-runners as predators and wild horses as prey is useful in understanding how brumby-running is currently influencing the wild horse population. Pople *et al.* (1998) used the same analogy for control of goats by aerial shooting. Predators have the effect of reducing the densities of prey animals. For example, densities of caribou (*Caribou tarandus*) populations with no wolf predation are two orders of magnitude higher than populations with high and constant predation (Seip 1991 in Caughley & Sinclair 1994) and red kangaroo and emu populations are higher when dingoes are scarce (Pople *et al.* 2000). Brumby-runners may play the functional role of a true predator. True predators kill their prey (Begon *et al.* 1990); brumby-runners remove them from the population, and may therefore have the same effect as a true predator.

Predator-prey theory relevant to this chapter covers two areas, firstly, the behaviour of the predators and the way this affects what and how much they consume, and secondly the consequences of predation in terms of the dynamics of the prey population (Begon *et al.* 1990). The dynamics of the brumby-runner population is not considered as brumby-runner abundance is controlled by many factors other than the abundance of their “prey” for food. When humans play the role of predators, their dynamics can be simplified because factors such as hunting permits regulate them (Caughley 1977a).

Aspects of predator behaviour combine to influence the population dynamics of their prey (Begon *et al.* 1990). Behaviour of brumby-runners may influence their efficiency at catching wild horses, and their selection for different animals within the population. Selective behaviour by a predator will have implications for the population dynamics of the prey. A classic example is evident with predation on musk-rat (*Ondatra zibethicus*) by mink (*Mustela*

vision) (Errington 1946). Mink killed musk-rats that were least likely to survive and reproduce: without territories, in poor condition or injured (the doomed surplus). Hence the effects of minks on the musk-rat population were far less than if they preyed randomly on the population.

Another aspect of predator behaviour relevant to the current project is strategies by social mammalian carnivores. It has been shown that lions (*Panthera leo*) (Caraco & Wolf 1975) and North American wolves have an optimal group size for foraging (Nudds 1978). Optimal group size is defined by a balance between the benefits of co-operative hunting and the energetic returns to individuals within the group. For example, a pack of 3-4 wolves benefit from co-operative success of capturing deer, while maximising the food available per individual (Nudds 1978). Smaller groups have less success as capturing animals, while individuals in larger groups are not likely to meet their energetic requirements from their portion of food. Brumby-runners may also have an optimal group size maximising the number of wild horses caught by co-operation.

5.1.3. Brumby-running in the context of wildlife harvesting and control

The limiting effect of a predator on a prey population is very similar to humans limiting wildlife populations by harvesting and control. Brumby-running can be viewed as a classic case of wildlife harvesting. The difficulty lies in determining whether it is a viable form of wildlife control. The central idea of wildlife harvesting is that offtake (or yield) is sustainable and does not jeopardise future yield (Caughley & Sinclair 1994). Wildlife harvesting is either for recreational, commercial or traditional reasons. Recreational harvesting is popular for a variety of reasons including quality of experience as much as meat or trophies. On the other hand commercial harvesting is for the end product, usually skins or meat (Caughley & Sinclair 1994). Another area of harvesting that has received interest is traditional hunting (for example Marsh *et al.* 1997).

Wildlife harvesting often leads to a decline in wildlife population size. When a stable population is exploited for the first time a reduction in population size is inevitable and if a growing population is harvested at a higher rate than its growth rate, it will also decline

(Sutherland 2001). Empirical examples of harvesting causing population declines include kangaroos (Sinclair 1977), white-tailed deer (Fryxell *et al.* 1991) and moose (Solberg *et al.* 1999). The challenges in wildlife harvesting have been the prevention of overexploitation (Ludwig *et al.* 1993, Lande *et al.* 1997) and optimising harvest (for example Kokko 2001, Saether *et al.* 2001).

The simplest harvesting strategy is to harvest (H) animals from the population at the rate equivalent to the rate the population would increase if it were not harvested (Caughley & Sinclair 1994). For example, if the population would increase at 10% per annum, then a sustained yield would be 10% of the population (Caughley 1977a). If the population is at half of its ecological carrying capacity, then yield is the maximum sustainable yield (MSY). However, if the harvest rate is above the MSY, then the population will eventually be driven to extinction (Caughley 1977a). It is inadvisable to harvest a population consistently at its MSY because it can easily lead to population collapse. If the population suffers from a natural decline under this strategy from an environmental event, the population will quickly be driven to a lower level (Caughley & Sinclair 1994). Thus, environmental variation reduces sustained yield (Beddington & May 1977, Bayliss 1989). Uncertainties of the biology of the species and difficulties in estimating parameters also make harvesting at MSY unadvisable (Sutherland 2001).

For large herbivores like horses, which are typically food limited (Sinclair 1989, 1996), a logistic growth model is used to calculate MSY (Caughley & Sinclair 1994). In general, MSY is fractionally higher than $r_m K/4$ (Caughley & Sinclair 1994). The logistic growth model and the modified generalised logistic growth model (Hone 1994) can be used with an additional term for offtake to predict the effect of harvesting. This offtake term (H) is equivalent to offtake by predators. Harvesting at MSY or at some other level below MSY (fixed-quota) is one of several harvesting systems. There are five other systems that can be used: free-for-all, fixed-effort, variable quota, fixed percentage or fixed escapement (sometimes called threshold harvesting) (Newton 1998). The last three systems have the advantage that they can prevent overexploitation, however, they require annual measurement of abundance.

Age-selective harvesting occurs in some species (for example moose, Solberg *et al.* 1999). Similarly, age-selective predation occurs in some predator-prey interactions. For example mountain lions prey on juvenile horses (Turner *et al.* 1992, Greger & Romney 1999), and coyotes (*Canis latrans*) kill more juvenile kit foxes (*Vulpes macrotis*) as density increases (White & Garrott 1999). Optimal harvesting strategies for moose involve higher harvest of young animals and adult males (Saether *et al.* 2001). Sensitivity analysis (as discussed in Chapter 4) can be used to determine the effect of age-selective harvesting or predation. Age-selective harvesting or predation has the effect of reducing the survival rate of an age class. That is, removing adults from a population is equivalent to decreasing adult survival, while removing young horses has the effect of decreasing juvenile survival. Sex-selective harvesting occurs for some species. Male saiga antelope (*Saiga tatarica*) are selected for based on their antlers and higher meat yields (Milner-Gulland 1994) and in kangaroos where males are larger and return more meat per individual. Optimal harvesting strategies for moose involve targeting males, whereas females should hardly be harvested (Saether *et al.* 2001).

The timing of harvest is another consideration in wildlife harvesting. It has an effect on the response of the population to the harvest in seasonal environments (Kokko 2001). The reason for this is that with seasonal breeders, reproduction and the majority of mortality tend to occur at different times of the year leading to annual fluctuations in population size. Consequently, removing an individual just prior to reproduction causes a larger reduction in the population than if the individual had been removed earlier. When mortality is density-dependent, population responses to time of harvest can be even more dramatic (Kokko 2001).

Wildlife control is similar to the concept of harvesting but is complicated by the objectives of the control (Caughley & Sinclair 1994). Where the objective of harvesting is to take a sustained yield, the objective of control is not in terms of pest animals removed, but the benefit derived from their removal (Caughley & Sinclair 1994). Control may aim to reduce or prevent predation of livestock or threatened species, the spread of infectious disease, loss of crop production or environmental damage (Hone 1994). It is important to note that a unit

change in pest abundance doesn't necessarily result in a unit change in damage- the relationship is rarely linear (Hone 1994). Another consideration in control is economic costs. Maintaining a wildlife population below its ecological carrying capacity through control has an ongoing cost (Hone 1994). This is in contrast to wildlife harvesting, where the ongoing removal of animals is perceived as a benefit.

Population control methods can reduce fertility and/or increase mortality (Caughley & Sinclair 1994). Fertility control is becoming popular as it is thought to be humane. Fertility control reduces fecundity, which is a less efficient form of control for wild horses than increasing mortality. A reduction in fecundity rates of wild horses in the Australian Alps would have to be twice that of adult survival rates to effect the same reduction in population growth (see the sensitivity analysis, Chapter 4). Garrott (1991) had the same findings for wild horses in Nevada, USA. Furthermore, immunocontraceptive techniques are expensive and labour intensive; each female must be administered an immunocontraceptive every one to two years from close range (Stafford *et al.* 2001, Turner *et al.* 2001). Fertility control is not considered to be a viable option for wild horse management at present in the Australian Alps (NSW National Parks & Wildlife Service 2002a), though a trial has been suggested (Walters & Hallam 1992). For more detailed assessment of fertility control refer to Garrott (1991), Kirkpatrick *et al.* (1997) and Cameron *et al.* (2001).

Wildlife control through increasing mortality (or permanently removing animals from a population) is common practice, for example kangaroos (Sinclair 1977) and deer (Gogan *et al.* 2001). It is a widespread method of wild horse control practiced in Australia (for example Dobbie *et al.* 1993), in the USA (for example Garrott *et al.* 1991a), Europe (for example Duncan 1992) and New Zealand (Department of Conservation 1995). Wildlife control through increasing mortality has the same effect on the target population as harvesting and predation.

Acceptable wild horse population size or impact have not been clearly defined for the Australian Alps national parks (for example Department of Conservation & Environment 1992, NSW National parks & Wildlife Service 2002a); therefore it is not possible to

determine whether brumby-running is likely to be successful in meeting control objectives. It is possible however to examine the likely population effects of brumby-running through harvest modelling.

5.1.4. Aims

The purpose of this chapter is to gain an understanding of the influence of brumby-running on the wild horse population using the Alpine National Park, Victoria as a case study. The aims are to:

1. Describe the demography of wild horses removed by brumby-runners and compare it to the demography of the populations described in Chapter 4,
2. Assess the behaviour of brumby-runners relevant to the type and quantity of wild horses caught,
3. Predict the likely effect of brumby-running on the wild horse population, and
4. Discuss the role of brumby-running as a form of management.

5.2. Methods

5.2.1. Demography of wild horses removed by brumby-runners

The Alpine Brumby Management Association (ABMA) and Parks Victoria from 1998 to 2002 inclusive supplied demographic data on wild horses caught and removed from the Alpine National Park, Victoria. Brumby-runners recorded the general location, number and type (stallion, colt, mare, filly) of all horses caught each financial year. In this context, colts and fillies generally refer to males and females respectively, which do not appear to have reached sexual maturity. The system was established in 1998-99 and the quality of information supplied has improved over time. The data for 2000-2001 and 2001-2002 are the most detailed including the month of capture and estimated age. Data have been presented under the categories of stallions, mares, colt and fillies to allow comparison between years. However data for 2000-2002 were reclassified into age classes to allow for comparison with age structure from the three key study areas (Chapter 4). Using data from people who harvest wildlife is common practice in wildlife management, for example kangaroo harvest (Sinclair 1977) and moose harvest (Solberg *et al.* 1999).

Body condition scores were given to 176 horses caught between autumn 1999 and spring 2001 inclusive based on photographic records kept by one brumby-runner (BR1). A comparison between body condition of horses caught to the body condition of horses from the three populations studied in Chapter 4 over the same time period were made to test the hypothesis that brumby-runners catch animals in poorer condition. Analysis of body condition in Chapter 4 found that there were differences in body condition between seasons, sexes, sites and years. To account for this variation, a three way fixed-factor ANOVA was used to test the difference in mean body condition between treatments (including wild horses from three sites and wild horses caught), sex and seasons (spring and autumn) (the effect of year was not tested). Mean body condition for each treatment were calculated from sample sizes of 122, 192, 304 and 219 for horses caught, and those at Cowombat, Big Boggy and Currango respectively. Then comparisons between sites and horses caught were made using the Tukey-Kramer procedure (Sokal & Rohlf 1995).

A four-way fixed-factor ANOVA was used to compare the effects and interactions of age (juvenile, adult), sex, season (autumn, winter, spring, summer) and year on the number of horses removed between June 2000 and July 2002 inclusive. Further analysis of season was made using the Tukey-Kramer procedure (Sokal & Rohlf 1995).

5.2.2. Behaviour of brumby-runners

Records were kept by one brumby runner (BR1) for all trips made including numbers of horses removed and numbers of brumby-runners on each trip between January 2000 and January 2001. On many of the trips there were also riders not actively brumby-running that were not recorded. Catch per unit effort (CPUE) was calculated for each month (for all months that trips were made) based on these records. A two way ANOVA was used to test whether there were effects of skill or season on catch per unit effort, and whether there is an interaction between skill and season. Monthly CPUE were used as replicates.

5.2.3. Predicting the effect of brumby-running on the wild horse population

Several basic models are outlined below, which describe the relationship between two trophic levels. These models aim to predict population size at some time in the future (N_t) and are based primarily on prior abundance (N_0), time (t), intrinsic rate of increase (r_m) and rate of offtake (H). Hone (1994) discusses the following models in more detail. The logistic model is one of the simplest population growth models. It has been used for modelling a variety of vertebrate pests (see Hone 1994: 127). Logistic models describe growth dampened by carrying capacity (K). The equation used for logistic growth with annual harvest is

$$N_1 = N_0 + r_m N_0 \left(1 - \frac{N_0}{K} \right) - H$$

It is assumed that $N_0 = 2000$ (population estimate for Alpine National Park, Chapter 3), and annual r_m is 0.2 (see Chapter 4). K was estimated for two scenarios. The first scenario is that 200 horses (H) have been removed annually from the population in perpetuity up until the present and the population is stable. This assumption is based on the rate of removal observed over 1998-2002 (Section 5.3.1). The second scenario is that 100 horses (H) have been removed annually from the population up until the present and the population is stable. This is based on removal estimates made by the Department of Conservation & Environment (1992) up until the early 1990s. If the harvest (H) has been constant, then the population will be stable (Caughley & Sinclair 1994) and $N_0 = N_t$. Therefore the logistic growth model was solved for K . When $H = 200$ in perpetuity, $K = 4000$, and when $H = 100$ in perpetuity, $K = 2667$.

A variation on logistic growth is the generalised logistic growth (Fowler 1988), the equation for which, with annual harvest is,

$$N_t = N_0 + r_m N_0 \left(1 - \left[\frac{N_0}{K} \right]^z \right) - H$$

The generalised logistic growth equation has an additional term z , a measure of curvilinear density-dependent effects on the ratio of N_0 to K (Hone 1994). When $z = 1$, the equation reduces to the basic logistic model. This modification was used by Fowler (1988) to reflect the observations in some large mammals where density-dependent growth is more severe close to K . A value of z has been estimated for several mammal species (after Hone 1994). These include $z = 7$ for badgers (*Meles meles*) in Europe (Anderson & Trewhella 1985), $z = 3$ for brushtail possums (*Trichosurus vulpecula*) in New Zealand (Barlow 1991), and $z = 11$ for elk (*Cervus elaphus*) in the USA (Eberhardt 1987).

A z value for wild horses in the Australian Alps was determined using the relationships between the inflection point of the logistic and generalised logistic models. Fowler (1988) described a regression of the inflection point (N/K) in logistic growth as a function of rate of increase (r_m) per generation ($T =$ generation interval). The regression was:

$$N/K = 0.633 - 0.187 \ln(r_m T)$$

Following Eberhardt (1987),

$$N/K = (1+z)^{-1/z}$$

In the case of wild horses, assume annual $r_m = 0.2$ (see Chapter 4), and $T = 9$ years (Big Boggy population, Chapter 4), therefore $z = 1.25$. This value is very close to 1. The marginal difference to the logistic growth model means that the use of the generalised logistic growth model will not contribute significantly to our understanding of the relationship between brumby-runners and wild horses, so the generalised logistic growth model is not discussed further.

Selective harvesting based on age was examined using Lande's (1988) equation (following on from Chapter 4) using the Alpine National Park, Victoria as a case study. The demographics for the Big Boggy, Cowombat and Currango populations are used as a baseline for determining the effect of selective harvesting of juveniles and adults on the rate of increase of a population. This model is density independent. Initial population size is assumed to be 2000 horses (based on the aerial survey estimate for the Alpine National Park,

Victoria), It is also assumed that 200 horses per year are harvested (based on current harvesting rates for the Alpine National Park, Victoria).

The effect of sex-selective harvesting by brumby-runners on the composition of the adult wild horse population was also modelled for the Alpine National Park. The model was based on several assumptions. Firstly, that initial adult population size is 1440 with equal sex ratio (total population size of 2000 and 72% adult). Secondly, that 30 males and 70 females were harvested annually (following the current harvesting strategy) and third that there is recruitment into the adult population of 100 individuals (with equal sex ratio because there is no sex-selection in harvest of younger age classes Section 5.3.1), which keeps the adult population stable.

5.3. Results

5.3.1. *Demography of horses caught by brumby-runners*

Between July 1998 and June 2002 inclusive, the ABMA recorded the capture of 683 horses in the Alpine National Park, Victoria. In 1998-1999, 224 horses were caught, in 1999-2000 only 78 horses were recorded as being caught. In 2000-2001 and 2001-2002, 174 and 207 horses were recorded as being caught respectively. The low numbers recorded in 1999-2000 are the results from only 5 members of the Association. More horses were caught but not recorded. The numbers of brumby-runners that caught and removed horses in 2000-2001 and 2001-2002 respectively were 15 and 18 respectively. The average number of wild horses caught per brumby-runner from 1999-2002 (n=3) was 13.

More females (mares and fillies) than males (colts and stallions) were caught in all years. The ratio of males to females of all wild horses caught is 0.67:1. This difference from parity is significant ($\chi^2 = 25.13$, d.f. = 1, $p < 0.005$), and there was no significant heterogeneity in this pattern across years ($\chi^2 = 0.32$, d.f. = 3, $p > 0.05$). The difference from parity was primarily because so few stallions were caught (Figure 5.2). Stallions represent less than 10% of horses caught in all years. Mares had the highest average capture rate across years (39%), followed by colts (32%) and fillies (21%) (Figure 5.2). More pre-reproductive horses (colts and fillies)

were caught than adults (stallions and mares) in all years except 1998-1999 (Figure 5.2). There is a trend over time where higher proportions of pre-reproductive animals were caught.

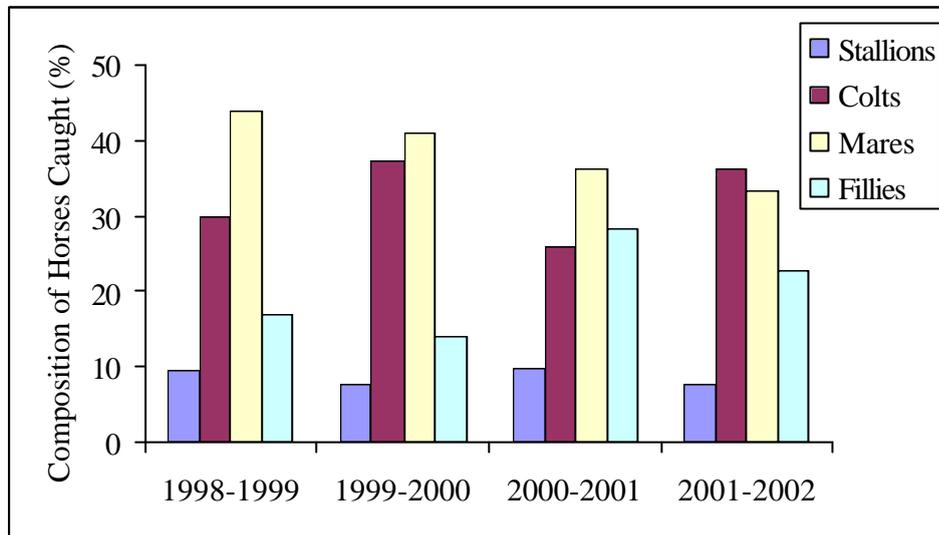


Figure 5.2: Composition of wild horses caught by members of the ABMA annually from 1998-2002.

The assignment of wild horses caught into categories of stallion, mares, colts and fillies can be misleading for interpreting age of horses, particularly males (Table 5.1). In 2000-2002, 20 of the 120 animals recorded as colts were three-year-olds, 4 were four-year-olds and 1 was a six-year-old. The classification of 3-year-old females and older as fillies is less common. Over 2000-2002, 3 of the 96 animals recorded as fillies were older than three (Table 5.1). Due to these discrepancies, all further analyses were based on age classes (Table 5.1).

Table 5.1: Number of stallions, colts, mares and fillies caught by brumby-runners between 2000 and 2002 using different classification systems. The ABMA classification is based on the classifications given by ABMA members. The age classification is based on estimated age (also by ABMA members) where colts and fillies are 0-2 years and mares and stallions are three years and older.

Classification	Stallions	Colts	Mares	Fillies
ABMA	33	120	132	96
Age	58	95	135	93

Brumby-runners caught relatively more younger wild horses. Forty-nine percent of all horses caught between July 2000 and June 2002 were younger than three-years-old. The average portion of horses under the age of three at each study site over the same period were 28, 28 and 27 % respectively for the Big Boggy Cowombat and Currango.

Over the period of 2000-2002, the ratio of males to females under the age of three ($n = 188$) caught by brumby-runners varied between being skewed towards females then towards males depending on age class (Figure 5.3). However when these three age classes were pooled, the ratio of males to females was 1.02:1, which is very close to parity. In adults (3 years and older), the sex ratio of horses caught is skewed towards females at 0.43 males to every female ($n = 193$) (Figure 5.3). This skew does not reflect any skew in the sex ratio of the total population, because the average ratio of males to females that were observed during surveys at the Big Boggy, Cowombat and Currango over the same period was 1.08:1. There was a significant difference in the sex of horses caught across all age classes ($F = 6.38$, d.f. = 1, 3, $p = 0.025$). There was also a significant interaction between sex and age ($F = 5.38$, d.f. = 1, 3, $p = 0.037$), which is a result of the relative number of adult females but not of females less than three-years-old. All other effects and interactions in the four-way ANOVA were non-significant.

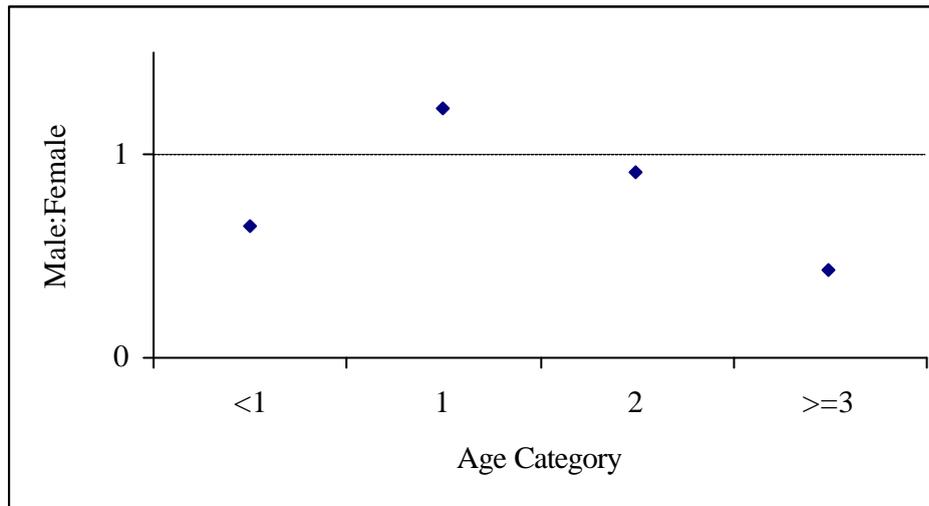


Figure 5.3: Sex ratio of wild horses caught by ABMA members between 2000-2002 for different age categories.

Seasonal trends of capture from July 2000-June 2002 show that most horses were caught in autumn (51 in 2000-01 and 70 in 2001-02), least in summer (30 in 2000-01 and 29 in 2001-02), while spring and winter were similar ranging from 40 to 61 horses (Figure 5.4). The effect of season was almost significant ($F = 3.25$, d.f. = 3, 3, $p = 0.057$). A Tukey-Kramer pairwise comparison showed that the difference was significant between autumn and summer ($p = 0.042$) but not for any other seasons.

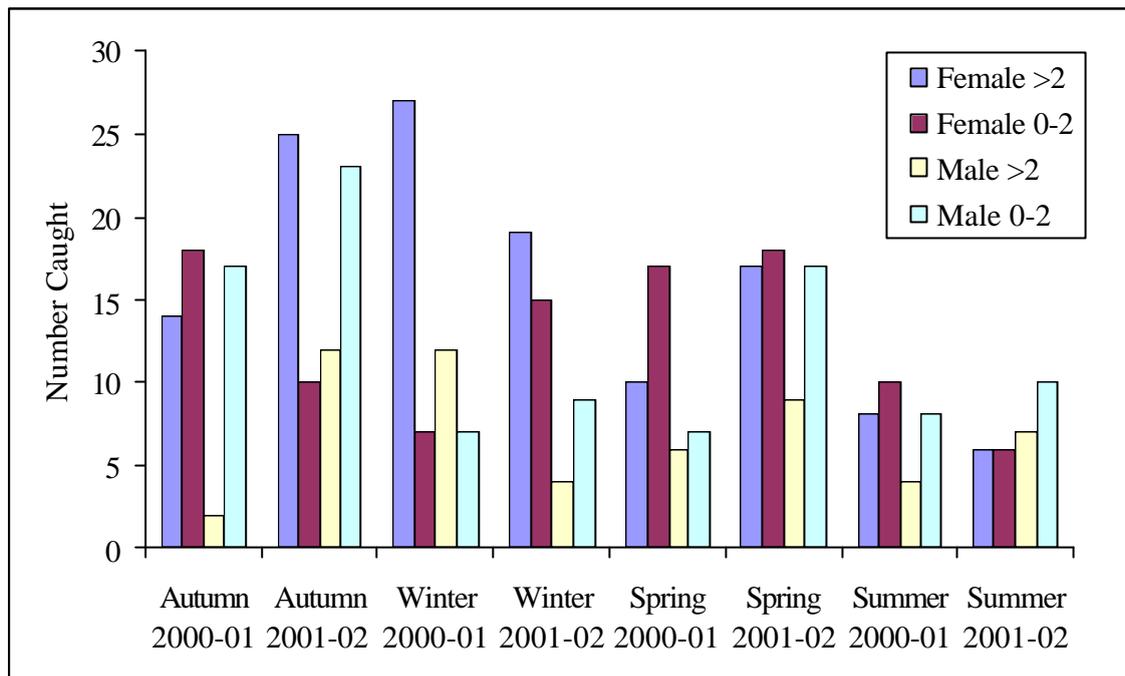


Figure 5.4: Wild horses caught by ABMA brumby-runners each season between July 2000 and June 2002 for age and sex classes.

The hypothesis that brumby-runners select horses in poorer condition is not supported. A significant component of the variation in body condition could be attributed to treatment ($F = 41.73$, d.f. = 3, 3, $p = 0.006$). Tukey-Kramer pairwise comparison among the 4 treatments (including the three study areas and horses caught by brumby-runners) revealed no significant difference between the body condition of horses caught by brumby-runners (2.42) and study populations at Big Boggy (2.71) and Cowombat (2.12) ($p = 0.090$ and 0.131 respectively). Populations at the Big Boggy and Cowombat were in the same geographical region as those caught by brumby-runners. Wild horses caught by brumby-runners had significantly poorer body condition than wild horses at Currango (3.00) ($p = 0.017$). The Currango study site contained horses which were in a geographically isolated population to those caught by the Victorian brumby-runners, so is not representative of animals available for capture. There was no significant interaction between treatment and season ($F = 2.24$, d.f. = 3, 3, $p = 0.263$) or treatment and sex ($F = 3.92$, d.f. = 3, 3, $p = 0.146$).

Difference in body condition between the sexes was significant ($F = 40.93$, d.f. = 1, 3, $p = 0.008$) as were differences in body condition between seasons (spring and autumn) ($F = 24.36$, d.f. = 1, 3, $p = 0.016$). These results mirror the results found in Chapter 4 for the three sites where females had poorer body condition than males and animals had poorer body condition in spring than in autumn. There was no significant interaction between sex and treatment ($F = 3.92$, d.f. = 3, 3, $p = 0.146$) or sex and season ($F = 0.30$, d.f. = 1, 3, $p = 0.623$) indicating that the same trend was evident in body condition across the three sites and horses caught by brumby-runners.

5.3.2. Behaviour of brumby-runners

Over a period of 22 months, (January 2000-October 2001), one brumby-runner (BR1) went on 30 trips. No trips were made in 6 of the 22 months. He spent an average of 3.45 days/month brumby-running. The average trip length was 2.93 ($\pm 0.4SE$) days, and the majority of trips were two days in duration. He was accompanied by other brumby-runners on most trips. BR1 put in a total effort of 76 days while the total effort for all brumby-runners accompanying him was 172 days. BR1 was more efficient at catching wild horses than his companions in all seasons (Figure 5.5). He caught 88 horses, so his mean CPUE was 1.16 horses/day. All other riders combined caught a total of 94 horses, so mean CPUE for all other riders was 0.55 horses/day. Skill had a significant effect on catch per unit effort in a two way ANOVA ($F = 6.83$, d.f. = 1, 23, $p = 0.016$) associated with BR1 having a higher catch per unit effort than other brumby-runners. Combining the data for all riders including BR1, CPUE is 0.73 horses/day.

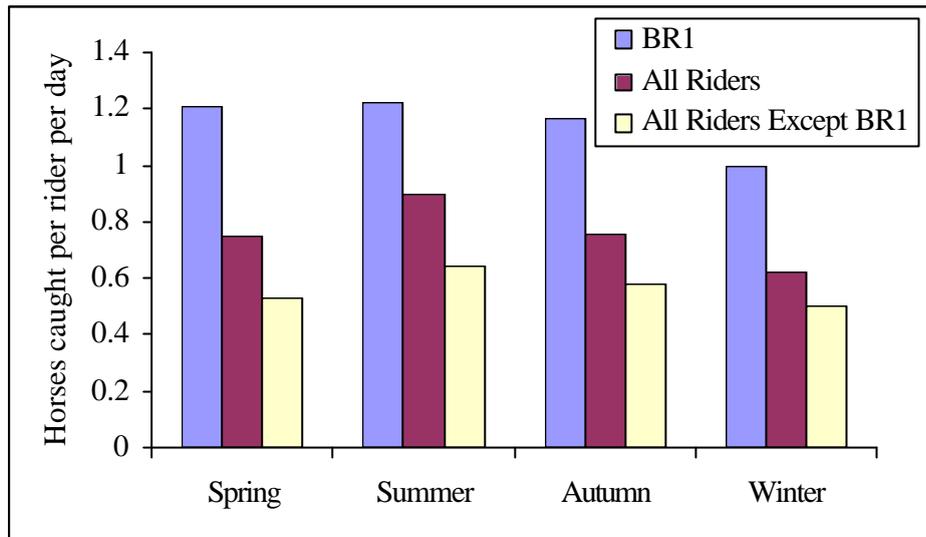


Figure 5.5: Mean catch per unit effort (horses/day) for BR1, all riders and all riders except BR1 in each season.

Catch per unit did not vary across seasons ($F = 0.34$, d.f. = 3, 23, $p = 0.800$), and there was no interaction between skill and season ($F = 0.28$, d.f. = 3, 23, $p = 0.842$). The percentage variation that could be attributed to the effect of skill and season on a month by month basis was low ($R^2 = 0.309$). The variation in monthly CPUE may be the result of many effects such as weather or numbers of horses sighted on a trip by trip basis.

Mean catch per unit effort (horses/brumby-runner/day) was highest when brumby-runners were in a group size of three (Figure 5.6). However, there was no significant difference in average CPUE between brumby-runner group sizes as assessed in a one-way ANOVA ($F = 0.83$, d.f. = 5, 24, $p = 0.53$). Because of small sample size, the ANOVA may have had limited power to detect any differences between means.

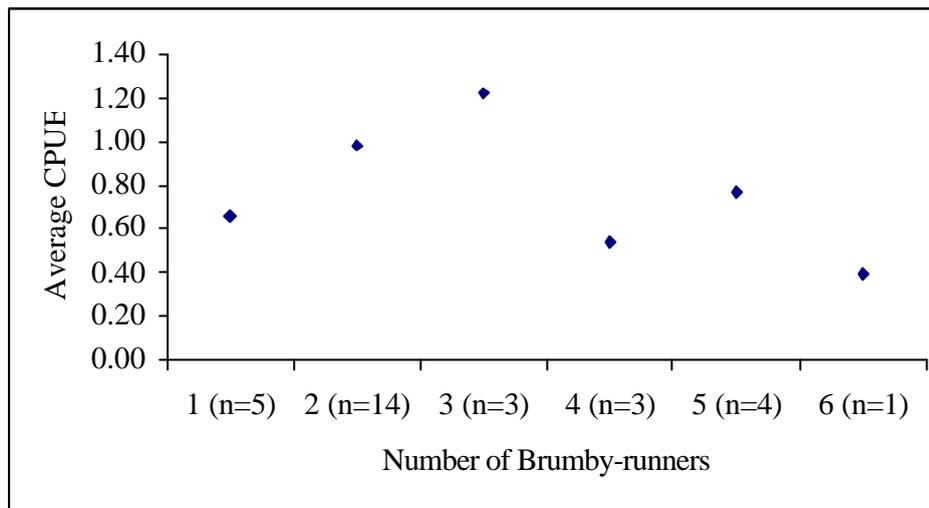


Figure 5.6: Average CPUE (horses/rider/day) for different group size of brumby-runners.

5.3.3. *Predicting the effect of brumby-running on the wild horse population*

Predictive modelling for the wild horse population assuming logistic growth shows that brumby-running is likely to have a limiting effect on the population. In the first scenario, carrying capacity was estimated assuming 200 horses had been removed annually by brumby-runners every year prior to the present (Figure 5.7). In this case, the model predicts that if brumby-running ceases and no other method of control is used, the population will increase until it reaches its carrying capacity of 4000 horses; twice the current estimated population size in under 20 years. If harvesting (H) continues at 200 horses per year under this scenario, then the model predicts that the population will remain stable at 2000 individuals. If harvesting occurs at levels greater than 200 horses per year then the model predicts that the population will become extinct. If the population is harvested at less than 200 horses per year, then population size will increase until it plateaus at a new stable density above 2000 (Figure 5.7).

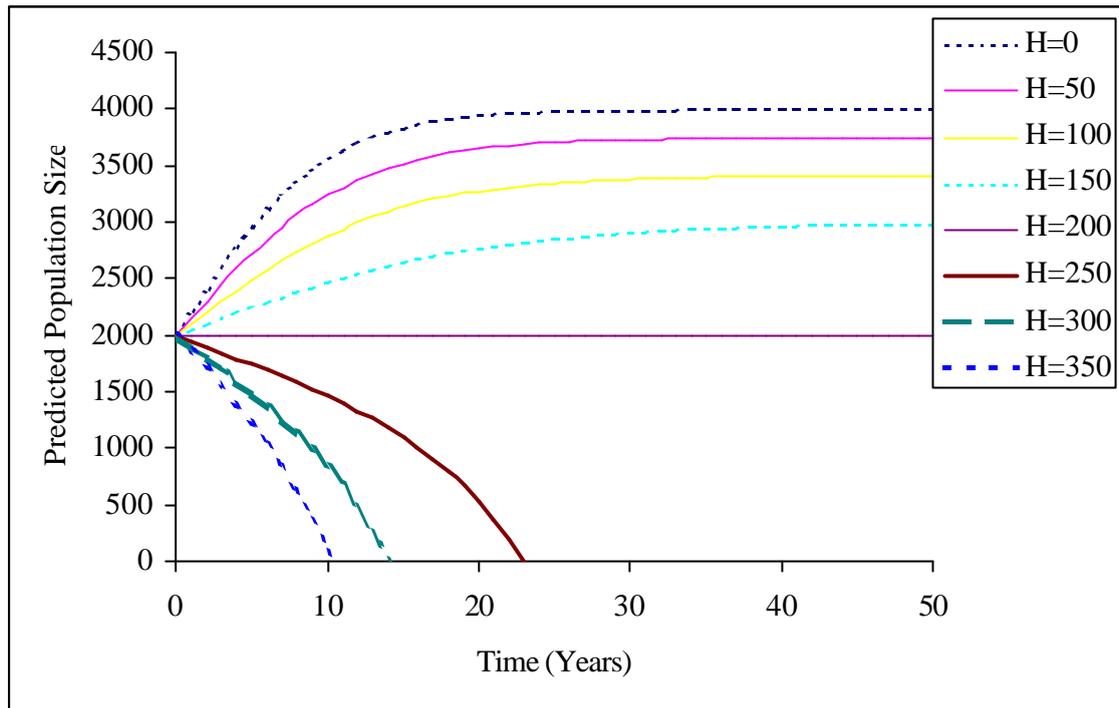


Figure 5.7: Scenario 1. Predicted size of the Alpine National Park wild horse population at different levels of harvesting (H) per year. The model assumes logistic growth with $r_m = 0.2$, $N_0 = 2000$ and $K = 4000$.

Under the second scenario, it is assumed that the population has been harvested (H) at 100 horses per year up until the present. In this case, if harvesting were to cease ($H = 0$), then the population would increase to a carrying capacity of 2667 horses in a period of about 15 years (Figure 5.8). If the aim of harvesting is to maintain populations at their current level, then harvesting rates should be 100 horses per year. If harvesting rate is above 100 horses per annum, then the model predicts that the Alpine National Park wild horse population will decline (Figure 5.8).

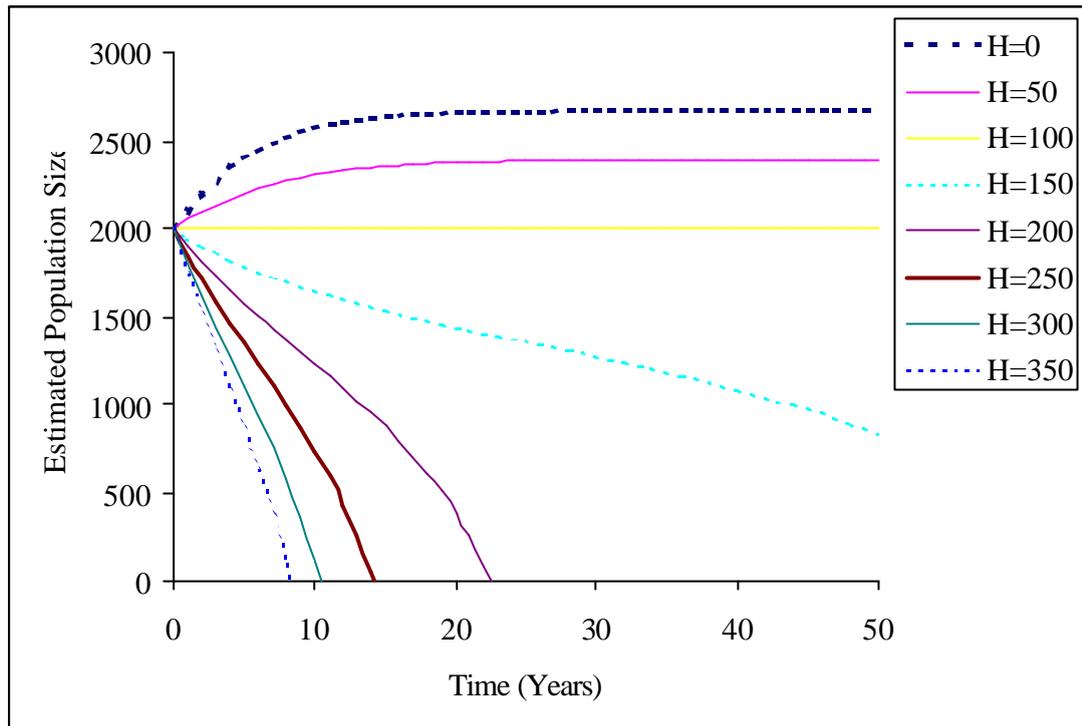


Figure 5.8: Scenario 2. Predicted size of the Alpine National Park wild horse population at different levels of harvesting (H) per year. The model assumes logistic growth with $r_m = 0.2$, $N_0 = 2000$ and $K = 2667$.

Rate of increase of the wild population is reduced as the proportion of adult (>3 years old) wild horses removed from the population increases (Figure 5.9). This trend is apparent based on the demographic parameters estimated for wild horses at Cowombat, Big Boggy and Currango (Figure 5.9). The rate of increase across all age ratios of horses removed was lowest based on the demographics for the Cowombat population. The slope of the relationship was similar for Cowombat (-0.006) and Big Boggy (-0.005) wild horse demographics, and flatter for Currango (-0.002). Annual rate of increase was positive ($\lambda > 1$) when all wild horses caught are <3-year-olds and the demographics are representative of Big Boggy and Currango (Figure 5.9).

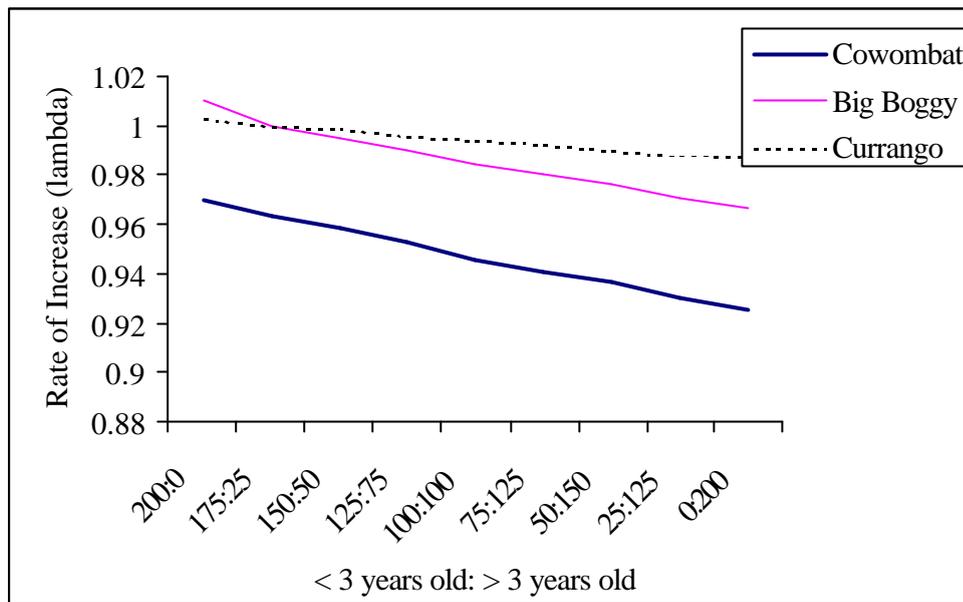


Figure 5.9: Predicted relationship between wild horses removed by brumby-runners < 3 years old: > 3 years old and annual finite rate of increase of the population assuming a population size of 2000 and 200 horses removed per annum. Demographic parameters are based on analysis for Cowombat, Big Boggy and Currango wild horse populations (Chapter 4).

Predictive modelling shows that the current pattern of sex-selective harvesting will lead to a bias in adult sex ratios of the remaining population (Figure 5.10). If brumby-runners practiced unselective harvesting of adults, then the proportion of males to females would stay constant. However Figure 5.10 predicts that the current bias in harvest will lead to a bias in the sex ratio of the breeding population. This could in turn lead to a reduction in the number of foals born per year because the breeding population will decrease from 720 females to 320 females a 2.25-fold decline. If fecundity per female remained constant, then such a decline in breeding females would result in a reduction of foals born from 374 per year to 166 per year.

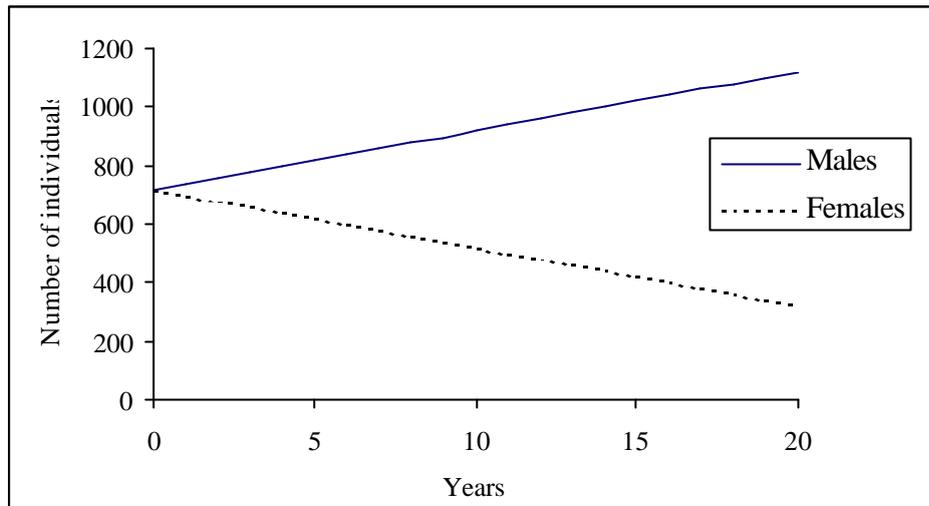


Figure 5.10: The predicted number of adult male and adult female wild horses in the Alpine National Park over time if the current pattern of adult sex-selective harvesting (0.43 males: 1 female) continues with constant recruitment of equal numbers of males and females. The model assumes adult population size remains constant at 1440, 100 horses are harvested per year and recruitment keeps the population stable.

5.4. Discussion

The information presented in this chapter is the first assessment of the demography of wild horses removed by brumby-runners, the behaviour of brumby-runners relevant to the wild horses that they remove and the likely influence that brumby-runners are having on a wild horse population. The key findings are that brumby-runners catch relatively more young animals and females, and they do not appear to select horses in poor condition. Brumby-runners are analogous to predators, exhibiting prey-selection and heterogeneity between predators. A limitation of this study is that I could not sample the harvested population for comparisons, only an unharvested population.

Predictive modelling found that it is likely that the level of brumby-running currently occurring in the Alpine National Park in Victoria is suppressing the population. However the selective removal of younger animals is likely to increase yield, with a less significant reduction in rate of increase than if harvested unselectively, while the selective harvesting of females reduced the breeding population when compared to unselective harvesting.

5.4.1. Demography of wild horses caught and removed by brumby-runners

The data collected by the ABMA has proven to be very useful in assessing the demography of wild horses removed by brumby-runners. The Department of Conservation & Environment (1992) identified the importance of collecting such data for evaluating the effectiveness of brumby-running as a control measure. Similar mandatory reporting of animals harvested provides valuable information for harvest management of big game species such as deer (Strickland *et al.* 1996) and moose (Solberg *et al.* 1999).

The quality of the four-year data set varied across years. There was a general improvement in data quality over time, particularly with respect to recording estimated age of animals. This reduced the confusion that can arise from assigning age as either stallion/colt for males and mare/filly for females. The accuracy of data is questionable in some instances for example some brumby-running occurs by non-ABMA members. In 2000-01, the ABMA knew of five people who caught wild horses but were non-members. The effect of not including all brumby-running data is that the total number of horses removed by brumby-runners is underestimated. Another source of error is that horses caught outside of the area designated for brumby-running are recorded as being caught within the area. This has the effect of overestimating the numbers of horses caught by brumby-runners in the area. These potential disadvantages of harvest reporting systems are common and difficult to overcome (Strickland *et al.* 1996). If brumby-runners are aware of the value of accurate harvesting data, then they are more likely to provide accurate reports.

It does not appear that brumby-runners are selecting for either sex in young age classes because the sex ratio of animals caught is close to parity; the sex ratio in the wild horse population should also be close to parity (Monard *et al.* 1997). Selection based on sex occurs mostly in adults where females are caught more frequently. This is in contrast to other large herbivores such as saiga antelope (Milner-Gulland 1994) and moose (Solberg *et al.* 1999) where males are selected. The selection of female wild horses by brumby-runners is likely to be a function of catchability. Adult male horses are typically harder to catch than females. They are difficult to get close to during a chase because they are generally strong and fit and they are difficult to handle because they are strong and aggressive. They also pose problems

during transport due to their aggressive habits towards one another (pers. obs. and Linklater *et al.* 1999, Linklater & Cameron 2000 for examples of stallion aggression).

Brumby-runners selected younger animals. Similar selection occurs in some other harvested species (for example moose, Solberg *et al.* 1999). Predators can also select young animals (for example mountain lions select foals, Turner *et al.* 1992, Greger & Romney 1999). The reason for this selection by brumby-runners is likely to be a combination of two factors. The first is that young animals are often easier to catch and handle than larger adults are. Young animals are also easier to train. The reason for selective harvesting of young moose was to optimise the yield of moose in conjunction with harvesting adult males and restricting the harvest of reproductive females (Solberg *et al.* 1999). The portion of the moose harvest comprised of yearlings was about 32%, while yearlings made up 12% of populations on average.

5.4.2. *Brumby-runner behaviour*

Catch per unit effort data collected in this study revealed the effectiveness of brumby-running. The number of horses removed per day by brumby-runners is low (approximately 0.7 horses/brumby-runner/day), especially when it is compared to management techniques such as aerial shooting. An aerial cull of horses in Guy Fawkes River National Park, New South Wales culled 606 horses in three days (202 horses/day) (English 2000). However aerial culling in NSW National Parks was consequently banned due to the practice being unacceptable to a large portion of the community (Gilligan 2002). There is general consensus that the local community should be involved in any wild horse control operations because the issue has strong social elements (English 2000, NSW National Parks & Wildlife Service 2002a). Brumby-running is a method of control that is suitable in a management strategy that requires ongoing low-level control because of the low catch per unit effort. It is unlikely to be effective if control objectives are to remove large numbers of wild horses quickly.

Skill improves catch per unit effort of brumby-runners (Figure 5.6). The specific skills required to undertake brumby-running successfully take a long time to develop and require a high level of fitness (personal experience). The implications for this are that if brumby-

running is to be used as a form of management then opportunities must be provided for inexperienced brumby-runners to gain experience in the company of skilled brumby-runners.

The group foraging size for brumby-runners with highest CPUE was three though the ANOVA had limited power to detect differences in group size. This is equivalent to the optimal group foraging size estimated for lions (Caraco & Wolf 1975) and North American wolves (Nudds 1978). However, when lions prey on large, abundant herbivores, they form groups apparently larger than would be predicted by maximum foraging efficiencies (Caraco & Wolf 1975). The same trend is likely to be true for brumby-runners (riders that were not brumby-running were not included in the analysis in section 5.3.3). Caraco & Wolf (1975) suggest that larger group size improves breeding success, survival of young and is important in defence of food against scavengers. Brumby-runners are often accompanied by horse-riders that are not brumby-running, or are learning how to brumby-run. Brumby-runners are likely to benefit from larger group sizes because skills of inexperienced brumby-runners are improved and extra riders assist in wild horses handling.

5.4.3. Predicting the effect of brumby-running on the wild horse population

Modelling is a valuable tool for exploring management strategies (Starfield 1997). Modelling was used as a tool for gaining an understanding of the effect of brumby-running on the wild horse population. Other authors have used modelling to explore management strategies. Fryxell *et al.* (1991) examined white-tailed deer harvests, Solberg *et al.* (1999) examined moose harvests and Milner-Gulland *et al.* (2001) used simulation modelling to test strategies for harvest management of saiga antelopes. The simple logistic growth model used in the current study suggests that the brumby-running could suppress the wild horse population in the Victorian Alps. This is not unexpected because harvesting can have this effect (Caughley 1977a, Sutherland 2001).

When model are used as a problem-solving tool, it is important to meet the assumptions and understand the limitations associated with the assumptions (Starfield 1997). The assumptions of the models used in the current study are: 1) that the population is subject to logistic growth, 2) that the population has an ecological carrying capacity, 3) that carrying capacity

was estimated correctly, 4) that initial population size (N_0) was estimated correctly, 5) that r_m is accurate and 6) the harvesting is completely random from a population.

The underlying assumption that the wild herbivore population is subject to logistic growth is acceptable for basic modelling (Caughley 1977a). The trends shown in Figures 5.7 & 5.8 are similar to the trends shown for harvesting by Caughley (1977) and Caughley & Sinclair (1994). Logistic density-dependence is a very simple way of including density-dependence in a model (McCallum 2000). Its simplicity makes it appropriate when details are either unavailable or unnecessary (McCallum 2000). Equid populations are like most large mammal populations (Sinclair 1989); they are affected by density-dependence (Duncan 1992 and Choquenot 1991). The generalised logistic model (Fowler 1988) is useful for some species because it accounts for the fact that most density-dependent change occurs at high population levels (Fowler 1981). However, estimates of z (see section 5.2.3) revealed that using the generalised logistic model had little benefit in the case of wild horses. When harvest per year is substantial then population growth can be described by exponential growth, as demonstrated for many large mammals (Eberhardt 1987).

Elementary in the logistic growth model is the concept of an ecological carrying capacity. Carrying capacity has been estimated given the variability in rainfall under natural conditions for some populations, for example elk in Yellowstone National Park (Coughenour & Singer 1996). However the concept of carrying capacity is not useful in environments with a high environmental variance (McLeod 1997). Many Australian environments have a high degree of unpredictable environmental variance. For example, McLeod (1997) showed that the highly stochastic environment (represented by seasonal rainfall variability) at Tibooburra, NSW precluded the use of the concept of carrying capacity because the plants and herbivores rarely, if ever reach an equilibrium. Tibooburra lies in an area characterised by high rainfall variability, whereas the Australian Alps falls in a zone of low to moderate rainfall variability (Bureau of Meteorology 2002). It is not clear whether it is appropriate to use the concept of carrying capacity in the Australian Alps. Environmental variability can be incorporated into logistic growth by assuming r_m is a variable (Beddington & May 1977), or assuming carrying capacity (K) is a variable. In the later case, $K = V/a$ where V is food availability and a is the

equilibrium ratio of food availability per head. Then the logistic equation becomes the ratio numerical response described in Chapter 4.

Carrying capacity is notoriously difficult to estimate (Sutherland 2001). Two levels of carrying capacity were estimated in the current study based on previous harvesting rates. In reality, harvesting rate is likely to have been variable with rates somewhere in between these two values. Estimates of carrying capacity based on a numerical response relationship were not possible due to lack of data on food availability in Victoria.

The initial population size was estimated from the aerial survey results in Chapter 3. It is the most accurate estimate of population size currently available. Maximum population growth rate (r_m) was based on the maximum growth rate from international studies on horses and is likely to be accurate because wild horse population dynamics in the Australian Alps are similar to those in other countries (Chapter 4).

The final assumption of the model that harvesting is unselective is clearly not true given the data presented in section 5.3.1; brumby-runners select young animals and females. Selecting younger animals has the effect of increasing the sustained yield (Figure 5.9), while selecting females has the effect of decreasing the sustained yield (Figure 5.10). Selective harvesting of young horses is also predicted to increase yield given the lower sensitivity of rate of increase to survival of younger animals (Chapter 4 of this thesis, Garrott & Taylor 1990, Gaillard *et al.* 1998, 2000). The sustained yield patterns from age- and sex-selective harvesting occur in other ungulates. For example harvesting strategies that aim to maximise sustained yield of saiga antelope (Milner-Gulland 1994) and moose (Saether *et al.* 2001) target adult males and young animals.

A limitation to potential application of the modelling results is that at high harvest rates, abundance declines to low levels, however at those lower horses would be harder and more costly to catch. Such increasing costs have been reported for control of feral goats (Pople *et al.* 1998) and feral pigs (Choquenot *et al.* 1999).

Finally, the model presented here does not consider timing of harvest. Timing of harvest can be important in seasonal environments (Kokko 2001) such as the Australian Alps. Brumby-running occurs in all seasons, but mostly in autumn and least in summer (Figure 5.4). Removing individuals just prior to the breeding season (spring and summer for wild horses in the Australian Alps) causes a larger reduction in population size than if they were removed in autumn (Kokko 2001). That is, harvesting may be compensatory to natural mortality if it occurs in autumn, but is additive if it occurs just prior to the breeding season. This is complicated by the fact that the magnitude of this effect is related to the density of the population. That is, when the population is close to carrying capacity, the timing of harvest has a greater effect on the harvested population size (Kokko 2001).

The model discussed here is one of the simplest forms of harvesting. Other forms of harvesting are better for establishing desired population sizes (Newton 1998). These include variable quota, fixed-percentage or fixed escapement (threshold-harvesting) (Newton 1998). These more sophisticated harvesting strategies are recommended for managing populations to prevent overexploitation and encourage optimal harvesting (Sutherland 2001).

5.4.4. Wild horse harvesting and control

Brumby-running is a form of recreational harvesting. The key question is whether it can meet control objectives. Brumby-running is used by managers in the Victorian Alps to control wild horses and is soon to be trialled in southern Kosciuszko National Park. The objectives of control operations in Victoria need clarification. Brumby-running was intended to maintain “effective population control” (Department of Conservation & Environment 1992). In southern Kosciuszko National Park, management objectives are to ensure the alpine area is free from all impacts, but levels of acceptable impacts have not been determined for non-alpine areas (where the majority of wild horse occur) (NSW National Parks & Wildlife Service 2002a). It is not possible to determine whether brumby-running can meet management objectives because they are vague.

The difficulty with setting control objectives has been partly due to the fact that it is based on a value judgement. The value of wild horses in the Australian Alps and acceptable impact is

different for different people. This issue is not unique to this situation. Caughley and Sinclair (1994) provide a structure for setting wildlife management objectives in such circumstances. Braysher (1993) and Dobbie *et al.* (1993) also outline principles and strategies for managers to address such issues. Once management objectives are set, management becomes a technical issue of achieving them (Caughley & Sinclair 1994). The importance of defining management goals and assessing whether management actions meet them is illustrated by ungulate control in New Zealand (Caughley 1982). Red deer, Himalayan thar (*Hemitragus jemlahicus*) and chamois (*Rupicapra rupicapra*) were heavily culled for decades on the assumption that it would slow the rate of flooding and river bed aggradation. When research finally tested this assumption, it was shown to be false.

Control and harvesting both act on the dynamics of the population. Thus harvesting can be used as a form of population control. It is easier to manage harvesting than it is to manage control because harvesting is concerned only with the harvested population while control is concerned primarily with the impact of the population (Caughley & Sinclair 1994). Reducing population size is likely to reduce the impact of the population, but the relationship between population size and impact is rarely linear (Hone 1994).

6. GENERAL DISCUSSION

The research undertaken here is the first detailed study of the population ecology of wild horses in the Australian Alps. The topics covered were necessarily broad and included distribution, abundance, population dynamics and potential limiting factors. Local land managers are already using the results (for example NSW National Parks & Wildlife Service 2002a), and it contributes to the international literature on ungulates. In this chapter, I discuss the key findings of the research that are pertinent to wild horses in particular and ecology and wildlife management in general. I then outline management recommendations and future research directions.

6.1. Wild horses

6.1.1. *Distribution*

Oral history was used to trace the history of wild horse distribution in the Australian Alps and determine what factors have influenced their distribution through time. Humans appear to be the primary factor defining the distribution of wild horses in the Australian Alps through their translocation and control efforts. This is true in general for horses around the world (for example Duncan 1992 & Kirsch 2002) including other parts of Australia (McKnight 1976), with more remote populations being manipulated less than those in areas populated by people. This close association with humans is in contrast to other introduced species such as foxes. Foxes threaten the long-term survival of native fauna and prey on livestock; they occur throughout most of southern Australia despite efforts to control them (Saunders *et al.* 1995).

Snow and drought play a role in defining wild horse distribution in some parts of the Australian Alps, as do geographical barriers and perhaps wild dogs and behaviour. Snow influences wild horse distribution at higher elevations in the Australian Alps in the winter months. Mortality occurs as a result of severe snow events or long periods of snow cover. Areas prone to such phenomenon tend to have lower densities of wild horses that may become locally extinct as described for animal populations by Andrewartha and Birch

(1954). Similar patterns have been observed for wild horses in Utah (Berger 1983, 1986). Drought acts on the distribution of wild horses in parts of central Australia (Dobbie *et al.* 1993) and in the rainshadow regions of the Australian Alps.

6.1.2. Density and abundance

The average density of wild horses, 1.8km^{-2} , in the Australian Alps calculated from the aerial survey was within the range predicted given their body mass (Damuth 1981, Freeland 1990). This density was an order of magnitude lower than densities observed in Europe (Duncan 1992, Koene pers. comm.) and Argentina (Scorrolli 2001), higher than densities observed in the Northern Territory (Skeat 1990, Black 2000), and similar to densities in Kaimanawa New Zealand (Cameron *et al.* 2001).

The survey to estimate density and abundance of wild horses across the Australian Alps was the first of its kind. The only other broad-scale estimate was by Dyring (1990) based on anecdotal evidence. If Dyring's estimates were accurate, then the wild horse populations in Victoria have not increased significantly, and the number of wild horses in Kosciuszko National Park has increased threefold. This assessment should be taken cautiously because the accuracy of Dyring's estimates is unknown, and the estimates of numbers in the current study are based on the average density across all areas. Nonetheless, an increase in population size in Kosciuszko and not in Victoria could be expected given that there has been no active management of wild horses in Kosciuszko in the last 20 years (NSW National Parks & Wildlife Service 2002b), while brumby-running continues in Victoria. The evidence of an increased distribution of wild horses in northern Kosciuszko National Park (Chapter 2) also supports this hypothesis.

6.1.3. Population dynamics

None of the three populations of wild horses studied at Cowombat, Big Boggy and Currango were increasing at the maximum annual intrinsic rate of increase calculated for other wild horse populations of $I = 1.2$ (Eberhardt 1987, Garrott *et al.* 1991a, Cameron *et al.* 2001). Furthermore, the Cowombat and Big Boggy populations may have been stable. The Currango

population was increasing ($I = 1.09$) associated with higher recruitment than in the other two populations. Adult survival was constant in all populations. The demographic rates estimated in this study were similar to those reported for wild horses elsewhere in Australia (Berman 1991), in the USA (Eberhardt *et al.* 1982, Berger 1986, Ganskopp & Vavra 1986, Garrott & Taylor 1990), in New Zealand (Cameron *et al.* 2001) and France (Duncan 1992).

The three populations were likely to be subject to different degrees of food limitation, with the Cowombat population affected the most. The latter population had horses in the poorest condition and pasture biomass was lowest. The Currango population appears to be affected least by food limitation because it is increasing at the fastest rate, the horses are in better condition and there was a higher pasture biomass. The Big Boggy population was intermediate. The estimated numerical response for the Big Boggy population summarised the positive relationship between population growth rate (I) and food availability per horse. This apparent pattern of food limitation across sites is also reflected in the large-scale population trends based on historical evidence. Population size in northern Kosciuszko (Currango) appears to have increased since Dyring's (1990) study, while Cowombat (in the middle of an expansive wilderness area) is part of a well-established population, which is more likely to be at equilibrium. The Big Boggy population is on the edge of the extensive southern Kosciuszko population and subject to more snow in winter.

6.1.4. *Brumby-running*

The relationship of brumby-runners to wild horses is analogous to a predator and its prey. Brumby-runners exhibit prey selection and heterogeneity between predators. Predictive modelling of brumby-running in the Alpine National Park shows that brumby-running is likely to be limiting the wild horse population.

6.2. Ecology and wildlife management

6.2.1. *Distribution*

The dependence of wild horse distribution on human actions means that they respond to management (or lack of) by humans. The approach to management of wild horse in the

Australian Alps has changed over time- this has resulted in a change in distribution over time. Horses do not occupy all suitable habitat at present, and they have the potential to move into new areas. If the predicted effects of a reduction in snow as a result of climate change (Whetton 1998) occur, then the spread of horses, if unmanaged, may continue into the delicate alpine areas. If populations are not managed, then the distribution of wild horses may increase in pulses as observed in wood bison (Larter *et al.* 2000).

6.2.2. Estimating density and abundance

This study showed that the method used to estimate abundance strongly influenced the result which has very important implications. Many aerial surveys use a 200m wide strip (Southwell 1989, Black 2000, Bayliss & Yeomans 1989b). We found here that sightability of horses dropped off dramatically after 50m and the commonly used mark-recapture analysis could not correct for the reported bias. Hence reported correction factors are likely to be too low. For species smaller and less visible than horses (for example goats) in similar habitat, this trend is likely to be even stronger.

6.2.3. Population dynamics

The dynamics of wild horses in the Australian Alps are typical of large mammalian herbivores with low density, low population growth rates, low fecundity, high adult survival and lower juvenile survival (Caughley 1976, Gaillard *et al.* 1998, 2000, Eberhardt in press). Wild horses in the Australian Alps appear to be food limited, which is also typical for large mammalian herbivores (Sinclair 1989, 1996). The estimated numerical response was similar in form but different in detail to that estimated for wolves and deer (Eberhardt & Peterson 1999). Food limitation may be most important similar to that for other ungulates in seasonal environments (Clutton-Brock & Coulson 2002). Estimates of population growth rate (I) from demographic data were more precise than from census data. This is consistent with the findings of Taylor & Gerrodette (1993) for small populations.

If managers aim to control the wild horse population, then efforts focused on removal of adults will have the greatest effect because population growth rate is most sensitive to

changes in adult survival. This is typical for large herbivores and is a function of life history patterns (Gaillard *et al.* 2000, Eberhardt in press).

6.2.4. Brumby-running

Brumby-running is a form of wildlife harvesting, with the hope that it is also an effective form of control (Department of Conservation & Environment 1992). The modelled suppression of population size by brumby-runners in Victoria is supported by empirical examples where wildlife harvesting suppresses population size and limits population growth in other ungulates such as white-tailed deer (Fryxell *et al.* 1991) and moose (Solberg *et al.* 1999) or predators limit prey such as wolves and caribou (Seip 1991 in Caughley & Sinclair 1994) and deer (Eberhardt & Peterson 1999).

Brumby-runners demonstrate prey selection and heterogeneity among predators similar to effects reported for predator-prey systems (Krebs 1994). The prey selection bias towards adult females means that harvest is not optimal. Optimal harvest would be biased towards adult males and young as reported by Saether *et al.* (2001).

6.3. Management of wild horses in the Australian Alps

Management of wild horses in the Australian Alps has been a contentious issue with no active management of the population in Kosciuszko National Park for the last 20 years (NSW National Parks & Wildlife Service 2002b) and brumby-running being the method of control in the Alpine National Park excluding wilderness areas. This study has stimulated debate about the management of wild horses and has raised the profile of wild horse management. I have been approached by staff from park staff and from the Australian Alps Liaison Committee to recommend approaches to management based on my research. Every population of wild horses within the national parks is connected to populations outside the national parks on forestry or private land. Furthermore, several populations span state and territory borders. Effective management will require co-operation across these borders.

A workshop on “Feral Horses in the Alps” (Walters & Hallam 1992) stated that the focus for the management of feral horses in the Alps is environmental protection. This is in keeping with the approach to managing vertebrate pests in general (Braysher 1993, Hone 1994). In the case of wild horses, this objective is tempered by the cultural value of wild horses (for example English 2000).

A simple management strategy that will protect environmental values is to prevent the distribution of wild horses from increasing from its current definition. This would contain the current impact of wild horses, and ensure that they do not damage new areas, and would have a minimal impact on the cultural values of wild horses.

A second objective of management should be to prevent wild horse populations from increasing in size. Again this objective should have little effect on cultural values of wild horses, but it will ensure that the impact of wild horses does not increase beyond its current level. The challenge in this objective lies in knowing the rate of increase of each population through time and to establish how many wild horses need to be removed per annum to maintain the current population size. Population trends can be monitored on a broad scale by repeating the aerial survey as established in this thesis. This technique is limited in that it will not pick up trends on a local scale and lacks precision.

There are some areas within the Australian Alps where wild horses are having an obvious impact on the vegetation and streams such as parts of the Pilot and Cobberas wilderness areas (pers. obs.). In addition, the horses are in very poor condition, which could be considered an animal welfare concern. In these areas I would recommend that population densities are reduced to levels where environmental impact is acceptable. The two main challenges in this management recommendation lie in defining acceptable impact levels, and determining the relationship between the impact and wild horse density. Estimating acceptable numbers of wild horses has occurred in the USA based on environmental concerns (Turner 1988) and conservation of genetic variation (Goodloe *et al.* 1991). I suggest that acceptable impact levels could be set in a public consultation process using indicators such as river health (CRC for Freshwater Ecology 2002) and vegetation cover (Dyring 1990).

A more contentious management option is eradication of smaller populations such as those in the Wonangatta-Moroka Unit of the Alpine National Park and Snowy Plains in Kosciuszko National Park. Removal of such populations would remove any environmental impact of horses and eliminate the need for ongoing commitment of resources for management. Such options would need to be considered on a case by case basis following criteria suggested by Braysher (1993). For successful eradication to occur six criteria must be met. Firstly, the population must be reduced faster than the natural rate of increase at all densities. This can become increasingly difficult at low densities. Secondly, immigration must be zero. Thirdly, all individuals in the population must be at risk from the control techniques. Fourthly, it must be possible to monitor the population at very low densities otherwise survivors may go undetected. Fifthly, the social-political environment must be suitable. Lastly, cost-benefit analysis should favour eradication over control. Eradication has high initial outlays, but, if it can be achieved, there are no ongoing costs.

Brumby-running is far less time-efficient than some other population control options. It is more closely tied to the culture of wild horses and thus appears to be a more socially acceptable form of control. Brumby-running may be used to achieve some of the control objectives outlined above. The current study highlighted the likely affect brumby-running is having on the wild horse population but future management should test these models with data and update projections.

6.4. Future research

At the commencement of this study there was no scope for manipulative experiments because wild horse management was such a politically sensitive issue in the Australian Alps. Now that there is a general acceptance that wild horses should be managed, there are opportunities to run research programs in conjunction with management to improve our understanding of the dynamics of wild horses and their response to management.

6.4.1. Distribution

Research in this study into historic distribution and its causes was brief. A more elaborate oral history study would be very useful in fleshing out the story presented in this thesis. Areas to focus on would be in Victoria around the Bogong High Plains where I spent very little time, and the Snowy River National Park. Another region that could be explored further is the western side of Kosciuszko National Park. I had little contact with people from the Tumut region, which have an association with wild horses in the mountains.

A landscape approach to assessing the distribution of wild horses would be valuable. Overlaying the map of wild horse distribution on maps of geology, soil, vegetation communities and topography and testing for associations may provide insights into the current boundaries of distribution. I looked briefly at such maps and couldn't detect a correlation, however it should not be dismissed. Such research would be an interesting example of a study on the edge of a distribution of the type described by Caughley & Sinclair (1994).

6.4.2. Abundance

The precision of the aerial survey should be improved by stratifying the area into habitat types (though the low numbers observed could limit this application). Other valuable topics to explore to improve the technique would be the use of a combination of mark-recapture and line transect techniques (for example Borchers *et al.* 1998) for which analysis software is currently being developed. Further research could also compare estimates of abundance using techniques in this study to estimates using thermal imaging- an emerging technology. For example thermal infrared sensing detected more white-tailed deer in tall reed beds and deciduous forests than normal sighting methods (Naugle *et al.* 1996).

6.4.3. Population dynamics

The current study represents a snapshot in time of a long-lived species. It would be beneficial to continue monitoring the three populations to better understand their dynamics.

The results of this study suggested that food is likely to be limiting unmanaged wild horse populations in the Australian Alps. Future research should include a more detailed study on the relationship between wild horses and their food supply. Such research should elaborate on the numerical response developed here and also describe the functional response (Caughley & Sinclair 1994). This will also be useful in addressing an important consideration- temporal environmental variability. The current study assumed a relatively stable but seasonal environment. If the environment is variable, then the concept of carrying capacity is less helpful and other models should be employed to describe the relationship between wild horses and their food supply (McLeod 1997).

Emigration and immigration are key components of population dynamics (Caughley 1977a) that were not addressed in this study. Assumptions were made about movement patterns (Chapter 4) that should be tested. Movement patterns of horses should also be considered when management is planned. For example, brumby-running is likely to lead to more movement of wild horses than trapping.

6.4.4. *Brumby-running*

If managers plan to continue to use brumby-running as a management tool, it is important that it is accompanied by monitoring to ascertain whether the projections made from the general models in this thesis are realistic and to update the projections.

For a more detailed understanding of the wild horse population in general and the effect of brumby-running on the wild horse population, there is some key information that brumby-runners could collect in addition to the information that they already collect. Catch per unit effort can provide an index of abundance of wild horses. Brumby-runners could include details of the number of days spent searching for horses. The crudest equation of the relationship is Schaeffer's $CPUE = qN$ where q is a catchability coefficient (Seber 1992). This is also a linear functional response in predator-prey theory (Begon *et al.* 1990). So, if wild horses are twice as common at one time compared to another, twice as many will be caught by the same means. Furthermore, in a declining population CPUE should decrease as animals become more difficult to catch. Conversely an increasing population should lead to

an increasing CPUE (Strickland *et al.* 1996). There are many problems associated with the use of CPUE as an index of abundance, the main one being that CPUE can usually be maintained or even increase as stocks decline, at least in the short term (McCallum 2000). CPUE is used a lot in fisheries research and this basic relationship has been modified and improved. For further information see Hilborn & Walters (1992).

It would be very useful if brumby-runners recorded body condition of horses caught (using Huntingdon 1991). The body condition scores could be used in two ways. If it was collected in conjunction with body condition data from the broader population it could be used to assess whether the finding (section 5.3.2), that BR1 and his companions do not select animals in poor condition, applies to all brumby-runners. If this is the case then the body condition score of horses caught by brumby-runners can be used to assess the health of the population as a whole. That is, if the mean body condition of horses caught by brumby-runners is low, then the entire population may be suffering from food shortages. This method was successfully employed by Solberg *et al.* (1999) to track the health of the moose population.

6.4.5. General

Many areas of wild horse ecology relevant to their management in the Australian Alps are poorly understood. Some of the key areas of research that would improve management would be assessment of the relationship between the density of wild horses and their impact. This relationship is unlikely to be linear (Hone 1994). A study could be done in conjunction with management if a goal of management is to reduce density (such as at the Big Boggy). Any such study would benefit enormously from an experimental control area (where density is not reduced) so that the effect of environmental variability would not confound results.

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