CASUARINA PAUPER (BELAH) WOODLANDS OF NORTHWEST VICTORIA: MONITORING AND REGENERATION

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ABSTRACT

Casuariha pauper (Belah) semi-arid woodland is one of the highest priority communities for conservation in Victoria. Little is known about its pre-European distribution and structure, and the long-term future of remnants is threatened through failure of regeneration of overstorey species and perennial understorey. Research was conducted to determine the original structure and distribution of *C. pauper* woodland in northwest Victoria, the effectiveness and cost of alternate methods of monitoring condition, and the regeneration requirements of perennial components.

Historical survey maps and other records were used to determine the likely distribution and structure of semi-arid woodlands at the time of settlement. High variability in woodland structure and composition at the time of survey (1860s–1930s) was found, suggesting the need for caution when applying benchmarks. More than 96 % of semi-arid woodlands on private land in the study area have been cleared, highlighting the conservation significance of remnants and the need to manage remnants to promote regeneration to ensure their continued survival.

Remote methods of vegetation condition assessment were compared with field-based methods to determine the most cost-effective approach. Only two condition classes (good and poor) were reliably mapped using the remote techniques investigated (overall accuracy 87.1–94.9%). Whilst field techniques gave greater differentiation at survey sites, when interpolated across the study area, maps were no more accurate. The normalised differential vegetation index (NDVI) was highly correlated with many of the field condition parameters including the field vegetation condition index (0.81), and provides the most promising technique for remote assessment of vegetation condition. Remote survey techniques are unlikely to replace field-based methods, but are likely to provide valuable information when used in conjunction with them.

Field and laboratory experiments were undertaken to examine the regeneration requirements of *C. pauper* woodland species. Abundant viable seed of *C. pauper* was found (>340 m^{-2}) within transient soil seedbanks that are replenished most years. A field investigation of simulated high rainfall coupled with glasshouse experiments suggests *C. pauper* regeneration is likely to be inhibited most years by insufficient soil moisture for seedling survival, although germination under relatively low soil moisture conditions (35 % FC) suggests that germination events may occur most years. Soil seedbank studies found only low numbers of germinants for most other perennial species suggesting that regeneration may be limited by transient soil seedbanks.

EXECUTIVE SUMMARY

Casuarina pauper (Belah) woodlands are extensively distributed through semi-arid south-eastern Australia. *C. pauper* woodlands have been used as indicators of suitable soils for cropping, and so have been preferentially cleared across their range. *C. pauper* woodlands have also been heavily utilised for stock grazing. The long-term future of *C. pauper* woodland remnants is now threatened through failure of regeneration of overstorey species and perennial understorey.

This study focused on *C. pauper* woodland in northwest Victoria, an area where the recent establishment of the Murray-Sunset National Park (MSNP) provides an ideal opportunity to manage these woodlands to promote regeneration. Some information on the closely related *Callitris gracilis – Allocasuarina luehmannii* (Pine-Buloke) woodlands has also been collected, as many key species occur in both these woodland types in the study area, and similar regeneration problems have been noted. Following declaration of the MSNP in 1991, priorities for gathering baseline data and establishing a monitoring program to determine the effects of management were identified (NRE 1996).

This research has investigated three main components of the ecology and assessment of semi-arid woodlands in northwest Victoria:

1. the pre-European distribution, structure and composition
2. methods for monitoring vegetation condition, comparing remote data sources with field survey techniques; and
3. the regeneration requirements of perennial components of *C. pauper* woodland;

The main objectives of this research were to:

1. determine the pre-European distribution, structure and composition of semi-arid woodland by reference to historical survey plans, and hence;
2. develop appropriate benchmarks for vegetation condition assessment;
3. investigate techniques for remote vegetation condition assessment, and determine the most cost-effective techniques;
4. determine conditions required for regeneration of perennials in *C. pauper* woodland through field and laboratory experiments;
5. investigate if replication of rainfall quantities of the early 1970s may promote regeneration of perennial species in *C. pauper* woodland; and
6. investigate the effects of soil disturbance and fire on the regeneration of perennial species in *C. pauper* woodland.

Pre-European distribution, structure and composition of *Casuarina pauper* woodland

To determine the likely pre-settlement distribution of *Casuarina pauper* and *Callitris gracilis* – *A. luehmannii* woodlands, historical survey maps and Parish plans (1860s–1930s) were used to create a geographical information system (GIS) database. Historical surveys of northwest Victoria were conducted principally to facilitate land selection and settlement, and now provide a wealth of information on the vegetation at the time of European settlement. Historical maps containing evidence of semi-arid woodland species were scanned and geo-registered to enable use in a GIS. This allowed production of maps showing woodland distribution and density and enabled calculation of the extent of clearing of semi-arid woodlands. The spatial database also provides a base to which any additional historical data could be added.

Tests of the accuracy of the survey plans and derived maps show that they correlate well with current vegetation patterns. Historical survey plans were found to be a valuable data source for determining the pre-European distribution and structure of semi-arid woodlands.

Maps produced from these data indicate the likely extent of semi-arid woodlands in northwest Victoria prior to European settlement. Results suggest over 250,000 ha (19 % of the study area) originally contained semi-arid woodland species. 'Pine' (*Callitris gracilis*) was the most frequently noted semi-arid woodland species.

The survey plans and other historical records show that semi-arid woodlands in northwest Victoria were variable in both composition and density. Historically, semi-arid woodlands supported higher species diversity, with many species, such as *C. gracilis,* less frequently occurring today. Woodlands at the time of settlement were more likely to be described as open, than dense, although a number of dense woodland areas were noted.

More than 96 % of semi-arid woodlands have been cleared in agricultural regions surrounding the MSNP. In addition, there has been an overall decrease in tree density within vegetation remnants throughout the study area, due to tree harvesting, thinning and senescence in the absence of regeneration. This highlights the need to conserve and manage remnants to ensure regeneration.

Vegetation condition assessment

Condition assessment is increasingly being used as a tool for monitoring vegetation that has been subject to disturbance. Remote methods of condition assessment (satellite imagery, aerial photography and the Treeden25 GIS layer produced by Department of Sustainability and Environment) were compared with field-based techniques to determine cost-effective methods for condition assessment in semi-arid woodlands.

Methods for remote vegetation condition assessment were developed through study of the literature, and investigation of a number of different analyses for each remote data source. Comparison of vegetation condition maps produced from remote assessment techniques showed that only two vegetation condition classes could be reliably mapped using the techniques investigated. However, maps of two vegetation condition classes provide limited information on the outcomes of vegetation management, and are unlikely to be sensitive to vegetation change.

The normalised differential vegetation index (NDVI), calculated from Landsat imagery, was found to be highly correlated with the field vegetation condition index, and many other field condition parameters. This is understood to be due to the nature of vegetation clearance and disturbance within the study area. High grazing pressure, tree clearing and thinning have all resulted in a reduction of perennial vegetation cover, along with a decrease in native species richness, inhibition of perennial recruitment, and increase in cover and abundance of exotic annual vegetation. In the absence of exotic perennials, or woody weed problems in the study area, high perennial vegetation cover is strongly correlated with good vegetation condition.

Costs for the remote condition assessments were very similar to that of the field based survey. This is in part due to the relatively low sample size for the field survey. However, the need for ground truthing of remote data to assess the accuracy of the analysis, and costs of data acquisition, pre-processing and analysis all contributed to costs of remote condition assessment. The main advantages of remote vegetation condition assessment over field assessment are greater ability to detect changes across the entire study area, and greater flexibility in assessment timing.

The Treeden25 layer, was the cheapest option for mapping vegetation condition, however, there are a number of limitations with using existing data sets. Replication of the assessment is reliant on other agencies to repeat their analysis, therefore repeat assessments cannot be undertaken when desired. As the data have been analysed on a state-wide basis, parameters have been developed that are not necessarily sensitive to the needs of assessment in semi-arid woodlands.

The NDVI shows potential as a monitoring tool to assist in detecting change in vegetation condition of semi-arid woodlands within the MSNP and surrounding remnants. It is likely that remote techniques could not replace field survey, due to the importance of perennial recruitment, and the difficulty of detecting recruitment using remote techniques. Remotely sensed assessment is expected to be of most benefit when used in conjunction with field assessment techniques.

Regeneration requirements of perennial species

Continuing regeneration failure, despite the occurrence of rainfall events, has often been attributed to high grazing pressure. However, there is little known about the likely success of regeneration under low grazing pressure. This research was conducted to investigate the conditions under which regeneration of perennial species of *C. pauper* woodland may occur.

Field and laboratory experiments were undertaken to examine regeneration processes and requirements of perennial components of *C. pauper* woodland, at the seed and early seedling establishment stages. Experiments and observations were made to determine; (i) the effects of watering, soil disturbance, seeding and fire on regeneration within a large grazing exclosure; (ii) if seed was being produced, the impact of granivores, and presence and quantity of seed in the soil seedbank; (iii) seed viability, dormancy characteristics, temperature and soil water requirements for germination; and (iv) soil water requirements for seedling survival.

Seed of the major overstorey species was produced each year from 2000 to 2003. Soil seedbank investigation revealed viable *C. pauper* seed was present in 63.8 % of soil and litter samples with seed densities of up to 399 m^{-2}. Seed viability was found to be a limiting factor for *Alectryon oleifolius*, but did not appear to inhibit regeneration of other species (percentage germination from 14–67 %). Significantly greater germination rate of *C. pauper* was found in summer temperatures, with gemination occurring at soil water contents from 9 %.

No *C. pauper* or other overstorey species were observed to germinate in the field experiment, although seedlings of *Enchylaena tomentosa*, *Olearia pimeleoides*, *Dodonaea viscosa* and *Acacia oswaldii* established under watered conditions in the exclosure.

Sufficient soil moisture for *C. pauper* seed germination is likely to occur most years. This research suggests that low soil water content is likely to be a major factor limiting seedling establishment, supporting findings of previous research that consecutive years of well above rainfall are required for regeneration.

STATEMENT OF AUTHORSHIP

Except where explicit reference is made in the text of the thesis, this thesis contains no material published elsewhere or extracted in whole or in part from a thesis by which I have qualified or been awarded another degree or diploma.

No other persons work has been relied upon or used without due acknowledgement in the main text and bibliography of the thesis.

Signed:

Katrina E. Callister

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TERMINOLOGY AND ABBREVIATIONS

To avoid potential problems with taxonomic revisions when discussing past studies of the vegetation, all plant names have been updated to the most recent as listed in Walsh and Entwisle (1994; 1996; 1999). Nomenclature issues of four dominant canopy species of *Casuarina pauper* woodlands are summarised below.

Current nomenclature	Previous nomenclature	Common names
Casuarina pauper ex L.A.S. Johnson	*Casuarina stricta* ssp. *pauper, Casuarina cristata, Casuarina pauper* ssp. *pauper, Casuarina lepidophloia, Casuarina cambagei*	Belah, Black Oak, Belar, Scaly-barked Casuarina, Scrub She-oak, Bulloak, Swamp Oak, Billa, Ngaree, Grey Buloke
Myoporum platycarpum ssp. *platycarpum* R. Br. *ssp. *perbellum* is distinguished by a narrower diameter of the flower.	*Myoporum platycarpum*	Sugarwood, False Sandalwood
Alectryon oleifolius ssp. *canescens* S.T. Reynolds *ssp. *elongatus* S.T. Reynolds has a similar range, but has not been collected in Victoria.	*Heterodendrum macrocalyx, Heterodendrum oleifolium; Heterodendrum oleifolium* var *macrocalyx*	Rosewood, Cattle Bush, Mindra, Bullock Bush, Western Rosewood, Bonaree
Callitris gracilis ssp. *murrayensis* (J. Garden) K.D. Hill	*Callitris robusta*, *Callitris propinqua*, *Callitris preissii*, *Callitris preissii* ssp. *murrayensis*, *Callitris preissii* ssp. *preissii*	Native Pine, Slender Cypress-pine, Southern Cypress-pine, Murray pine

Alectryon oleifolius ssp. *canescens*, *Callitris gracilis* ssp. *murrayensis*, *Myoporum platycarpum* ssp. *platycarpum*, will be referred to as *Alectryon oleifolius, Callitris gracilis* and *Myoporum platycarpum* unless the subspecies is required for taxonomic clarity.

Abbreviations	
AGL	Above ground level
ASL	Above sea level
DSE	Department of Sustainability and Environment
EVC	Ecological Vegetation Class
FIS	Flora Information System
LCC	Land Conservation Council
MSNP	Murray-Sunset National Park
NRE	Department of Natural Resources and Environment
NDVI	Normalised Differential Vegetation Index
SFAP	Small-format Digital Aerial Photography
y BP	Years Before Present

Style follows the current style of the CSIRO journal, *Australian Journal of Botany*.

Vegetation community nomenclature

Many Ecological Vegetation Classes (EVCs) were described in the study area by the Land Conservation Council (LCC 1987). Community names and descriptions have been updated in recent 1:25,000 EVC mapping of private land in northwest Victoria.

The current study has focused on *Casuarina pauper* (Belah) woodland, describing woodland dominated by one of a number of species including *Casuarina pauper*, *Alectryon oleifolius*, *Callitris gracilis*, *Myoporum platycarpum* and *Hakea leucoptera*. This broad definition of *C. pauper* woodland encompasses a number of EVCs from past and more recent mapping. The use of a broader community definition is in accord with findings of Beadle (1948), Porteners *et al.* (1997) and Westbrooke (1998) that local dominance of overstorey species and differences in grazing history occur within this variable community.

The closely related *Callitris gracilis – Allocasuarina luehmannii* (Pine-Buloke) woodlands are indistinguishable from *C. pauper* woodlands by most remotely sensed data sources. They were also not adequately separated in most historical records. Therefore, *C. gracilis – A. luehmannii* woodlands have been included in these sections of the thesis. The term semi-arid woodland has been used to describe both *C. pauper* and *C. gracilis – A. luehmannii* woodlands in the study area.

1. INTRODUCTION

1.1 VEGETATION IN THE SEMI-ARID ZONE OF SOUTH-EASTERN AUSTRALIA

1.1.1 Aridity in the Australian environment

Aridity is commonly defined in terms of precipitation. The arid zone in Australia is defined as the area where the annual rainfall is less than 250 mm (Barlow 1994), although this may be extended to 350 mm in the north where higher temperatures lead to greater evaporation (Beadle 1981). More than a third of the Australian environment is currently arid, with further large areas seasonally arid (Grove 1977; Barlow 1994). Semi-arid regions occur where the rainfall is between 250 and 500 mm (Meigs 1953). Arid and semi-arid lands represent 69.5 % (5,294,000 km^2) of Australia's landmass (ADTR 1982). For the purposes of this review, the term arid will be used to encompass both arid and semi-arid areas unless otherwise specified.

Precipitation in arid regions of Australia, as in many other arid regions in the world, is highly variable in time and space (Stafford Smith and Pickup 1990). In the south eastern arid zone of Australia, the majority of rain falls in winter, compared with predominantly summer rainfall in the north (Beadle 1981). High evaporation rates are a feature of Australian arid lands, with annual evaporation commonly exceeding annual rainfall values.

Changing aridity in south-eastern Australia

Current aridity in south-eastern Australia is unrepresentative of the long-term climate, having existed for approximately 5 % of the last 120,000 years (Bowler 1990). Arid climates were evidenced across Australia between 20,000–13,000 y BP (years Before Present) (Grove 1977), coinciding with the peak of the last ice age (Bowler 1980; Barlow 1994). Active dune building and lowered lake levels were a feature of this arid period which was quite extensive across Australia, incorporating areas outside the current limits of the arid zone (Bowler 1976; Grove 1977).

Following this arid period, conditions began to alter, with temperatures slowly rising and increased precipitation from approximately 15,000 y BP (Bowler 1976; Grove 1977). There is some disagreement about the date of culmination of this pluvial period, which is suggested to have occurred somewhere between 15,000 y BP (Bowler 1976; Beard 1982) to 3,500 y BP (Barlow 1994). However, it is widely agreed that during this period temperatures were slightly higher (2–4°C) with rainfall 150 % higher than present (Grove 1977; Beard 1982; Kershaw and Nix 1988; Bowler 1990; Barlow 1994).

From the pluvial period to the present day, conditions have became cooler and drier (Bowler 1976; Beard 1982). Whilst there have been fluctuations in climate, they have been on a much smaller scale (Grove 1977). Recent arid periods have been noted between 6,000 to 3,000 y BP and the last 2,000 years (Bowler 1976). However, these periods have not been more extreme than the current climatic range (Bowler 1976).

Effects of climate change on vegetation

Changes in climatic conditions as outlined above have had a significant impact on the evolution and distribution of present day flora and fauna in south-eastern Australia. It is thought that current day arid flora may have evolved 100,000 to 1,000,000 y BP (Barlow 1994; Dodson 1994). Changes in climate within the last few thousand years have also had a large influence on the distribution of flora in the Australian landscape (Barlow 1994; Dodson 1994).

Casuarinaceae and Poaceae were found to be dominant in the pollen record from Lake Tyrell in northwest Victoria during relatively dry conditions between 10,000 and 6,600 y BP (Luly 1993). From 6,600–2,200 y BP, an increase in pollen from *Callitris* and *Eucalyptus* was found during this relatively pluvial phase. A brief period of decreased rainfall between 2,200–800 y BP led to an increase in Casuarinaceae, but *Eucalyptus* remained dominant. From 800 y BP to the present, a return to slightly wetter conditions has led to the establishment of dense mallee vegetation, with scattered patches of *Callitris gracilis* (Luly 1993). Sluiter and Parsons (1995) suggest that the Casuarinaceae pollen is likely to have been *Casuarina pauper*, possibly with *Allocasuarina luehmannii* present in small numbers.

There is evidence from pollen records in Lake Frome, South Australia, of increased numbers of trees and tall shrubs during a period of higher rainfall between 9,500 to 8,000 y BP Singh (1981). Trees and tall shrubs appeared to have declined until another wet period from 7,000 to 4,200 y BP when the proportions of Casuarinaceae increased. From this time, the climate appears to have been relatively dry, with increased evidence of fire (Singh 1981).

Evidence of a relatively recent change in the climate of arid eastern Australia is significant in explaining current day distributions of perennial overstorey species. It is possible that the full effects of historical climate change are still to be expressed. If the more recent date of 3,500 y BP is accepted as the date of change from more pluvial conditions to current climatic conditions, it is possible that distributions of these species will still be in a state of flux. For long-lived species with a potential life span of over 300 years, this period may only represent 10 generations. Many of these species may regenerate under less favourable conditions by clonal growth. For example, most regeneration of *Alectryon oleifolius* since European settlement appears to be due to vegetative re-sprouting from root disturbance (Westbrooke 1998). It is possible that this has been a long-term survival mechanism with minimal regeneration from seed over a number of centuries.

It is feasible that some species may have had limited successful seed regeneration since the change in climatic conditions at approximately 3,500 y BP. Moreover, climatic fluctuations have been noted during this period, and so populations potentially may have been sustained by occasional infrequent periods of increased rainfall allowing rare events of successful seedling establishment.

1.1.2 Adaptations of arid vegetation

Adaptations of flora to arid zone conditions may include behavioural, morphological, and physiological mechanisms (Heathcote 1983). All stages, from germination and seedling establishment to growth and flowering, exhibit adaptations to aridity.

Four broad response categories of organisms to arid conditions provide a framework for understanding adaptations to aridity (Schantz 1956). Australian arid plants use all four drought response categories: (i) drought escaping annuals grow during moist seasons and live through drought conditions in the seed stage; (ii) drought evading plants make economical use of limited soil moisture supply through wide spacing, reduced leaf and stem surface; (iii) drought resistant plants employ mechanisms such as deep-roots, or succulent leaves to enable them to survive low water conditions; and (iv) drought enduring, or drought-dormant plants aestivate when droughts occur, and continue to grow when moisture is available. This last category includes many prominent arid seed plants and also algae, lichens, mosses and ferns.

Many adaptations of arid vegetation relate to regeneration, as juveniles are most susceptible to desiccation. Many arid zone flora species use vegetative regeneration as a means of establishing new populations (Noble 1984a). Vegetative regeneration allows the developing sucker to utilise the resources and protection of established trees enabling more photosynthetic tissue to be produced, and hence greater seedling growth by utilising deeper moisture reserves than newly establishing seedlings can access (Maconochie 1982). For many species regeneration by clonal growth may exceed that by seed (Gijsbers *et al.* 1994).

Whilst clonal growth reduces the need for seed germination and seedling establishment, regeneration by seed is essential if new genetic populations are to be formed. Regeneration of the perennial component in arid regions appears to be a rare event (Gupta 1979; Maconochie 1982). Successful regeneration and seedling establishment is dependent on favourable climatic conditions, with moisture the major limiting resource. Some species rely on a trial and error strategy, attempting regeneration most years in the hope of adequate rainfall (Gupta 1979). For other species, various adaptations of flowering, seed dormancy and germination link seed regeneration events to periods of increased soil moisture (Gupta 1979). However, some Australian arid species have rarely been observed as seedlings, and mechanisms for seed germination have eluded researchers (Beadle 1981).

1.1.3 Status of arid woodlands in south-eastern Australia

Since European settlement, there has been a widespread conversion of semi-arid woodlands to agricultural areas. Woodlands often occur on better soils for agriculture, and so were preferentially cleared. Southern Australian woodlands in particular have been extensively cleared with between 40–90 % of woodlands cleared (McIntyre *et al.* 2002). In Victoria, it is estimated that 92 % of woodlands had been cleared by 1987 (Woodgate and Black 1988).

In addition to widespread clearing, thinning of existing remnants for pasture improvement, timber harvesting, firewood collection and lopping trees for stock feed has altered the structure of many woodland remnants (Allen 1983; Westbrooke *et al.* 1988; Mott and Tothill 1993; ASOEC 2001). Rising groundwater tables, salinity, agricultural chemicals and fertilisers also place additional pressure on remnant vegetation in an agricultural matrix. These problems have been compounded by continuing natural or modified processes, including fire, grazing by invertebrates and vertebrates, mistletoe, , natural tree deaths due to old age and severe droughts, storms and frosts (Allen 1983). Whilst all of these factors have affected semi-arid woodlands in south-eastern Australia, the impacts of clearing, thinning and grazing have been most extensive (Porteners *et al.* 1997).

Change in grazing regimes have had a variable impact on semi-arid woodlands, with almost complete inhibition of perennial regeneration occurring in many lower rainfall systems (Crisp and Lange 1976; Crisp 1978; Wisniewski and Parsons 1986; Cheal 1993). This is in contrast to woody weed problems occurring in higher rainfall areas where perennial regeneration has been favoured by disturbance (Harrington *et al.* 1984). These problems have been exacerbated by changes in fire regimes, which may have contributed to woody weed regeneration and survival (Harrington *et al.* 1984).

Grazing

There have been many studies on the effects of rabbits, sheep, and native herbivores on the regeneration and diversity of remnant vegetation in both arid and temperate regions of south-eastern Australia. These studies have highlighted the susceptibility of Australian woodlands to grazing by introduced herbivores.

Grazing and browsing have numerous direct effects on native vegetation. Decreases in individual plant height, growth rate, seed production, competitiveness and regeneration have been observed (Burrows 1990; Hulme 1996). Community composition and diversity are also affected by grazing (Pigott 1983; Gibson and Kirkpatrick 1989; Kinucan and Smeins 1992; Cheal 1993; Pettit *et al.* 1995; Prober and Thiele 1995). Indirect effects of grazing include change in soil nutrients, compaction, erosion, soil disturbance and dispersal of weedy exotics by herbivores (Gibson and Kirkpatrick 1989; McIntyre and Lavorel 1994; Prober and Thiele 1995; Yates and Hobbs 1997a).

Rabbits

Rabbits were first successfully introduced in Australia at Geelong in 1859 (Stodart and Parer 1988). Following initial introduction, and possibly assisted by additional concurrent introductions in other areas, rabbits rapidly spread across Australia. By the late 1870s, rabbits had reached semi-arid south-eastern Australia (Rolls 1969). By 1881 the rabbit problem was so severe that farms were being abandoned in northwest Victoria (Myers *et al.* 1989).

It was not until the early 1950s, with the introduction of myxomatosis, that any significant long-standing impact on rabbit numbers was achieved (Coman 1999). However, even this breakthrough had a limited impact in arid regions where the disease vector, the mosquito was limited due to lack of water. The European and Spanish rabbit fleas were introduced in an attempt to overcome this problem with limited success (Coman 1999).

Recently rabbit numbers has been reduced through the introduction of rabbit haemorrhagic disease (RHD), first detected in rabbit populations in China in 1984. The virus was brought into Australia in 1991 for investigation as a potential biological control (Cooke and Fenner 2002). In 1995, RHD was observed in wild rabbit populations on mainland Australia prior to planned introduction. Following this accidental introduction, RHD has spread throughout much of southern Australia with a significant reduction in rabbit numbers reported in some areas (Coman 1999; Cooke and Fenner 2002).

Rabbits have been implicated in the widespread inhibition of woody plant regeneration in Australia, including *C. pauper*, *Acacia ligulata*, *A. loderi* and *Alectryon oleifolius* (Onans and Parsons 1980; Lange and Graham 1983; Chesterfield and Parsons 1985; Cooke 1987; Auld 1993, 1995a). This is due in part to their characteristic browsing behaviour. Rabbits will severely prune seedlings, repeatedly removing all foliage and stems to almost ground level (Lange and Graham 1983). Small saplings that manage to grow in good years, or in protected sites, are still vulnerable to rabbits stripping bark leading to ringbarking and subsequent death of saplings (Lange and Graham 1983). Rabbits are also highly selective browsers capable of eliminating preferred species, even when present in low numbers (Lange and Graham 1983).

There are many reports of regeneration of woody perennial species following the reduction of rabbit grazing, and many landscapes bear evidence of the introduction of myxomatosis and subsequent regeneration events in the 1950s. *Maireana pyramidata*, *M. brevifolia*, *Myoporum platycarpum* and *Atriplex* species were all found to regenerate extensively following extensive warren ripping in semi-arid South Australia (Mutze 1991). In northwest Victoria following RHD, increased survival rates of vegetative suckers of some species was found (Sandell 2002).

Kangaroos

The creation of artificial water sources, clearing and thinning of overstorey vegetation to improve pastures and reduction in dingo numbers has created ideal kangaroo habitat, and populations have increased accordingly (Robertson *et al.* 1987). Despite increased kangaroo numbers their impact on the regeneration of perennial overstorey species appears to have often been overlooked. This is believed to be based on two underlying assumptions; firstly, that kangaroos are preferentially grazers, and secondly that arid vegetation in Australia has evolved with kangaroos, and therefore would be expected to be well adapted to this form of disturbance (Griffiths and Barker 1966).

Kangaroos are preferentially grazers, and the majority of their diet is based on grasses and herbs (Edwards *et al.* 1995). However, kangaroos will browse and destroy perennials in poor seasons when grass and herbs are in limited supply, when kangaroo populations exceed available grass and herb resources, or simply as a variant in the diet (Edwards *et al.* 1995). Kangaroos have been observed to consume young seedlings of woody plants, and strip leaves from older woody seedlings (Auld 1993; Tiver and Andrew 1997). Whilst a number of studies have found kangaroos do not totally eliminate regeneration (Auld 1995a; Tiver and Andrew 1997), increased seedling mortality has been noted (Auld 1993; Stoneman *et al.* 1994).

Stock

Stock grazing continues to be one of the most extensive and constant disturbances on many arid woodlands. Sheep have been shown to have a major impact on the regeneration of many arid perennial species including *Acacia melvillei* (Batty and Parsons 1992), *Acacia burkittii* (Crisp and Lange 1976; Woodell 1990), *Acacia aneura* (Crisp 1978), and *C. pauper* (Chesterfield and Parsons 1985). Lange and Wilcocks (1980) demonstrated that sheep would selectively browse on a favoured element, even when it represents a small proportion of available feed. Over-grazing by stock has had a large impact on some areas, particularly around watering points (Pickup 1994). Following overgrazing, minimal regeneration is observed even following destocking (Hall *et al.* 1964).

Soils

Soils have also been affected by stock grazing in the semi-arid zone. Decreased ability to trap resources is apparent in increased water runoff, and occurs due to reduced cover of perennial ground cover, litter and cryptogams (Greene *et al.* 1994; Ludwig *et al.* 1997; Yates *et al.* 2000). This in turn can lead to greater erosion by wind and water (Ludwig *et al.* 1997). Stock grazing leads to changes in soil nutrients from urine and faeces (Yates *et al.* 2000), resulting in a less favourable environment for native plant establishment and encouraging weed invasion (Greene *et al.* 1994).

1.1.4 Current condition of arid woodlands in south-eastern Australia

Widespread clearing and thinning, altered grazing regimes, and other deleterious disturbances, have changed arid woodlands across south-eastern Australia to the extent that it is now difficult to determine the pre-European distribution, community structure and function (Gillison 1994; Pickard 2002). There has been little regeneration of many overstorey species since European settlement, and many remaining populations are approaching senescence (Lange and Purdie 1976; Westbrooke 1998). Despite exceptional rainfall in the early 1970s, little successful regeneration of perennial woodland species was observed due to high grazing pressures. There are few areas where fortuitous circumstances have combined to allow some successful regeneration (Stannard 1958; Woodell 1990).

1.1.5 Requirements for regeneration of woodland perennials

Across the arid woodlands of south-eastern Australia the extent of decline is not homogenous. There is a range of decline in woodland condition from areas where few if any indigenous species remain, to areas where some overstorey species are not regenerating, but the community remains largely intact (Westbrooke *et al.* 1988; Porteners *et al.* 1997; Sluiter *et al.* 1997). A single management response is unlikely to lead to successful recovery in all situations. A range of responses will be required to match the degree of decline.

For successful natural seed regeneration in the Australian arid zone, six requirements have been noted (Hall *et al.* 1964):

(i) source plants are present;
(ii) the species flowers and fruits readily;
(iii) the seeds remain viable until a suitable season for germination;
(iv) the seeds have a relatively high germination rate;
(v) there is a suitable seedbed for establishment; and
(vi) seedlings are relatively unpalatable to introduced grazers such as rabbits and sheep.

For seedlings that are palatable to introduced and native grazers, low grazing levels may be the key to regeneration. The potential for recovery of vegetation following cessation of grazing will depend on a number of factors. The resilience will be related to the timing and extent of disturbance, as well as the inherent resilience of the system (Hacker 1989). In less degraded areas, removal of the disturbing influence may be sufficient for natural regeneration to occur. Grove (1977) suggested that grazing exclusion should lead to successful recovery of semi-arid vegetation, but there is limited support for this statement following grazing exclusion trials.

Mutze (1991) found regeneration of many perennial shrubs occurred following destocking and rabbit control in South Australian rangelands in a rare example of rapid recovery. However, there are many more examples of continued regeneration failure than regeneration success following grazing cessation (Hacker 1989). Following cessation of grazing on previously cleared, grazed and cropped land, minimal regeneration of mallee eucalypts was found in northwest Victoria (Onans and Parsons 1980), and even after 50 years minimal mallee eucalypt regeneration was found (Liangzhong and Whelan 1993). Long-term ecological research has been conducted at Koonamore Station in South Australia, which was previously heavily over grazed. In 1925, stock were removed and natural regeneration monitored (Wood 1936). Following 26 years of grazing exclusion no significant regeneration of overstorey species was observed (Hall *et al.* 1964). However, these examples did not adequately control or monitor rabbit numbers.

Past over-grazing events have been shown to reduce the ability of systems to respond to conditions favourable for regeneration (MacLeod *et al.* 1993; Yates and Hobbs 1997a). However, some researchers have found an increase in resilience following grazing exclusion where increased species diversity endowed greater adaptation to environmental fluctuations (Carr and Turner 1959; Tilman and Downing 1994). Where resilience is low, restoration strategies to capture and retain resources, such as ripping, furrowing, cultivation and pitting may be required (Yates *et al.* 2000). These treatments have been shown to enhance trapping and maintenance of resources including water, seed and organic matter (Cunningham *et al.* 1976; Hobbs and Mooney 1993; Stoneman *et al.* 1994; Friedel *et al.* 1996; Kotanen 1997; Mnene *et al.* 1999).

1.1.5.1 Rainfall as a trigger for episodic regeneration

Within semi-arid environments, soil moisture, or rainfall is commonly the limiting resource for plant growth and regeneration. Rainfall in arid Australia is not only low, but is highly variable both temporally and spatially. High daytime temperatures lead to high evaporation rates relative to precipitation (Stafford Smith and Morton 1990). Water availability is dependent on precipitation, evaporation, infiltration into the soil profile, and runoff (Verstraete and Pinty 1991). Only a small proportion of soil moisture is available for plant growth, as the roots are limited in their ability to extract water held in low concentrations within the soil (Verstraete and Pinty 1991). The precipitation rate will also impact on water availability for plants as less water will infiltrate the soil during a heavy rainfall event, however, greater evaporation will occur from frequent light showers (Verstraete and Pinty 1991).

Above average rainfall is frequently linked to regeneration events in many Australian arid species. High rainfall is required for many stages in the reproductive cycle, from seed set, to seed growth and viability, seed fall and germination, seedling establishment and growth. Although the requirement for high rainfall is recognised, the exact quantities and timing remains unknown for the majority of species, although a minimum of 25 mm has been suggested for germination of many arid plants (Gupta 1979).

Precise rainfall requirements of species are likely to be dependent on other factors such as soil type, grazing pressure and competition. Chesterfield and Parsons (1985) found localised high regeneration of *C. pauper* following three years of above average rainfall in the early 1970s. On one site *C. pauper* regeneration was linked to an event where 120 mm rain fell in one day, resulting in five days of flooding (Chesterfield and Parsons 1985).

Indirect effects of high rainfall may also assist regeneration. Flooding of watercourses may reduce rabbit grazing by drowning warrens, or provide a short-term protective barrier to islands of vegetation (Woodell 1990). Stock numbers will also be reduced during periods of prolonged flooding. However, it is unlikely that these effects will provide protection for seedlings over the years necessary to escape grazing. Grazing impacts may also be reduced through extensive grass and herb growth following rainfall, reducing the grazing pressure on developing seedlings. Tall grasses may afford some physical protection, with seedlings less visible through a thick ground layer (Woodell 1990). Raised water tables following extensive rains may also assist regeneration (Woodell 1990).

Extensive root systems are a common adaptation of arid plants. However, the costs of allocating resources to extensive root systems are slower growth rates, and hence longer exposure to vertebrate herbivores (Stafford Smith and Morton 1990). High rainfall may help provide the competitive edge that these species require to establish (Stafford Smith and Morton 1990).

Long-term studies where the number of seedlings per season can be correlated with seasonal rainfall is an effective way of determining the relationship between regeneration and rainfall (eg. Bowers 1994). However, infrequent regeneration events, and poor survival of seedlings under grazing are significant limitations for this type of study. Alternatively, many studies have investigated tree-rings or diameter at breast height (DBH) data to establish if a relationship occurs between year of seedling establishment and rainfall received (Crisp and Lange 1976; Auld 1986; Woodell 1990; Westbrooke 1999). In Australia, this approach is limited by the availability of rainfall records, which have been recorded at best for the last 150 years. In addition, this approach may not show the year of germination, but the year when sufficient rainfall enabled substantial growth of seedlings germinated over many years (Watson *et al.* 1997a).

Another approach that has rarely been used in the study of regeneration, but commonly used to investigate soil erosion, is simulated, or artificial rainfall (Eldridge and Kinnell 1997). This approach was trialed in the Negev desert in southern Israel to attempt to promote germination of arid zone species without success. Germination did not occur for any species despite regular and substantial watering (Evenari *et al.* 1971).

1.1.5.2 Fire

Fire is known to be a significant factor in the regeneration of many Australian vegetation communities. However, the extent to which fire facilitates regeneration in semi-arid woodlands is less certain. Only following high rainfall years is the ground layer vegetation dense enough to provide sufficient fuel to carry a fire (Leigh and Noble 1981). Even during these rare years of abundant ground cover, the rapid grass fire produced leads to a relatively low intensity fire. Possible exceptions to this may occur where litter build up may produce patches of higher intensity fire (Hobbs and Atkins 1988; Hodgkinson and Oxley 1990). Litter build up leads to spatial heterogeneity of fire. Patchy heating allows temperatures to reach the required level for regeneration of many different species across the landscape (Hobbs and Atkins 1988).

In arid areas, fires naturally occur during the first dry season following years of high rainfall (Leigh and Noble 1981). Therefore, summer and autumn fires are more frequently observed.

Hodgkinson (1991) found highest seedling regeneration of perennial shrubs following summer fires and lowest following winter fires.

Seven fire adaptive features enhancing seed reproduction have been suggested by Hodgkinson and Griffin (1982) and include:

(i) fire-enhanced flowering (rare);
(ii) early reproductive maturity (eg. *Dodonaea viscosa* and *Senna artemisioides*);
(iii) heavy seed production;
(iv) long-lived seed store;
(v) protected seed embryos;
(vi) fire-stimulated seed fall; and
(vii) fire promoted germination (common in *Dodonaea*, *Senna* and *Acacia*).

The extent to which germination is enhanced by fire depends on the species involved, the extent of heating and timing of the fire. Fire promoted germination has been demonstrated through chemical effects from smoke compounds, ash and through direct heating effects. The extent of heating the seeds receive will affect the germination rate. The extent of heating depends on the fire intensity, duration, soil moisture and depth of seed burial (Hodgkinson and Oxley 1990). Heat promoted germination is common in the families Caesalpiniaceae, Fabaceae, Mimosaceae and Sapindaceae (Hodgkinson and Oxley 1990).

Indirect effects of fire occur through change in the vegetation structure and changes to the soil and seedbed. The community structure is altered through tree thinning (Wilson and Mulham 1979) and removal of competition (Cluff and Semple 1994). This leads to changes in diurnal light and soil temperatures (Auld and Bradstock 1996). The soil may be altered by changes to soil nutrients (Raison 1980; Humphreys and Craig 1981; Jensen *et al.* 2001), removal of allopathic effects, reduction in levels of soil CO_2 and the addition of leachates from charred wood (Keeley 1987).

1.1.5.3 Soil disturbance

Soil disturbance has been identified as an important factor in the regeneration of many plant species. Soil disturbance has been shown to increase soil nutrients, soil moisture, water infiltration, and increase physical space availability through reducing competition (Cunningham *et al.* 1976; Eldridge and Robson 1997; Kotanen 1997; Rokich *et al.* 2001). Soil disturbance may trigger germination of some species, and altered microtopography can provide safe sites for germination (Hobbs and Atkins 1988). By increasing soil macropores, soil disturbance can enable more water to be stored within the soil. Soil water is also made more accessible to plants, by decreasing soil compaction, and resistance to roots (Stoneman *et al.* 1994).

It has been proposed that deep disturbance will lead to seed germination of sexually reproducing species by increasing water infiltration and loosening soils (Belsky 1986; Kotanen 1997). Shallow disturbance will favour regeneration of resprouters, by disturbing, but not removing roots (Belsky 1986; Kotanen 1997). Increased germination rates of semi-arid woodland species including *Acacia aneura*, *C. pauper*, *Alectryon oleifolius* and *Myoporum platycarpum* following soil disturbance have been observed (Walker *et al.* 1995; Tiver and Andrew 1997). Cunningham and Walker (1973) found regeneration of Mulga woodlands only occurred on sites where soil disturbance treatments had been undertaken.

Negative effects of soil disturbance are also observed. It is often found that weeds are the dominant species regenerating following soil disturbance (Hobbs and Atkins 1988; Pierson and Mack 1990; McIntyre and Lavorel 1994; Prober and Thiele 1995; Kotanen 1997). Early colonising species are favoured by soil disturbance, potentially at the cost of long-lived perennial species.

Much of the soil disturbance currently observed in the arid zone is due to anthropogenic disturbance, such as road grading and firebreaks. Prior to European settlement, native fauna such as Bettongs and Bilbies may have been responsible for more extensive soil disturbance (Martin 2003). Small mammals can create soil disturbances that have been shown to improve water infiltration of soils, and trap litter, and seeds (Laudre 1993; Whitford and Kay 1999). Small mammals may also act as a vector for mycorrhizal fungi, many of which play a vital role in facilitating plant establishment and growth (Claridge *et al.* 1996). Many species of small mammal have become extinct, or persist only in much reduced numbers in arid areas since European settlement. It is now difficult to estimate the impact of the extinction of small mammals on arid plant communities. Estimates from relict burrows in western NSW of up to 70 *Bettongia lesueur* (Burrowing Bettongs) per km^2 (Noble 1995), indicate that small mammals were previously abundant and may have had a significant effect on soil, and subsequently plant communities.

1.2 THE *CASUARINA PAUPER* WOODLANDS OF SOUTH-EASTERN AUSTRALIA

The *C. pauper* woodlands of south-eastern Australia are a structurally and floristically diverse community, distributed from northwest Victoria, through southeast South Australia, western New South Wales (NSW) and into central western Queensland. *C. pauper* woodlands largely occur in semi-arid environments with an average rainfall of between 1000 mm in the north (Queensland) to 250 mm in the south (Beadle 1981).

Within Victoria *C. pauper* is usually the dominant overstorey species, but *A. oleifolius*, *Callitris gracilis*, *M. platycarpum*, *Hakea tephrosperma* or *H. leucoptera* may be locally dominant. Pure stands of a single overstorey species often occur (Beadle 1948). Differences in overstorey species dominance within *C. pauper* woodlands is likely to have occurred due to a competitive lottery effect where species within the same broad regeneration functional group are favoured by their differing response to environmental conditions (Lavorel 1999). Therefore, under certain conditions, regeneration of one species is favoured over another, leading to changes in community dominance and species relative abundance.

In their intact form, *C. pauper* woodlands contain a diverse shrubby understorey including species of *Acacia*, Atriplex, *Senna*, *Olearia*, *Eremophila*, *Maireana* and *Dodonaea*. Tall shrubs commonly comprise an important strata in northern woodlands, decreasing in frequency to the south (Beadle 1948). Tree density, previous management (Williams 1979) and soil type also influence the extent of shrubby understorey, with almost continuous cover on loamy soils, dominated by *Maireana* species when shallow and *Atriplex* when deep (Boomsma and Lewis 1980). Where the woodland is very open, perennial tussock grasses tend to dominate the groundlayer (Beadle 1948; Boomsma and Lewis 1980), whilst in very densely treed areas almost complete suppression of the understorey may occur (Chesterfield and Parsons 1985).

Tree density in *C. pauper* woodlands varies with trees or small clumps of trees spaced up to 1 km apart, or as little as 4–5 m apart in more dense areas (Cunningham *et al.* 1981). Hall *et al.* (1964) observed 109 overstorey individuals ha^{-1} at Koonamore. Up to 400 stems ha^{-1} have been measured in dense *C. pauper* woodlands (Chesterfield and Parsons 1985).

C. pauper woodlands occur largely on calcareous soils, usually brown calcareous earths, solonized brown soils (Beadle 1948), or calcareous red earths (Cunningham *et al.* 1981). Underlying limestone (Beadle 1948; Hall *et al.* 1964) or sandstone (LCC 1987) formations are common. Beadle (1948) states that *C. pauper* alliances are restricted to alkaline soils. *M. platycarpum* has been found to be dominant on areas of shallow loamy soil, with *A. oleifolius* preferring more fertile soils on low dunes, ridges and floodplains (Hall *et al.* 1964).

Current condition of *Casuarina pauper* woodland

C. pauper woodlands, like many arid communities, have been subject to widespread clearing and thinning, and altered grazing and fire regimes. This has led to scalding, soil erosion, tree dieback and lack of regeneration (Morgan and Terrey 1992; Porteners *et al.* 1997). Less than 1 % of pre-European *C. pauper* woodlands are now regenerating (McLennan and Cooke 1991). As remaining woodlands reach senescence, the likelihood of regeneration further decreases.

Floristic diversity and cover is significantly reduced in most *C. pauper* woodland remnants (Westbrooke *et al.* 1988; Cheal 1993). In addition, the ground flora exhibits a high level of weed infestation consistent with overgrazing (Cheal 1993).

***Casuarina pauper* woodlands in north west Victoria**

Similarly to many semi-arid woodlands across south-eastern Australia, woodlands in northwest Victoria have been identified as having been degraded by heavy grazing pressure, tree clearing and thinning. Lack of regeneration of perennial species, decrease in perennial species diversity and abundance, and invasion of weeds have resulted from past land management (Cheal 1993; NRE 1996). In 1991, following recommendations of the Land Conservation Council (LCC 1989), much of the remaining semi-arid woodlands in northwest Victoria were conserved within the Murray-Sunset National Park (MSNP), providing an opportunity to manage *C. pauper* woodlands for conservation and to study regeneration issues. This research has focused on woodlands in northwest Victoria, although it is expected that results may be more widely applicable.

1.2.1.1 Casuarina pauper

C. pauper is a tall shrub or tree. In open stands trees grow from 5–7 m tall (Beadle 1948; Cunningham *et al.* 1981), reaching up to 15–20 m in more dense stands on heavy clays and floodplains (Cunningham *et al.* 1981). The bark is dark, rough and fissured. Few large branches support many slender, spreading branchlets. Leaves are reduced to small pointed scales occurring at the joints of branchlets (Boomsma and Lewis 1980; Cunningham *et al.* 1981; Entwisle 1996). Hall *et al* (1964) report variation in foliage colour with changing moisture availability.

Flowering usually occurs in summer and autumn, however, it has also been recorded in winter (Cunningham *et al.* 1981). *C. pauper* is dioecious, with male flowers in small terminal spikes 1–3 cm long and females clustered in heads (Cunningham *et al.* 1981).

The fruit is a woody cone containing many winged seeds 6–11 mm long. Large seed crops have been recorded (Chesterfield and Parsons 1985), and seed may be viable for many years (Godfrey 1979). *C. pauper* is easily propagated by seed in nurseries and seedlings are commonly planted in revegetation works (Cunningham *et al.* 1981). Despite this, few natural seedlings are observed and regeneration appears to occur primarily from root suckering (Cunningham *et al.* 1981).

C. pauper is readily browsed by stock, goats, rabbits and kangaroos (Cunningham *et al.* 1981). In northwest Victoria, Chesterfield and Parsons (1985) found *C. pauper* juveniles only in areas where stock were absent, such as fenced roadsides. Auld (1995a) found at Kinchega National Park, seedlings and suckers that were protected from browsing had a significantly higher survival rate. Sucker survival was found to be significantly greater than that of seedlings (Auld 1995a). Within a large stock exclosure at Koonamore Station in South Australia, no *C. pauper* seedlings were observed in over 35 years (Hall *et al.* 1964).

Seedling occurrences of *C. pauper* have largely been related to years of exceptionally high rainfall, with continuing high soil moisture apparently necessary for seedling survival (Cunningham *et al.* 1981; Auld 1995a). Relatively slow growth rates (after three and a half years, a seedling reached 15 cm in height) increase the time suckers and seedlings remain vulnerable to grazing (Auld 1995a). Observations of *C. pauper* by Sims (1951) seedlings in northwest Victoria suggests that seedlings occur in open, grassed patches in timber reserves, but not within more dense, mature stands.

1.2.1.2 Alectryon oleifolius

A. oleifolius is a shrub or small tree with height varying from 3–10 m. It is found in numerous vegetation communities including *C. pauper* and mallee woodland communities. It is most commonly found on sandy soils containing limestone nodules, although it may be found on other soil types (Cunningham *et al.* 1981). It is usually bushy with weeping or erect branches, and pale grey or brown, rough, fissured bark. Leaves are grey-green, alternate, lanceolate or linear-lanceolate with prominent veins (Cunningham *et al.* 1981).

Flowers are usually bisexual, with flowering reported from September to February (Duretto 1999). Flowers are small, inconspicuous and cream coloured. The fruit is composed of between one and four globular, single seeded lobes, the outer casing of which splits irregularly at maturity (Cunningham *et al.* 1981). The seeds are glossy black, with a red fleshy aril covering half the seed.

Beadle (1981) states that all attempts to germinate seed under laboratory conditions failed, however, in subsequent research Wisniewski and Parsons (1986) achieved high germinability (88–93 %) with scarified seed. Burbidge (1960) found that a high ambient temperature is required for optimal germination. Seed viability may decline rapidly and significant proportions of seed may be destroyed by insects contributing significantly to the low germination rates recorded in many years (Wisniewski and Parsons 1986). Spasmodic flowering and variable numbers of filled seed may also limit seed availability in some years (Wisniewski and Parsons 1986).

Despite abundant seed, *A. oleifolius* seedlings are rarely observed (Chesterfield and Parsons 1985; Wisniewski and Parsons 1986; Auld 1995b), with most regeneration appearing to rely on root suckering (Cunningham *et al.* 1981). Suckering is observed commonly following fire and land clearance (Chesterfield and Parsons 1985; Wisniewski and Parsons 1986) with occasional dense suckering forming low, monospecific stands (Beadle 1948). Some researchers have suggested that soil disturbance is the main factor encouraging suckering (Tiver and Andrew 1997).

A. oleifolius is browsed by stock, goats, rabbits and kangaroos and is often lopped to provide drought forage for stock (Cunningham *et al.* 1981). Mature trees in high grazing pressure situations commonly exhibit a browsing line up to 2 m high, giving them a characteristic umbrella shape. Regeneration by both seedlings and suckers is severely affected by stock grazing (Chesterfield and Parsons 1985; Auld 1995a). Tiver and Andrew (1997) also found kangaroo browsing was correlated with a lack of *A. oleifolius* regeneratión.

A. oleifolius appears moderately fire tolerant. Chesterfield and Parsons (1985) report a 43–45 % mortality following fire. Wisniewski and Parsons (1986) found only 19 % of mature trees were killed during a high intensity fire. Of those killed many were smaller trees with a DBH of less than 8 cm. In both cases, suckering was observed post fire along damaged roots, and at the base of affected trees (Chesterfield and Parsons 1985; Wisniewski and Parsons 1986).

1.2.1.3 Callitris gracilis

C. gracilis is a conical tree with a single straight trunk and rough furrowed bark to 20 m. It has aromatic dark green foliage, occasionally glaucous, with leaves consisting of many tiny scales covering branchlets. It occurs in open woodland on sandy ridges, in almost pure stands, or in association with *C. pauper* (Cunningham *et al.* 1981; Entwisle 1994).

North of the riverine region of NSW, *C. gracilis* is largely replaced by *C. glaucophylla* (Cunningham *et al.* 1981). Hybridisation may be found between *C. gracilis* and *C. glaucophylla* in the Murray River area (Cunningham *et al.* 1981; Entwisle 1994). As an important forestry species, and woody weed in higher rainfall areas of NSW, significantly more research has been performed on *C. glaucophylla* (Johnston 1968; Lacey 1972; Horne 1990; Johnston and Jennings 1991; Read 1995).

C. gracilis produces a woody globular cone, occurring either individually or in groups of two or more (Cunningham *et al.* 1981). Cones may be tuberculate, but not covering the entire cone scale as in *C. verrucosa.* Cones rarely remain on the tree long after maturity. Seeds are hard and compressed with two small wings (Cunningham *et al.* 1981).

C. gracilis numbers are reported to be substantially decreased since European settlement (Cunningham *et al.* 1981). The problems of extensive clearing and lack of regeneration of *C. gracilis* in Victoria were first recognised in the early 1930s (Zimmer 1942, 1944). During the extreme drought of 1943–1945, *C. gracilis* in northwest Victoria were reported to have been nearly wiped out through the combined effects of clearing, lack of regeneration and death of mature trees (Patton 1951).

A similar situation exists for *C. glaucophylla*, with these communities amongst the most threatened and vulnerable in western NSW due to rabbit and stock grazing limiting regeneration, widespread clearing, modification of understoreys, and weed invasion (Porteners *et al.* 1997).

Mature *Callitris* foliage is rarely eaten by stock, however, *C. gracilis* juveniles appear highly susceptible to grazing, with reports of much regeneration being eliminated by grazing pressure (Johnston 1968; Lacey 1972). *Callitris* seedlings appear particularly susceptible to rabbit grazing (Cochrane and McDonald 1966; Johnston 1968). Age classes of *C. glaucophylla* have been examined and establishment related to periods of decreased rabbit abundance, such as at the introduction of myxomatosis (Mitchell 1991).

Adults of *C. glaucophylla* appear relatively fire tolerant (Lacey 1973), although fire is reported as being fatal to most other *Callitris* species (Enright and Hill 1995). Fire results in high mortality rates for juvenile *C. glaucophylla*, with regular burning by aborigines believed to have played an important role in structuring open *Callitris* woodlands prior to European settlement (Wells 1974). Fire is thought to have a role in thinning dense regeneration following years of above average rainfall (Wilson and Mulham 1979).

Above average rainfall is a consistently reported requirement for the establishment of *C. gracilis* and *C. glaucophylla* (Stannard 1958; Lacey 1972; Cunningham *et al.* 1981; Mitchell 1991). Germination of *C. glaucophylla* is commonly triggered by a rainfall event in autumn, winter or spring, with summer temperatures inhibiting germination (Lacey 1972). However, the crucial factor for successful seedling establishment of *C. gracilis* and *C. glaucophylla* is survival through the first summer (Zimmer 1944; Lacey 1972). Additionally, Read (1995) suggests that *C. glaucophylla* regeneration in a semi-arid region of South Australia occurred only following at least two years of well above average rainfall. Young *C. gracilis* trees were observed at Walpeup in 1950, regenerating in and around depressions that occasionally held water (Sims 1951).

1.2.1.4 Myoporum platycarpum

M. platycarpum is a small tree to about 10 m with rough fissured bark. It may occur as an overstorey species within mixed *C. pauper* woodland or in pure stands. Branches are tuberculate and viscid, particularly on terminal branchlets. Leaves are alternate, linear-lanceolate to elliptic, usually with a few fine serrations on the upper leaf margins (Cunningham *et al.* 1981; Jeanes 1999).

Flowering has been reported from June through to December (Hall *et al.* 1964; Cunningham *et al.* 1981; Chesterfield and Parsons 1985; Jeanes 1999). Inflorescences consist of 4–11 flowers either white or tinged purplish-pink, spotted or unspotted (Jeanes 1999). The fruit is a dry, flattened ovate drupe. Large numbers of fruits are observed during most years with numerous fruits on the soil surface during seeding (Chesterfield and Parsons 1985). Despite abundant fruits, however, seed set is often minimal (Hall *et al.* 1964; Chesterfield and Parsons 1985).

Germination of *M. platycarpum* seed is reported to occur frequently, however only occasional seedlings were observed at Koonamore (Wood 1936; Hall *et al.* 1964). Chesterfield and Parsons (1985) found regeneration of *M. platycarpum* at least once each decade between the 1950s to 1970s, with regeneration often following heavy summer rainfall (Hall *et al.* 1964). *M. platycarpum* regeneration has been observed on both disturbed and undisturbed sites, with more regeneration noted in open sites, compared with densely treed sites (Chesterfield and Parsons 1985).

Foliage of mature trees is eaten by stock and often used as forage, however some juveniles have been observed to survive in areas of high stock grazing pressure (Chesterfield and Parsons 1985). Juvenile *M. platycarpum* appear less palatable than many other arid zone perennials, however, they appear highly susceptible to rabbit grazing (Sinclair 1984; Westbrooke 1998). Few *M. platycarpum* seedlings survived at Koonamore until rabbit numbers decreased in the 1950s (Hall *et al.* 1964), and similar effects of rabbit grazing on *M. platycarpum* regeneration were found in southwest NSW (Chesterfield and Parsons 1985). Whilst evidence suggests that *M. platycarpum* seedlings are relatively unpalatable, where grazing pressure is high, and particularly in the presence of high rabbit numbers, *M. platycarpum* seedlings are eliminated through browsing.

M. platycarpum trees appear to be relatively fire tolerant. Mature *M. platycarpum* have been observed to survive low intensity fires, and only few mortalities have been observed following moderate intensity fires (Lay 1976; Chesterfield and Parsons 1985). Juvenile plants are also relatively fire tolerant, with post-fire regrowth from rootstock (Cunningham *et al.* 1981; Chesterfield and Parsons 1985). However, no post-fire regrowth has been observed from mature trees following fire (Chesterfield and Parsons 1985). During drought, *M. platycarpum* may become totally defoliated, but recovers in following rains (Wood 1936; Hall *et al.* 1964).

1.3 VEGETATION CONDITION ASSESSMENT

Given the extent of landscape decline in condition across semi-arid woodlands in Australia, tools are required, not only for restoration, but also for monitoring. Prior to undertaking management activities, it is necessary to establish a methodology to determine baseline measures of the vegetation, and for monitoring vegetation change. This is essential to target management, and determine the effectiveness of management. Vegetation condition assessments have been used for this purpose in rangelands since the early 1900s, and are increasingly used across many ecosystem types today (Spencer *et al.* 1998; Parkes *et al.* 2003).

Whilst condition is a term frequently used in vegetation studies, rarely is it well defined. In many published papers of condition assessment, authors have neglected to define what they mean by condition, beyond the title and description of methodology (Kharuk *et al.* 1991; Fuls 1992; Belnap 1998; Ladson and White 1999). Vegetation condition has been used to describe biodiversity, tree dieback, plant moisture content, presence of pollutants, vegetation productivity and vegetation cover (Kharuk *et al.* 1991; Minor *et al.* 1999; Ritchie *et al.* 2001). The term "range condition" is often used to refer to forage availability (Wilson and Tupper 1982; Frost and Lamar Smith 1991; Jordaan *et al.* 1997).

Condition is also often used as a simile for concepts of ecosystem health, or productivity (Smith 1989). Additionally, there is much overlap with measures of ecosystem degradation or decline, recovery or restoration or simply vegetation change (Kelly and Harwell 1990; Aronson *et al.* 1993).

Smith (1989) provides a succinct summary to condition assessment for rangelands, which is applicable to other vegetation types and land uses:

> Condition "is not a characteristic ... which can be measured directly. Attributes such as plant cover or density, standing crop, soil texture, can be measured or estimated in the field. Trend in these parameters over time can be measured. Range condition, however, is an interpretation of these data in light of what is assumed to be possible or desirable. The selection or weighing of attributes chosen for measurement reflects the values and objectives of the person or agency making the evaluation".

Thus, the assessment of condition of an area describes the vegetation relative to defined goals or ideals. The vegetation is in good or poor condition for some purpose or species. It is possible that a vegetation community may be in good condition for one goal, but poor condition for another. For example, the vegetation may be in good condition for stock grazing, but poor condition for habitat of a threatened species. This has implications for design of condition assessments, which must relate to management goals and identified issues for a site.

Four main components necessary for designing a vegetation condition assessment have been identified (Pickup *et al.* 1994). Each of these components requires some subjective choices to be made, based upon the values and objectives of the person or agency performing the assessment. The components are:

(i) a theoretical model of vegetation dynamics in the region of interest;

(ii) a benchmark or reference system and defined long-term goals;

(iii) vegetation measures or indicators able to measure the current state of the vegetation, detect change and determine direction of change; and

(iv) sampling, measurement, and analytical procedures that can distinguish between short-term natural fluctuations and long-term change.

These components form the framework to the design and understanding of vegetation condition assessment, and each will be discussed in more detail.

1.3.1 Theoretical models of vegetation dynamics

The theoretical framework underlying vegetation condition assessment has implications for the measures chosen to indicate condition change, the use of benchmarks and predictions of vegetation decline or recovery. Therefore, the theoretical framework that best represents vegetation change in the locality must be taken into consideration (Walker 1988). An accurate model of vegetation dynamics enables the condition assessment to provide insight into likely past and future vegetation change (Friedel 1991).

Successional models

Clements' (1916) theory of vegetation succession provided the first widely accepted model of vegetation change. In this model, seral stages towards climax vegetation were recognised. The model is based on the occurrence of ecosystem equilibrium, initially observed in temperate regions.

The initial link between successional theory and rangeland assessment was made in 1917, however, it was not until the late 1930s that the term "condition" was introduced to describe vegetation successional stages in rangeland assessments (Humphrey 1949; Joyce 1993). Many different methods for rangeland condition assessment were developed, based on successional theory. By 1949, in the face of many competing methods for assessing range condition, Dyksterhuis (1949) called for a common methodology underlain by quantitative ecological methods. He proposed a technique based on percentages of plants classified into one of three condition classes, increasers, decreasers and invaders, compared with a climax community. In this method, community composition of these increasers, decreasers and invaders was the main variable measured.

Whilst Dyksterhuis's approach was widely accepted, there has been increasing criticism of many aspects. The successional model has been largely rejected within arid and semi-arid ecosystems, with non-equilibrium systems being accepted as a better model of vegetation dynamics within an environment structured by episodic events (Westoby *et al.* 1989; George *et al.* 1992; Milton *et al.* 1994). However, many methods for condition assessment are still based on successional theory, with continued acceptance of this model by some researchers (Pieper and Beck 1990; Jordaan *et al.* 1997).

Limitations of successional models

It has been suggested that critics of the successional model take an extreme and inflexible interpretation, and that recent alternatives only present new interpretations of the original model (McIntosh 1980; Smith 1989). However, a number of intractable problems with successional theory for vegetation condition assessment have been identified.

Successional models predict a linear pattern of return to climax vegetation following removal of disturbance, such as grazing. However, recovery following cessation of grazing is frequently not observed in arid environments where episodic events such as rainfall may have a greater impact on vegetation change than disturbances such as grazing (Laycock 1991; Oba *et al.* 2000). The assumption of vegetation change as predictable, linear and reversible is not always observed (Laycock 1991).

Dominance by annual species, an indication of an early seral stage, is often viewed as a sign of degradation under successional models. However, annual plants provide an important forage source, and the proportion of annual species to perennials in arid environments varies seasonally with rainfall (Friedel 1991; Oba *et al.* 2000). Climax vegetation is often not the optimal state to provide maximum forage, particularly relevant in areas where the climax vegetation is not grassland (Friedel 1991).

Major natural disturbance events such as fire may have an apparently catastrophic effect on native vegetation condition. Following a fire, low vegetation cover, species richness and abundance, and dominance by early successional species occurs. However, in time, many of these measures may show higher values than a neighbouring long unburnt site. In addition, by attributing low condition values to sites disturbed by fire, the value of maintaining a mosaic of patches in different condition states across the landscape is ignored (Frost and Lamar Smith 1991; McIntyre and Barrett 1992).

Non-equilibrium models

Non-equilibrium models have been proposed as an alternative to successional models, suggesting that ecosystems are not commonly in a state of equilibrium. They describe thresholds of change and sudden, potentially irreversible, discontinuous change related to episodic events (Westoby *et al.* 1989; Friedel 1991). Vegetation change in non-equilibrium models is not assumed to be linear or necessarily reversible and multiple stable states may occur (Laycock 1991).

The state and transition model has gained increasing support (Yates and Hobbs 1997b). State and transition models are based on the identification of a number of stable states, separated by thresholds. These thresholds represent a spatial and temporal boundary between vegetation states (eg. between grassland and woodland states) (Westoby *et al.* 1989; Laycock 1991; Yates and Hobbs 1997b). States may be relatively stable and removal of disturbances or rehabilitation attempts may not lead to alternative stable states returning to the original climax vegetation (Griffin and Friedel 1985; Rapport and Whitford 1999). Positive change between stable states is unlikely to occur within the lifetime of most management plans without significant management or restoration input. However, there have been some criticisms of this approach, for example, Oba *et al.* (2000) state that the role of herbivory in structuring ecosystems is overlooked in many non-equilibrium models including state and transition models.

In arid ecosystems, reliance on co-occurrence of multiple rare events, such as high rainfall, with sufficient follow-up rain dictates the few regeneration events and hence enables only slow recovery (Noble 1984b). Temporal change in the order of hundreds of years is difficult for even the most long-term research to encompass and provides a conceptual problem for theories of vegetation dynamics. Whilst successional theory recognises even very slow vegetation change as leading towards the climax community, a state and transition model places more emphasis on vegetation change within a human scale. Long-term ecological research is required to determine if very slow recovery may occur on highly degraded sites that may appear as stable states.

In summary, to reliably predict all aspects of vegetation change, models must be specific to the region and vegetation community. It is apparent that models for vegetation dynamics in temperate regions do not necessarily apply to arid regions, and models that are appropriate where the vegetation is adapted to grazing are not necessarily appropriate in a system where grazing has not historically been an element of the disturbance regime (Oba *et al.* 2000). A consensus has not been reached on the vegetation models best suited to different environments, but in many semi-arid and arid regions, non-equilibrium models appear to assist in predicting and explaining vegetation dynamics.

A model of vegetation dynamics for northwest Victoria

A model of vegetation dynamics that reflects the nature of vegetation change in northwest Victoria should be chosen prior to determining a method for vegetation condition assessment. Insufficient evidence exists to date to fully determine processes of vegetation change, however, within semi-arid areas of Australia, a state and transition model seems likely to provide an explanatory model of vegetation dynamics.

Implications for vegetation change represented by a state and transition model include the potential for an alternative stable state despite change in management, such as removal of or reduction in grazing. If stable alternative states are identified, significant management action or disturbance may be required to affect vegetation change, and, in the absence of this, vegetation change may be very slow. Therefore, a condition assessment should be able to recognise areas of persistent alternative states.

1.3.2 Long-term goals and benchmarks

Determination of vegetation condition is commonly reliant on a benchmark or reference site against which to compare a site and define long-term goals. However, there has been much debate over how to determine reference systems (Cairns 1990; Howe 1994; Pickett and Parker 1994; Aronson *et al.* 1995; Hobbs and Norton 1996). Choice of reference system is also related to the underlying theoretical framework for the condition assessment. Using a benchmark of climax vegetation is appropriate under a successional framework, but may be less relevant in a state and transition model.

Three types of reference system have been identified to benchmark vegetation condition, (i) historical systems, (ii) desired systems, and (iii) potential and threshold systems (Walker and Reuter 1996). Selecting a reference based on historical systems is not possible for many Australian systems due to the fragmented and degraded state of much remnant vegetation. Moreover, determining what defines an historical system in Australia is a controversial issue. Australian vegetation has been shaped by anthropogenic disturbance for over 40,000 years. Should an ecosystem of reference be based on a pre-European or pre-human system? Regardless of the answer, historical re-creation is unlikely to be a realistic goal in many Australian systems due to the extent of landscape modification and the ongoing disturbance from surrounding agricultural processes (Werner 1990; Hobbs and Mooney 1993).

Desired or notional reference systems relate to management goals for a specific site. Management goals, such as maximising biodiversity, forage productivity, or habitat value can be used to determine a theoretical desired system. Examples of the desired system may or may not exist in the natural environment. Managing for a desired system involves working towards a value or a range of values that are selected from management goals.

Potential levels and thresholds are defined through research, and also relate to management goals (Walker and Reuter 1996). For example, the threshold of minimum habitat area for a threatened species may be used. The difficulty with this type of indicator is obtaining the underlying knowledge, with insufficient data being available to set threshold levels for habitat for many species.

If a reference system involves an on-ground site, there are a number of potential problems. Natural disturbance such as fire may affect the validity of the reference site, however, exclusion of disturbances may equally affect the site. Regardless of management, ecosystems will continue to change over time leading to a change in both the reference and the compared site (Landres 1990).

Benchmarks and goals for management of the Murray-Sunset National Park

To determine parameters for vegetation condition assessment in the semi-arid woodlands of northwest Victoria, management goals must be considered. The vast majority of semi-arid woodlands remnants in northwest Victoria now occur within National Parks. Under Section 17 of the National Parks Act and the Parks Regulations 1992, each Park must be managed to:

- preserve and protect the park in its natural condition;
- maintain or where possible enhance wilderness values;
- allow natural environmental processes to continue with the minimum of disturbance; and
- maintain biodiversity.

Goals for management of parks and reserves in northwest Victoria are outlined in the Mallee Parks Management Plan (NRE 1996). Management directions state that as far as practicable, areas of the Parks that have been degraded through past land management activities will be rehabilitated by:

- reducing the impact of pest animals and plants on native species and communities;
- addressing the current imbalance of kangaroo populations;
- restoring a more natural hydrological regime within all parks;
- active revegetation within areas of localised extinction and rarity; and
- adopting manipulative fire regimes where they can be demonstrated to be of value to the Parks environments and ecosystems.

In addition, specific threats and goals for management of semi-arid woodlands have been identified including (NRE 1996):

- lack of regeneration of the overstorey and tree senescence;
- paucity of woody perennial cover and species richness in the understorey; and
- scalding, erosion and dune mobilisation.

The imbalance of kangaroo populations and other grazing animals has been identified as an important issue within parks and reserves in the Mallee region (LCC 1987; NRE 1996). It is important to note, however, that condition assessment does not directly identify the causes for change in condition. Attributing causes for change in condition is difficult without assessment of disturbances and threats.

To assess vegetation condition a better understanding of pre-European vegetation condition would assist in determining appropriate benchmarks. Insufficient evidence currently exists to determine benchmarks from historical data, and there has been little exploration of historical vegetation data in the study area. The extent of disturbances to vegetation in the region, particularly to semi-arid woodlands has resulted in very little vegetation remaining in an undisturbed state. This makes determining a benchmark based on existing remnant vegetation a difficult process. Desired or notional systems, informed by available historical evidence may be a better choice in regions such as this, where little native vegetation remains undisturbed.

1.3.3 Condition indicators

An indicator is defined as "any expression of the environment that estimates the condition of ecological resources, magnitude of stress, exposure of a biological component to stress, or the amount of change in a condition" (Breckenridge *et al.* 1995). Thus indicators are shorthand descriptors of aspects of the environment (Walker and Reuter 1996). Indicators may be based on field data, remotely sensed data, or compilation of existing data (Walker and Reuter 1996).

Condition assessment involves selection of a range of indicators relevant to the goals for vegetation management at the site (Smith 1989). The choice of indicator will depend on economics, assessor expertise, logistics, repeatability and goals of the assessment (Cairns 1986, 1990). In addition, indicators must be sufficiently scientifically sound, easily replicable, reliable and sensitive to change (Kelly and Harwell 1990; ANZECC 2000). Measures should be site and goal specific, but also need to provide data for comparison across sites. As no single indicator is likely to have all these properties, a range of indicators is usually required (Noss 1990).

Numerous types of indicators have been proposed. Walker and Reuter (1996) describe two types of indicators: (i) condition indicators assess the state of the system relative to a reference system; and (ii) trend indicators assess how the system has changed. Bertollo (1998) describes two alternative categories of indicator: (i) ecological indicators describe the elements of an ecosystem such as flora, fauna, water, air and soil; and (ii) environmental indicators present the broader picture illustrating both the current condition of the ecosystem, as well as current stresses on the system.

The use of keystone species and indicator species has also been proposed as a method for assessment of condition (Majer and Nichols 1998; Krogh *et al.* 2002). Majer (1983) proposes the use of ants as indicator species for assessment of condition. Invertebrates have been linked to ecosystem function, structure and composition and ant re-colonisation has been positively correlated with plant species richness and diversity (Majer 1983).

A number of studies have used measures of actual or potential threats as indicators of vegetation condition, for example, presence of grazing animals, scat counts, or land management (eg. stocking levels) (Belnap 1998). Measures of threats may give some indication of the causes for change in condition, or of the potential for further change in condition, however, should not be confused with measurement of vegetation condition.

1.3.4 Techniques to distinguish between short-term fluctuations and long-term change

It is important that temporal and spatial scales are recognised in developing vegetation indicators and vegetation condition assessments (Kelly and Harwell 1990; Parker 1997). Determining appropriate sampling, measurement and analytical techniques to distinguish between short-term fluctuations and long-term change is particularly relevant in arid areas where high temporal variability make detection of long-term change difficult. For example, unless temporal fluctuations in rainfall quantity are considered, the condition assessment may result in a measure of how much rainfall the site has received.

Within arid areas, spatial distribution of vegetation is variable and non-randomly distributed which also limits choice of vegetation assessment techniques and requires care with appropriate sampling methodology (Verstraete and Pinty 1991). Appropriate sampling must consider the landscape scale to ensure that the assessment is representative (ANZECC 2000).

To effectively detect long-term change, variability within indicators must also be considered (Landres 1990; Bastin *et al.* 1993a). For most indicators, the degree of natural variation is unknown, and variation may differ between ecosystem types. Long-term research is required to fully understand variation within indicators, and how they relate to other factors within the ecosystem (Landres 1990).

The way data are presented and analysed may also impact on the ability to separate short-term fluctuations from long-term change (Cairns *et al.* 1993). Indicator values may be combined into a single condition score, or viewed individually. Combining values into a single score allows for simple comparison across sites, however, it may also lead to loss of pertinent information. Measures that allow comparison of individual components of the condition assessment are therefore more sensitive in assessing vegetation change (Walker and Reuter 1996).

A number of techniques have been employed in arid vegetation assessment to limit variability due to short-term fluctuations and better distinguish long-term change (Friedel 1990). One method used is to assess at the same time of year to limit seasonal variation. The main limitation of this approach in arid environments is the low seasonality of rainfall. To address this problem, the response of vegetation to a rainfall event of similar proportions has been measured (eg. Pickup *et al.* 1998). This reduces effects of seasonality, but also limits the timing of assessment, for example, vegetation condition cannot be assessed during drought conditions. Another option is to assess a parameter that is less seasonally variable, such as cover of woody perennials (McCloy and Hall 1991). Again, this limits the scope of vegetation condition assessment, providing limited information on the annual component of the vegetation.

1.3.5 Field survey techniques for vegetation condition assessment

The traditional method of vegetation condition assessment is to visit the study site and perform a series of objective measures of the vegetation status. Quantitative ecological methods for vegetation assessment are detailed in many texts (eg. Greig-Smith 1983; Brower *et al.* 1990). Common measures of vegetation and components of the vegetation that they may be applied to are listed in Table 1.1. Measurement of any of these elements of the vegetation can provide a baseline measure or a picture of the vegetation at a point in time, however, to determine condition, analysis of the data with comparison to a benchmark is required.

Several limitations of ground-based assessment techniques have been noted in the literature. The major limitation of field survey is cost. It is both time consuming and expensive to perform large-scale field work (Gullan 1991; Milham *et al.* 1996; Mumby *et al.* 1999; Ritchie *et al.* 2001). Costs include travel, accommodation and wages for field staff, data entry, analysis and reporting. High costs limit sample size, with most studies sampling only a small percentage of the total area (Milham *et al.* 1996; Bird *et al.* 2000).

Table 1.1. Potential direct measures of vegetation condition.

Measures of vegetation	Vegetation components
Cover	Overstorey
Species abundance	Shrub layer
Species richness	Understorey
Diversity	Ground layer
Strata intactness	Vines / lianes
Habitat value	Epiphytes
Regeneration	Moss / lichens
Condition / dieback	Native species
Age class structure	Exotic species

Example of a field condition assessment

Victoria's native vegetation framework assessment, based around the "habitat hectare" is an interesting example of a field based condition assessment currently in use. The primary goal identified for the management of vegetation in Victoria is "a reversal, across the entire landscape, of the long-term decline in the extent and quality of native vegetation, leading to a Net Gain"(DNRE 2002). The Net Gain principal aims to ensure overall gains in native vegetation outweigh any losses. This is achieved through an assessment of vegetation quality and quantity, requiring relative gains in reserved native vegetation as offsets for permitted clearing or other negative impacts (DNRE 2002).

The habitat hectare refers to the habitat value and area of the site. Two main determinants of the vegetation quality of a site are outlined in the methodology: (i) inherent site condition involves a comparison of the condition of a site with a notional optimum (benchmark); and (ii) viability in the landscape context examines how the patch is situated in the broader landscape (Parkes *et al.* 2003). Variables used to estimate inherent site condition include presence of large old trees (for woodlands and forests), intact tree canopy, cover and diversity of understorey, presence of recruits, weediness and presence of litter and logs. These measures are compared to a benchmark of a mature, long-undisturbed site of the same vegetation type (Parkes *et al.* 2003).

There are a number of limitations with the habitat hectare assessment, and many of these relate to the choices or value judgements made by the Department of Sustainability and Environment (DSE). This highlights the nature of condition assessment, which is a value judgement, based upon values and objectives of the agency (DSE) making the decisions. Some of these limitations or value judgements include:

- Sites following fire or other disturbance may be downgraded on many measures, when compared to the notional optimum, although evidence of episodic events required for regeneration is noted. Therefore the potential value of patchiness of condition or disturbance across the landscape is not considered.

- Measures were chosen to enable assessment by a range of natural resource managers who may not have detailed botanical skills. This may limit the ability to detect rare or threatened flora and communities, or potential weed problems.

Despite these limitations, the habitat hectare assessment provides a method of quantifying the vegetation condition of a site with an emphasis on habitat values.

1.4 REMOTELY SENSED DATA FOR VEGETATION CONDITION ASSESSMENT

Ground survey has long been the traditional method for vegetation condition assessment. However, since the first aerial photographs were used for vegetation mapping in 1913, remotely sensed data have provided an alternative method for vegetation assessment (Wolf and Dewitt 2000). With the rapid advance of remote sensing technologies, satellite imagery and aerial photography have increasingly been used for vegetation condition assessment.

Whilst it is not always possible to monitor the same parameters as used in ground-based surveys, various indicators of vegetation condition have been used. Remote methods for vegetation condition assessment have not been widely adopted, and so literature for this review has included studies on vegetation indices, classification, ecosystem health and productivity. This is in keeping with the definition of vegetation condition as interpretation of vegetation parameters in light of what is assumed desirable.

Remote sensing is the process of obtaining information about an object using a sensor which is physically separated from the object (Harrison and Jupp 1989). Remote sensors commonly detect electromagnetic radiation from the sun, which has been reflected off the surface of the object, however, they may also detect other energy sources such as sound waves, magnetic energy, or gravity (Harrison and Jupp 1989). Energy sources may be either passive, such as reflected light from the sun, or active such as microwave or laser. Remote sensors respond to the electromagnetic radiation or other energy source by producing a voltage signal. A wide range of remote sensors are available, based on both aerial and space borne platforms. These sensors include cameras, multispectral sensors, hyperspectral sensors and radar systems (Harrison and Jupp 1989).

The earliest types of remote sensors were simple cameras. The first aerial photographs were taken in 1885 from a hot-air balloon in France (Mikhail and Bethel 2001). In 1909, the first photos were taken from an aircraft and in 1913 aerial photographs began to be used for mapping purposes (Wolf and Dewitt 2000). Aerial photography was used extensively for topographic mapping between the first and second World Wars, and rapid advancements since have led aerial photography to become an indispensable tool for vegetation mapping and monitoring (Wolf and Dewitt 2000).

Following the launch of the first satellite, Sputnik, in 1957 observing the earth from space became a possibility. Thus, satellite imagery was conceived as an indirect result of the 1960s space race. The first photographs of the earth taken from space were taken from the satellites Mercury, Gemini and Apollo (Curran 1985). TIROS 1 was the first satellite designed for monitoring the earth surface, and was launched in 1960. The next stage in monitoring was to devise a multispectral sensor to enable a more automated approach to image classification. Landsat 1 was the first satellite to carry a multispectral scanner (MSS) and was launched in 1972 (Mikhail and Bethel 2001). Many different sensors are now available and may be mounted on aircraft, or satellite platforms.

The framework for vegetation condition assessment by remote methods remains unchanged from that outlined above. The main difference compared with ground-based methods is in the choice of indicators and methods for data analysis. Many different types of remotely sensed data are currently available. Depending upon the goals of assessment, different types of imagery will be better suited to the condition assessment. Some of the types of data that are available will be briefly outlined in the following sections.

1.4.1 Passive sensing systems

Aerial photography produced from vertical black and white, or colour photographs has been used for many years in map production and monitoring of landscape features (Milham *et al.* 1996; Fensham *et al.* 1998; Lobo *et al.* 1998; Rowe *et al.* 1999; Fensham and Fairfax 2002). The high resolution of aerial photography and the ability to produce stereo-pairs has enabled measurements of many vegetation parameters. Aerial photography is one of the most commonly used data sources for vegetation mapping with small aircraft, radio-controlled model aeroplanes (Thome and Thome 2000), blimps (Murden and Risenhoover 2000) and helicopters having been used as platforms for cameras, and other sensors (Tueller *et al.* 1988; Ritchie *et al.* 2001). Many different types of sensors can be carried on an aircraft platform. These include standard metric cameras, small-format cameras, video cameras, and hyperspectral sensors such as Compact Airborne Spectrographic Imager (CASI).

Small-format digital aerial photography (SFAP) is a low-cost alternative to standard aerial photography. The format size of the camera refers to the dimensions of the film or charge-coupled device (CCD) (Rees 1999). A metric aerial survey camera uses a 230 mm square format (9 inch), whilst small-format aerial photography utilises cameras with smaller formats, with the main advantages being cost and flexibility (Rowe *et al.* 1999; Abd-Elrahman *et al.* 2001).

Small-format cameras are considerably cheaper to purchase than metric cameras, and are also cheaper to mount within single engine light aircraft which are commonly employed for this type of imagery (Rowe *et al.* 1999). SFAP also allows for flexible image acquisition at whatever scale is required. The date of imagery acquisition can be chosen, as well as weather conditions (Rowe *et al.* 1999). Colour and colour infrared are also easily obtainable with SFAP (Rowe *et al.* 1999).

The main limitations of SFAP occur through the use of standard cameras without lens calibration, or film flattening devices leading to potential for image distortion (Rowe *et al.* 1999). In addition, positional accuracy may be relatively low compared with the relatively high resolution of these images (Abd-Elrahman *et al.* 2001). Despite these limitations, SFAP has been used for a number of vegetation mapping projects, and has been adopted in the forestry industry (Warner 1994; McCormick 1999; Rowe *et al.* 1999; Abd-Elrahman *et al.* 2001).

Airborne video is an alternative cost-effective method for recording and storing remotely sensed data. Advantages of airborne video are ease of use and immediate availability of data (Hutchinson and Schoengerdt 1990). To maximise use of video cameras for remote sensing applications, increased spectral capabilities are required and various systems have been devised to enable video cameras to record images in multiple bandwidths (Hutchinson and Schoengerdt 1990). Airborne video can be viewed on screen or converted into a series of digital images (Tickle *et al.* 1998).

Compact Airborne Spectrographic Imager (CASI) is a hyperspectral imaging sensor or imaging spectrometer developed for use with an aircraft platform (Lewis 2000). CASI sensors provide many more, narrow spectral bands than typical broadband multi-spectral sensors (Lewis 2000). The hyperspectral capabilities of CASI make these data ideal for separating different vegetation communities or land cover types at a small-scale. This capability has been used to accurately determine vegetation composition in sparsely vegetated arid ecosystems (Lewis 2000).

Multispectral Scanner (MSS), Thematic Mapper (TM) and Enhanced Thematic Mapper (ETM) sensors have been used with the Landsat satellite platforms, with the first Landsat satellite launched in 1972 (Mikhail and Bethel 2001). Since then, there have been a further six Landsat satellites launched. Spatial and spectral resolutions have improved from the initial MSS, to the TM, and the recently launched ETM sensor. Spatial, spectral and temporal resolution details are presented in Table 1.2.

Table 1.2. Spatial, spectral and temporal resolution, and image size of Landsat Missions (Williams 2003).

Landsat Missions	Spatial resolution (pixel size m)	Spectral range µm	Number of bands	Temporal resolution (days)	Image size km
MSS	79	0.5–1.1	4–5	16–18	185 x 185
TM	30 120 (thermal)	0.45–12.5	7	16	185 x 172
ETM	15 (panchromatic) 30 (colour) 60 (thermal)	0.45–12.5	8	16	183 x 170

Some limitations of Landsat data have been observed. Landsat MSS has limited use in local land management, only being able to distinguish broad vegetation types, and usually unable to distinguish the composition of different communities (Catt *et al.* 1987; Hobbs *et al.* 1989). Landsat wavebands are often highly correlated in arid rangelands limiting the ability to separate features (Graetz and Gentle 1982; Pech *et al.* 1986).

SPOT (Systeme Probatoire d'Observation de la Terre). SPOT 1 was launched in February 1986. Since then, a further four SPOT satellites have been launched (Table 1.3). The enhanced spatial resolution of SPOT data, when compared with Landsat data, enables mapping of finer spatial grain in vegetation communities. However, the spatial resolution of SPOT data is limited by a reduced spectral resolution, which has been found to diminish the discrimination of different vegetation types (Franklin *et al.* 1993; Harvey and Hill 2001).

Table 1.3. Spatial and spectral resolution, and image size of SPOT Missions (Spot Image 2000).

SPOT Missions	Spatial resolution (pixel size m)	Spectral range (µm)	No. bands	Image size (km)
1–4	20 5	0.5–1.75	4 multi-spectral 1 panchromatic	60 x 60–80
5	5–10 2.5	0.5–1.75	4 multi-spectral 1 panchromatic	60 x 60–80

Advanced Very High Resolution Radiometer (AVHRR) is a mechanically scanned imaging radiometer carried on NOAA and TIROS-N satellites (Rees 1999). The broad-scale of this sensor leads to efficient measurement at the landscape scale. AVHRR data have been found useful in broad-scale studies such as drought detection, determining vegetation biomass and crop productivity (Liu and Kogan 1996; Minor *et al.* 1999).

1.4.2 Active sensing systems

LIDAR (light detection and ranging) sensors measure distance or elevation using Laser (Light Amplification by Stimulated Emissions of Radiation), in the infrared spectrum (Mikhail and

Bethel 2001). LIDAR is commonly used to produce digital elevation models (DEMs), however, LIDAR has also been used to measure vegetation height, canopy cover and distribution, and surface roughness (Ritchie *et al.* 1992; Mikhail and Bethel 2001). Analysis of LIDAR data using fractals has been shown to distinguish between vegetation types by determining unique patterns of vegetation clumping within different semi-arid vegetation types in western United States rangelands (Ritchie *et al.* 2001).

RADAR (radio detection and ranging) sensors detect microwave electromagnetic radiation. Pulses of microwave energy are transmitted and then scatter as they hit the ground, with only a small proportion returning to be recorded by the radar receiver (Mikhail and Bethel 2001). The return time of the signal and strength of the signal are recorded, with the pixel value determined by the strength of the signal, and an estimate of elevation measured by the return time. Radar sensors may be used on either satellite or aircraft platform. Synthetic aperture radar (SAR) provides a two-dimensional data source that can be used to generate images (Mikhail and Bethel 2001). One of the main advantages of radar imagery is that it is unaffected by weather, eliminating the need for cloud free imagery (Smith *et al.* 1995).

1.4.3 Combined data sets

Remote data sets may be combined to achieve advantages of higher spatial or spectral resolution. The increased spatial resolution of SPOT imagery is often used in conjunction with Landsat imagery to produce an image with greater spatial resolution than the original Landsat image (Gibson and Power 2000). However, Green *et al.* (1998) found that merged SPOT and Landsat TM imagery actually led to a decrease in spectral resolution and hence reduced accuracy of the final output.

Smith *et al.* (1995) investigated the use of radar and visible infrared sensors in assessing rangeland condition. Radar imagery used together with SPOT or Landsat imagery was shown to give a more accurate vegetation classification than either data set used individually (Smith *et al.* 1995).

The main limitation with combined data sets is the potential doubling in cost resulting from purchasing two or more image types, and despite the increase in cost there is no guarantee of improved accuracy in the final map product (Gibson and Power 2000).

1.4.4 Comparison of remote data sources

Choice of remote data source will depend on the requirements for spectral and spatial resolution, cost and data availability for the assessment (Pech *et al.* 1986; Patil and Myers 1999). For example in a small forestry plot, precise data regarding tree height and canopy dimensions may be required, however, particularly if only one species is involved, only minimal spectral detail may be necessary. Few studies have investigated more than one remote data source for the same assessment, so direct comparison of data sources is more easily made on the grounds of resolution as shown from the sensor specifications. Resolutions of current satellites sensors commonly used for environmental monitoring are listed in Table 1.4. Relative costs of medium and high-resolution data are presented Table 1.5.

Table 1.4. Satellites commonly used for environmental monitoring grouped according to spatial resolution (ERSC 2000).

Broad scale	Medium resolution	High resolution	Radar
ERS-1 & 2 (multi-spectral)	CBERS-1	EROS-A1	ESR-1 & 2
NOAH-14, 15 & L	EO-1	IKONOS	JERS-1
Orb View-2	IRS-1B, C, D & P4		RADARSAT
RESURS-01-3	Landsat 5 & 7		
Terra	SPOT-2 & 4		

Table 1.5. Indication of relative costs of medium and high resolution data (Geoimage 2003; Geoscience Australia 2003a; Raytheon 2004).

Imagery	Scene footprint	Area of Scene km^2	Cost per scene	Cost km^2	Approx cost for 6,300 km^2
MEDIUM RESOLUTION					
Landsat ETM	185 x 185	22,052	$ 1,500	$ 0.07	$ 431
SPOT 5	60 x 60				
10 m colour		3,600	$ 4,700	$ 1.31	$ 8,264
5 m B&W		3,600	$ 5,200	$ 1.44	$ 9,143
2.5 m B&W		3,600	$ 8,600	$ 2.39	$ 15,122
HIGH RESOLUTION					
IKONOS					
1 m B & W		121	-	$ 14	$ 88,620
4 m multispectral	11 x 11	121	-	$ 14	$ 88,620
1 m colour		121	-	$ 15	$ 97,482

For detailed mapping or assessment, airborne, rather than satellite remote-sensing imagery is more likely to give an accurate result (Graetz 1987; Mumby *et al.* 1999; Hyyppa *et al.* 2000). A comparison was made of satellite imagery (SPOT, Landsat, ERS-1/2SAR PRI and SLC), ground-based methods, aerial photography and radar imagery for the assessment of forest attributes (stem volume, basal area and mean height) in dense boreal forest in southern Finland. The accuracy of profiling radar was found to be equivalent to ground-based survey for these measures. Hyperspectral aerial imagery and aerial photography returned similar results and were both significantly better than satellite imagery (Hyyppa *et al.* 2000).

For large areas, such as the MSNP (6,330 km^2), medium-scale data such as Landsat, or SPOT may be applicable. For an area the size of the MSNP the cost of some smaller scale imagery types will exclude complete coverage of the study area. However, interpolation procedures may enable production of a map of the study area based on a sampling strategy.

1.4.5 Image enhancements and analyses

Once the type of imagery has been chosen, there are many options for data analysis to optimise identification of the relevant features from the data. The choice of analyses defines the condition indicator, and so the same factors of economics, ease of use, repeatability and sensitivity to change must be considered (Cairns 1986, 1990; Kelly and Harwell 1990; ANZECC 2000). Options for analysis of multi-spectral data include contrast enhancements, transformations, classification, and vegetation indices. Visual interpretation can also be performed, particularly with aerial photography and other high-resolution data.

Image enhancement

To provide the best display of the imagery, image enhancements are often required. Whilst an 8-bit sensor can record 256 levels of brightness, each individual band of an image commonly uses only a small portion of this. Stretching the data over the full available 256 levels enhances the contrast, enabling features to be more easily identified. Different stretching options include contrast stretching, linear stretches, and histogram equalisation stretches (Gibson and Power 2000).

Filtering procedures can also be applied to enhance the imagery. Filtering involves mathematical calculations on a matrix of numbers (Gibson and Power 2000). The matrix is passed over the image, resulting in the assignment of calculated pixel values to a new image. In the new image, pixel values are based on the original pixel value and those of its immediate neighbours. Filters are commonly applied to remove noise in the data, to sharpen or enhance edges (Gibson and Power 2000).

Vegetation indices

Vegetation indices are some of the most commonly used remote indicators of vegetation condition (Warren and Hutchison 1984; Huete and Jackson 1987; Graetz *et al.* 1988). Vegetation indices are intended to normalise for the effects of soil background colour, shadowing and illumination to produce an index of vegetation (Huete and Jackson 1987; Franklin *et al.* 1993). Many types of vegetation indices have been developed for use in different vegetation communities and with different types of imagery. Vegetation indices have been developed to closely correlate with green vegetation cover, green biomass, rangeland productivity, crop production and leaf area index (LAI) (Justice *et al.* 1985; Kogan 1990; Liu and Kogan 1996; Coops *et al.* 1997; Schmidt and Karnieli 2002).

Two types of spectral vegetation indices commonly used in arid areas are brightness and greenness indices (Duncan *et al.* 1993). Brightness indices are based on the finding that vegetation cover in arid and semi-arid zones results in a darkening of the naturally bright reflectance of soils in satellite imagery (Graetz and Gentle 1982; Duncan *et al.* 1993). Greenness vegetation indices are calculated by a linear combination or ratio of a near infrared waveband and a red (chlorophyll absorption) waveband (Eastman 2001).

Duncan *et al.* (1993) found that brightness in the red band of SPOT imagery was significantly correlated with vegetation cover in arid rangelands in New Mexico. Vegetation thinning was able to be detected using a brightness index in western New South Wales (Collerton 1999).

Wallace and Furby (1994) were able to distinguish good and poor condition areas of Wandoo and mallee woodland and heath land using a brightness index based on Landsat TM and MSS data. However, the brightness index was unable to distinguish between good and poor condition areas of *Eucalyptus loxophleba* ssp. *loxophleba* (York Gum) and *Acacia acuminata* (Jam), dominated woodland (Wallace and Furby 1994).

Greenness vegetation indices include the ratio vegetation index (RVI), transformed vegetation index (TVI), perpendicular vegetation index (PVI), weighted difference vegetation index (WDI), the soil-adjusted vegetation index (SAVI) and the normalised differential vegetation index (NDVI) (Eastman 2001). The NDVI was derived to spectrally separate green vegetation from background soil brightness (Rouse *et al.* 1974). This is the most commonly used vegetation index as it minimises topographic effects and produces a linear measurement scale (Kogan 1990). It is calculated as the difference between the near infrared and red bands normalized by the sum of those bands (Gibson and Power 2000):

$$\mathrm{NDVI} = \frac{\text{near infrared} - \text{visible red}}{\text{visible red} + \text{near infrared}}$$

Negative and low values of the NDVI are associated with cloud, bare soil, low green vegetation density and senescent vegetation (Belward 1991). Higher values relate to photosynthetic active cover such that the higher the value, the higher the green biomass (Sellers 1985). The NDVI has been used for estimating drought impacts (Liu and Kogan 1996), vegetation classification, monitoring rainfall and crop yields, net primary production, leaf area index and above ground total dry-matter accumulation (Tucker *et al.* 1983; Kogan 1990). The NDVI has also been found to change with anthropogenic impacts on the landscape such as irrigation, fertilisation, changes in land management practices and soil disturbance (Minor *et al.* 1999).

A limitation of the NDVI is that it tends to produce a saturation response to high green leaf densities (Eastman 2001). The NDVI, like most green vegetation indices is dependent on soil background, and at low vegetation densities, this can affect the accuracy of the index (Huete *et al.* 1984). Dead vegetation has also been found to reduce the NDVI and RVI values (Sellers 1985).

Limitations with vegetation indices for condition assessment

Vegetation indices have the potential to be used as a surrogate for vegetation condition assessment, and have been used in this capability (Wallace and Furby 1994). Vegetation cover or productivity is an important component of vegetation condition, and decline in condition is often correlated with low native vegetation cover and low productivity. However, a number of potential problems with its use as a surrogate for condition have been noted (Foran 1987; McCloy and Hall 1991):

(i) changes in species composition may not occur together with changes in percentage cover. However, it has been found that changes in species composition occurring through degradation of semi-arid areas commonly leads to a general decrease in vegetation cover (Bastin *et al.* 1993a);

(ii) high seasonal variations in annual cover are common, annual cover may increase following rainfall, without necessarily indicating a long-term change in condition; and,

(iii) the extent of annual cover may also be partially or totally covered by the perennial layer, making changes in annual cover difficult to detect via remote data sources.

Classification

Classification of images results in a clustering of pixels that represent a single feature type on the image. This enables simplification of the image so that features can be more easily viewed and analysed (Gibson and Power 2000).

Supervised classification involves the user identifying known areas of land cover classes on the image, and developing a signature that defines the spectral characteristics of each land cover class. A search for pixels matching the spectral characteristics is then undertaken based on a mathematical classifier (Gibson and Power 2000).

Unsupervised classification groups the image pixels into clusters based on an iterative process where the pixels are initially randomly assigned to a defined number of groups. The process is repeated by reallocating pixels to the group mean to which they are closest. Re-allocation to groups continues until the movement of pixels between groups reaches a set threshold value (Gibson and Power 2000).

Classifications are commonly used to distinguish land cover types, for example, agricultural land, urban land, and rivers. This can be used to produce a thematic map of land cover types which can then be used to analyse change in land cover over time (Diouf and Lambin 2001; Lobo *et al.* 2002).

1.4.6 Other potential condition indicators

Heterogeneity index

Tanser and Palmer (1999) used a Moving Standard Deviation Index (MSDI), based on a standard deviation filter of the red band of Landsat TM imagery. The MSDI was trialed to detect landscape heterogeneity, as an indicator of landscape changes leading to decline in condition. A high MSDI was found to be correlated with the NDVI and extent of decline in South African arid ecosystems (Palmer and Tanser 1999).

Perennial cover

To minimise the issue of seasonal fluctuations in annual cover, assessment of perennial cover provides an alternative. McCloy and Hall (1991) used this to their advantage by assessing cover of woody perennials to determine condition status. However, careful interpretation of results is required, as in some sites increased canopy cover is indicative of recovery of natural vegetation and improved condition, whilst in other areas it indicates invasion of woody weeds and decreased condition status. The determination of ideal woody canopy density is a controversial issue in some areas, with little evidence remaining to indicate pre-European densities. Vegetation canopy has been proposed as a useful indicator of total ecosystem condition as it is influenced by both climate and anthropogenic impacts (Belward 1991).

Understorey vegetation

Presence of understorey vegetation has potential as a condition indicator, as understorey is important for biodiversity, habitat values and forage. Generally, results have not been encouraging with Landsat MSS and TM data. Stenback and Congalton (1990) found they were able to determine presence or absence of understorey in a mixed conifer forest with accuracy varying between 55–69 % depending on density of the overstorey. It is likely that the use of higher resolution data may lead to a better outcome in detecting presence of understorey vegetation.

Bare ground

Exposed bare soil is often indicative of the condition of both soil and vegetation in arid communities and could be used as a remotely sensed indicator (Franklin *et al.* 1993). A combined perennial vegetation and organic soil cover measure was recommended to detect change in rangeland condition using remotely sensed data (Warren and Hutchison 1984).

Biodiversity

Nagendra and Gadgil (1999) investigated the use of Indian Remote Sensing (IRS) 1B LISS 2 imagery for large-scale monitoring of biodiversity. They performed a supervised classification of the imagery to differentiate landscape elements. They found a different suite of species within each classified area, and determined this was an adequate measure of broad scale biodiversity.

Grazing gradient

Within Australian arid rangelands, the resilience or grazing gradient method is the most widely published remotely sensed rangeland condition assessment (Bastin *et al.* 1993b; Pickup *et al.* 1994; Bastin *et al.* 1996; Pickup 1996; Pickup *et al.* 1998). This method is aimed at identifying trends of land degradation in non-equilibrium rangelands. To enable vegetation change due to grazing and management to be isolated from natural fluctuations and seasonal variability, a number of steps are performed. Paddocks are stratified according to vegetation type and distance from water, thus calculating "grazing gradients". Imagery is acquired six weeks following high rainfall, so that pasture cover is at a maximum and response to rainfall for different grazing gradients can be established. The response to rainfall or resilience of the area is equated to the degree of decline in condition. Those areas that have not responded to rainfall are expected to be more degraded than those where vegetation cover has improved in response to rainfall (Pickup *et al.* 1994).

Whilst this method has proved useful in rangeland management in Central Australia, it is not easily applicable to condition assessment for conservation of native biodiversity. Native and exotic species are not distinguished in this assessment, and it is probable that regeneration in response to rainfall in some degraded areas may be due to growth of exotic species. Shrubs and areas of woodland appear as low cover areas and therefore change over time in these areas cannot be closely monitored. Whilst a measure of change in resilience appears to be a useful concept for distinguishing between natural variations and long-term trends in condition (Pickup *et al.* 1994), it seems unlikely this particular methodology could easily be adapted for use in reserve management.

1.4.7 Change detection using remotely sensed data

As part of a monitoring strategy, not only is an accurate assessment of the vegetation condition required, but also a reliable method for change detection is essential to determine the effects of management, and the direction and location of changes in condition. "In the case of assessing the condition of vegetation in a Park, the principal hypothesis to be tested is the change in condition over time" (Parks Victoria 1998).

Change detection techniques have been used with all types of remote imagery, and there is a large and rapidly increasing body of literature on detecting land cover changes (Lunetta and Elvidge 1998; Johnson *et al.* 2000; Diouf and Lambin 2001; Fensham and Fairfax 2002). Fewer studies exist on methods for detecting vegetation condition changes, however similar concepts apply (Furby 1994; Wallace and Furby 1994).

Methods for spectral change identification involve either post-classification techniques or pre-classification techniques. Post classification techniques rely on classifying each of the original images individually and then determining areas of change. However, results using post classifications are often unsatisfactory for change detection, as errors in classification or registration are compounded by use of multiple imagery (Pilon *et al.* 1988).

Pre-classification techniques involve some form of image subtraction or image ratio where a single image of change is created from the original images (Lunetta and Elvidge 1998). Radiometric changes between the images limit simple image subtraction techniques, with radiometrically normalised image subtraction proving more accurate (Lunetta and Elvidge 1998). Hybrid methods use image differencing to detect areas of change then apply post-classification to determine cover classes on the areas of change (Pilon *et al.* 1988; Eastman 2001). The benefits are that thresholds of change may be applied, minimising errors of commission in change detection (Lunetta and Elvidge 1998).

A number of potential problems with change detection between imagery of different dates have been identified (Lunetta and Elvidge 1998):

(i) cloud cover on any of the images;

(ii) phenological variation – whilst this is less of an issue in semi-arid Australia, major variations in annual vegetation and vegetation greenness may occur following rainfall (Wallace and Thomas 1999);

(iii) radiometric performance of satellite sensors;

(iv) solar irradiance, solar zenith angle solar azimuth, cloud cover or change in atmospheric conditions will affect shadowing and scene brightness (Graetz *et al.* 1988; Pilon *et al.* 1988; Duggin and Robinove 1990; Davis *et al.* 1991); and

(v) registration errors (Walsh 1989).

These problems can usually be minimised with appropriate image acquisition, radiometric normalisation and co-registration of imagery (Lunetta and Elvidge 1998).

Potential problems with remote sensing techniques in arid environments

Remote sensing techniques present certain difficulties in arid environments. In areas of sparse vegetation cover, soil is found to have the highest reflectance, except in times of rapid growth of green vegetation following high rainfall. Arid environments often have complex underlying soil associations which mix with the vegetation reflectance (Graetz and Gentle 1982; Graetz 1987; Huete and Jackson 1987). Therefore, unsupervised classifications of Landsat data in arid areas tend to distinguish major soil types, rather than vegetation types (O'Neill and Eldridge 1990).

Low proportions of plant cover make estimating cover proportions difficult and detecting change in cover less reliable. Assessment of plant cover is made more difficult by low plant reflectance values (Graetz and Gentle 1982). Low reflectance of many arid species is due to low photosynthetic activity, but may also be related to leaf surface characteristics such as glaucousness or hairiness (Graetz and Gentle 1982). Generally, the typical response of high infrared and low red response of green vegetation occurs only during short growth periods following rain.

High temporal and spatial variability in arid environments may lead to difficulties in distinguishing between natural fluctuations in climate, rainfall, and vegetation and anthropogenically induced change (Pilon *et al.* 1988; Davis *et al.* 1991).

1.5 RESEARCH OBJECTIVES

The MSNP was gazetted in 1991, conserving much of the remaining semi-arid woodlands in northwest Victoria. Following this, priorities for gathering baseline data on vegetation condition and establishing a monitoring program were identified (NRE 1996). Three main areas of research were determined, based around the need for monitoring and understanding processes of vegetation change of semi-arid woodlands in northwest Victoria.

1. The pre-European condition of *C. pauper* woodlands in Victoria is largely unknown. Historical survey plans were analysed to provide information on the original distribution of semi-arid

woodlands, and pre-European structure and composition, to assist in developing appropriate benchmarks for vegetation condition assessment (Chapter 3).

2. To obtain baseline vegetation condition data and establish a monitoring program for the MSNP, an investigation of the most cost-effective techniques for condition assessment was required. Remote sensing techniques including satellite imagery, aerial photography and an existing GIS layer were compared to field survey condition assessment (Chapter 4).

3. To better understand regeneration requirements, and therefore processes of change in semi-arid woodlands field and laboratory experiments were undertaken. Regeneration within a grazing exclosure in the MSNP was investigated to determine the effects of water addition, fire, and soil disturbance. Soil seedbank studies, germination and dormancy experiments, and investigation of water requirements for germination and seedling survival were carried out. These experiments attempt to gain a greater understanding of regeneration requirements for perennial trees and tall and small shrubs in *C. pauper* woodland in northwest Victoria (Chapter 5).

2. THE STUDY AREA

2.1 PHYSICAL FEATURES

Study area

Casuarina pauper woodlands in Victoria are found only in the far northwest of the state, within the area north of the Mallee Highway and west of Manangatang, extending to the South Australian border in the west, and the Murray River in the north (NRE 2000a). This area lies within the Mallee Region of Victoria, which is the area bounded by the state borders of NSW and SA in the north and west, a line approximating the 36th parallel to the south and the Loddon River to the east (LCC 1987).

The term mallee is derived from the aboriginal word "Mali" referring to the water mallee tree, the roots of which aboriginal people in the area would derive water (Tindale 1974). Mallee is also used to describe the characteristic low and multi-stemmed growth form of eucalypt species within the region (Noble *et al.* 1990).

The study area for this research is bounded by the Mallee Highway to the south, the Victorian - South Australian border to the west, the Murray River to the north and the Calder Highway to the east. This forms an area of approximately 14,250 km^2 (Figure 2.1).

2.1.1 Geology

The study area lies within the Murray Basin, a large sedimentary unit consisting of sedimentary accumulations of Early Tertiary to Recent age. This sedimentary sequence overlies Palaeozoic and Proterozoic basement rocks (Wasson 1989). Fluctuation in sea level in these areas of low surface relief has been recorded in the sedimentary layers, with three major depositional events during the Tertiary Period (Brown and Stephenson 1991).

Two major fault lines occur within the study area. They are the Danyo Fault which is aligned in a north easterly direction from near Murrayville and extends to Rocket Lake; and the Murrayville Fault which extends northwest to the South Australian border (Figure 2.2) (Macumber 1991).

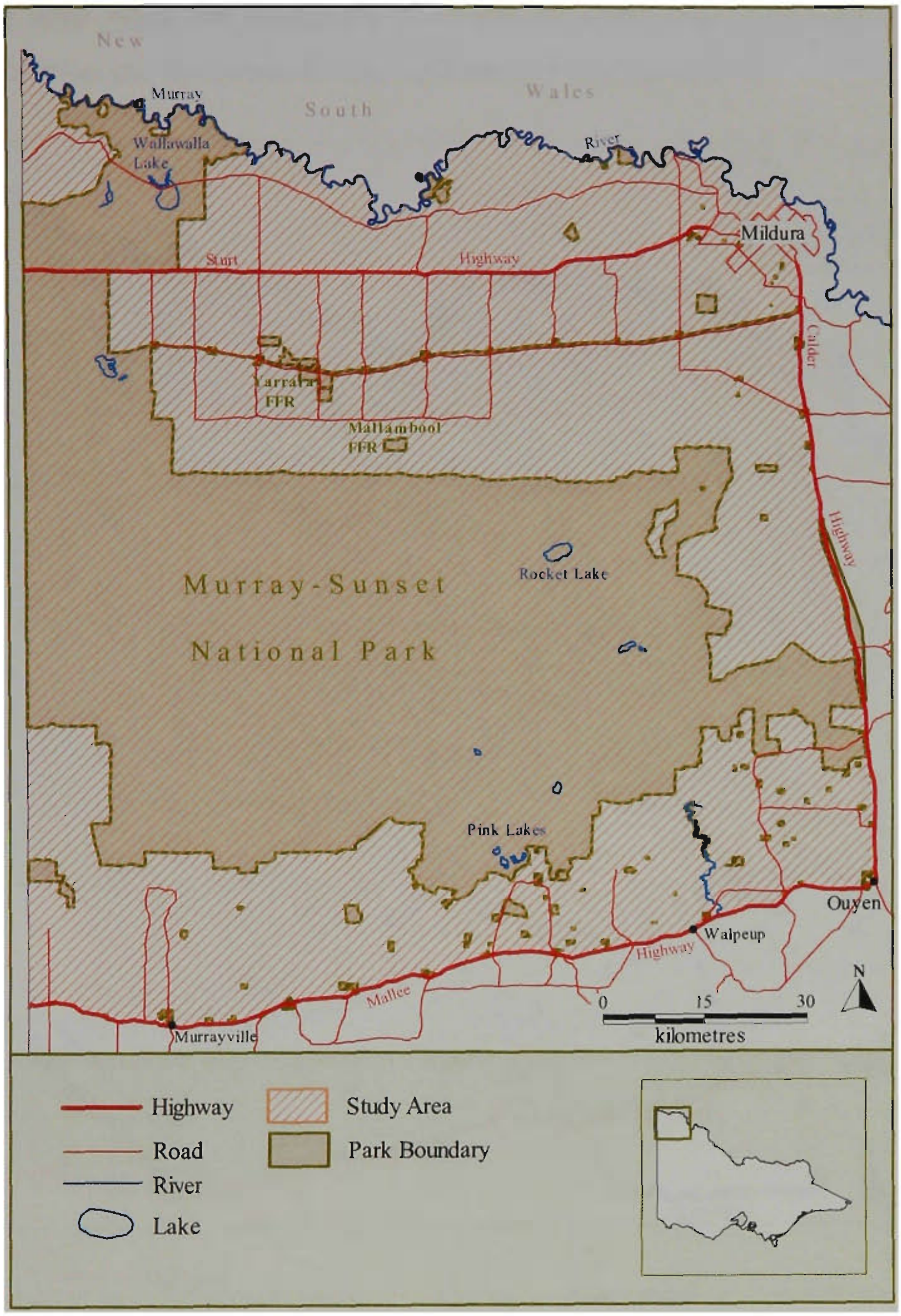


Figure 2.1. Map of the study area in northwest Victoria.

The major landform and geological features of the present day are dominated by the events of the last five million years, as Late Miocene-Pliocene sea levels retreated leaving sheets of quartzitic Parilla Sands over formerly marine and fluvio-lacustrine sediments of Palaeocene-Miocene age (Brown and Stephenson 1991). On the Pliocene strand-plain the step-wise retreating of the sea left behind a series of beach ridges of generally northwest-southeast trending alignment (Lawrence 1966).

Present day landforms of the area belong to two main geomorphic types, aeolian and alluvial. Alluvial landforms dating from the Quaternary occur along the Murray River forming extensive flood plains. However, the study area is dominated by Pleistocene age aeolian landforms. Two major aeolian dune types are recognised as occurring within the study area. These are the Woorinen Formation and the Lowan Sands (Lawrence 1966) (Figure 2.2).

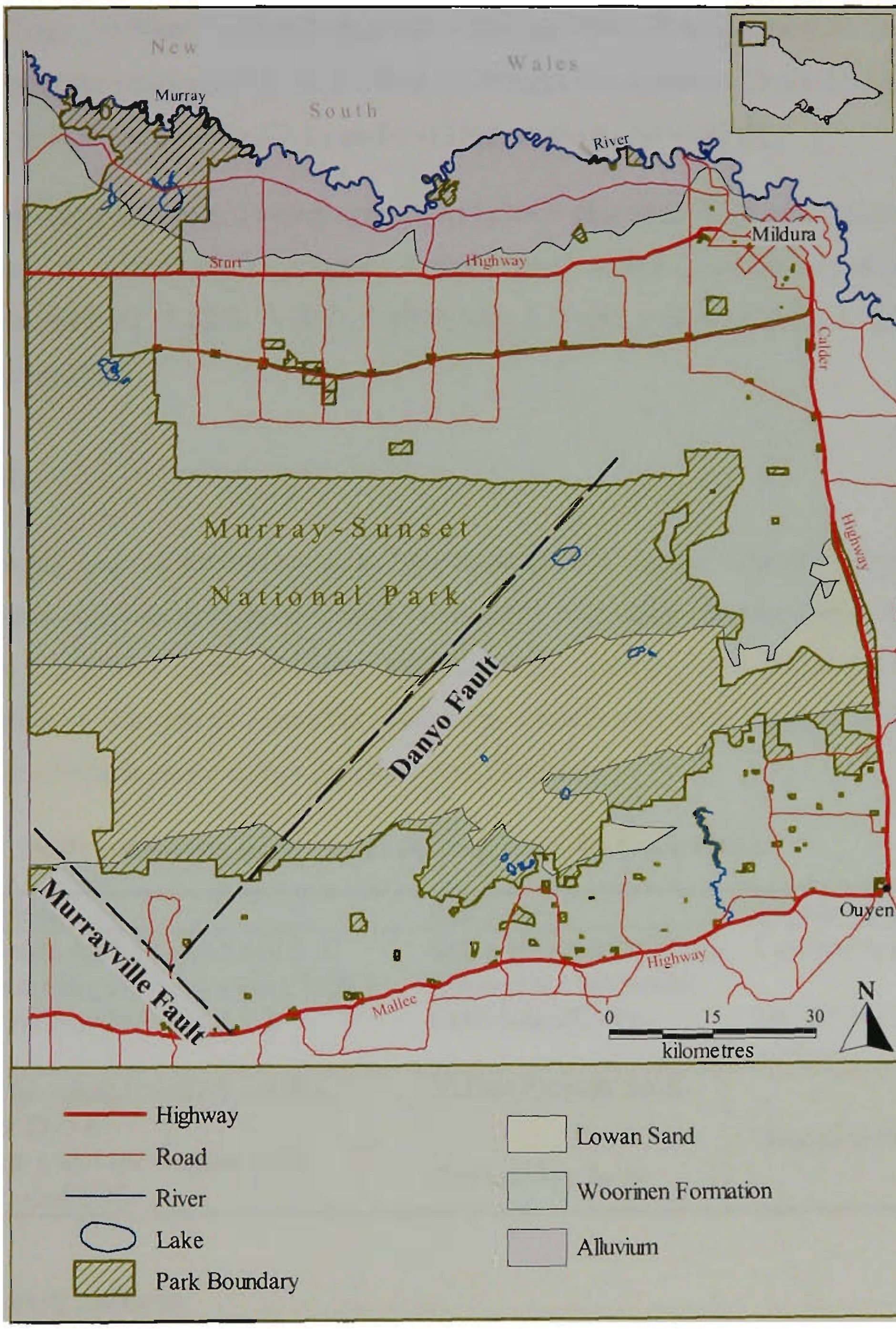


Figure 2.2. Geology of the study area (adapted from LCC 1987).

The Woorinen Formation dune type consists of short, low-linear dunes of east-west alignment and with moderately high clay and calcareous contents (Lawrence 1966). Areas of undulating plains also occur within the Woorinen dune fields. The Lowan Sands consist of steep and high-crested parabolic or sub-parabolic dunes of curved to irregular shape. Sediments are siliceous with very small clay and calcareous contents (Lawrence 1966). The Lowan Sand dune fields are also extremely prone to mobilisation following vegetation removal (Rowan and Downes 1963).

2.1.2 Land systems

Land systems have been described in Victoria as a simple method for classifying areas of similar climate, lithology, landform, soil and vegetation (Rowan 1990). Four geomorphic units have been described within the study area (Rowan 1990): (i) Present floodplain (4.1); (ii) Older alluvial plain (4.2); (iii) Low calcareous dunes (5.1); and (iv) High siliceous dunes (5.2).

From these geomorphic units, 37 land systems have been described. Twelve of these land systems are described as containing *C. pauper*, *A. luehmannii* and *C. gracilis* woodlands (5.2Pfc2, 5.1RPEfc3, 5.1RPEfc2, 5.1Rf3, 5.1Rf2, 5.1PREfcl2, 5.1Pcz2, 5.1Lf3, 4.2Lf2, 4.1Lf2, 4.1FWcf2, 4.1Lf3) (Rowan 1990).

2.1.3 Soils

In a broad-scale map of Australian soils, Northcote (1960) mapped five soils within northwest Victoria; brownish sands, grey-brown calcareous earths, and sandy alkaline apedal mottled-yellow duplex soils, calcareous earths and grey self-mulching cracking clays. Badawy (1982) mapped soils of northwest Victoria at a smaller scale with seven major soils mapped within northwest Victoria, five of which occur within the study area (Table 2.1).

Table 2.1. Soils within the study area (adapted from Badawy 1982).

Soil description	Soil type	Texture profile
Brownish sands; Uc5.11, 5.12 and 5.13	Sands (Coarse textured)	Uniform texture profiles
Grey self-mulching cracking clays; Ug5.2	Clays (Fine textured)	
Calcareous earths; Gc 1.1 and 1.2	Calcareous Earths	Gradational texture profiles
Sandy alkaline apedal mottled-yellow duplex soils; Dy5.6	Yellow Duplex Soils	Duplex texture profiles
Hard alkaline pedal red duplex soils; Dr2.13, 2.23 and 2.33	Red Duplex Soils	

Brownish sands uniform

Brownish sands occur extensively throughout the study area on dunes and sand sheets. They are generally of low fertility, and high permeability, although permeability may be limited in some soils with subsoil hardpans (Northcote *et al.* 1975). Low clay content in these soils limits their water holding capacity, and water is easily lost through evaporation and drainage (Badawy 1982).

Grey self-mulching cracking clays

Clays are largely found on river flood plains in the study area. These areas are generally moderately fertile. Root development is dependent on the subsoil structure, which may contain up to 70 % clay (Northcote *et al.* 1975). Cracks develop in these soils from a minimum of 6 mm wide and 30 cm deep. The cracking nature of these soils, whilst encouraging rapid re-wetting and permeability, can lead to poor moisture retention (Badawy 1982).

Calcareous earths

Calcareous earths occur across much of the study area on undulating plains and dunes. These soils have been formed from unconsolidated calcareous sediments, modified by aeolian activity (Northcote *et al.* 1975). These soils tend to be of low to moderate fertility, with high permeability. The soil moisture-holding ability depends on the quantity of clay in the soil, with higher clay content giving better moisture-holding capacity. Clay content in these soils increases gradually with depth (Badawy 1982).

Sandy alkaline apedal mottled-yellow duplex soils

These soils are distributed across much of the study area, particularly within the Murray-Sunset National Park (MSNP). Low inherent fertility is a feature of these soils. Permeability of these soils is high, however, some have decreased water permeability when dry (Northcote *et al.* 1975). High salt content may be found in subsoils (Badawy 1982). Ironstone gravel may occur in these soils, as well as nodules of "coffee rock", and segregations of carbonates (Badawy 1982).

Hard alkaline pedal red duplex soils

Red duplex soils occur in the southwest of the study area. Dr2.23 soils are commonly used for wheat growing, although they are inherently of low to moderate fertility (Northcote *et al.* 1975). Surface soils tend to be hard setting, limiting soil permeability. Subsoil salinity is usually high (Badawy 1982). Soil moisture properties of the soils are affected by the soil structure and clay content of subsoils (Badawy 1982).

2.1.4 Vegetation

The first detailed descriptions and mapping of vegetation in the study area was undertaken between 1985 to 1987 by the Land Conservation Council (LCC 1987). Thirty communities were described in the Mallee Parks of northwest Victoria. Of these, 16 vegetation communities were mapped within the study area (Table 2.2). In 2002 to 2003, more detailed mapping of vegetation communities in the study area was undertaken. Ecological vegetation classes (EVCs) of the study area were mapped to 1:25,000, and pre-European vegetation was modelled to complete the EVC100 GIS layer (Appendix 1). The original 1985 to 1987 EVC mapping undertaken by the LCC was used in the current study.

Table 2.2. Ecological Vegetation Classes (EVCs) of the study area.

Ecological Vegetation Class	Area (km^2)
Alluvial Plain Shrubland	176
Belah Woodland	41
Black Box-Chenopod Woodland	179
Broombush Mallee	119
Chenopod	1,520
Gypseous Plain Woodland	55
Lake Bed Herbfield	7
Lowan Sands Mallee	1,562
Pine-Buloke Woodland	53
Red-Swale Mallee	70
Riverine Grassy Forest	81
Saline Shrubland	280
Sandplain Grassland	24
Savannah Woodland/Savannah Mallee/Grassland Mosaic	1,019
Woorinen Sands Mallee	2,544
Woorinen Sands Mallee/Grassland Mosaic	2

Prior vegetation mapping of public land within the study area was undertaken in 1987 and the resulting EVC map (LCC 1987) has five classes that contain semi-arid woodland species; Belah Woodland, Gypseous Plain Woodland, Pine-Buloke Woodland, Sandplain Grassland and Savannah Woodland / Savannah Mallee / Grassland Mosaic.

Mallee woodlands are the most extensive community within the study area, commonly supporting an understorey of chenopods or *Triodia irritans* in deeper sands (LCC 1987). *C. pauper* and *C. gracilis–A. luehmannii* woodlands in the study area commonly occur on lunettes and ridges in the study area, but may occur on all landscape positions (Rowan and Downes 1963).

Alluvial terraces and riverine areas along the Murray River to the north of the study area support *Eucalyptus camaldulensis* (River Red-gum) forests and *Eucalyptus largiflorens* (Black Box) woodlands along with chenopod shrublands dominated by *Atriplex vesicaria* and *Maireana pyramidata* (Figure 2.3). Extensive evaporative basins (Boinkas) occur in the south, and east of the study area around Pink Lakes and the Raak plain. These support shrublands dominated by *Halosarcia* spp. and *Austrostipa* spp. grasslands.

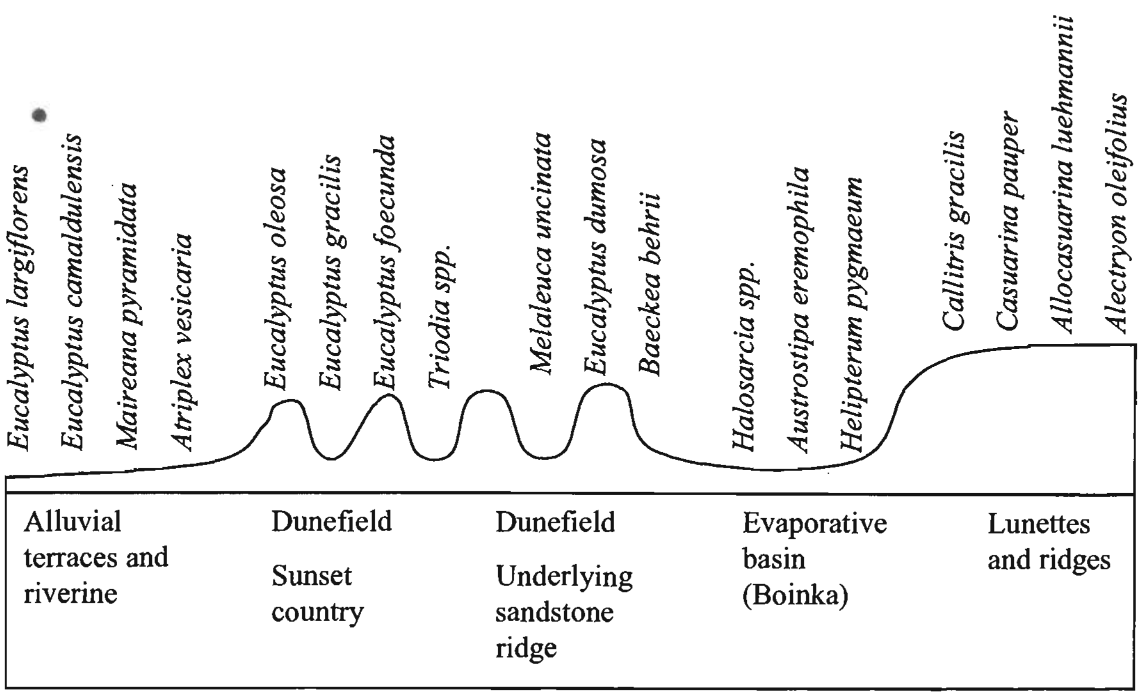


Figure 2.3. Dominant species occurring on different landforms in the study area.

2.2 CLIMATE

2.2.1 Temperature

Summers are hot with mean maximum temperatures in Mildura averaging 32.8°C in January (Figure 2.4). Summer maximum temperatures may commonly exceed 40°C in summer, with 46.9°C the hottest day on record at Mildura Airport. Winters are generally cool to mild with average minimum temperatures of 4.4°C (BOM 2000). Average daytime winter temperatures average 15°C in July. Frosts are relatively common, occurring 13 times per year on average at Walpeup Research Station and 20 times per year at Mildura Airport (Table 2.3).

Table 2.3. Mean number of days frosts may occur in the study area (Clewett *et al.* 2003).

Days < 2.2°C	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Year
Mildura Airport	-	-	-	-	1	6	8	4	1	-	-	-	20
Ouyen	-	-	-	-	2	6	8	5	3	-	-	-	24
Walpeup Research	-	-	-	-	-	3	5	3	2	-	-	-	13

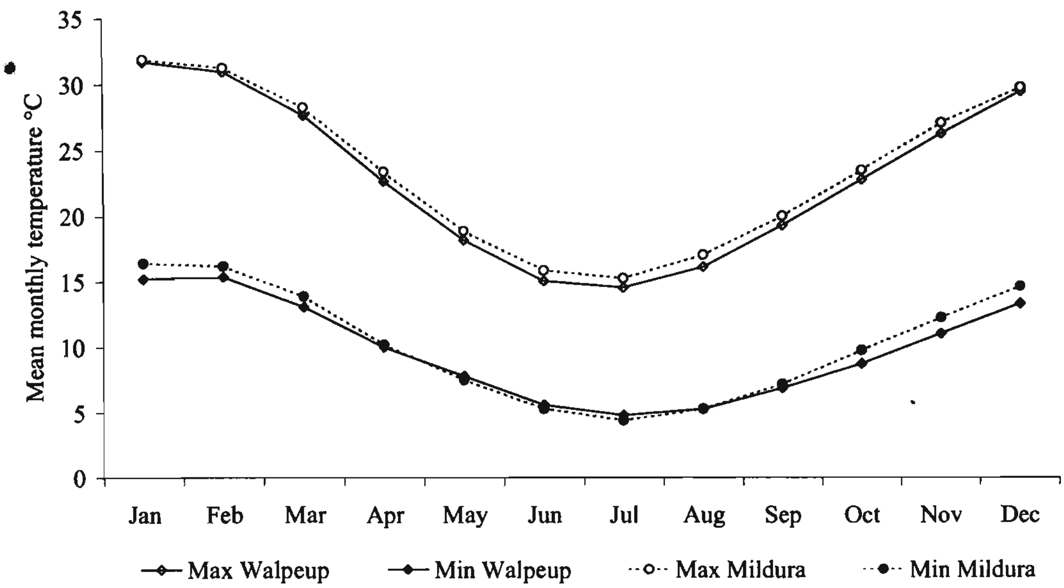


Figure 2.4. Mean monthly temperature at Walpeup and Mildura (BOM 2000).

2.2.2 Evaporation

Evaporation rates greatly exceed rainfall during most months of the year. Evaporation is particularly high during summer (Clewett *et al.* 2003). Therefore, the effects of light rainfall are short lived and only during larger rainfall events that exceed evaporation rates for a number of days, are much contribution to soil moisture made. Pan evaporation for three locations in the study area is shown in Figure 2.5.

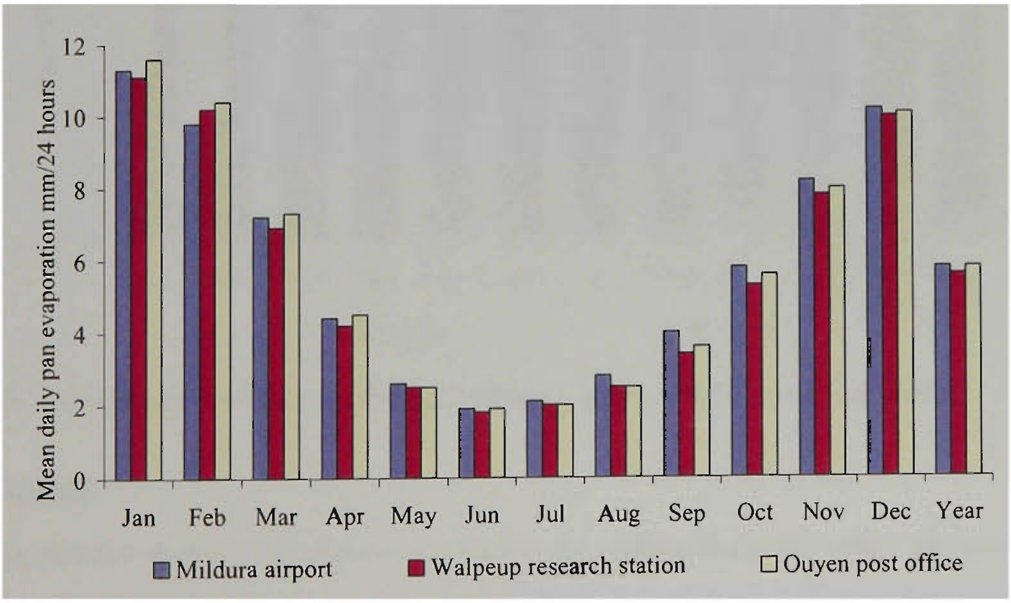


Figure 2.5. Mean daily pan evaporation at Mildura airport, Walpeup research station and Ouyen post office (Clewett *et al.* 2003).

2.2.3 Rainfall

Source of rainfall to northwest Victoria

Sources of rainfall change seasonally. The effect of northern tropical cyclones or thunderstorms can bring rainfall to the study area in late summer, as well as occasionally in October (Gentilli 1972). These summer rains are the most variable. Effective rains are slightly more reliable in spring and autumn, with winter rains, being the most reliable in this region (Gentilli 1972).

In winter, depressions moving east, from south of Australia bring rain to southern areas of the continent. However, heavy winter rains are only received in the northwest when anticyclones cross the coast at around 30 degrees latitude. Dry years have occurred when anticyclones repeatedly crossed the coast at 32 degrees or greater (Gentilli 1972). From midwinter, the path of anticyclones shifts further to the south, such that only the most southern areas of Victoria and Tasmania receive rain from these depressions (Gentilli 1972). The impact of variable, occasionally high rainfall events in summer is illustrated in the difference between mean and median rainfall, as occasional, high rainfall events raise the mean summer rainfall (Figure 2.6).

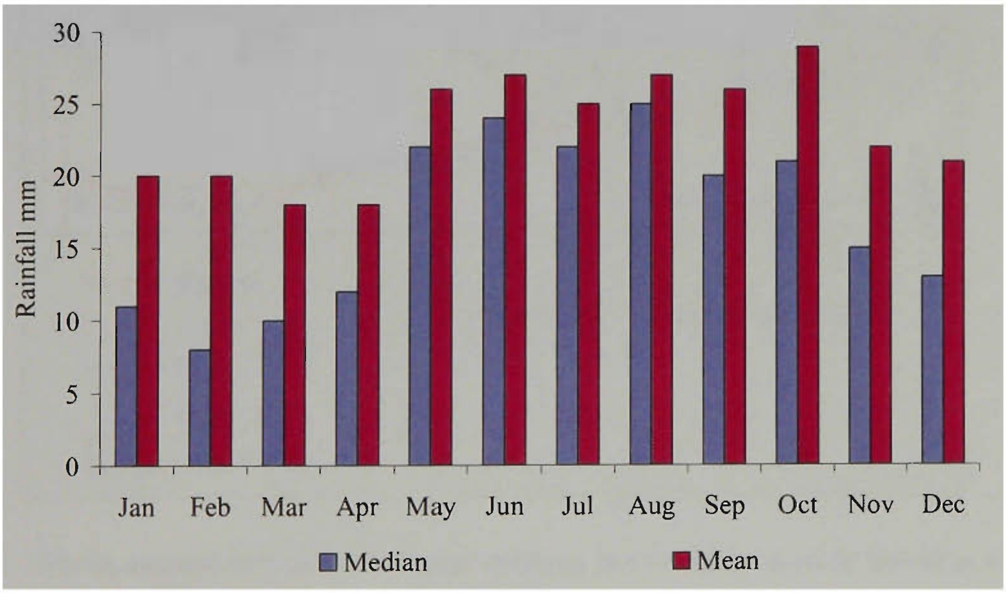


Figure 2.6. Mean and median rainfall at Mildura airport (Clewett *et al.* 2003).

Annual rainfall

On average, rainfall is relatively evenly distributed throughout the year, with a slight bias toward falls in the winter months (Clewett *et al.* 2003). Median annual rainfall in the study area ranges from 344 mm at Ouyen in the southeast, to 257 mm at Lindsay Point in the northwest (Clewett *et al.* 2003) (Figure 2.7). The longest rainfall record in the area is from Murray Lock, where records commenced in 1877 (Table 2.4).

Figure 2.7. Mean annual rainfall at rainfall stations in the study area (Clewett *et al.* 2003).

Rainfall variability

Despite minimal regular seasonal variation occurring within the study area (LCC 1987), significant variation occurs between years. Mean rainfall figures are frequently not realized, with higher mean rainfall resulting from rare heavy rainfall events. Consecutive years of well above average rainfall were experienced in the study site during 1889/90, 1910/11, 1973/4/5, and 1992/3 (Figure 2.8).

Drought years are frequent within the study area. Drought is defined as "a prolonged, abnormally dry period when there is not enough water for user's' normal needs" (BOM 2000). A severe rainfall deficiency exists in a district when rainfall for three months or more is in the lowest 5 %

of records (BOM 2000). Ten extended drought periods, lasting for more than 24 months, have been recorded in Mildura. These occurred during Jan 1895 to Feb 1899, Jun 1900 to Feb 1903, Apr 1913 to Jul 1916, Jul 1918 to Jul 1920, Jun 1925 to Jul 1930, Feb 1937 to Jan 1939, Jul 1942 to Feb 1946, Mar 1981 to Aug 1983, Jan 1997 to Dec 1998, and Nov 2000 to Jan 2003.

Table 2.4. Rainfall stations in northwest Victoria, showing mean and median rainfall, number of years and dates stations have been operating (Clewett *et al.* 2003).

Rainfall station	Mean rainfall	Median rainfall	No. years	Records from	Records to
Carwarp	288	278	65	1915	1979
Cowangie Post Office	320	326	88	1913	2004
Galah	323	324	90	1911	2004
Irymple (Highgate)	279	287	92	1909	2004
Kurnwill School	279	283	61	1930	1990
Lindsay Point	262	257	119	1882	2004
Merbein CSIRO Research Station	284	284	70	1921	2004
Meringur Post Office	282	277	73	1928	2004
Merrinee	287	288	76	1925	2004
Mildura Airport Amo Composite	280	277	112	1890	2004
Morkalla North	276	273	72	1929	2004
Murray Lock No.9 Composite	281	277	124	1877	2004
Murrayville	326	320	81	1911	1991
Ouyen Post Office	334	344	89	1912	2004
Walpa	308	309	85	1916	2004
Walpeup Post Office	339	342	89	1912	2004
Werrimull	276	269	74	1927	2004

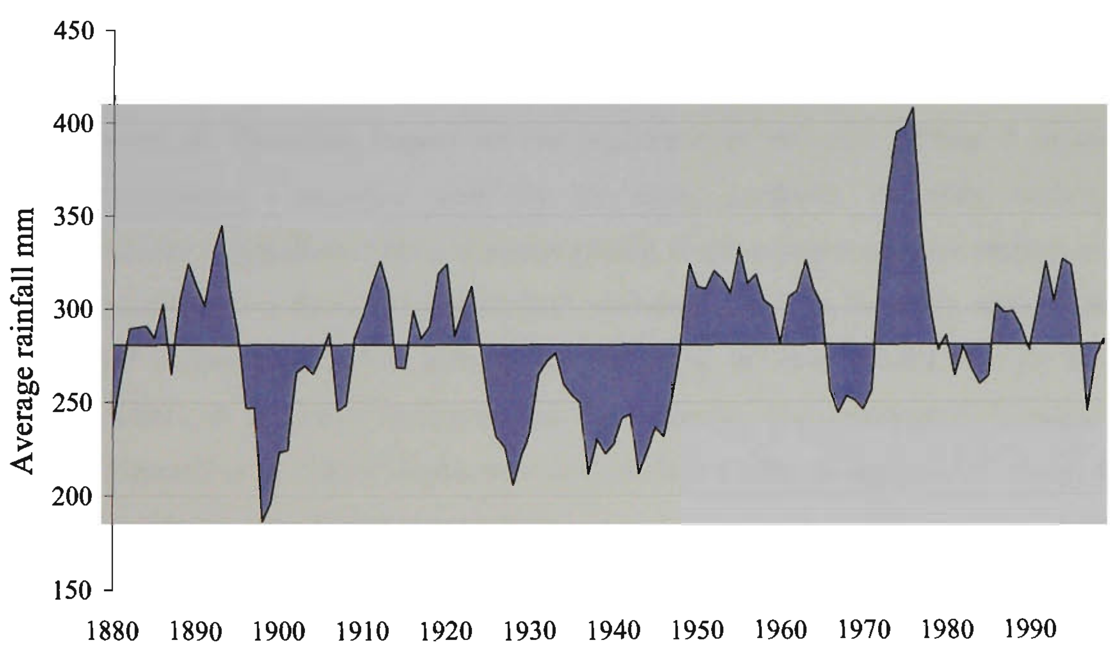


Figure 2.8. Five-year moving average of rainfall at Lindsay Point (Clewett *et al.* 2003).

2.3 HISTORY AND LANDUSE

2.3.1 Aboriginal occupation

Many burial grounds, middens and scar trees provide evidence of a long history of aboriginal occupation in the study area. The earliest evidence of aboriginal occupation of the study area comes from the Murray River Valley and suggests aborigines were living in this area up to 16,000 y BP (Ross 1981). Away from the permanent water source of the Murray River, there is evidence of aboriginal activity 7,600 y BP at the Raak Plains. Other aboriginal sites have been found near Murrayville and the Pink Lakes district (Ross 1981). Despite this long history, there is little record of aboriginal life in the Mallee at the time of European exploration and settlement. This has been attributed, in part, to small pox which is believed to have decimated tribes throughout Australia in 1789 and 1831 to 1832, prior to any European contact in the region (LCC 1987).

Scant evidence remains on how tribes managed to live in the harsh, dry conditions that would have prevailed over most of the Mallee area. Desert tribes living in the area did not have unlimited access to the Murray River, as other tribes occupied this territory. It is reported that the 'desert' aborigines used *Eucalyptus gracilis* (Yorrel), *E. oleosa* (Oil Mallee), and *Hakea leucoptera* ssp. *leucoptera* (Needlewood) to obtain water (Morris 1942; Tindale 1974). Crab holes, gilgais and other ephemeral water sources were also well known and utilised by aborigines in the area (Massola 1966).

Little is known of aboriginal impact on the vegetation of the area. Whilst it is known that Australian aborigines commonly used fire for many purposes, including management of vegetation density (Nicholson 1981), it is not known if these practises were employed to ignite *C. pauper* woodlands for the derivation of food resources. There is, however, some evidence that fire was used to manipulate plant growth for harvesting in other woodland types in Australia (Nicholson 1981). It is likely, however, that the generally low understorey biomass of these woodlands (Sandell *et al.* 2002) would have limited their ability to carry a fire except following exceptional rainfall events (Hodgkinson 1991).

2.3.2 European history

Exploration

The first European to view the Victorian Mallee was Captain William Charles Sturt, who viewed the northern boundary of the Victorian Mallee from near the Murray River in 1830. Joseph Hawdon, an early settler/explorer, drove cattle along the Victorian bank of the Murray River, on an overland trek from Sydney to Adelaide in 1838. Also in that year, Edward John Eyre attempted to cross the Victorian Mallee from the south to the north, but was thwarted due to lack of water and forced to return just north of Lake Hindmarsh. He subsequently followed Hawdon's route along the Murray to Adelaide (Kenyon 1982).

Surveys of the Victorian Mallee began in the 1840s. E. R. White, as assistant surveyor, explored and surveyed extensively within the Mallee between 1849 to 1851, including the border between South Australia and Victoria, and the 142nd Meridian road, sections of which are still in use today (LCC 1987).

European settlement

Kenyon (1982) outlines three major stages in the European settlement of the Mallee:

(i) squatting (1847 to 1879) – squatters took up large holdings, initially on better land in the southern Mallee, and along the Murray River frontage, and gradually moving into poorer country, as less land was available;

(ii) settlement (1870 to 1880s) – selectors began to move in and land was selected and settled; and

(iii) wheat (1890s to 1920s) – establishment of the mallee as a wheat-growing region, accelerated under 'Soldier Settlement Schemes' after World War I.

These three stages outline the development and major changes of the Victorian Mallee from aboriginal occupation to European settlement. Selection in the northern Mallee, including the Millewa took place some 30 to 40 years later than Kenyon's schedule, as much of this land was originally considered unsuitable for agriculture. Therefore, in these areas the settlement and wheat phases developed concurrently.

Squatting

By 1846, pastoralists had begun to colonize the northern boundaries of the Murray River, and by 1847, squatters occupied the southern banks of the Murray (Ward 1988). Joseph Hawdon was the first grazier to take up leases along the Murray River frontage, which he later sold to John Crozier. Crozier acquired further leases, and by the 1860s had become the largest leaseholder in the region (Ward 1988).

Rapid expansion of squatters in the Victorian Mallee continued, and sheep numbers and land tenure reached a peak in the Mallee in 1871, with 425,000 sheep grazing on over 3 million ha of land (LCC 1987). However, due to rabbit and wild dog problems, numbers began to fall rapidly, and by 1882, there were 145 holdings on 1.4 million ha, with approximately 122,000 sheep in the Victorian Mallee (Roberts 1968).

As early as 1892, negative impacts on the land were beginning to be recognised, especially problems with weeds, rabbits and erosion (Dixon 1892). This is evidenced on early parish plans, with numerous records of Tobacco Bush (*Nicotiana glauca*) and records of Barley Grass (*Hordeum* spp.) and other introduced plant species (eg. Anon 1896; Turner and McGauran 1908).

Settlement and wheat

Two main hurdles initially limited the closer settlement and cropping of the northwest Mallee. Prior to the development of the Mallee roller, jump stump plough and improvements in "dryland farming" techniques in the 1870s, it was too costly to undertake clearing of the Mallee country for cropping (Holt 1947). Secondly, until railways were built, there was no feasible or cost effective way for farmers to transport their wheat.

The first railway in the area was the railway from Melbourne to Mildura, which opened in 1903, thus opening large areas of the Mallee for wheat growing (Condon 1980). The railway from Ouyen to Murrayville was then opened in 1912, as the country from Ouyen to the South Australian border was opened for selection (McLean 1975). In 1923, the railway west from Red Cliffs to Meringur was constructed, opening further land for cropping in what was to become the Millewa farming district.

By 1927, the northern Millewa area was largely settled, and the Victorian Government decided to open the southern Millewa area or Sunset Country, much of which consisted of lighter soils and so was expected to yield poorer crops (George 1990). In addition to poorer soils, extreme difficulties were faced in providing these areas with water. To open more country for development the construction of 30,000 gallon water tanks through the Millewa and Sunset Country was funded by the State Government (McLean 1975). The last railway to be constructed in the northwest Victorian Mallee was the Nowingi Line, running west from Nowingi to the Raak Plain. Despite the provision of water tanks and transport, this country proved unsuitable for cropping, and the railway was used to transport gypsum mined from the Raak Plain (McLean 1975).

The original surveyed farm blocks in most of the Mallee were 640 to 800 acres. The Millewa farming blocks opened for selection in 1924 were 800 acres in size. It was quickly realised that these blocks would not provide sufficient income for a family to subsist (Holt 1947). In 1934, compensation was offered to the largely English migrant settlers for misrepresentation of the value of the land. The vacated land was then divided into larger holdings of 6,000 acres or more (George 1990).

Clearing and thinning

Widespread clearing of woodlands in northwest Victoria began in the late 1800s, reaching a peak in 1907 to 1914 after the construction of the railway to open up the country for wheat growing (Condon 1980). Nearly 200,000 ha of woodlands were cleared in the southern Victorian Mallee region during this period (Condon 1980). Initially clearing was done by hand, however, with more powerful machines, larger areas could be cleared. Using powerful tractors, chaining became the preferred method of clearing. By dragging a chain between the two tractors just above ground level, up to 16 ha per day was able to be cleared (Condon 1980). Similar clearing was undertaken in woodlands in western New South Wales both for cropping and to improve pastures for grazing (Cunningham *et al.* 1981).

2.3.3 Land use

Current land use within the Mallee region is largely agricultural (LCC 1987). There are over 2,000 agricultural properties within the Mallee statistical division, covering a total area of 10,507,010 ha (ABS 2001). Dry land cropping is the most extensive land use in the region, followed by grazing by sheep and to a lesser extent, cattle.

The irrigation district in Mildura is one of Australia's most productive agricultural centres. The development of irrigation in the Mildura area began with the arrival of the Chaffey brothers in 1886 (Condon 1980). Wine grapes, dried fruits, olives, citrus fruits, avocadoes and vegetables such as asparagus, carrots and potatoes are now produced in the region (ABS 2001).

A forestry industry was centred on a number of reserves of *C. gracilis* in the study area, however, this had largely ceased by the 1940s. Forestry reserves of *C. pauper* woodlands were set aside in the early 1900s, however, were increasingly depleted due to a number of processes. *C. gracilis* timber was in demand, and in some early survey maps, it was reported by the 1920s that "all the good timber has been cut out" (Victorian Railways 1921). In the 1930s, a combination of drought and disease or insect attack led to the death of many remaining stands of *C. gracilis* – particularly in the Sunset Country and Millewa farming district. The lack of regeneration of *C. gracilis* was noted at this time and forestry activity was limited (Zimmer 1942; Sims and Carne 1947). Interestingly, a number of reserves were also established during the initial parish surveys for the production of eucalyptus oil. Today no commercial harvesting of *C. pauper* or *C. gracilis* timber takes place, although some fire wood collection may take place on private land (LCC 1987).

Other land uses in the region include gypsum mining and salt harvesting from saline discharge areas, as well as more recently, mineral sands mining. The latter industry, although new, is likely to have a significant and increasing presence within the region due to the large quantity of economic deposits now known to exist there.

2.3.4 Vegetation conservation

Early priorities in vegetation management in the northwest Mallee were overridingly for vegetation clearance. Despite the rush to maximise cultivation, it was recognized very early on that some native vegetation should be retained. Landowners were encouraged not to clear vegetation on top of sand dunes and under the *Closer Settlement Act* 1904, landowners were required to leave 3 % of trees on their property (Roberts 1968; Torpey 1986). These requirements were often ignored, however, in a zealous effort to cultivate. Patches of trees and shrubs were thought to harbour weeds and vermin, and were often removed due to this perceived threat (LCC 1987).

Some initial vegetation conservation resulted through the reservation of recreation reserves; some areas of *C. gracilis* timber were set aside by the Victorian Forestry Commission for harvesting, and some areas of mallee were set aside for eucalyptus oil production (Anon 1896; Turner and McGauran 1908). All of these measures resulted in only relatively small vegetation reserves, but a well vegetated network of road reserves was retained which provides clues as to the original vegetation types of the area.

Whilst many early settlers wrote of their great appreciation for the beauty of this country (Torpey 1986), no formal arrangement for conservation of any of the land was made until 1979 with the proclamation of the Pink Lakes State Park (507 km^2). This followed the recommendations of the Land Conservation Council (LCC), (established under the *Land Conservation Act* 1970). The investigation of the LCC into the use of public land in the Mallee commenced in 1972. Final recommendations following these investigations were released in 1977 (LCC 1977).

Grazing began to be phased out within the Pink Lakes State Park after the 1977 LCC recommendations (LCC 1987). Thirty-three additional Flora and Fauna reserves were also created on land containing rare or endangered species or communities otherwise poorly represented in other conservation reserves (LCC 1987). However, grazing continued on many of these reserves.

Following concern over clearing in the Mallee for agriculture, the LCC review was commenced in 1985, and extensive surveys of the ecology and agricultural use of the Victorian Mallee were undertaken (LCC 1987). Final recommendations resulting from this investigation were published in 1989 (LCC 1989).

Resulting from the LCC final recommendations, the Sunset Country and much other remaining public land, together with the former Pink Lakes State Park was incorporated into the MSNP

(6,330 km^2) (LCC 1989). The MSNP is Victoria's second largest national Park, and is listed in Category II (National Parks) of the IUCN United Nations' List of National Parks and Protected areas. Category II areas are managed primarily for ecosystem conservation and appropriate recreation (NRE 1996). The MSNP covers a broad range of vegetation communities, dominated by dune fields of greater than 2 % slope vegetated by mallee species.

Whilst much of the land that was unsuitable for agriculture within the MSNP has remained in close to pristine condition, substantial areas of more fertile soils are severely degraded. Of the significant vegetation types (EVCs) that have been identified within the MSNP, four of these contain *C. pauper* as an important constituent plant species. These EVCs and threatening processes are listed in Table 2.5.

Table 2.5. EVC descriptions, significance in Mallee Parks and area of EVC mapped in public land within the study area.

EVC description (LCC 1987)	**Significance to Mallee Parks** (NRE 1996)	**Area** (ha)
Belah Woodland – Dominant species *Casuarina pauper*. Diverse shrub layer including tall shrubs and ground layer consisting of herbs, sub-shrubs and perennial grasses.	Substantially threatened due to small size of remnants.	4,073
Gypseous Plain Woodland – Overstorey of scattered *Myoporum platycarpum*. Understorey of *Sclerolaena* spp., native and introduced annuals. Occasional *Maireana sedifolia* and *Nitraria billardierei*.	Within Victoria is endemic to MSNP. The only remnants are grossly disturbed such that the original community is now believed to be extinct within Victoria.	5,460
Pine-Buloke Woodland – Dominated by *Allocasuarina luehmannii* and/or *Callitris gracilis* with a number of small trees eg. *Pittosporum phylliraeoides*, *Acacia oswaldii* and *Santalum acuminatum*. Understorey dominated by perennial grasses and herbs.	Largely cleared for agriculture and identified as the most threatened community within the Victorian Mallee.	5,271
Sandplain Grassland – Occasional scattered woodland trees. Understorey dominated by perennial grasses and native annual herbs.	Endemic to the MSNP within Victoria. Among the largest and least disturbed native grasslands in the state.	2,422
Savannah Woodland / Savannah Mallee / Grassland Mosaic – mosaic of very disturbed, probably anthropogenic communities. Dominated by *Callitris gracilis*, *Alectryon oleifolius*, and *E. socialis* or *E. gracilis*.	Severely disturbed remnants of other woodland communities.	101,803

3. INVESTIGATION OF CHANGE IN DISTRIBUTION AND COMMUNITY STRUCTURE OF SEMI-ARID WOODLAND IN NORTHWEST VICTORIA

3.1 INTRODUCTION

Extensive modification of *Casuarina pauper* (Belah) and *Callitris gracilis–Allocasuarina luehmannii* (Pine-Buloke) semi-arid woodlands has occurred due to clearing, thinning and grazing since European settlement (LCC 1987; Westbrooke *et al.* 1988; Westbrooke 1998). Due to the extent of modification, determining the distribution and community structure of semi-arid woodland at the time of European settlement in northwest Victoria must rely on either modelling or historical records. Historical records have been used in few studies in Australia, despite their potential value in describing the distribution and structure of pre-European vegetation (Lunt 2002).

Historical information provides a useful option for determining benchmarks for vegetation condition assessment, where sufficient information is available. Even where historical data are limited, gaining a better understanding of historical systems can be useful in informing desired or notional benchmarks.

Understanding the distribution and structure of pre-European vegetation can also benefit revegetation and restoration efforts. Whilst return to pre-European conditions may not always be the goal, knowledge of the extent of historical variability is valuable in informing restoration and management (Swetnam *et al.* 1999; Foster 2000). It has been suggested that historical conditions should be considered prior to any ecological interpretation of current ecosystems or determining management protocols (Foster 2000).

Use of historical data can also assist in determining the extent of change since European settlement (Brown 1998), enabling an understanding of the impacts of past management regimes on the native vegetation. Future changes in ecosystems can also be predicted with a greater understanding of past conditions (Foster 2000; Mladenoff *et al.* 2002).

Many types of historical information have been used to provide information on ecosystems at varying temporal and spatial scales. These include, but are not limited to: survey plans and historical maps; diaries and observations of explorers, early settlers and travellers; aerial photography (Fensham and Fairfax 2002); repeat photography (Pickard 2002); pollen analysis from sediments (Harle *et al.* 2002); Packrat middens (Rhode 2001); dendrochronology (Kipfmueller and Swetnam 2001; Pearson and Searson 2002); analysis of fire scars (Kipfmueller and Swetnam 2001); and stable carbon isotopes (Witt 2002). The type of historical data used will depend on the historical information required, data availability and accuracy and spatial and temporal requirements.

Interpretation of historical information is a subjective process relying on largely qualitative data (Swetnam *et al.* 1999). Subjectivity is inherent in placing vegetation boundaries to map pre-European vegetation from survey plans and surveyor's notes (Brown 1998). Confusion may also arise when trying to determine plant species referred to on survey plans. Many plans allude to now obscure common names, and generic names that do not allow isolation of separate species (Mladenoff *et al.* 2002).

Difficulty in interpreting subjective descriptions such as 'dense forest' or 'open plains' on survey plans has frequently been encountered (Norris *et al.* 1991; Lunt 1997; Fensham and Holman 1998). Fensham and Holman (1998) concluded that structural descriptors on survey plans could indicate density, providing they were carefully calibrated. However, Lunt (1997) found the terms written on survey plans relating to vegetation density were generally unreliable. Further insight into this problem is given by the definition of Forest Land in 1805 "such as abounds with Grass and is the only Ground which is fit to Graze; according to the local distinction, the grass is the discriminating character and not the Trees, for by making use of the former it is clearly understood as different from a Brush or Scrub" (Ramson 1988 p.257).

Another potential difficulty in interpretation can be absence of a feature from the map. Absence of a feature does not necessarily indicate the feature was not present at the time of survey (Pickard 1994). Depending on the cause for the survey and the interest and knowledge of the original surveyor, certain features may have been recorded less frequently or omitted entirely. Details recorded and reliability of the data are limited in part because these data were never expected to provide a complete ecological record (Mladenoff *et al.* 2002). To obtain greater confidence in historical data sources, a number of historical sources used in conjunction with more current data such as field surveys, or more recent mapping is recommended (Swetnam *et al.* 1999; Egan and Howell 2001).

Analysis of survey plans can also be very time consuming, requiring overlaying survey plans with current maps to determine precise location of features. Geographical information system (GIS) software has the potential to greatly improve the efficiency of overlaying maps, creating spatial databases, spatial analysis, and map production.

Despite these limitations, survey plans and historical maps provide one of the few spatially precise sources of data on the extent and structure of vegetation communities at the time of European settlement. A number of studies have been conducted in northern USA where extensive and detailed survey records exist (Bourdo 1956; Barnes 1989; Frelich 1995; Brown 1998; Whitney and DeCant 2001), however, there have been few similar studies in Australia.

Jeans (1978) details the use of the survey plan record of New South Wales (NSW) for vegetation mapping. Survey plans have been used to investigate pre-European vegetation distribution in the Midlands area of Tasmania (Fensham 1989), the Wimmera Plains of Victoria (Morcom and Westbrooke 1998) and the Darling Downs region of Queensland (Fensham and Fairfax 1997). Changes in distribution and structure of vegetation communities have also been investigated using the survey plan record (Oxley 1987; Norris *et al.* 1991; Lunt 1997, 1998).

To date there is only limited information available on the pre-settlement distribution and structure of semi-arid woodland in northwest Victoria (LCC 1987), and there has been limited research into possible changes since European settlement (Westbrooke *et al.* 1988). Therefore this study investigated historical maps and survey plans, which provide a thorough spatial coverage of the study area. Historical survey plans for the study area date back to the 1860s, the time of first European settlement of the study area.

The objectives of this study were to:

(i) assess survey plans as a data source for mapping pre-European vegetation distribution, structure and composition;
(ii) determine the likely pre-European distribution, structure and composition of semi-arid woodland in northwest Victoria;
(iii) determine if sub-communities, based on overstorey species (*C. pauper*, *C. gracilis*, *Alectryon oleifolius* and *Allocasuarina luehmannii*) can be defined in the semi-arid woodlands of northwest Victoria; and
(iv) determine the extent and nature of change in semi-arid woodland since European settlement.

3.2 METHODS

3.2.1 Analysis of survey plans and compilation of GIS database

Historical data sources

Historical records were consulted to determine the likely pre-European distribution and structure of semi-arid woodland in northwest Victoria. Information sources included historical maps, survey plans and written material from early 20th century botanists.

Locating survey plans

Land surveys of northwest Victoria were commenced in the 1850s with the survey of the border between Victoria and South Australia (Chappel 1966). In 1885 and 1886 the Victorian Mallee was surveyed by T. H. Turner and E. J. Nankinell into the A and B blocks as recommended by the Royal commission in 1878 (Kenyon 1982). From this survey the map "Mallee block surveys, counties of Millewa and Weeah surveyed by T. H. Turner, contract surveyor, Nhill, 9th May 1887" was produced, and today remains a valuable record of the vegetation of the area (Appendix 2).

Surveys were conducted to provide information to early settlers and the government on the quality of the land for agriculture. Data on vegetation and soil types were important in determining agricultural values of the land and so detailed notes were written on many survey plans. These survey plans now provide detail about vegetation in areas that have been cleared for cropping, or modified by over 100 years of stock grazing.

Historical maps and early survey plans were located at the Land Victoria office of the Department of Natural Resources and Environment (NRE), and at the State Library of Victoria. Survey plans at the Land Victoria office are stored on microfiche, with two main registers of early survey plans. The "historical plans" series includes surveys that contributed to the development of the cadastral system, and other ancillary plans (Cabena *et al.* 1992). These plans contain some vegetation information, but did not provide such a spatially complete record as the "Put Away" series and so were not used in this study.

The "Put Away" series contains superseded parish plans and record plans. The scale of the surveys ranges from small-scale plans of individual allotments to large-scale plans containing many parishes (Cabena *et al.* 1992). All survey plans of the 79 parishes in the study area were examined from the "Put Away" series of Plan Registers (see Appendix 3 for list of parishes in the study area). Plans of Allotments and Subdivision of Allotments from the "Put Away" series containing fewer than three references to semi-arid woodland vegetation were not used. Survey plans containing reference to semi-arid woodland species were printed from microfiche (Appendix 2).

Derivation of GIS map layers from survey plans

Digital images of each survey plan were required to enable use in a Geographical Information System (GIS) and plans were scanned and saved as Joint Photographic Experts Group (JPEG) files using an Epsom 1650 scanner. Plans from the State Library of Victoria were photocopied then scanned as JPEG files using a Hewlett Packard Scan jet 3c scanner.

The resulting digital files were registered against current day topographical features (Road25, Hydro25, Cont25) (Appendix 1) that could be matched to those on the survey plans. Road intersections and distinguishing corners in roads were the most easily identifiable features used for registration of plans. Other features including lakes, tanks, drainage lines, contours, and hilltops were also used to assist registration. Between five and 35 control points were used to register each survey plan.

Vegetation notes relating to semi-arid woodland species were identified, and polygons created around the text map vegetation extent. Polygons were deemed to be more useful and representative than point data as the text was usually written in lines or blocks across the survey plan, to indicate areas where the species were found.

On some survey plans, further indication of the likely extent of the vegetation was provided. For example lines were drawn, occasionally with text, indicating vegetation boundaries such as "edge of good pine country". Often pine ridges were drawn onto the survey plan indicating the entirety of the ridge. Boundaries were occasionally drawn around other areas of vegetation, particularly areas of dense vegetation such as dense mallee, pines or Belah. The extent of any vegetation boundary was recorded in the database (Table 3.1). Boundaries provided on the survey plans were used in defining polygons. Where no boundary was provided, vegetation boundaries were determined through examination of topographical features, and the area the surveyor appeared to be indicating on the survey plan.

Table 3.1. Vegetation boundary information from historical survey plans.

Complete	Almost complete	Incomplete	Absent
A complete polygon drawn on the survey plan enclosing the vegetation notes.	An almost complete polygon, indicating more than ¾ of the vegetation boundary.	Some indication of a vegetation boundary given.	No indication of vegetation boundary.

For some areas, the vegetation text was not specific to one vegetation community, for example, "Broken saltbush plains with belts of Mallee, Pines & Scrub", whereas in other areas the text was very specific, such as "Pine Country sparsely timbered". Where a distinct community appeared to be identified by the surveyor, the community was recorded in the database. The term mosaic was used to describe all areas where Mallee was listed with other tree species of Belah and Pine-Buloke woodlands (*M. platycarpum, C. gracilis,* or *Hakea* spp.), except where Mallee was listed near the end of a list of semi-arid woodland trees. It is not uncommon for a few scattered Mallee Eucalypts to occur in an area dominated by semi-arid woodland.

All areas where only woodland overstorey species were listed (*C. pauper, C. gracilis, Myoporum platycarpum, Hakea* spp., and *A. oleifolius*) were described as semi-arid woodland, due to the difficulties in distinguishing Ecological Vegetation Classes (EVCs), or separating any other sub-communities. *A. luehmannii* woodland areas were also separated where references to "Oak" and "Bulloak" were found.

Understorey vegetation was also described on some survey plans. Understorey was recorded as one of four categories in the GIS database:

(i) Shrubby – scrubs, scrubby, shrubs, shrubby, Hop bush or undergrowth noted on survey plans;
(ii) Grassy – references to grass or grassy vegetation on survey plans;
(iii) *Maireana* species – Bluebush noted on survey plans; and
(iv) *Atriplex* species – Saltbush noted on survey plans.

Descriptors describing tree density were included on some survey plans. Descriptors were interpreted to refer to either dense or open vegetation and this information was recorded in the GIS database. Terms thought to relate to dense vegetation included dense, close, thickly timbered and forest. Terms describing open vegetation included open, lightly timbered and sparsely timbered. A few vegetation notes were described as both dense and open, for example "Large Open Mallee Dense Pines and Bulloaks".

A GIS database was created to store information on each vegetation polygon including the original surveyors notes on the vegetation, soils and landforms and current scientific names, after the Flora of Victoria (Walsh and Entwisle 1994, 1996, 1999).

Four GIS map layers were derived from the historical maps and survey plans, each containing data as listed above. Each of the map layers was created from individual historical surveys undertaken at different times, and often for different purposes. The Mallee 1864 and Nowingi map layers were derived from survey plans located at the State Library of Victoria. The Parish Plan and Feature Survey map layers were compiled from survey plans located at Land Victoria (Appendix 2). The derived map data were stored in separate layers to allow analysis of each individual data source.

3.2.2 Pre-European distribution of semi-arid woodland

The distribution of semi-arid woodland species was compiled in a GIS database from the survey plans. Vegetation communities were determined from these species lists, and the total area of semi-arid woodland, Pine-Buloke woodland and semi-arid woodland mosaic was calculated. The area supporting each of the dominant tree species was also calculated.

3.2.2.1 Accuracy assessment of maps derived from survey plans

Accuracy of the map layers derived from the survey plans data was assessed by comparison with three main data sources:

(i) field survey of vegetation remnants;
(ii) Ecological Vegetation Class (EVC) mapping; and
(iii) Flora Information System (FIS), a geographic database of flora records in Victoria (NRE 2000a).

The majority of remnant vegetation is found within public land, which therefore forms the basis for much of the accuracy assessment.

A field survey was undertaken to compare maps derived from survey plan data with remnant vegetation in the study area. Vegetation notes from survey plans that contained reference to overstorey density were selected to allow comparison of current structure with historical structure of remnants. Quadrats were located where vegetation polygons from historical survey plans intersected with a known Park or Reserve at a point less than 200 m from roadsides. Where possible, quadrats were located near the centre of the polygon. Where vegetation at these sites was not representative of surrounding vegetation, the quadrat was moved no more than 150 m to the nearest representative vegetation. All perennial species were recorded and percentage cover estimated within a 20 x 50 m quadrat. A count was made of overstorey individuals (including stumps and dead trees) within the 20 x 50 m quadrat. Tree density descriptors on the survey plans were compared to the number and cover of trees ha^{-1} measured from field survey.

A soil sample from the top 10 cm was collected at each site, and a rapid assessment of soil type performed. Four major soil groups supporting semi-arid woodlands were identified from the literature and distinguishing features of each soil type were noted (Stace *et al.* 1968; Northcote *et al.* 1975; Badawy 1982). Three main discriminating features were identified from the literature (Table 3.2);

(i) A field effervescence of carbonate in fine earth assessment using two to three drops of 1-molar HCl was performed (McDonald and Isbell 1990). If the surface soil was found to be calcareous, the soil was assumed to be a calcareous earth;

(ii) The presence of a soil crust was investigated by digging lightly into the surface soil to observe if segregations occur. If a crust was identified it was assumed to be a duplex soil;

(iii) Soil was augured to a depth of at least 35 cm to determine if a bleached A2 horizon was present. If a bleached A2 horizon was found, the soil was assumed to be a sandy mottled duplex; and

(iv) If none of the above occurred, it was assumed to be brownish sand.

Table 3.2. Major soil groups supporting semi-arid woodland in north west Victoria and distinguishing features of each soil type.

	Calcareous earths	Red and yellow duplex	Sandy mottled yellow duplex	Brownish sands
Calcareous surface soil	Y	N	N	N
Thin surface crust	N	Y	N	N
Bleached A2 horizon	N	Possible	Y	N

Y=yes; N=no

The correlation between the map layers derived from survey plans and the EVC map layer was assessed by overlaying the two data sources. This allowed assessment of the area of correlation, and comparison of community classification.

Maps derived from survey plans were also assessed against the FIS (NRE 2000a). The most spatially accurate of the FIS quadrats, the 1-minute grid points, were selected within the study area to assess errors of omission and commission. Semi-arid woodland quadrats were identified from the FIS by selecting all quadrats containing semi-arid woodland overstorey species.

An error matrix was created to assess the accuracy of the semi-arid woodland map. The error matrix is created as a cross-tabulation, with column headings (by convention) representing the reference data (FIS), and rows representing the classification classes of the imagery or map.

Overall accuracy of the semi-arid woodland map was calculated by dividing the number of FIS quadrats correctly assigned to semi-arid woodland, or non-semi-arid woodland, by the total number of FIS quadrats assessed (Congalton 1991).

$$\text{Overall accuracy} = \frac{\text{Correctly assigned quadrats across all categories}}{\text{Total quadrats assessed}}$$

Producer's accuracy, or errors of omission were calculated by dividing the number of correctly assigned quadrats within a category, by the total number of quadrats measured within the category on the ground (Congalton 1991).

$$\text{Producer's accuracy} = \frac{\text{Correctly assigned quadrats within a category}}{\text{Total quadrats assessed within the category (column total)}}$$

Producer's accuracy shows how well notes from the survey plans can identify the total amount of semi-arid woodland on the ground, for example, a producer's accuracy of 80 % suggests that most areas of semi-arid woodland have been correctly identified.

User's accuracy or errors of commission were also calculated by dividing the total number of quadrats correctly assigned to a category on the map by the total number of quadrats that were assigned to the category on the map (row total).

$$\text{User's accuracy} = \frac{\text{Correctly assigned quadrats within a category}}{\text{Total quadrats assigned to the category from the map (row total)}}$$

The user's accuracy indicates the probability that a point on the map represents the correct category on the ground. The user's accuracy reveals how well the classified data relates to what is on the ground, for example, a user's accuracy of 80 % for semi-arid woodland suggests that when someone visits an area classified as semi-arid woodland, 80 % of the time they will find it is semi-arid woodland.

3.2.3 Community structure and composition

Community structure and composition of pre-European semi-arid woodland were investigated by examining the survey plans and reports of early 20th century botanists. Whilst these reports were made after the first impacts of European settlement, they represent some of our earliest records of semi-arid woodland in northwest Victoria.

The results from the historical data were compared to field data collected during the current study, and data from the FIS. Two hundred and fifty-nine semi-arid woodland quadrats were analysed within the study area. These consisted of 102 quadrats from the FIS, which were selected from the one-minute grid quadrats containing one or more semi-arid woodland overstorey species, and 157 quadrats undertaken during the current study (see Section 4.2.1 for description of methods).

All quadrats were filtered to exclude any quadrats containing *Eucalyptus* or *Halosarcia* species that may represent ecotones between communities, such as mallee shrubland or saline shrubland. High variability in rainfall, and the extended drought during the second half of this study had a large impact on the presence of annual and short-lived perennial species. To reduce the effects of seasonal climatic conditions on community definitions, annuals and short-lived perennials were excluded from the analysis, and presence-absence data were investigated.

A Multidimensional Scaling Semi-Strong-Hybrid (SSH) analysis was performed on the 259 quadrats using Pattern Analysis Package (PATN) to examine associations between quadrats and between species presence/absence in a sites x species matrix (Belbin 1995). The SSH was used to calculate the "distance" between species. An initial random configuration of points was accepted. SSH then calculates the distance between points, and a regression is performed to examine the degree of association between points. Estimates of the input association values are calculated to determine an estimate of stress. The process continues iteratively until a minimum value of stress is achieved (Belbin 1995). The results for species data were displayed as points mapped on x and y-axes.

A flexible Unweighted Pair-Group Method Average (UPGMA) or group average fusion with a beta value of -0.10 was undertaken based on Bray & Curtis Index values (Belbin 1995). An agglomerative hierarchical fusion strategy was performed using PATN to form associations of the quadrat data and species data. The results for quadrat and species data were displayed in dendrograms.

3.2.4 Estimation of vegetation clearance woodland since European settlement

Changes in distribution of semi-arid woodland in northwest Victoria were explored through comparison of survey plans with the Tree100 map layer indicating tree presence / absence as determined from satellite imagery in 1990 and 1993 (Appendix 1).

Areas where trees were mapped as absent (largely corresponding with cleared vegetation) were subtracted from the map layers generated from survey plans. The total area of original semi-arid woodland where trees are now absent was calculated. Whilst this does not equate precisely with vegetation clearance it gives a strong indication of the extent of clearing of semi-arid woodlands within the region.

The study area was divided into three areas; north of the Murray-Sunset National Park (MSNP), south of the MSNP, and the MSNP (Figure 3.1). Treeless areas were calculated within each of these three sections.

3.3 RESULTS

3.3.1 Survey plans and compilation of derived maps

Maps of semi-arid woodland derived from early survey plans

References to semi-arid woodland vegetation were identified on 63 survey plans of the "Put Away" series. References to semi-arid woodland vegetation were also found on the Mallee 1864, and Nowingi Line maps (Appendix 2). Survey plans and maps used in creating the derived maps covered 91.3 % (12,722 km^2) of the study area (Figure 3.2).

Figure 3.1. Division of the study area into three areas; north and south of Murray-Sunset National Park and the National Park area.

A total of 1,256 references to semi-arid woodland species were identified on all maps and survey plans. These references sometimes consisted of one word, being the common name used for the species, for example "Pines", however, sometimes relatively long descriptors were used including reference to soils and landforms such as "Undulating country with a few salt bush plains and sandy ridges, Red and brown loamy soil on flats, timbered with large mallee with some pines". Figure 3.3 shows a section of the survey plan for the parish of Murrnroong with references to semi-arid woodland species.

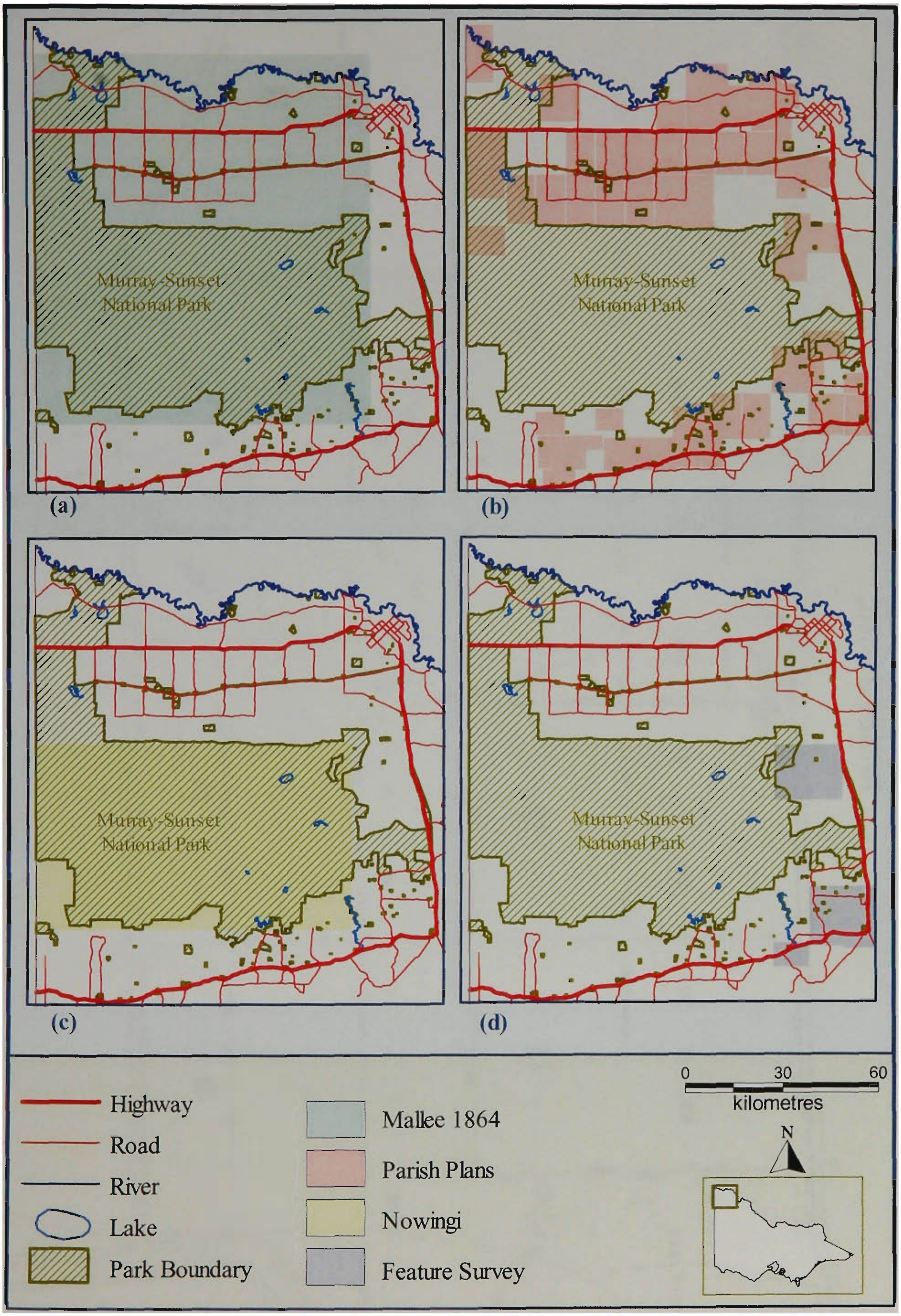


Figure 3.2. Coverage of survey plans containing *Casuarina pauper* woodland species. (a) Mallee 1864 (b) Parish Plans (c) Nowingi (d) Feature Survey.

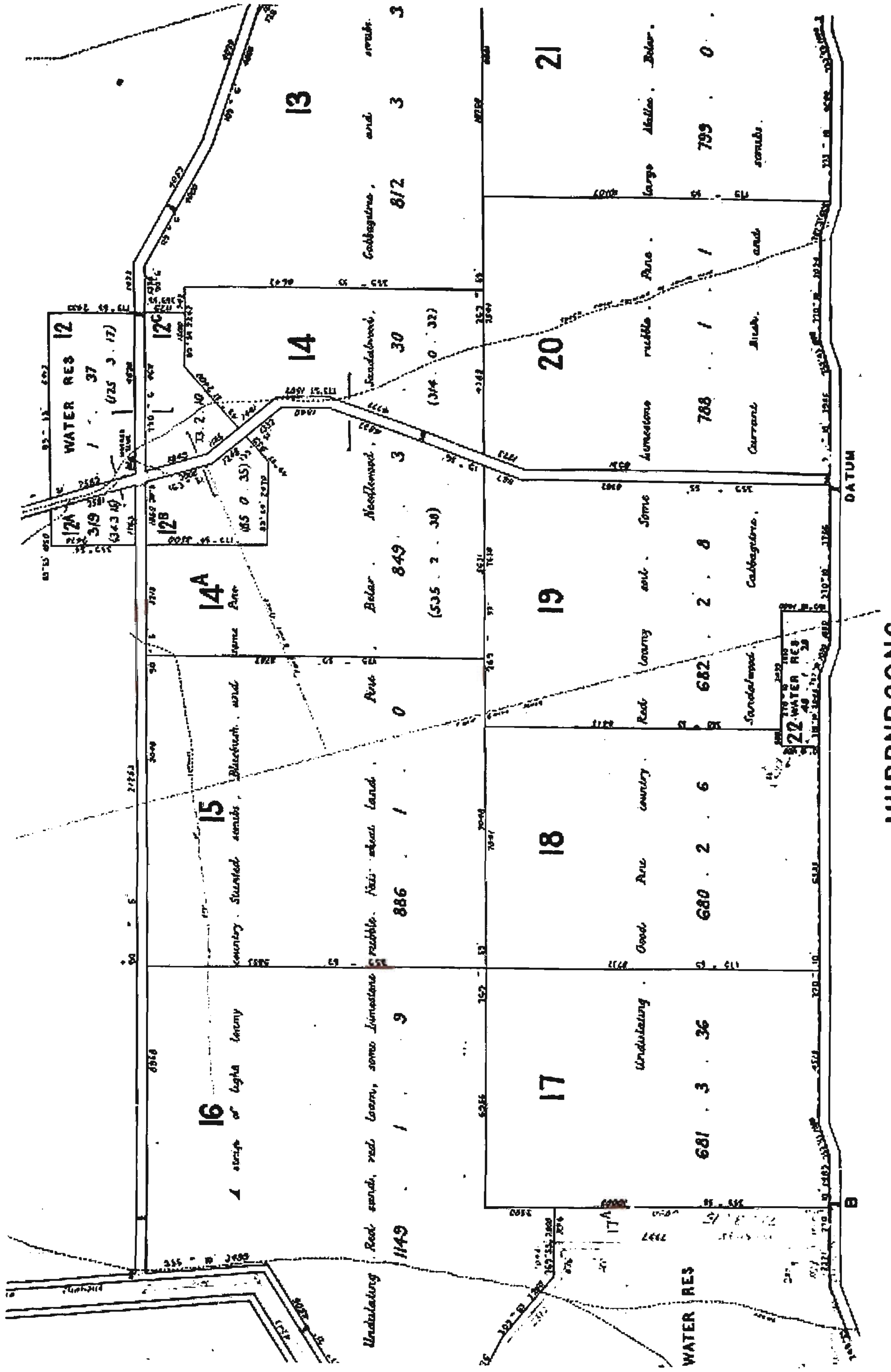


Figure 3.3. **Section of Parish plan for Murrnroong showing typical notes on vegetation and soils.**

Map scale ranged from 1:3,200 to 1:130,000 with a scale of twenty chains to one inch (1:15,840) commonly used. Surveys used to create the derived maps were undertaken from 1864 to 1935 with the mean date of surveys 1916 (Table 3.3), although for some surveys the precise date of survey was difficult to determine.

Table 3.3. Detail of survey plans from which maps were derived, showing mean map scale, date compiled and registration error.

Derived map layer	Mean map scale	Mean date compiled	Mean registration error (pixels)
Parish plan	1:18,100	1919	6.2
Nowingi	1:126,700	1921	7.9
Mallee 1864	Approx. 1:130,000	1864	11.6
Feature survey	1:15,800	1916	12.4

Registration errors were calculated in MapInfo (MapInfo Corporation) by determining the relative positions of the control points, based on the map coordinates used. MapInfo (MapInfo Corporation) then determines where each control points should be relative to other control points entered. Generally, registration of the images was successful with registration errors in MapInfo (MapInfo Corporation) calculated at less than 15 pixels. The Mallee 1864 and Feature survey maps showed the highest registration errors due to the age of the maps and the change in roads since the time of these surveys (Table 3.3).

Vegetation boundaries were indicated on many survey plans, however they were almost entirely absent on the Mallee 1864 and Nowingi map layers (Table 3.4).

Table 3.4. Information provided on survey plans indicating vegetation boundaries.

	No. references to semi-arid woodland vegetation			
Map layer	Boundaries absent	Boundaries complete	Boundaries incomplete	Boundaries almost complete
Parish Plan	447	59	82	16
Feature Survey	150	10	49	
Mallee 1864	39			
Nowingi	403			1
Total no. references	1039	69	131	17

3.3.2 Distribution of semi-arid woodland

Areas containing semi-arid woodland species were identified from the historical survey plans across much of the study area. Areas where semi-arid woodland were not mapped, or were sparsely mapped include the southwest corner of the study area, the centre of the MSNP and the area north of the Sturt Highway (Figure 3.4). There is a concentration of semi-arid woodland references around Mallanbool and Yarrara Flora and Fauna reserves (Figure 3.4). Few areas were mapped as Buloke from the historical survey plans and these occur generally to the south and the east of the study area. Table 3.5 indicates the area of semi-arid woodland, Mosaic, Pine-Buloke and combined Buloke and Belah mapped from the survey plans.

Table 3.5. Area (ha) of semi-arid woodland mapped within GIS map layers derived from survey plans.

Community	Parish Plan	Nowingi	Feature Survey	Mallee 1864
Semi-arid woodland	64,263	28,481	2,756	2,665
Semi-arid woodland Mosaic	172,538	8,926	1,474	364
Pine-Buloke woodland	616	-	202	-
Belah and Buloke	160	-	104	-

3.3.2.1 Accuracy assessment

Comparison with Ecological Vegetation Class (EVC) data

Seventy-six percent of all Belah Woodland EVC occurred within the area mapped as semi-arid woodland and mosaic from the survey plans. The area of intersection of derived maps with EVC mapping is shown in Table 3.6, indicating errors of omission (areas of semi-arid woodlands not classified) and commission (other communities classified as semi-arid woodland). This shows that errors of commission are relatively low, with less than 10 % of any community misclassified as semi-arid woodland, with the exception of Alluvial Plain Shrubland, and areas that are cleared or severely disturbed. Areas of omission are higher for woodland EVCs other than Belah Woodland.

Table 3.6. Area (ha) of overlay of EVCs with map layers derived from survey plans.

EVC within historical polygons	Total area EVC	Area classified semi-arid woodland	% classified semi-arid woodland
Alluvial Plain Shrubland	17,597	2,943	16.7
Belah Woodland	**4,073**	**3,095**	**76.0**
Black Box-Chenopod Woodland	17,908	196	1.1
Broombush Mallee	11,920	77	0.7
Chenopod	151,901	15,185	10.0
Cleared/Severely Disturbed	2,872	1,122	39.1
Gypseous Plain Woodland	**5,460**	**1,948**	**35.7**
Lowan Sands Mallee	156,101	1,347	0.9
Pine-Buloke Woodland	**5,272**	**880**	**16.7**
Private Land	1,468	21	1.5
Red-Swale Mallee	7,000	176	2.5
Saline Shrubland	28,011	2,256	8.1
Sandplain Grassland	**2,423**	**209**	**8.6**
Savannah Woodland/Savannah Mallee/Grassland Mosaic	**101,803**	**24,229**	**23.8**
Unknown/Unclassified	1,149	24	2.1
Water Body-Fresh	181	2	0.9
Woorinen Sands Mallee	254,230	9,598	3.8

Bold – Semi-arid woodland communities

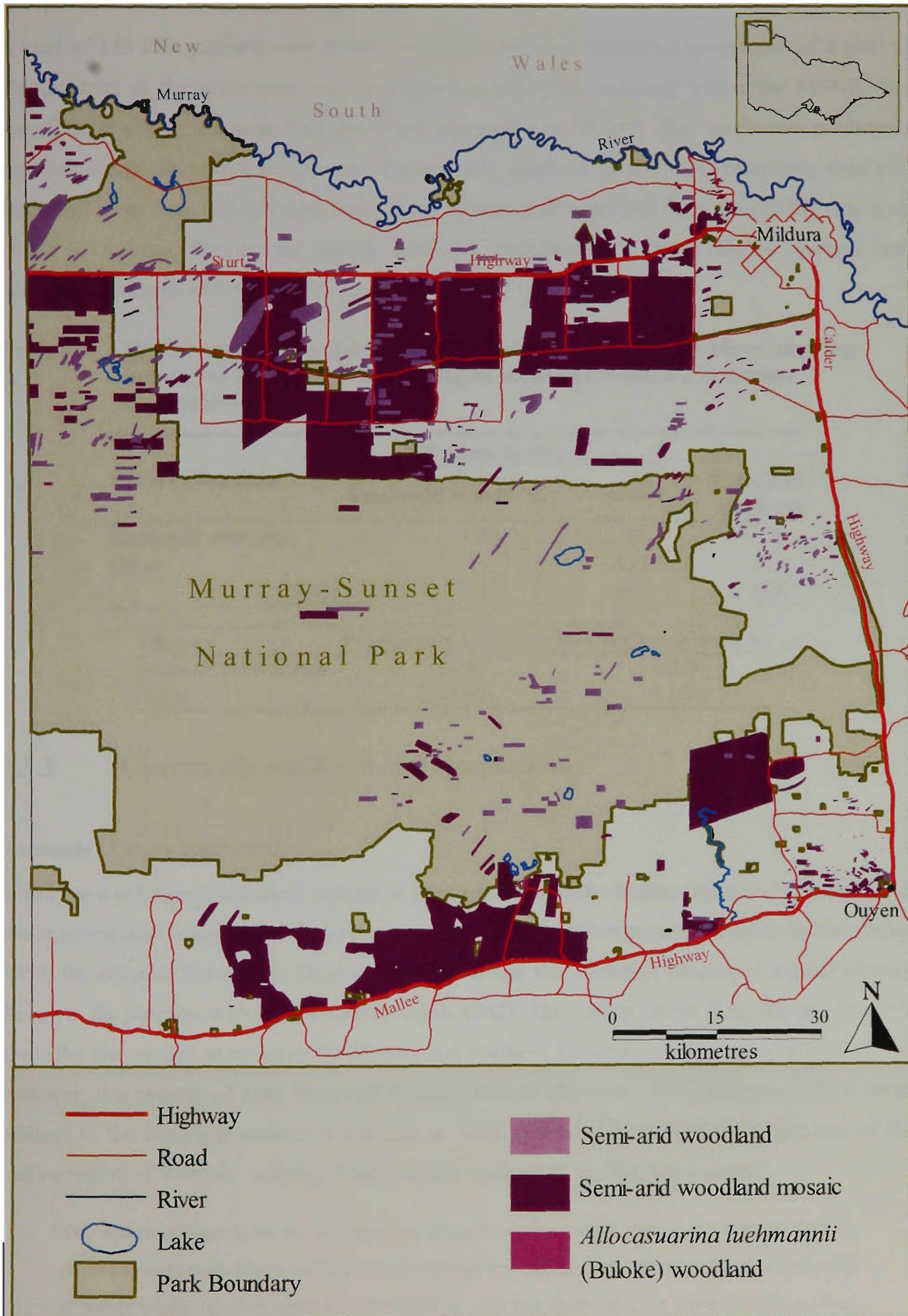


Figure 3.4. Distribution of semi-arid woodland, semi-arid woodland mosaic and *Allocasuarina luehmannii* woodland from all derived map layers.

Comparison with Flora Information System (FIS) data

A total of 143 FIS quadrats were found to contain semi-arid woodland species out of a total of 606 quadrats in the study area, with 446 quadrats (73.6 %) occurring within the MSNP. The overall accuracy of the semi-arid woodland mapping was 78.9 %. The user's and producer's accuracy values show that whilst some success was achieved in correctly classifying semi-arid woodland areas from survey plans, many areas of semi-arid woodland were not classified as such (Table 3.7). Areas that do not contain semi-arid woodland were more reliably mapped, and accounted for much of the overall accuracy.

Table 3.7. Error matrix for semi-arid woodland mapped from survey plans, showing number of FIS quadrats occurring in semi-arid woodland and other communities.

Survey plan data	FIS quadrat data		Total FIS quadrats
	Semi-arid woodland	Other	
Semi-arid woodland	45	30	75
Other	98	433	531
Total	143	463	606

Class	Producer's accuracy %	User's accuracy %
Semi-arid woodland	31.47	60.00
Other	93.52	81.54

3.3.3 Community structure and composition

Accounts of early vegetation

There are a number of historical reports on the vegetation of the Mallee that give information on the structure and composition of semi-arid woodland before European settlement. In November 1853, the botanist Baron Von Mueller travelled across the Victorian Mallee, along the Murray River, to the junction with the Darling (Kenyon 1982). The curator of the Botanical gardens, Mr Dallachy also visited areas along the Murray and southern Victorian Mallee between 1858–1860, however, few records of their botanical findings remain (Kenyon 1982). Kenyon (1982), in an address to the historical society of Victoria in 1912, gave this picture of the vegetation of the mallee region of Victoria, referring to semi-arid woodland as the "broken country".

> The eleven million acres are not covered with dense low scrub, nor is the country flat, as popularly supposed. There are four general divisions – the scrub country, generally consisting of sandy ridges running easterly and westerly, with red loam flats, all covered with mallee, from 8 to 20 feet in height, probably 50 per cent. of the whole; the broken country, consisting of open grassy plains, pine and belar ridges, big mallee up to 60 feet in height and all sorts of smaller trees such as quandongs, pittosporums, hakeas, grevilleas, heterodenrums,

> myoporums, eremophilas, acacias, &c., mostly peculiar to the area as far as Victoria is concerned; the heath country of infertile white sands, generally in steep blown hills, covered with epacridae and sandy forms of vegetation and stunted forms of mallee; the frontage country to the Murray River, with its large box and red gum flats and bordering pine and hop bush country. [sic] (Kenyon 1982).

By the 1930s, more detailed botanical surveys were made of the region, with species lists and frequency values compiled for "vegetation type 2 – Pine-Belar-Buloke" suggesting that Zimmer (1937) did not find consistent sub-communities. Whilst much clearing and modification of the environment had occurred by this stage, the work of Zimmer (1937) provides a more precise botanical description of semi-arid woodland at the time of European settlement.

> Pine-Belar-Buloke areas; ... comprises extensive areas carrying principally *Callitris* spp. and *Casuarina* spp. with scattered *Myoporum platycarpum*. There is a lower canopy of various small trees and shrubs. Mixed with the Pine-Belar-Buloke belts, patches of Tall Mallee (*Eucalyptus oleosa-Eucalyptus gracilis-Eucalyptus incrassata*) occur.... The chief species of undergrowth to Pine-Belar-Buloke vegetation are *Pittosporum phillyreoides, Fusanus acuminatus*, *Hakea vittata*, *Hakea leucoptera*, *Eremophila oppositifolia*, *Eremophila maculata*, *Eremophila glabra*, *Exocarpos aphylla*, *Grevillea Huegelii*, *Beyeria Leschenaultii*, *Templetonia egena*, *Acacia colletioides*, *Acacia Oswaldii*, *Acacia ligulata*, *Acacia obliqua*, *Acacia sclerophylla*, *Heterodendron oleifolium*, *Scaevola spinescens*, *Olearia Muelleri*. In this area, Pine (chiefly *Callitris robusta*) and Belar (*Casuarina lepidophloia*) do not usually appear as species mixed in the same area, but form stands solely of the one genus – that is to say, they mingle only where the change from one species to another takes place...Belar patches generally do not carry the variety of smaller plants that are usually met with under *Callitris*. The absence of grasses is particularly marked and no doubt this is due to the enormous amount of surface-feeding roots put out by the *Casuarina lepidophloia* [sic] (Zimmer 1937).

Only three of the species recorded by Zimmer appear to have increased in frequency in the study area, with 24 species showing a decline in reported frequency (Appendix 4). See Appendix 5 for current scientific names of species listed in historical descriptions and survey plans.

Later, in the early 1950s, Patton (1951) writes of the Pine-Belar association in the Victorian Mallee;

> Sometimes these two trees form a mixed forest and have as an understorey a large number of other trees, *Eremophila oppositifolia*, *Fusanus acuminatus*, *Hakea leucoptera*, *Heterodendron oleifolium*, *Myoporum platycarpum* and *Pittosporum phillyreoides*. At other times, each of the dominants forms a forest by itself, but generally the pine is on the highest sandiest red soil, while belar seeks the heavier lower elevations. The development of the shrub layer is very irregular and does not appear to be wholly controlled by the density of the

> crown canopy. Among the most interesting are the spinescent *Acacia colletioides* and *Scaevola spinescens*, the former being very conspicuous when covered with the very bright flowers of *Loranthus preissii*. *Olearia pimeleoides* ... is very striking when in bloom. Both species of Cassia are present and are very attractive. The number of species of annuals is not great, as is commonly the case in forests, but the floor of a pure pine forest is most attractive in a good year when carpeted with masses of small annuals, mostly species of Compositae. [sic] (Patton 1951)

Numerous accounts were made of Pine ridges amongst otherwise seemingly drab scrub; "Here and there the monotony of the scrub was broken by groups of fine pine-trees" referring to country near Tiega – 28 miles from Mournpall (Neumayer 1858-64 cited in Kenyon 1982). And, "In the Millewa certain tracts of loamier soils originally bore pine vegetation" (Holt 1947).

These descriptions all point towards the diversity of semi-arid woodland. There are a number of tree species present and many shrubs and annual species mentioned. There is significant variety between woodland patches, with differences in species dominance and structure noted.

Vegetation density

Forty-two references to semi-arid woodland vegetation density on the survey plans were found to intersect with current parks and reserves. Fourteen of these sites were recorded as dense and 28 recorded as open. Areas where fewer than 50 trees ha^{-1} were found (including dead trees and stumps) were reliably referred to as "open" by the early surveyors, whereas areas with greater than 300 trees ha^{-1} were described as "dense" (Figure 3.5). Tree canopy cover estimates in the field did not appear to align with descriptions of vegetation density used by the early surveyors (Table 3.8).

Table 3.8. Percentage of field survey quadrats classified as dense or open compared with current canopy cover classes (Walker and Hopkins 1990).

Canopy cover categories	% Cover	Open	Dense
Very sparse (open woodland)	<10	82.1	28.6
Sparse (woodland)	11–30	14.3	64.3
Open forest	31–46		7.1

References to vegetation density occurred on just over 20 % of vegetation notes, of these open vegetation was recorded nearly three times more than dense vegetation (Table 3.9). Dense woodland is recorded around the Yarrara, Mallanbool and Meringur Flora and Fauna reserves, with other smaller areas scattered throughout the study area (Figure 3.6).

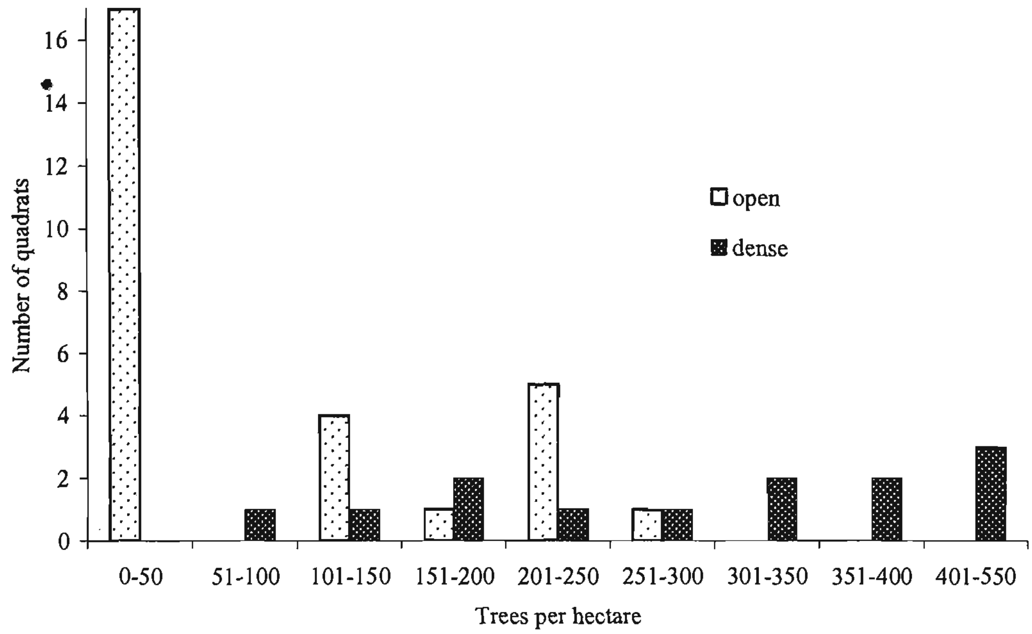


Figure 3.5. Percentage of quadrats described as open and dense by early surveyors, compared with current tree density.

Table 3.9. Frequency of occurrence of notes relating to vegetation density on survey plans.

Vegetation density	No. references	% total vegetation references from survey plans
Total	261	20.8
Open	195	15.5
Dense	69	5.5

Overstorey species listed on the survey plans were compared with species found on field survey. At 50 % of quadrats, all of the overstorey species listed on the survey plans were found during field survey. At least one of the species listed in the survey plans was recorded on all but one of the field survey quadrats (Table 3.10).

Table 3.10. Percentage correlation of tree species from survey plans with field survey observations.

% Correlation of overstorey species on survey plans with field survey	% quadrats
0	2.4
1–20	0
21–40	4.8
41–60	35.7
60–80	7.1
100	50.0

At 79.4 % of field sites where remnant trees were present, the first listed species in the derived maps was the dominant species located on the ground when stumps and dead trees were taken into consideration. This dropped marginally to 73.5 % when only living trees were considered.

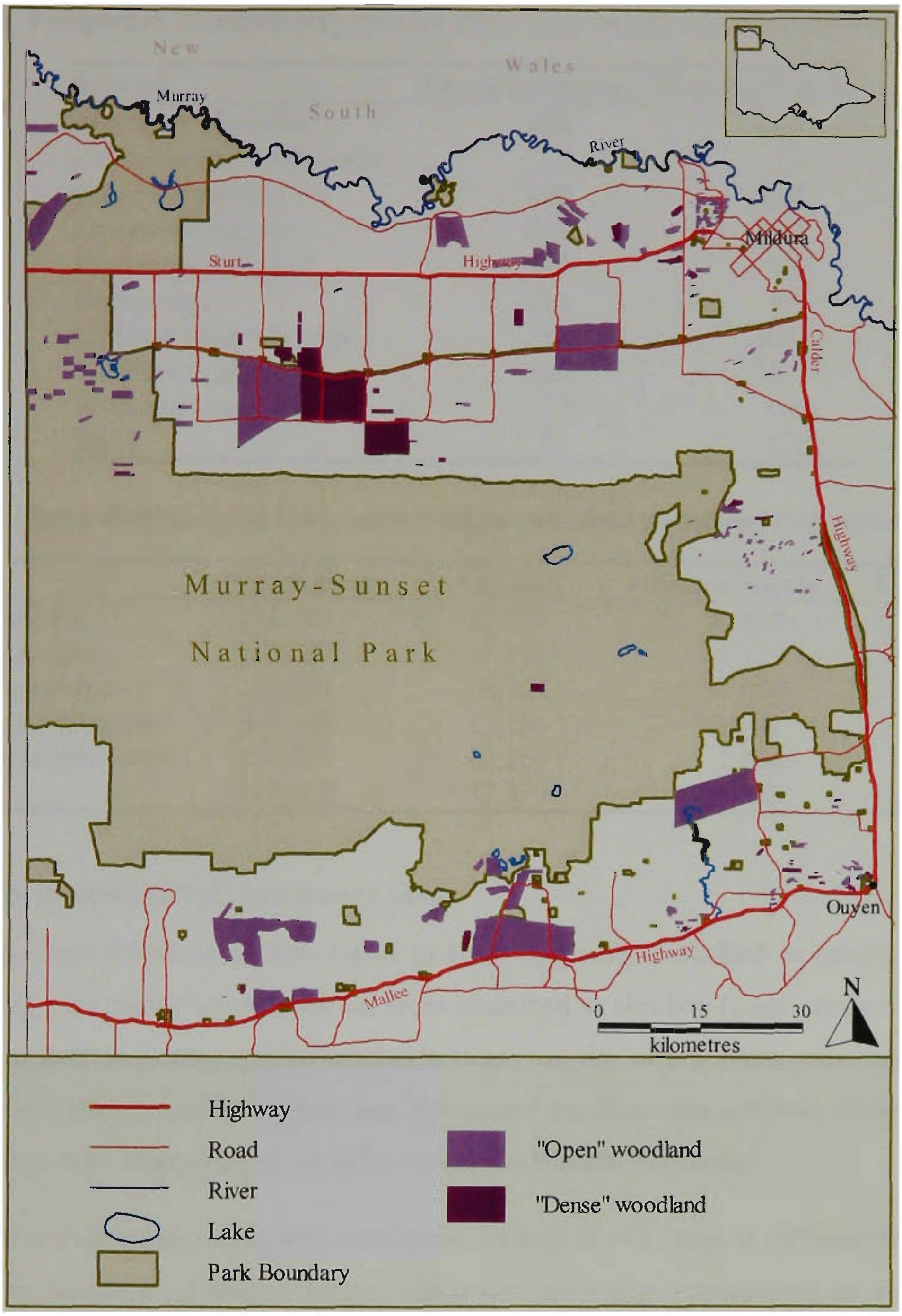


Figure 3.6. Map of pre-European semi-arid woodland tree density based on descriptive measures from survey plans.

In five out of the 42 (11.9 %) field survey quadrats, species mentioned on the derived map layer were represented only by stumps or dead trees in the field. *C. gracilis* was represented only by dead trees or stumps at four sites, with *M. platycarpum* only represented by dead trees at one site.

Overstorey species recorded on survey plans

C. gracilis was the most frequently recorded species on the semi-arid woodland survey notes, followed by *M. platycarpum* and Mallee *Eucalyptus* spp. (Table 3.11). A similar pattern is observed for the area mapped from the survey plans for each species (Table 3.12).

Table 3.11. Frequency of overstorey species references on the historical survey plans.

Species	No. of references	% of references
Alectryon oleifolius	209	16.5
Allocasuarina luehmannii	27	2.1
Callitris gracilis	830	65.7
Casuarina pauper	299	23.7
Exocarpos aphyllus	45	3.6
Hakea spp.	125	9.9
Mallee *Eucalyptus* spp.	391	30.9
Myoporum platycarpum	396	31.3
Santalum acuminatum	7	0.6
Myall	21	1.7

Table 3.12. Area (ha) mapped from survey plans containing semi-arid woodland species.

Species	Parish Plan	Nowingi	Feature Survey	Mallee 1864
Callitris gracilis	213,163	22,027	4,412	2,483
Casuarina pauper	145,623	11,169	1,119	1,241
Alectryon oleifolius	116,827	8,726	1,658	273
Mallee *Eucalyptus* spp.	205,102	13,186	1,991	502
Myoporum platycarpum	137,984	12,169	1,006	298
Total	237,962	37,379	5,132	3,028

Understorey species recorded on survey plans

On average, lower shrub cover was found in areas originally described as grassy (mean shrub cover 18.8 %), and higher shrub cover on areas described as shrubby (mean shrub cover 27.5 %). In areas described as having a bluebush understorey on the survey plans, only one site visited during the field survey had an understorey dominated by *Maireana sedifolia* (bluebush), whilst remaining sites were dominated by *Atriplex vesicaria* (Bladder Saltbush).

From the soil survey of semi-arid woodland sites, 36 were found to occur on brownish sands, nine were found to occur on calcareous earths, whilst the remaining site occurred on a sandy duplex soil. All references to dense and shrubby vegetation occurred on brownish sands.

Understorey vegetation was recorded much less frequently than overstorey with only 162 (12.8 %) of references indicating understorey species. Woodlands were more frequently recorded with a shrubby understorey, closely followed by bluebush and grassy understoreys (Table 3.13). From the quadrats surveyed in the field, *Enchylaena tomentosa* and *Atriplex vesicaria* were the most frequently occurring understorey species (Table 3.14).

Table 3.13. Frequency of references to understorey types in the survey plans.

Understorey	No. references	%
Bluebush	45	3.6
Grassy	42	3.3
Saltbush	10	0.8
Shrubby	65	5.1
Total	162	12.8

Table 3.14. Most frequently occurring shrub species recorded during field surveys.

Species	% Frequency
Enchylaena tomentosa	93.6
Atriplex vesicaria	63.1
Sclerolaena obliquicuspis	56.1
Sclerolaena diacantha	54.8
Maireana brevifolia	43.9
Chenopodium desertorum	33.8

3.3.3.1 Semi-arid woodland community analysis

Five large groups and two clusters of smaller groups were identified from the fusion analysis dendrogram when analysed with a stress value of 1.0 (Figure 3.7). Species lists, species frequency, and number of quadrats appearing in these groups are listed in Appendix 6.

Whilst weak community associations were found within semi-arid woodlands, species lists show that these groups are not distinguishable based on overstorey species or other character species. All five of the major overstorey species were found in four of the seven groups with two exceptions; (i) *A. luehmannii* did not occur in the two groups where *M. platycarpum* was the most frequently occurring tree species; and (ii) group three, a species depleted community characterised by *C. gracilis* and grazing tolerant shrubs with occasional *M. platycarpum*, but no other overstorey species (Appendix 6).

Group 1 consists of *C. pauper* woodland with species richness higher than in any other group. The list of species present is similar to that of quadrats in Group 2, all of which contain *C. gracilis* and are relatively species rich. Group 3 appears very similar to Group 2, also characterised by *C. gracilis* but with only half the species richness. The overstorey of Group 4 is characterised by *A. oleifolius* and *C. pauper*, however all other major overstorey species are also represented. Group 5 consists of *C. gracilis–A. luehmannii* woodland with grazing tolerant shrubs. Groups 6 and 7 are characterised by *M. platycarpum* in the overstorey with low species richness and a number of species commonly occurring on calcareous or clay soils.

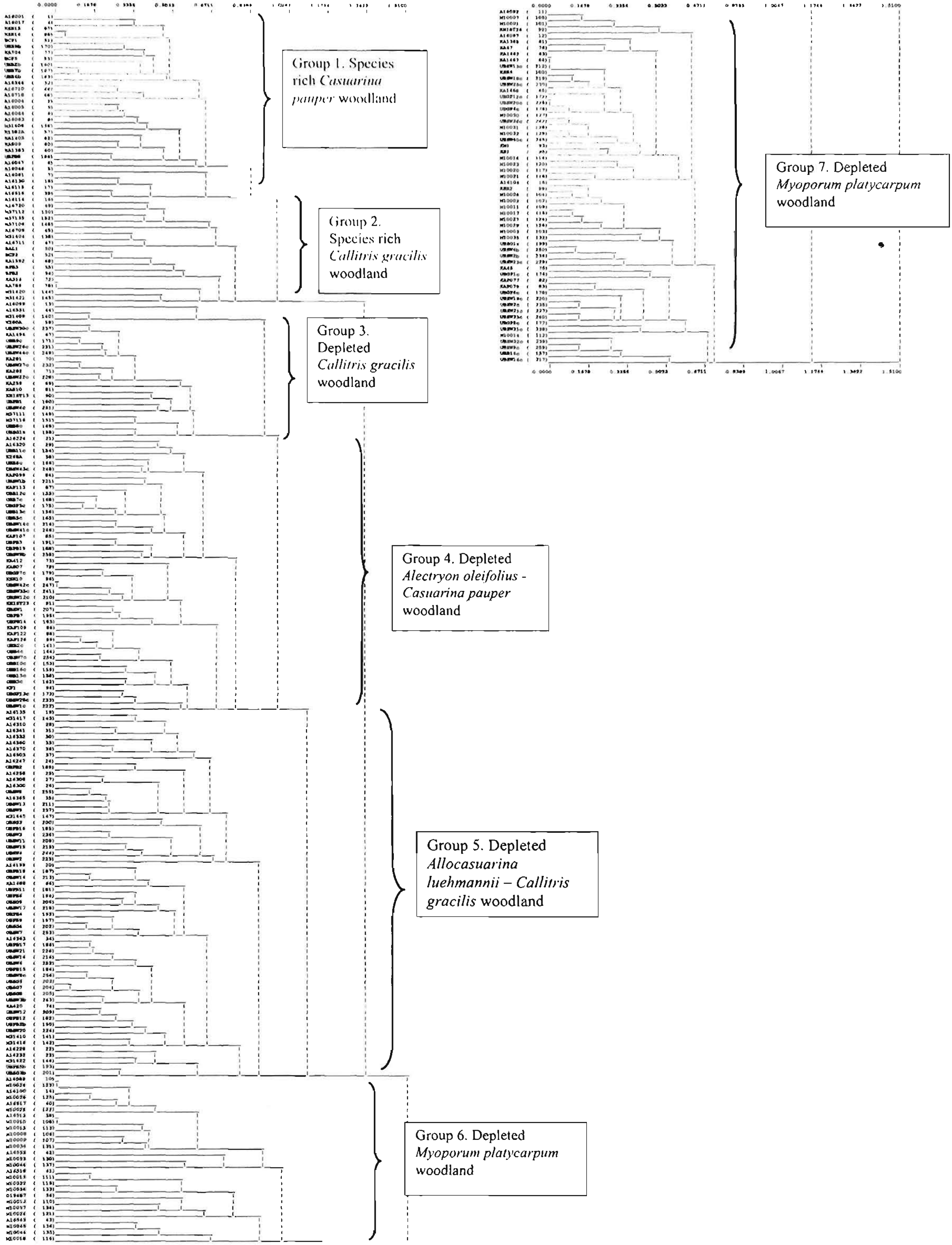


Figure 3.7. Dendrogram based on fusion analysis of 259 quadrats of perennial species in semi-arid woodland in northwest Victoria.

By combining these groups, a situation that is more reflective of what is observed in the field can be obtained. Groups 1–5 appear to represent what was originally a species rich shrubby community with an overstorey dominated by *C. pauper* or *C. gracilis.* A number of other trees were probably also present, along with tall and small shrubs. To the southeast of the study area *C. pauper* is replaced by *A. luehmannii.* In the northwest of the study area *M. platycarpum* dominates on calcareous and heavier soils with the understorey characterised by grazing tolerant small shrubs and fewer tall shrubs.

The groups formed from the fusion analysis based on species did not appear to relate to any observed communities or any environmental gradients (Figure 3.8). This supports the results from the fusion analysis of quadrats that community associations in the semi-arid woodlands are at best weak (Figure 3.7).

3.3.3.2 Multidimensional Scaling Semi-Strong-Hybrid (SSH) analysis

The SSH analysis on species also suggests there is no obvious separation of communities based on overstorey species. However, a stress value of 0.29 suggests that results should be interpreted with caution (Belbin 1995). All of the main overstorey species appear close together near the centre of the X and Y axes (Figure 3.9). Around the base of the cluster are a few species that may occur on heavier clay or calcareous soils such as *Maireana sedifolia, Lawrencia squamata, Nitraria billardierei* and *Frankenia foliosa*, however, they are interspersed with species occurring on deeper sands such as *Callitris verrucosa* and *Acacia ligulata.* No reliable environmental gradients or community associations were detected from the SSH analysis on species.

3.3.4 Extent of change in the semi-arid woodland since European settlement

Estimation of the extent of vegetation clearance

The area mapped from the historical map layers as containing semi-arid woodland is significantly greater to the north of the MSNP (Figure 3.10). The total area of treeless vegetation within each area was calculated from the Tree100 map layer, and shows a higher rate of clearance within semi-arid woodland than surrounding vegetation (Table 3.16). High rates of vegetation clearance also appear to occur within the MSNP.

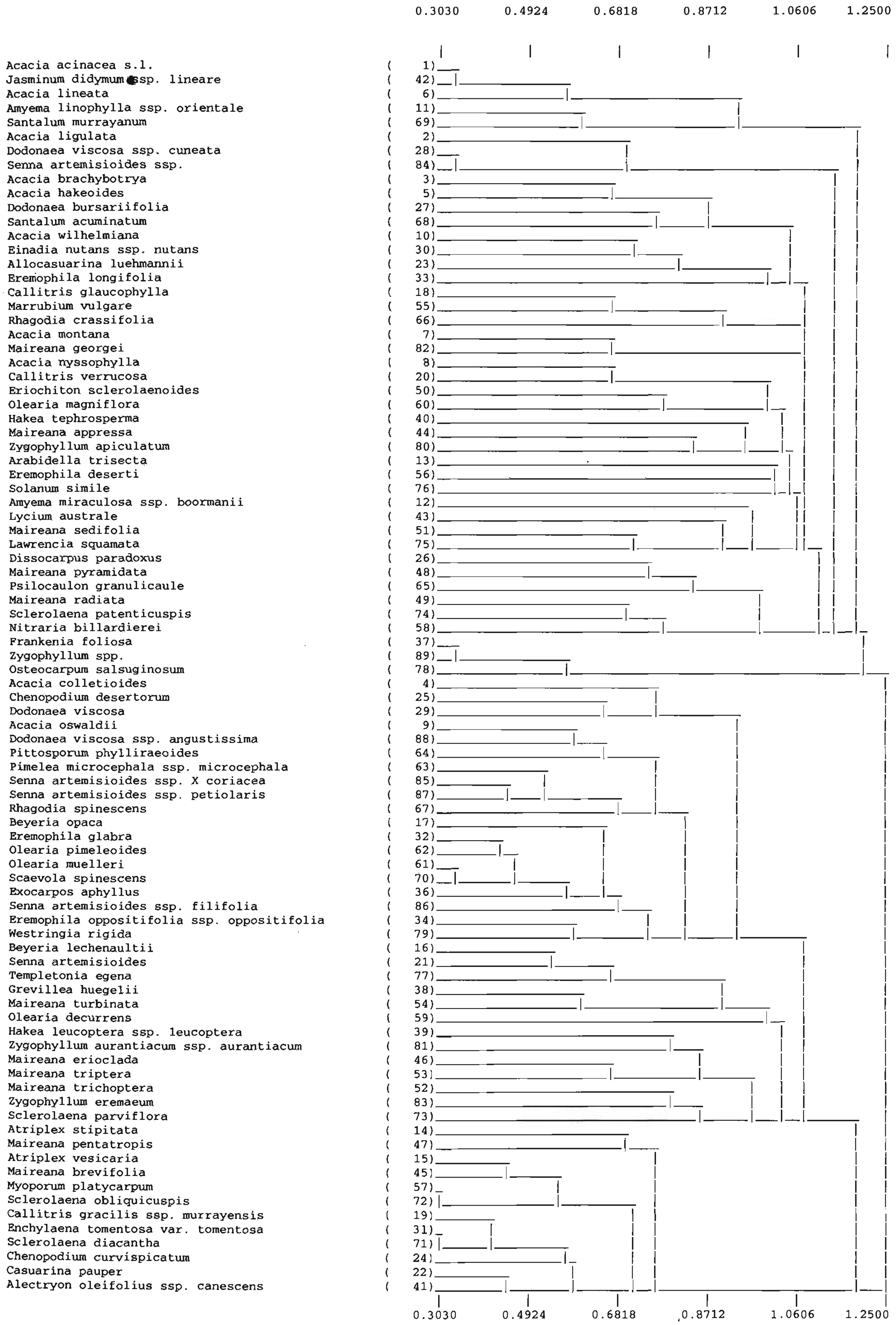


Figure 3.8. Dendrogram based on fusion analysis of perennial species of semi-arid woodlands in northwest Victoria.

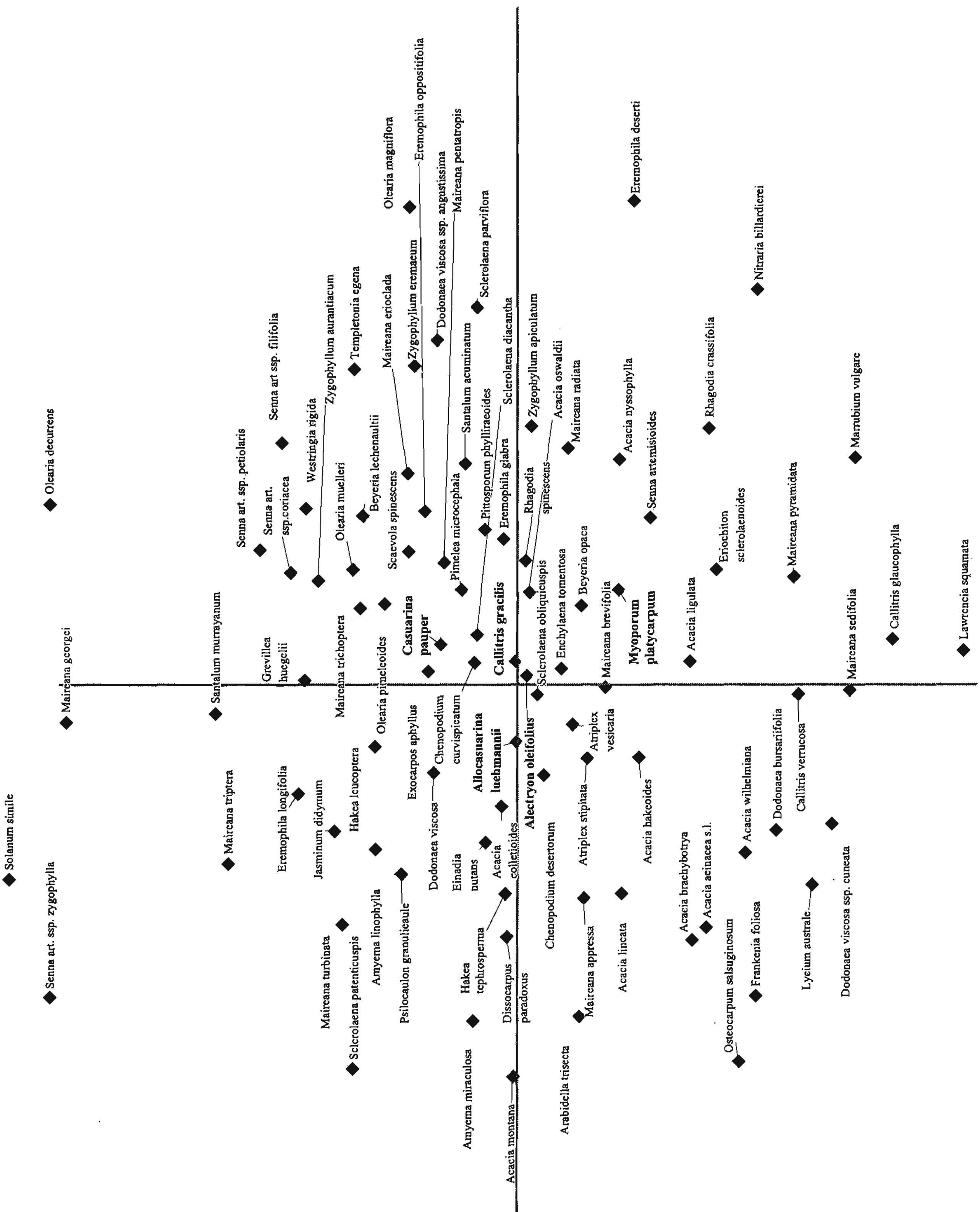


Figure 3.9. Multidimensional Scaling Semi-Strong-Hybrid (SSH) analysis of perennial species of semi-arid woodlands in northwest Victoria

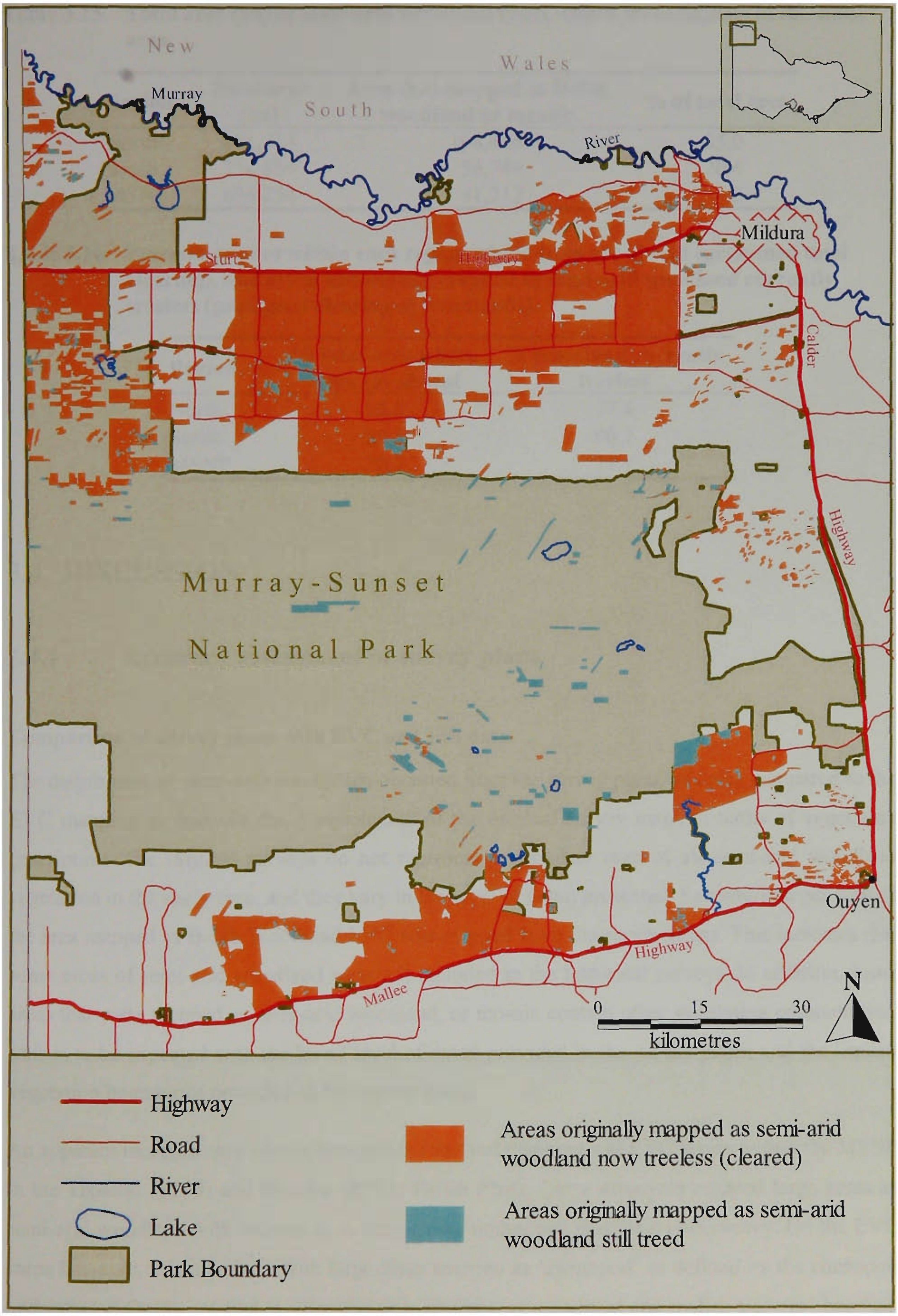


Figure 3.10. Extent of clearing of semi-arid woodland calculated from Tree100 data and historical survey plans.

Table 3.15. Total area (ha) of semi-arid woodland types within three regions of the study area.

Region	Total area (ha)	Area (ha) mapped as Belah woodland or mosaic	% of total area
North	481,714	168,446	35.0
South	274,854	56,749	20.6
MSNP	636,254	41,212	6.5

Table 3.16. Percentage area within each region where trees are absent (presumed total clearing), and areas historically mapped as semi-arid woodland currently treeless (presumed clearing of woodlands).

Region	% total area where trees are absent	% woodland currently treeless
North	93.6	97.4
South	89.8	96.7
MSNP	20.7	71.1

3.4 DISCUSSION

3.4.1 Accuracy assessment of survey plans

Comparison of survey plans with EVC and FIS data

The distribution of semi-arid woodlands obtained from the survey plans was also compared to the EVC mapping to measure the completeness of the original survey maps in terms of vegetation description. The original surveys do not represent a complete map of all semi-arid woodland vegetation in the study area, and they vary in the level of detail presented. Seventy-five per cent of the area mapped as Belah Woodland EVC was mapped from the survey plans. This indicates that some areas of semi-arid woodland were not included in the historical surveys. In addition, some areas that were mapped as semi-arid woodland, or mosaic contain other vegetation communities. This is to be expected with the broad level of detail provided in the survey plans, and the limited vegetation boundaries provided on the survey plans.

An apparent inconsistency occurs between the derived mapping and EVC layers within the MSNP in the Taparoo (T309) and Boorlee (B791) Parish Plans. Early surveyors mapped large areas as semi-arid woodland with references to very sparse timber and Bluebush understorey. On the EVC maps however, this intersects with large areas mapped as 'chenopod' as defined by the chenopod understorey. On the ground, a very sparse overstorey of woodland and mallee trees is observed, often with less than 1 tree ha^{-1}. It appears this is due to a difference in classification techniques with early surveyors more focused on the overstorey component of the vegetation. More recent EVC mapping has classified these areas according to the dominant vegetation layer, the shrub layer. It is unlikely that this apparent discrepancy is due to any major change in the vegetation since the time of the early surveys, as there are few dead trees or stumps in this area.

EVC mapping was undertaken within the public land area (mostly consisting of the MSNP), whilst the greater efforts of the original surveys were in the land surrounding the MSNP. This is likely to be due to the relatively low agricultural value of the land now within the MSNP, which is associated with a lesser amount of semi-arid woodland within this area.

Overall accuracy calculated from the Flora Information System (FIS) quadrats was 78.9 %, however, this is largely due to the low rate of errors of commission. Despite 606 FIS quadrats being located within the study area, less than 7 % of the quadrats that did not contain semi-arid woodland species occurred within areas mapped as semi-arid woodland or woodland mosaic. However, errors of omission, calculated from the FIS data were quite high, with a number of areas containing semi-arid woodland not noted on the survey plans within public land.

EVC mapping is currently only available for public land, as are the majority of FIS quadrats, in contrast to the survey plans that provided greater detail within private land areas. Errors within the EVC mapping, which relied largely on aerial photo interpretation was a limitation with comparisons of survey plan data to remnant vegetation. It is also possible that change during the intervening 70–140 years may have altered vegetation, rendering comparisons invalid.

Despite these limitations, remnant vegetation provides us with one of the best verification of records from the early survey plans, and has been used in previous studies to compare with early survey data (Oxley 1987; Fensham and Holman 1998).

3.4.2 Structure and composition of semi-arid woodland in northwest Victoria

Overstorey

Descriptions of dense and open semi-arid woodland vegetation from the survey plans did not reliably describe current day tree density of remnants. The best separation of the categories for dense and open vegetation was measured when stumps and dead trees were included.

Areas where fewer than 50 trees ha^{-1} including mature and dead trees and stumps were measured were consistently recorded as open in the original surveys. The early surveyors consistently recorded areas with more than 300 trees ha^{-1} as dense. However, areas of between 51–300 trees ha^{-1} were described as both dense and open. This may be due to inconsistencies in how early surveyors used these measures, or it may be related to loss of evidence of some trees since the original surveys.

Previous studies have indicated that little overstorey regeneration has occurred within the study area since European settlement (Westbrooke *et al.* 1988; Cheal 1993), and so it is likely that the vast majority of mature trees recorded during the recent field survey would have established during or prior to the wet years of the 1870s and 1889/90 (Westbrooke 1998). Evidence of stumps and dead trees is likely to remain, as these semi-arid woodlands rarely contain enough fuel to burn, *C. gracilis* timber is termite resistant, and *C. pauper, A. luehmannii* and *C. gracilis* are all durable timbers (Cunningham *et al.* 1981; Doran and Turnbull 1997). Therefore, by including all mature trees, dead trees and stumps the best measure of original tree density (as observed by the surveyors) is obtained. It is probable, however, that in the intervening 70 to 140 years since the original surveys, some evidence of original tree density will have been lost.

The correlation of surveyors' descriptions with current tree densities indicates that the early surveyors were using these terms in much the same way as we would today. Fensham and Holman (1998) also found tree density descriptors reliable in a study in southern Queensland where descriptors were compared with canopy cover calculated from current aerial photography of undisturbed remnants. However, there is evidence that terminology usage has altered over the last century, and density descriptors must be interpreted carefully (Lunt 1997; Griffiths 2002).

As well as information on tree density, the survey plans have provided data on species present in the overstorey and their relative dominance. The first listed overstorey species in the survey plans appears to represent the dominant species found on the ground. When compared with the field survey quadrats, 79 % of quadrats showed the same dominant overstorey species, indicating that early surveyors approached the surveys in a similar fashion as we would today, listing species in order of dominance.

Information on the understorey was also available from the survey plans. Shrub cover, on average was lower on sites originally described as "grassy" than those described as "shrubby". This indicates that the pattern of some woodlands supporting a grassy understorey, and others a shrubbier understorey persists today.

Analysis of vegetation communities revealed weak community associations forming seven semi-arid woodland types within the MSNP. These woodland types differed according to the overstorey species present, and in the species richness. Whilst it is possible that differences in species richness may be due to natural variation in sites, previous studies across southeast Australia suggest that these changes are a result of previous land use (Westbrooke 1998).

Descriptions from early botanists

Descriptions from early botanists in the region all point towards the diversity of semi-arid woodland. The understorey is described as containing "a large number of other trees" (Patton

1951). Such diversity is apparent in few remnants in northwest Victoria today, and many of the species listed as frequently occurring in the 1930s are rarely seen (Appendix 4). Early descriptions of the vegetation indicate a wide variety in the presentation of semi-arid woodlands in terms of tree density varying from open woodland to forest and understorey composition dominated by annuals in some areas and perennial shrubs in others. Certainly semi-arid woodlands at the time of settlement were far from uniform.

Notes on the survey plans indicating open vegetation were much more frequent than those indicating dense vegetation suggesting that historically, semi-arid woodland commonly supported fewer than 300 trees ha-1. Whilst small dense patches were recorded throughout the study area, the largest area is centred on the Yarrara Flora and Fauna reserve. This area has been used as a benchmark of *Casuarina pauper* woodland as it represents an area with relatively high diversity of shrub species, and a relatively intact overstorey. However, the map of semi-arid woodland tree density indicates that this area was probably not representative of the majority of *C. pauper* woodland in the study area at the time of settlement.

C. gracilis was the most frequently referred to semi-arid woodland species on the survey plans, followed by *M. platycarpum* and *C. pauper.* Mallee eucalypt species were also commonly recorded in areas that also contained some semi-arid woodland species. Infrequently recorded species were Myall, *Santalum acuminatum* and *Exocarpos aphyllus.*

Myall

Twenty-one references to Myall were recorded around the Nowingi area. Myall is the common name used to refer to a number of species occurring in northwest Victoria including *Acacia loderi, A. pendula, A. ancistrophylla* (Dwarf Myall) and *A. stenophylla* (River Myall) (Entwisle *et al.* 1996). *A. melvillei* has also been referred to as Myall on historical surveys in southern Queensland (Fensham and Fairfax 1997). Soil and hydrology in this location suggest it is most likely the surveyors were referring to *A. loderi* or *A. melvillei.*

There are a few records for *A. melvillei* near Mildura, but it has not been recorded elsewhere in the study area (NRE 2000a). *A. loderi* is now restricted, within the study area to a few remnant stands also near Mildura (Entwisle *et al.* 1996). *A. melvillei* and *A. loderi* are listed as vulnerable in Victoria (NRE 2000a). *A. loderi* shrublands are also listed under Schedule 1 of the NSW Threatened Species Conservation Act, 1995.

During field surveys of the area, no evidence was found of *A. melvillei, A. loderi* or any other uncommon tree or shrub species. Myall may have been lost either through harvesting for firewood along the Nowingi railway (*A. melvillei* is listed as a good firewood and was used for fence posts), through grazing pressure limiting regeneration (Cunningham *et al.* 1981), or a combination of both.

Callitris gracilis **(Pine)**

Another species that appears to have been depleted since European settlement is *C. gracilis*, the most frequently recorded semi-arid woodland species on the survey plans. This is in stark contrast to the FIS data, which shows *C. gracilis* occurring in 52 of 606 quadrats (8.6 %) in the study area.

Frequent dead standing and fallen trees, numerous stumps and widespread lack of regeneration point towards a decline in *C. gracilis* throughout remnants in the study area. In fact the impact of vegetation clearance on *C. gracilis* was noted as early as the 1940s "Owing to the advance of settlement in the far north-west of Victoria, large belts of Murray Pine [*C. gracilis*] have been lost in the process of clearing land for the cultivation of cereals" (Zimmer 1944). In the 1930s, the combination of severe drought and insect attack led to the death of many pines (Zimmer 1944; Sims and Carne 1947). It has been suggested that many of the stumps observed today were cut following tree death in the 1930s drought.

3.4.3 Change in semi-arid woodland extent since European settlement

Estimations of the extent of clearing of semi-arid woodland outside of the MSNP indicates more than 96 % of areas originally supporting semi-arid woodland have been cleared. These figures also suggest clearing within the MSNP. Evidence of stumps shows that some clearing and thinning has occurred within the Park, however, the satellite imagery interpretation, which was used to calculate tree presence and absence, only detects foliar cover greater than 10 %. The extent of clearing within the MSNP is believed to be much less than indicated by the Tree100 layer, with these figures providing an indication of extensive areas of very open woodland. As mentioned above, the original surveyors described these areas of sparse trees as woodlands.

It must be noted that all historical vegetation surveys took place during early settlement, from 1864 to 1930s. The vegetation was no longer pristine at this time, with impacts of grazing, timber harvesting for fences, building materials and fuel. There are notes on many of the survey plans relating to burnt vegetation. Fires used to assist with the clearing of land frequently escaped to burn large areas within the study area (LCC 1987). Despite these impacts, we can assume that the distribution, composition and structure were relatively unchanged at the scale described. This information contributes towards an understanding of the pre 1750 vegetation.

3.5 CONCLUSION

Historical survey plans and maps contribute unique information to our understanding of the distribution, composition and change of semi-arid woodlands in northwest Victoria. In areas such as this, where vegetation disturbances have been great and little native vegetation remains, historical survey plans can contribute greatly to an understanding of the pre-European vegetation. Tests of the accuracy of the survey plans and derived maps show that they correlate with current vegetation patterns, however, accuracy of woodland classification was limited when compared to existing EVC data. Compiling and analysing the historical survey plans using GIS facilitated spatial analyses and map generation.

The survey plans and other historical records show that semi-arid woodlands in northwest Victoria were always variable in both composition and density. Historically semi-arid woodlands supported higher species diversity, with many species, such as *C. gracilis* less frequently observed. Woodlands at the time of settlement were more likely to be described as open, than dense, although a number of dense woodland areas were described.

More than 96 % of semi-arid woodlands have been cleared in agricultural regions surrounding the MSNP. In addition, there has been an overall decrease in tree density within vegetation remnants throughout the study area, due to tree harvesting, thinning and senescence in the absence of regeneration. This highlights the need to conserve and manage remnants to ensure regeneration.

4. DESIGN AND EVALUATION OF FIELD AND REMOTE METHODS FOR VEGETATION CONDITION ASSESSMENT OF SEMI-ARID WOODLAND

4.1 INTRODUCTION

Condition assessment is increasingly being used as a tool for monitoring vegetation that has been subject to disturbance. Condition assessment has been used since the early 1900s for monitoring rangelands, and is increasingly being developed as an assessment tool in riparian areas (Dyksterhuis 1949; NRE *et al.* 1997; Spencer *et al.* 1998). The National Framework for the Management and Monitoring of Australia's Native Vegetation states that monitoring change in vegetation condition is an essential component of native vegetation management. In Victoria, vegetation condition assessment is being used to assess impact of disturbances, particularly grazing, within National Parks (eg. CEM 1998; Gibson *et al.* 1999; Westbrooke *et al.* 2001; Gowans and Westbrooke 2002; Leversha and Gowans 2003b, 2003a).

Since aerial photography first became available in the early 1900s, remotely sensed data have been investigated and used for vegetation monitoring. Techniques for monitoring forestry activity and rangeland productivity have been widely published (Smith and Woodgate 1985; Tickle *et al.* 1998; Taube 1999; Wallace and Thomas 1999; Hyyppa *et al.* 2000; Pickup *et al.* 2000). However, few studies have utilised remotely sensed data for vegetation assessment of conservation reserves (Wallace and Furby 1994; McCormick 1999).

Potential advantages of using remotely sensed data include reduced costs, complete coverage of an area as opposed to representative sampling, and the ability to produce maps of vegetation condition. In addition, temporal coverage is high for many remote data sources, enabling repeatability of assessment and change detection, both in the recent history and future.

The major goal of vegetation condition assessment within National Parks is to acquire data on parameters that directly relate to park management (Parks Victoria 2000). To be of use in park management, remotely sensed vegetation condition assessments must produce data that can be directly related to management outcomes, in a cost-effective manner.

Cost effectiveness of condition assessment is vital to ensure sufficient data from parks can be collected as frequently as required. However, given the limits of funding for condition assessment, rarely is there the opportunity to perform repeated assessments of the one reserve to determine the cost-effectiveness of various methods.

This study examined vegetation condition assessment using four types of data; (i) field data (ii) satellite imagery (iii) aerial photography, and (iv) pre-existing data. The objectives for this research were to:

(i) determine suitable techniques and parameters for field condition assessment of vegetation condition;
(ii) determine parameters that best describe the variability in vegetation condition in the semi-arid woodlands of northwest Victoria;
(iii) determine suitable techniques for remotely sensed condition assessment using satellite imagery;
(iv) determine suitable techniques and parameters for remotely sensed condition assessment using aerial photography;
(v) determine suitable techniques for remote condition assessment using existing data sources;
(vi) determine relative costs of acquiring and using remotely sensed data sources for vegetation condition assessment; and
(vii) perform a cost-benefit analysis to determine the most appropriate remotely sensed data for assessing semi-arid woodland vegetation condition.

Each data source has been analysed individually and the method, results and discussion, of each technique are presented separately in sections 4.2–4.5. A combined cost benefit analysis, discussion and conclusion follow the sections on individual techniques. The objectives of this study were to determine appropriate methods for vegetation condition assessment. Therefore, only brief results of the condition of the Murray-Sunset National Park are presented here. Further information on the condition of the Murray-Sunset National Park is presented in the unpublished report 'Vegetation condition assessment of the semi-arid woodlands of Murray-Sunset National Park.' (Westbrooke *et al.* 2001)

4.2 FIELD SURVEY CONDITION ASSESSMENT

4.2.1 Methods

In 2000, Parks Victoria commissioned a report to assess the vegetation condition of the semi-arid woodlands of Murray-Sunset National Park (MSNP) (Westbrooke *et al.* 2001). The main goals identified were to determine a quantitative baseline measure of vegetation condition, and to assist in determining the need for kangaroo population management. Providing data for long-term monitoring of vegetation change was another important objective. The data collected during this ground assessment provided an opportunity for comparison with alternative remotely sensed condition assessments. Methods were based on a previous vegetation condition assessment in northwest Victoria to enable comparison between assessments (Miller *et al.* 1998).

4.2.1.1 Sampling strategy

Sampling was undertaken between November and December 2000 within the five Ecological Vegetation Classes (EVCs) containing semi-arid woodlands; Belah Woodland; Pine-Buloke Woodland, Savannah Woodland / Savannah Mallee / Grassland Mosaic, Gypseous Plains Grassland and Sandplain Grassland. The study area was the MSNP, with sites for benchmarks chosen from within the park or neighbouring reserves (Westbrooke *et al.* 2001).

A stratified random sampling approach was undertaken with quadrats randomly selected within each EVC, the number of quadrats within each EVC being approximately proportional to the area of the EVC. A Global Positioning System (GPS) was used to locate quadrats in the field. Quadrats were located in the field at the woodland tree closest to the randomly generated co-ordinates and the tree was marked with an aluminium tag to assist in relocation of the quadrat. The quadrat was oriented to provide the best example of the EVC. If an appropriate tree could not be located within 100 m of the random co-ordinates, then an alternative site was selected from the randomly generated points. Quadrat size was 20 x 50 m based on recommendations from previous studies (Miller *et al.* 1998). The sample size was limited by the project budget, enabling 115 quadrats and six benchmark sites to be sampled. Table 4.1 shows the level of sampling within each EVC.

4.2.1.2 Benchmarks

Three benchmark sites were chosen for each of the Belah Woodland and Pine-Buloke Woodland EVCs. Benchmark sites were chosen using local expert knowledge to select areas where both current and historical disturbances were known to be minimal and perennial species richness high (Westbrooke, M. [University of Ballarat] 2000, pers. comm., 15 November). Evidence suggests that other woodland EVCs are not distinct communities, but are variants of these two communities (Westbrooke 1998). Therefore, all quadrats were compared to benchmarks of either Belah or Pine-Buloke, depending upon locality and dominant overstorey species. All quadrats within the southeast of the study area supporting either *Callitris gracilis* or *Allocasuarina luehmannii* were compared to the Pine-Buloke reference sites. Elsewhere, all quadrats were compared to the Belah Woodland EVC, unless supporting *A. luehmannii.*

Table 4.1. Summary of sampling in each EVC within the MSNP (excluding benchmark sites) for the field based condition assessment.

Ecological Vegetation Class	Area (ha)	Area (%)	Quad. (No.)	Quad. (%)	% area sampled
Pine-Buloke Woodland	1,274	2	20	17	0.16
Belah Woodland	4,006	6	20	17	0.05
Gypseous Plain Woodland	5,438	8	8	7	0.01
Sandplain Grassland	2,414	4	10	9	0.04
Savannah Woodland/Savannah Mallee/Grassland Mosaic	50,730	80	57	50	0.01
Total	63,862	100	115	100	0.02

4.2.1.3 Parameters used to determine condition index

Six parameters were chosen to best meet the objectives of the condition assessment. Parameters were chosen based on a review of parameters used in similar studies (Miller *et al.* 1998; Parks Victoria 1998; Gibson *et al.* 1999). To compare quadrats to the benchmark sites, the value of the parameter at each quadrat was divided by the benchmark value for that EVC resulting in a number from 0 to 1. If a field value was greater than the benchmark value, the score was truncated to a maximum value of one. Benchmark values for each EVC are listed in Table 4.2.

Table 4.2. Benchmark values for condition parameters derived from reference sites and from fixed values based on expert opinion.

Vegetation parameter	Source of benchmark	Benchmark values	
		Pine-Buloke	Belah
Native perennial species richness	Benchmark site	15	26
% Indigenous species shrub cover	Benchmark site	50	65
Regeneration of perennial shrub species	Benchmark site	10	12
Strata intactness	Fixed	4	4
Tree condition	Benchmark site	3.52	3.70
Dominant strata age classes	Fixed	3	3

The parameters were:

Native perennial species richness – the total number of woody perennial species (trees, tall shrubs and small shrubs) within the quadrat was counted and compared to the maximum perennial species richness measured in the benchmark sites.

$$\text{Score} = \frac{\text{Total number of woody perennial species}}{\text{Maximum number from benchmark sites}}$$

Native shrub cover – the projected foliage cover of native tall and small shrubs was visually estimated for the quadrat and compared to the maximum shrub cover of the benchmark sites.

$$\text{Score} = \frac{\text{Percentage cover of woody perennial species}}{\text{Maximum percentage cover from benchmark sites}}$$

Regeneration of perennial shrub species – the presence of juveniles of shrub species, estimated to be less than two years of age was recorded. Juveniles of sucker and seedling origin were not separated, as origin cannot be determined without destructive sampling (Westbrooke 1998). The proportion of regenerating shrubs was calculated by dividing the number of regenerating shrub species on the plot by the maximum number of regenerating shrub species found on the benchmark sites.

$$\text{Score} = \frac{\text{Number of regenerating shrub species within a quadrat}}{\text{Maximum number of regenerating shrub species from the benchmark sites}}$$

Strata intactness – strata intactness was calculated for each of four strata identified in benchmark sites. Values were based on expert opinion (Westbrooke, M. [University of Ballarat] 2000, pers. comm., 15 November).

- Tree strata – intact if two or more mature trees present (stem diameter >10 cm), with a tree condition score greater than two;
- Tall shrub layer – intact if cover is greater than 5 % and species richness greater than two;
- Small shrub layer – intact if cover is greater than 10 % and species richness greater than five; and
- Ground layer – intact if annual native species cover is greater than 10 %.
- An intact stratum scored a value of one, with a maximum score of four if all strata were intact.

$$\text{Score} = \frac{\text{Number of intact strata}}{\text{Number of expected strata (four)}}$$

Tree condition – A visual estimate of tree health or condition was performed for each individual on the following five-point scale:

4 – Healthy, well formed crown, no dead branches within the canopy

3 – Well formed crown but dead branches projecting from the canopy

2 – Irregular crown, many dead branches projecting from the canopy

1 – Less than 25 % of original tree mass alive

0 – Dead (Excluded from calculation of age classes and species richness)

An average of all trees in the quadrat was then calculated:

$$\text{Score} = \frac{\text{Average tree condition on quadrat}}{\text{Average tree condition from benchmark sites}}$$

Overstorey strata age classes present – The number of age classes present in the overstorey layer was determined through measurement of stem diameter. Size classes were determined from the stem diameter measures based on expert opinion (Westbrooke, M. [University of Ballarat] 2000, pers. comm., 15 November). Stem diameter was measured at approximately 50 cm above ground level, or where possible below the point of stem branching. As root suckering is a feature of many woodland tree species, all multiple branches emerging from the ground adjoining the main stem were measured, and the largest stem was used in determining age class.

$$\text{Score} = \frac{\text{Average number of age classes}}{\text{Expected average number of age classes (three)}}$$

Condition index

A condition index was calculated to combine the score for each parameter to create an overall condition score. The resulting values for each of the six parameters were summed and then divided by the number of parameters (six) to produce a condition index scaled from 0 to 1.

$$\text{Condition Index} = \frac{\text{Sum of parameter scores}}{\text{The number of parameters assessed (six)}}$$

4.2.1.4 Additional measures

In addition to measures used to determine the condition index, other measures were taken to provide long-term monitoring data.

All woody perennial vascular plants within the quadrat were recorded, identified to species (nomenclature following Walsh and Entwisle (1994; 1996; 1999) and categorised according to life forms as identified in the Flora Information System (FIS) (NRE 2000a). Annual herbs and grasses were not recorded, as unpredictable fluctuations following rainfall are problematic for comparative studies in semi-arid areas (Pickup 1996). However, dominant ground layer native and introduced herbaceous species were recorded. Visual estimates of cover of litter, bare ground and cryptogamic cover were also made.

To assist future re-location of quadrats, each corner tree was marked with an aluminium tag. The bearing of the quadrat and the corner of the quadrat were recorded, for example, the northwest corner marked of an eastwest quadrat. A photo was taken at each site to provide a record of the appearance of the site. The date, recorders names, a unique quadrat number, Australian map grid

(AMG) co-ordinates, description of site location and observed EVC were recorded. A qualitative condition score served to crosscheck the quantitative assessment. Estimation for the qualitative assessment was made on a ten-point scale between 1 and 10 based on surveyor experience of the site, compared to known benchmark sites.

Condition classes

To create a simplified categorical condition class, three condition groups were derived. Condition classes were chosen to best represent the observed and calculated condition of sites. The upper range of condition index (CI) values (CI greater than 0.6) represented areas in good condition. The mid range (CI between 0.35 and 0.6) represented vegetation in moderate condition, whilst the lower range (CI less than 0.35) represented vegetation in poor condition. Three condition classes were initially selected as it was thought that this was the maximum number of categories likely to be separable. Two condition classes were also formed from the field condition scores with condition indices greater than or equal to 0.5 representing good condition and indices less than 0.5 representing poor condition.

4.2.1.5 Condition mapping

Interpolation of quadrat data were used to create a model of semi-arid woodland condition across the MSNP according to the method used by Westbrooke *et al.* (2001). The area assessed consisted of all EVCs containing semi-arid woodlands within the MSNP. A raster condition surface was calculated at 20 m resolution, using the "inverse distance weighted" (IDW) interpolation algorithm of the INTERPOL module within Idrisi32 (Clark Labs).

The IDW is one of the more commonly used algorithms for interpolating point data (Eastman 2001). To calculate the condition of an area the module performs a search for data points, increasing the search radius until all data points are located. A weighted average of these points is used create the resulting score.

The IDW algorithm predicted the condition score for every 20 x 20 m pixel throughout the study area, using the following equation:

$$C = \frac{\sum_{1}^{n} \frac{C_i}{d_i^{\beta}}}{\sum_{1}^{n} \frac{1}{d_i^{\beta}}}$$

Where:

C the condition score for the pixel currently being interpolated.

n the number of quadrats (81).

C_i the condition of quadrat i.

d_i the distance between the pixel currently being interpolated and quadrat i.

β the distance weighting exponent (2).

The distance-weighting exponent (β) determines the extent to which the condition of nearby points influences the interpolation model. The higher the value of β, the greater the relative influence of nearby points. A β value of two was used to generate both condition maps.

4.2.1.6 Accuracy assessment

To determine the validity of maps determined from remotely sensed data or interpretation of other data, accuracy assessment must be performed. Accuracy assessment involves "ground truthing" or sampling on the ground to assess whether areas have been correctly assigned to classes. Accuracy assessment was performed for all techniques investigated.

Sampling for accuracy assessments required a pragmatic balance between acquiring an adequate and representative sample for reliable calculation of map accuracy, and on ground costs in obtaining the data. Costs dictated no more than five days could be spent on determining accuracy of mapping. It has been suggested that a minimum sample of 50 quadrats per class is required (Congalton 1991), but due to the amount of data to be collected at each site, this was not feasible.

To maximise the sample size, a rapid method of condition assessment based on the same parameters used in the initial field was derived. The method needed to be comparable to the initial condition assessment used, but as rapid as possible to enable the maximum number of sites to be assessed. To reduce the time taken in the field, DBH and condition of each tree was not recorded, and visual estimates were made of tree health and number of age classes present. All other measures were recorded in the same manner as in the original assessment.

Stratified random sampling was applied within a subset of the study area that corresponded to the extent of the 2001 Landsat satellite image (Section 4.3). To allow a greater number of samples to be performed in a limited period, sites were selected within 100 m of tracks and roads. The map of three condition classes produced by supervised classification of Landsat data (Section 4.3.2) was used to enable stratified random sampling in areas of good, moderate and poor condition (Table 4.3).

Samples were selected with the aim of achieving an equal number of samples in each condition class with quadrats at least 200 m apart. Quadrat co-ordinates were loaded into a GPS to enable location in the field. As with the field assessment, at least one woodland tree was required to fall within the quadrat. If no suitable trees were present within 100 m of the quadrat site then another quadrat was selected.

Table 4.3. Sampling effort for accuracy assessment within condition classes.

Condition class	No. quadrats sampled within classes determined by satellite image classification.	No. quadrats where condition class calculated from field condition index.
Good condition	25	16
Moderate condition	25	34
Poor condition	28	28

The parameters measured were modified from the original field assessment to enable greater speed of assessment. Estimates of the number of age classes of each tree species and the average condition of all trees in the quadrat were substituted for measurements of individual trees to reduce survey time. Benchmarks used were those of the initial field assessment (Table 4.2) and the same calculations performed (Section 4.2.1.3). Condition indices were calculated and condition classes formed as outlined above (Section 4.2.1.3).

Measures of map accuracy

The interpolation model was reduced to three classes to facilitate calculation of an error matrix. The error matrix is created as a cross-tabulation, with column headings (by convention) representing the benchmark data, and rows representing the classification classes of the remotely sensed imagery or map. Values in each cell represent the number of points in each class on the image and in the benchmark data.

Overall accuracy of the interpolation classes was calculated by dividing the number of correctly assigned quadrats with the total number of quadrats assessed (Congalton 1991). Producer's accuracy, and user's accuracy were also calculated (Congalton 1991) (see Section 3.2.2.1 for a description of producer's and user's accuracy).

Statistical measures of map accuracy

One of the most commonly used measures of statistical agreement is the Kappa, which measures the agreement between two classifications, such as agreement between a classified satellite image and ground data. Kappa values indicate the extent of agreement between two data sets, greater than that expected by chance. A value of one indicates a perfect agreement, with all observations falling on the diagonals of the error matrix and a value of zero indicates agreement no better than chance (Agresti 1990). Negative values occur when agreement is less than expected by chance. Kappa values were calculated from the error matrix where:

Pij is the number of observations in row i, column j

Pi+ is the sum of row i

P+i is the sum of column i, and

n is the total number of observations

$$P_o = \sum p_{ii}$$

$$P_e = \sum p_{i+} + p_{+i}$$

$$\kappa = \frac{\sum p_{ii} - \sum p_{i+} + p_{+i}}{1 - \sum p_{i+} p_{+i}} = \frac{P_o - P_e}{1 - P_e}$$

Z scores can be used to test the hypothesis that the value of K is greater than that expected by chance where:

$$\sigma^2(\kappa) = \frac{1}{n} \left\{ \begin{array}{c} \frac{P_o(1-P_o)}{(1-P_e)^2} + \frac{2(1-P_o)\left[2P_oP_e - \sum p_{ii}(p_{i+} + p_{+i})\right]}{(1-P_e)^3} + \\ \frac{(1-P_o)^2\left[\sum\sum p_{ij}(p_{j+} + p_{+i})^2 - 4P_e^2\right]}{(1-P_e)^4} \end{array} \right\}$$

and

$$Z = \kappa / \sigma^2$$

To test the hypothesis that agreement between two data sets (eg. reference data and classified image) is greater than that expected by chance, a test for significance difference between Kappa values is employed where:

$$Z \sim \frac{\kappa_1 - \kappa_2}{\sqrt{\sigma_1^2 + \sigma_2^2}}$$

A potential limitation with this technique is that accuracy assessment of two classification techniques based upon the same ground truthed data does not represent independent samples, but paired data. The formula above to calculate Z scores for Kappa is appropriate for independent samples, but with paired data, may over-estimate the standard error, resulting in a greater chance of accepting the null hypothesis (type two error) (Agresti 1990).

An alternative approach, to examine the overall proportion of correct and incorrectly assigned pixels, is the McNemar test (Agresti 1990). The McNemar test is a nonparametric test for two related dichotomous variables and was used in the study to detect if a significant difference occurred in proportions of correct and incorrectly classified pixels between classification techniques.

4.2.1.7 Parameters contributing to variation in vegetation condition

Understanding parameters that account for most of the variability can assist in choosing parameters that are priorities for assessment, and which parameters may be less important and potentially discarded. The field condition assessment data for both the rapid assessment for ground truthing undertaken in April 2003 and the field condition assessment undertaken in November to December 2000 were combined. Spearman's Rank Order Correlations were performed to determine which parameters were correlated with the condition index. Assumptions of normality and homoscedasticity were tested by inspecting Q-Q plots and the Levene statistic.

4.2.1.8 Costs of field survey condition assessment

Costs to perform the field based condition assessment were calculated based upon actual figures from the field condition study in the MSNP where possible. This assumes approximately 100 random quadrats over a total area of 6,363 km^2. The area of interest, semi-arid woodlands, totalled 638 km^2. Travel costs have also been included as costs of accessing remote sites are commonly high and must be taken into consideration. Time taken on some tasks was not recorded, and so estimates were based on total hours recorded for the project. Costs have been based on current rates of the Centre for Environmental Management, an environmental consultancy operating from the University of Ballarat.

Costs for report writing have been excluded, as it would be expected that the report costs would remain constant regardless of the type of condition assessment performed. All costs exclude any set up costs such as computers and software.

4.2.2 Results

4.2.2.1 Summary of parameters

Native perennial species richness

Native perennial species richness was consistently lower than the benchmark sites for both Belah and Pine-Buloke woodlands (Figure 4.1). The number of perennial species in Belah woodland ranged from three on more degraded sites, to 26 on one of the benchmark sites. The minimum number of perennial species found in degraded Pine-Buloke woodland was also three, with a maximum of 15 on one benchmark site.

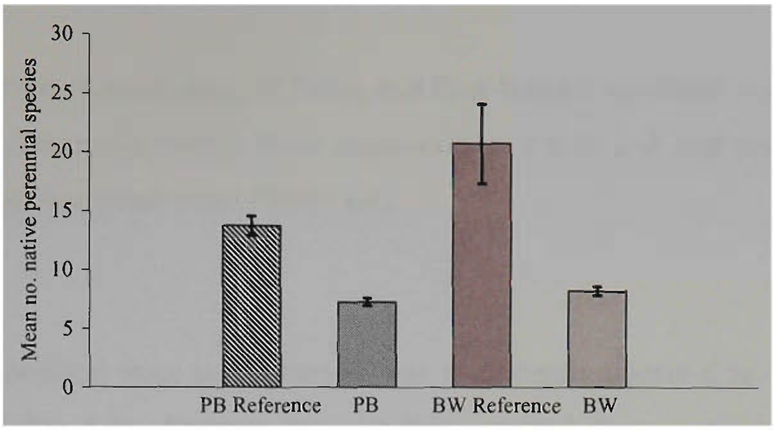


Figure 4.1. Mean (error bars represent ± 1 SE) native perennial species richness of Pine-Buloke and Belah woodland sites and benchmark sites.

Native shrub cover

Native shrub cover was consistently lower than the benchmark sites for both Pine-Buloke and Belah woodland (Figure 4.2). Shrub cover for Pine-Buloke woodlands ranged from 2 % on degraded sites to 50 % on more intact sites, with shrub cover on Belah woodlands ranging from 1 % to 65 %.

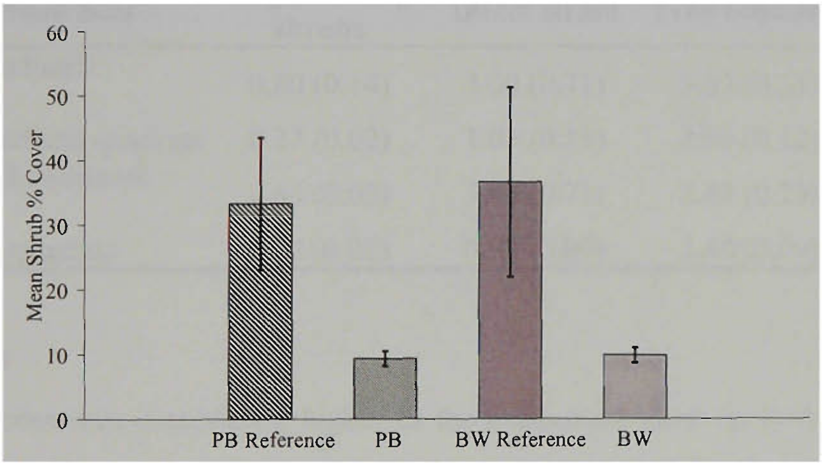


Figure 4.2. Mean (error bars represent ± 1 SE) native shrub cover of Pine-Buloke and Belah woodland sites and benchmark sites.

Regeneration of perennial shrub species

Table 4.4 shows mean and standard error of the proportion of regenerating shrubs on Pine-Buloke and Belah woodland field sites, and benchmark sites. The number of shrubs regenerating was greater on benchmark sites for both Pine-Buloke and Belah woodland.

Strata intactness

The degraded condition of many areas of Belah and Pine-Buloke woodland is clearly reflected in the results for strata intactness, which show mean values of 0.92 and 1.05 respectively out of a possible maximum of four intact strata (Table 4.4).

Tree condition

The condition of woodland trees in the survey was consistently poorer than those measured in benchmark sites (Table 4.4). Average tree condition ranged from a minimum of one to a maximum of four, with only three quadrats supporting trees with an average condition of four.

Overstorey age classes present

The number of age classes present was consistently low for both Pine-Buloke and Belah woodland sites in both sample and benchmark sites (Table 4.4).

Table 4.4. Mean (± 1 SE) of vegetation condition parameters for Pine-Buloke and Belah woodland field sites and benchmark sites.

Semi-arid woodland sites	Regenerating shrubs	Intact strata	Tree condition	Tree age classes
Pine-Buloke woodland benchmark sites	0.80 (0.14)	3.00 (0.71)	3.03 (0.51)	2.17 (0.10)
Pine-Buloke woodland quadrats	0.27 (0.02)	1.05 (0.13)	2.59 (0.12)	1.61 (0.09)
Belah woodland benchmark sites	0.64 (0.03)	3.00 (0.71)	3.44 (0.29)	2.42 (0.26)
Belah woodland quadrats	0.26 (0.02)	0.92 (0.09)	2.45 (0.09)	1.76 (0.08)

Condition index

The condition index was consistently higher in the benchmark sites for both Pine-Buloke and Belah woodland (Figure 4.3). Similar mean values were obtained for Pine-Buloke (0.41) and Belah woodland (0.37).

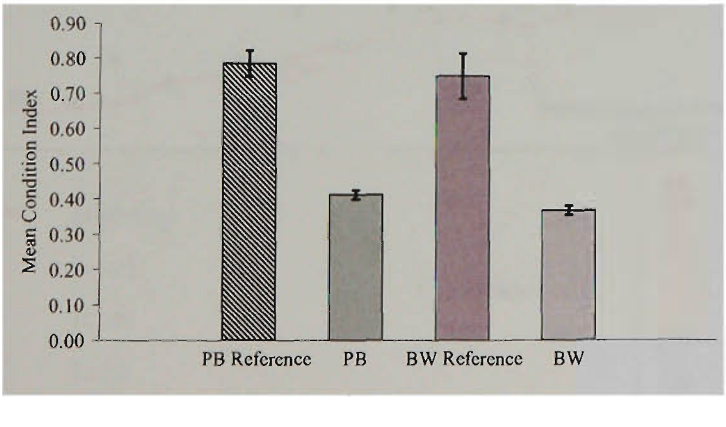


Figure 4.3. Mean condition index (error bars represent ± 1 SE) for Pine-Buloke and Belah woodland sites in the MSNP and benchmark sites.

4.2.2.2 *Map of semi-arid woodland condition derived from field survey*

An interpolation procedure was used to model the condition of semi-arid woodland across the MSNP (Figure 4.4). To enable comparison with other methods of vegetation condition assessment and to assist in accuracy assessment, the results of the interpolation model were reduced to three condition classes (Figure 4.5).

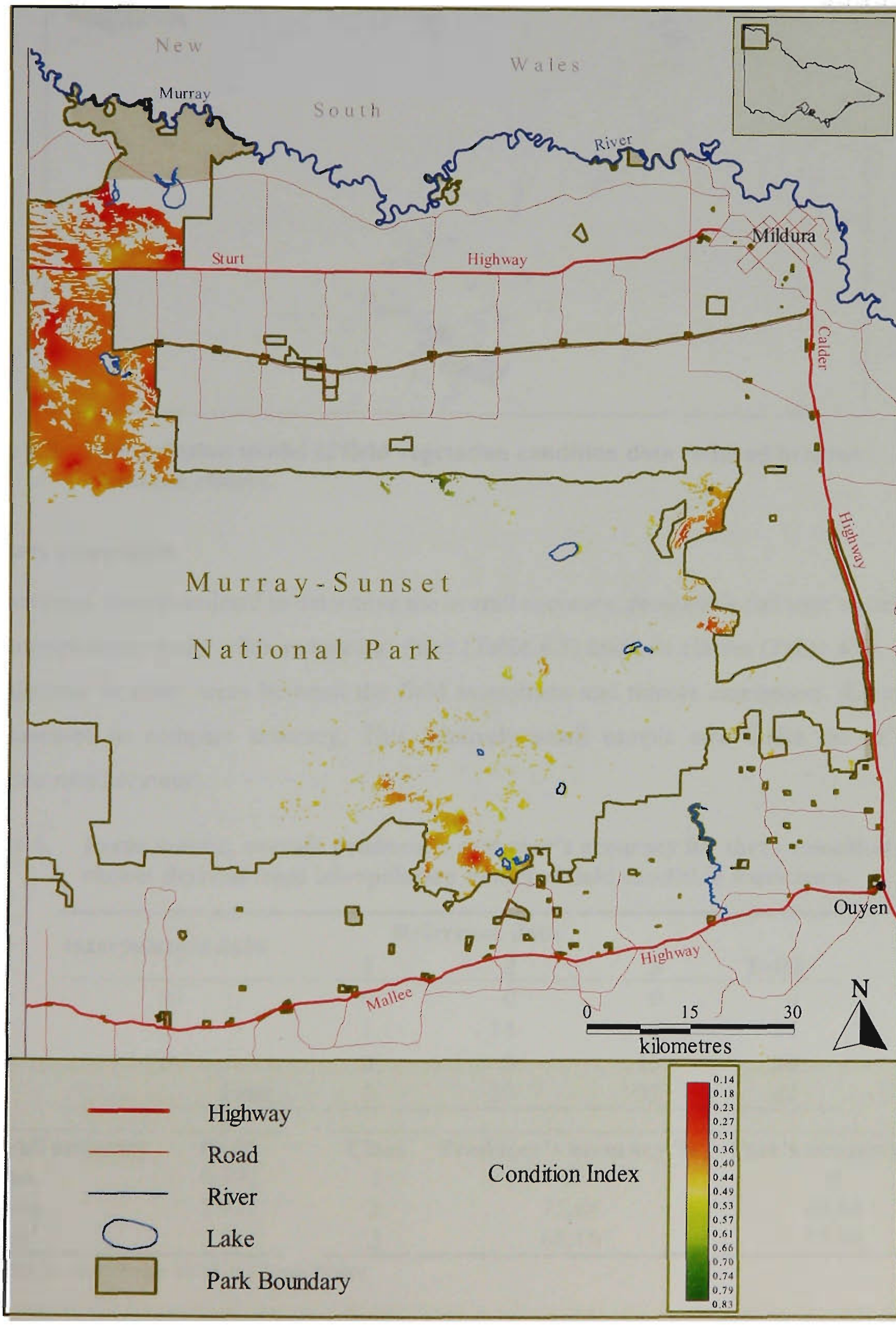


Figure 4.4. Interpolation model of vegetation condition based on field assessment.

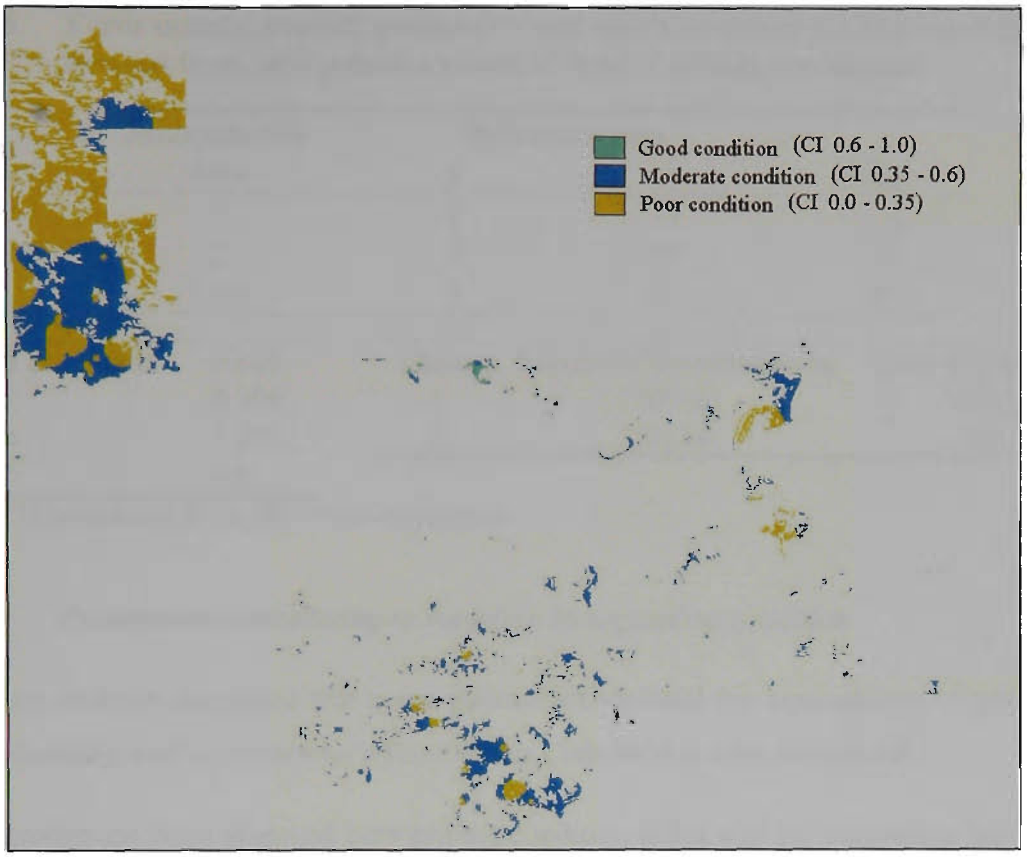


Figure 4.5. Interpolation model of field vegetation condition data reduced to three condition classes.

Accuracy assessment

Error matrices were calculated to determine the overall accuracy, producer's and user's accuracies of the interpolation maps when reduced to three (Table 4.5) and two classes (Table 4.6). Due to the difference in study areas between the field assessment and remote assessment, 42 quadrats were assessed to compare accuracy. This relatively small sample size limits the ability to determine map accuracy.

Table 4.5. Error matrix, overall, producer's and user's accuracy for three condition classes derived from interpolation model of field condition assessment.

Interpolation data	Reference data 1	2	3	Total
1	0	0	0	0
2	1	14	7	22
3	0	5	15	20
Total	1	19	22	42

Overall accuracy	69.05
Kappa	0.397
Z score	2.932
Result[a]	S

Class	Producer's accuracy %	User's accuracy %
1	0	0
2	73.68	63.64
3	68.18	75.00

[a]At the 95 % confidence level, S =Significant.

Table 4.6. Error matrix, overall, producer's and user's accuracy for two condition classes derived from interpolation model of field condition assessment.

Interpolation data	Reference data 1	2	Total
1	1	0	1
2	4	37	41
Total	5	37	42

Overall accuracy	90.48
Kappa	0.306
Z score	1.287
Result[a]	NS

Class	Producer's accuracy %	User's accuracy %
1	100.00	20.00
2	90.24	100.00

[a]At the 95 % confidence level, NS = Non-Significant.

4.2.2.3 Parameters contributing to variation in vegetation condition

Preliminary analysis suggested that some parameters violated the assumptions of normality and homoscedasticity, and so Spearman's Rank Order Correlations were performed.

Strong correlations were observed between the condition index and the percentage tree cover, tall shrub species richness, average number of tree age classes present and average tree condition (Table 4.7). In addition, many correlations were found between variables, with all variables showing significant correlations with the most other variables. In particular, many strong and moderate correlations are observed with the percentage cover of trees, and with the percentage cover of bare ground.

4.2.2.4 Costs of field survey condition assessment

Table 4.8 indicates approximate costs of performing a field based condition assessment at a remote site such as the MSNP. Costs are based upon the current study and assume performing 100 randomly allocated quadrats across an area the size of the MSNP (6,363 km^2). The number of quadrats performed was determined by the project budget. With 115 quadrats of 0.1 ha each, 0.018 % of the semi-arid woodlands was sampled. It is difficult to estimate the number of samples that would ideally be required due to the large number of factors that affect the required sample size. Factors including the heterogeneity of the site or surface variation, distribution of the sampling (random or systematic) may influence sample size (Haining 1993).

Table 4.7. **Spearman's correlations of parameters measured from 119 quadrats within the Murray-Sunset National Park and surrounding reserves.**

	%cover trees	%cover tall shrubs	%cover small shrubs	%cover ground layer	%cover exotic ground layer	%cover cryptogams	%cover litter	%cover bare ground	tree age classes	tree condition	tree richness	tall shrub richness	small shrub richness	regenerating shrubs	Condition Index
%cover trees	1	0.387 **	0.369 **	-0.402 **	-0.404 **	0.415 **	0.354 **	-0.56 **	0.407 **	0.304 **	0.328 **	0.471 **	0.425 **	0.368 **	0.705 **
%cover tall shrubs	0.387 **	1	0.436 **	-0.202 **	-0.227 **	0.408 **	0.316 **	-0.572 **	0.184 **	0.161 *	0.12 *	0.791 **	0.259 **	0.447 **	0.545 **
%cover small shrubs	0.369 **	0.436 **	1	-0.146 *	-0.092	0.316 **	0.109	-0.378 **	0.15 *	-0.113	-0.105	0.336 **	0.139 *	0.185 **	0.391 **
%cover ground layer	-0.402 **	-0.202 **	-0.146	1	0.893 **	-0.314 **	-0.228 **	0.335 **	-0.091	-0.237 **	-0.252 **	-0.269 **	-0.352 **	-0.421 **	-0.452 **
%cover exotic ground layer	-0.404 **	-0.227 *	-0.092	0.893 **	1	-0.345 **	-0.221 **	0.377 **	-0.131	-0.303 **	-0.351 **	-0.33 **	-0.413 **	-0.451 **	-0.549 **
%cover cryptogams	0.415 **	0.408 **	0.316 **	-0.314 **	-0.345 **	1	0.234 **	-0.444 **	0.169 *	0.19 **	0.097	0.442 **	0.217 **	0.326 **	0.476 **
%cover litter	0.354 **	0.316 **	0.109	-0.228 **	-0.221 **	0.234 **	1	-0.297 **	0.026	0.086	0.099	0.464 **	0.243 **	0.117	0.247 **
%cover bare ground	-0.56 **	-0.572 **	-0.378 **	0.335 **	0.377 **	-0.444 **	-0.297 **	1	-0.33 **	-0.441 **	-0.364 **	-0.54 **	-0.41 **	-0.465 **	-0.656 **
tree age classes	0.407 **	0.184 **	0.15	-0.091	-0.131	0.169 *	0.026	-0.33 **	1	0.27 **	0.028	0.216 **	0.143 *	0.18 *	0.558 **
tree condition	0.304 **	0.161 *	-0.113	-0.237 **	-0.303 **	0.19 **	0.086	-0.441 **	0.27 **	1	0.198 **	0.227 **	0.213 **	0.252 **	0.572 **
tree richness	0.328 **	0.12	-0.105	-0.252 **	-0.351 **	0.097	0.099	-0.364 **	0.028	0.198 **	1	0.191 **	0.383 **	0.401 **	0.343 **
tall shrub richness	0.471 **	0.791 **	0.336 **	-0.269 **	-0.33 **	0.442 **	0.464 **	-0.54 **	0.216 **	0.227 **	0.191 **	1	0.345 **	0.46 **	0.609 **
small shrub richness	0.425 **	0.259 **	0.139	-0.352 **	-0.413 **	0.217 **	0.243 **	-0.41 **	0.143 *	0.213 **	0.383 **	0.345 **	1	0.479 **	0.532 **
regenerating shrubs	0.368 **	0.447 **	0.185 **	-0.421 **	-0.451 **	0.326 **	0.117	-0.465 **	0.18 *	0.252 **	0.401 **	0.46 **	0.479 **	1	0.598 **
Condition Index	0.705 **	0.545 **	0.391 **	-0.452 **	-0.549 **	0.476 **	0.247 **	-0.656 **	0.558 **	0.572 **	0.343 **	0.609 **	0.532 **	0.598 **	1

Table 4.8. Costs to undertake a field based condition assessment of approximately 100 random points across an area of 6,000 km^2.

Task / Item	**Resources**	**Unit**	**Unit Type**	**Cost per unit**	**Total Cost**
Project Development					
Define study area & method	1 scientist	2	days	$ 680	$ 1,360
Preparation for field work	1 scientist	2	days	$ 680	$ 1,360
	1 assistant	1	days	$ 430	$ 430
Sub-total					$ 3,150
Field Survey					
Field Survey	1 scientist	14	days	$ 680	$ 9,520
	1 assistant	14	days	$ 430	$ 6,020
Travel	travel	3,000	km	$ 0.55	$ 1,650
	Meals/accom	13	nights	$ 220	$ 2,860
Sub-total					$ 20,050
Data Analysis					
Plant ID	1 scientist	2	days	$ 680	$ 1,360
Data entry	1 assistant	3	days	$ 430	$ 1,290
Analysis of data	1 scientist	5	days	$ 680	$ 3,400
Sub-total					$ 6,050
Total cost					**$ 29,250**

To illustrate the effects of increasing the number of quadrats on the project costs, costs associated with sampling 0.1 % (5.5 times more quadrats) of the study area have been calculated at $123,000. As expected, a large increase in the costs, particularly those associated with the field survey component, occur with an increase in sampling rate.

4.2.3 Discussion

The method for field survey of vegetation condition described above has attempted to address the goals of park management in a replicable manner that will enable long-term monitoring of the site. Particular consideration has been given to minimising seasonal fluctuations by largely excluding annual vegetation from the condition score. However, some seasonal and rainfall effects may still influence the score for ground layer strata intactness, which is considered intact if native ground cover is greater than 10 %. In 2003, following drought conditions native ground cover did not exceed 2 % at any of the 80 quadrats assessed. This had a minimal effect on the condition score, reducing it by a maximum of 0.04.

Whilst parameters were equally weighted for the condition score calculations, effective weighting occurred through the choice of parameters. A number of parameters relied on the presence of trees, including tree strata intactness, average tree condition and average number of tree age classes. A number of parameters relating to species richness were also scored, including native perennial species richness, regeneration of native shrub species and strata intactness of small and tall shrubs. Weighting species richness in condition assessment of areas managed for conservation is a useful measure to ensure protection and management for biodiversity. Effective weighting of the tree layer is also considered appropriate when lack of regeneration and senescence have been identified as major threats for semi-arid woodlands in northwest Victoria.

Advantages of field survey of vegetation condition

Data were collected at a very fine scale (0.1 ha), enabling assessment of many parameters unlikely to be obtainable by any form of remote data, including regeneration of trees and understorey species, and species richness.

Field survey also enabled assessment of some of the potential impacts upon vegetation condition such as observation of grazing animals, scats and browse damage on plants. New threats could also be discovered during field survey, for example, observation of a new pest species.

Limitations of field survey of vegetation condition

The main limitation of field survey as a method for vegetation condition assessment is cost. Field survey, particularly in remote areas, is costly due the amount of time required and travel costs so therefore only a small proportion of the study area can be directly assessed. Costs for assessing 0.1 % of the study area were estimated at $123,000. Some studies have recommended sampling rates of 2 % (Bird *et al.* 2000), which would be expected to result in greatly increased costs in a remote area such as the MSNP.

Interpolation model

Interpolation procedures can be used to create models of vegetation condition from quadrat data, however, the accuracy of the model is dependent upon many factors including the heterogeneity of landscapes and disturbances, sampling regime and number of samples taken (Haining 1993). The reliability of interpolation models reduces with greater distance between adjacent quadrats and without further accuracy assessment the reliability of the model is unknown.

The accuracy assessment of the interpolation model was limited by the number of quadrats that occurred within the bounds of the interpolation procedure. Too few quadrats were assessed within the good condition classes to fully determine the accuracy of the interpolation model. As expected, greater accuracy was found when only two condition classes were derived from the interpolation model.

4.3 USE OF LANDSAT SATELLITE IMAGERY FOR CONDITION ASSESSMENT

4.3.1 Methods

Landsat imagery has a number of potential benefits for use in remote vegetation condition assessment. Imagery has been archived from the early 1970s, enabling assessment of the last 30 years of a site. Compared with many other remote data sources, Landsat imagery is relatively cheap, with good return-time frequency, relatively small pixel size and high spectral resolution (Lunetta 1998). All image analyses were undertaken using Idrisi32 (Clark Labs) with MapInfo (MapInfo Corporation) used for final GIS operations and production of maps.

4.3.1.1 Study site

For initial investigation of remote condition assessment methods, a smaller trial site was chosen within the study area. This site was chosen to include a number of different semi-arid woodland condition and community types. Mallanbool, Yarrara and Meringur Flora and Fauna Reserves represent areas of woodland dominated by *Casuarina pauper* in good condition. Cleared land surrounding the reserves is likely to have once been semi-arid woodlands and therefore may illustrate woodlands in the worst possible condition. At least two intermediate condition classes based on woodland structure were also included within the image. Semi-arid woodlands north of the Sturt Highway provided an example of a more open, degraded site, with few remaining understorey species. Semi-arid woodlands to the south of the Copi Plains region of the MSNP also support fair shrub diversity, but are not equal to the condition of those at Yarrara. In addition to diversity in woodland condition, by including private land and roadsides, a large variety of land management was able to be incorporated into the study area.

4.3.1.2 Image acquisition

The Landsat ETM image used was a subset of the Landsat scene 95/84 acquired on 7th January 2001, 11:38 am. For change analysis procedures this was compared to a subset of an earlier Landsat scene 95/84 234853 acquired on 25th April 1997 1:48 pm. Images were acquired during summer to autumn to simplify analysis by minimising green annual vegetation and therefore limiting seasonal changes (Pilon *et al.* 1988; O'Neill and Eldridge 1990; Wallace and Thomas 1999). With the sun near the azimuth at this time in southern Australia, shadowing is also limited (Pech *et al.* 1986). Images were selected to ensure minimal cloud cover.

4.3.1.3 Pre-processing

Prior to image analysis, geo-registration, co-registration and radiometric correction are required to correct any potential error sources, and to maximise spatial accuracy of the images.

Registration

During an earlier study (Kerr *et al.* 1999), the 1997 Landsat TM image was geo-registered with ground control points collected across all major roads throughout the study area using differential GPS. Co-registration of the 1997 and 2001 images was performed to better enable detection of change between the two images.

The 2001 ETM image was co-registered to the 1997 image using eight control points evenly spaced throughout the image. A linear polynomial equation describing the change in location of points from the 1997 image to the 2001 image was calculated using the RESAMPLE module in Idrisi32 (Clark Labs). A nearest neighbour resampling (interpolation) procedure was used to re-assign pixel values whereby the value of the relocated pixel is derived from the closest pixel. The control point with the largest deviation was omitted to reduce the overall root-mean-square error (Gibson and Power 2000).

Radiometric calibration

Radiometric calibration is required to overcome potential errors due to sensor degradation, change of satellites, presence of cloud, water vapour, and other changes in atmospheric conditions (Lunetta 1998; Gibson and Power 2000). Calibrations can be undertaken either to determine true reflectance values, or to make the radiometric properties of images comparable, which is referred to as "like-values" calibration. Calculation of reflectance values requires knowledge of atmospheric conditions as well as satellite sensor parameters whereas calibration to like-values uses invariant targets to calculate gain and offset between different images (Eastman 2001).

Images were calibrated to radiometric like-values using two methods, to enable comparison between the 1997 and 2001 images. Firstly, offset and gain were calculated from 21 invariant points selected from throughout the image where the slope of the regression equation is the gain and the Y intercept is the offset (Eastman 2001). The input image is then multiplied by the gain and the offset is added to the result to produce the calibrated image. Points included mallee vegetation with a dense canopy, a salt lake, wide road intersections, the Murray River and associated lakes, and highly reflective rooftops. Up to three outliers were removed from each band for the calculation of line of best fit based on a least squares regression.

Offset and gain were also calculated using the CALIBRATE module in Idrisi32 (Clark Labs) using the 2001 image as a reference image. Using this module, a linear regression is performed to determine how the reference image would appear if there were no change, other than that due to the change in sensor gain and offset (Eastman 2001). This method was used for all radiometric calibration in the study. The images were also inspected for other potential problems such as line-banding and line dropouts.

4.3.1.4 Classification

Classification enables reduction of image data into discrete classes that are easier to interpret visually (Gibson and Power 2000). The first stage of investigating vegetation condition was to create a mask to separate semi-arid woodlands from other vegetation and land cover types. Existing vegetation maps do not cover private land within the study area and therefore it was necessary to create a map of land cover classes.

Unsupervised classification

To create a map of land cover classes within the study area an unsupervised classification was first performed on the 2001 Landsat image using the ISOCLUST module in Idrisi32 (Clark Labs). A seeding process is performed using a colour composite image to locate initial clusters. Pixels are then assigned to the nearest cluster mean using a maximum likelihood procedure. The mean of each class is then updated and pixels are reassigned. This is repeated until no further significant change in classes or pixel assignment occurs (Eastman 2001).

To investigate the impact of the algorithm used, CLUSTER, an alternative clustering procedure was also tested. The CLUSTER module uses a histogram peak technique, based on the method described by Richards and Xiuping (1999). The CLUSTER procedure creates a three-dimensional histogram based on the three bands of a composite image. A composite image was created using a linear stretch with a saturation level of 1 % of bands 3, 4 and 6. Peaks occur in the histogram where frequently occurring values lie. These peaks are used to form classes and pixels are assigned to the nearest peak. The midpoint between peaks is used to separate classes (Eastman 2001).

Supervised classification

A supervised classification approach was also undertaken. Six classes were identified within the image; (i) semi-arid woodland, (ii) mallee, (iii) water, (iv) agricultural land one, (v) agricultural land two, and (vi) agricultural land three. The three agricultural land classifications appeared spectrally distinct on the image, but were not identified on the ground.

Training sites were selected using quadrat data from the field condition assessment (Section 4.2), prior knowledge of the area and visual inspection of the image. Semi-arid woodland quadrats were overlain on a composite image of bands 3, 4 and 6 for the blue, green, and red bands respectively. Polygons were drawn around quadrats and adjoining pixels in areas of homogenous pixel colour. Quadrats that occurred in heterogenous areas or near the edge of change in pixel spectral characteristics were not used. Polygons for three classes of agricultural land were based on visual inspection of the image, and drawn around areas of similar spectral characteristics on the composite image. Polygons for mallee training areas were drawn on sections of the MSNP known to contain continuous mallee vegetation and polygons for water were drawn over Lake Wallawalla and the Murray River. Three polygons for each land cover class were digitised and bands one to six were used to extract the signature files or spectral response patterns for each class. Plots of signature files were examined to determine separation of land cover classes on the Landsat bands 1 to 6.

A maximum likelihood procedure was used to classify the image, using the signature files. In this procedure, pixels are assigned to a class based on probability contours from the training sites (Gibson and Power 2000). Whilst information on the number of pixels expected to belong to each class can be incorporated into this function, the default of equal class membership probabilities was used due to a lack of information on the likely extent of each cover class within the image.

Visual inspection of the classified image revealed a lack of separation between degraded areas of semi-arid woodland and cleared agricultural land. The signature files were then modified to improve the classification result by adding an additional agricultural class. The maximum likelihood classifier was applied to the modified signature file, however, this did not fully rectify the problem and so additional tree cover data were used to separate these land cover classes.

The Tree100 layer was used to delete treeless areas that occur outside of parks and reserves. The approach was based on an assumption that treeless areas within the park will largely represent sparsely treed semi-arid woodlands, whilst treeless areas outside of the park will mostly represent cleared land. The limitation with this approach is that some areas of sparsely treed semi-arid woodlands outside of the MSNP may have been misclassified as agricultural land.

To assess accuracy of the semi-arid woodland classification, the classification result was compared to the EVC map of public land. A Boolean image of tree presence was created using the Tree100 layer to mask all private land areas classified as semi-arid woodland that did not contain trees. The original classification was applied within the boundaries of the MSNP. The resulting modified land cover classification was then used to create a mask of semi-arid woodland within the study area.

4.3.1.5 Condition assessment

The mask was used to remove all land cover classes except semi-arid woodland from the image. Classification techniques were first explored to determine if condition classes were separable. Vegetation and brightness indices were then examined.

Classification of condition classes

To determine if condition classes could be separated classifications were performed. A number of unsupervised classifications were performed using the ISOCLUST procedure, investigating the result from different number of clusters.

The condition classes distinguished in the field assessment (Section 4.2) were used to assist in identifying training sets for supervised classification. Quadrats were overlain on a composite image of bands 3, 4 and 6 for the blue, green, and red bands respectively. Polygons were drawn around quadrats and adjoining pixels in areas of homogenous pixel colour. Quadrats that occurred in heterogenous areas or near the edge of change in pixel spectral characteristics were excluded from the signature files. At least three polygons of each condition class were created to define training sites. The image was classified into three condition classes (good, moderate and poor) based on condition classes determined from the field condition indices. The image was then classified into two condition classes (good and poor).

A maximum likelihood classifier was used in the supervised classification approach. The degree of separation of signature files was checked using scatter plots of various band combinations, which show a square for each signature centred about the mean for each band and bounded by two standard deviations. This illustrates the separability of the signatures on the two bands plotted on the x and y-axes. Plots of mean reflectance for each signature file were also assessed.

Vegetation and brightness indices

Two vegetation indices and a brightness index were investigated as potential indicators of vegetation condition (see Section 1.4.5 for outline of vegetation indices). The two vegetation indices chosen were the normalised differential vegetation index (NDVI) and tasselled cap greenness indices.

The NDVI is one of the most commonly used vegetation indices, formed by a simple ratio calculated from Landsat bands where:

$$\mathrm{NDVI} = \frac{\text{near infrared} - \text{visible red}}{\text{visible red} + \text{near infrared}}$$

The NDVI produces an index scaled between 1 to -1, with zero representing no vegetation and negative values representing non-vegetated land cover classes (Eastman 2001).

The Tasseled Cap is produced by a de-correlation of the original Landsat bands through orthogonalization to extract three new bands (Eastman 2001). Coefficients are used to weight the original digital counts to generate the new transformed bands. Negative weights on the visible bands are used to minimise the effects of the background soil, while positive weights on the near infrared bands emphasize the green vegetation signal (Eastman 2001). Similar to a principal components analysis (PCA), the first two new bands that are produced (greenness and brightness) explain approximately 95 % of the vegetation information. The greenness band produced is free of soil background effects, whilst the brightness band describes soil characteristics. The green vegetation index (greenness) of the Tasseled Cap was determined using the TASSCAP module in Idrisi32 (Clarke Labs) where:

Green vegetation index = (TM 1 × -0.2848) + (TM 2 × -0.2435) + (TM 3 × -0.5436) +(TM 4 × 0.7243) + (TM 5 × 0.0840) + (TM 7 × -0.1800)

Soil Brightness = (TM 1 × 0.3037) + (TM 2 × 0.2793) + (TM 3 × 0.4343) + (TM 4 × 0.5585) + (TM 5 × 0.5082) + (TM 7 × 0.1863)

Moistness = (TM 1 × 0.1509) + (TM 2 × 0.1793) + (TM 3 × 0.3299) + (TM 4 × 0.3406) + (TM 5 × -0.7112) + (TM 7 × -0.4572)

A simple brightness index formed by addition of bands 3, 5 and 6 was also investigated.

The final step to produce a smoothed image for map production was to apply a median 5 x 5 filter. The data were then converted to vector form and exported into MapInfo (MapInfo Corporation).

4.3.1.6 Accuracy assessment

To determine the most accurate classification of vegetation condition, an accuracy assessment based on ground truthing data were performed (Section 4.2.1.6).

Of the 80 quadrats assessed for ground truthing, 78 intersected with the area mapped in the satellite imagery interpretation. Error matrices were calculated for supervised and unsupervised classification of vegetation condition based on two and three condition classes. Overall accuracy, producer's and user's accuracy were calculated.

Kappa scores were calculated for the classifications to two condition classes to test the hypothesis that the observed agreement with ground data was greater than that expected by chance. Kappa scores were also calculated to determine if there was a significant difference between the results of the classification techniques.

4.3.1.7 Change detection

Change detection has been identified as a critical aspect of vegetation condition assessment within the National Parks monitoring framework (Parks Victoria 2000). Therefore, it was important to investigate techniques for change detection using satellite imagery. Whilst data were not available to ground truth the results of change detection methods, this provides an investigation of the types of techniques that can be applied to detect change in condition over time.

Two techniques of change detection were investigated:

(i) radiometrically normalised image differencing; and

(ii) a hybrid method using image differencing to detect areas of change then applying a post-classification to determine cover classes (Pilon *et al.* 1988; Eastman 2001).

Radiometrically normalised image differencing

This technique involves subtracting the earlier image from the later image to produce a difference image. Radiometric normalisation reduces the degree of difference between the two images that is attributable to changes in sensor and atmospheric conditions. The resultant difference image indicates areas of change and the extent of change between the two images (Yuan *et al.* 1998). Radiometrically normalised image subtraction was performed between the 2001 and the 1997 NDVI images. Areas of change were identified as pixels exceeding one standard deviation from the mean, a commonly used figure for distinguishing actual change from error sources (Yuan *et al.* 1998).

Hybrid post-classification method

Radiometrically normalised image differencing was applied between all 2001 and 1997 Landsat bands. Areas of change were described as pixels greater than one standard deviation from the mean. A Boolean image of unchanged pixels was then produced.

To classify the 1997 image, the signature files used to classify the 2001 image were updated using the Boolean image to exclude any changed pixels. The maximum likelihood classification using the updated signature files was then applied to the 1997 image. To determine the extent of change in terms of condition classes, the 1997 image was subtracted from the 2001 image. A median 5 x 5 filter was applied to the change image to enhance edges and remove noise.

4.3.1.8 Costs of remote condition assessment

Costs to undertake an assessment of vegetation condition in the study area using Landsat ETM imagery were calculated based on costs associated with the current study, excluding time for developmental work. This assumes a study area of 6,363 km^2, and ground truthing of approximately 80 points.

4.3.2 Results

4.3.2.1 Pre-processing

Registration

Co-registration of the 2001 image to the previously geo-registered 1997 image was performed. Table 4.9 shows the control points used and the residuals, which represent the deviation of each point from the polynomial equation. Control point five had the highest residual, and by omitting this point, the overall root-mean-square (RMS) was reduced. An overall RMS error of 13.56 m was obtained, less than half the unit distance of a Landsat pixel (30 m), indicating a good correspondence, with registration of the two images within half of one pixel (Gibson and Power 2000).

Table 4.9. Correspondence points and residuals used to co-register Landsat images from different dates.

	1997 image		2001 image		
Point no.	**X**	**Y**	**X**	**Y**	**Residual**
1	539154	6192671	539122	6192685	10
2	564071	6206781	564048	6206780	6
3	514993	6204027	514940	6204032	10
4	568587	6181020	568591	6181046	16
5	521062	6178110	521004	6178068	omitted (23)
6	545522	6181030	545495	6181018	21
7	530675	6207060	530645	6207049	15
8	557039	6196318	557011	6196327	9

The Road25 layer (Appendix 1) was overlayed on the 2001 image for a visual check of registration accuracy and a good match of roads and intersections was found between the imagery and the Road25 layer.

Radiometric calibration

Offset and gain values were calculated from invariant targets for each band of the 1997 image from the 2001 image (Table 4.10). Similar results were obtained using the CALIBRATE method. The CALIBRATE method was chosen and all bands were calibrated using this technique.

Table 4.10. Gain and offset values calculated from invariant targets for radiometric calibration of Landsat satellite imagery.

Band no.	Offset	Gain	R^2	No. invariant points
1	1.42	7.79	0.82	18
2	2.36	20.51	0.80	21
3	2.70	10.69	0.94	19
4	1.48	10.46	0.89	20
5	1.83	12.75	0.97	19
6	2.36	11.83	0.98	19

Figure 4.6 shows band 3 of the 2001 image compared to the 1997 image before and after radiometric calibration using the CALIBRATE method.

4.3.2.2 Classification

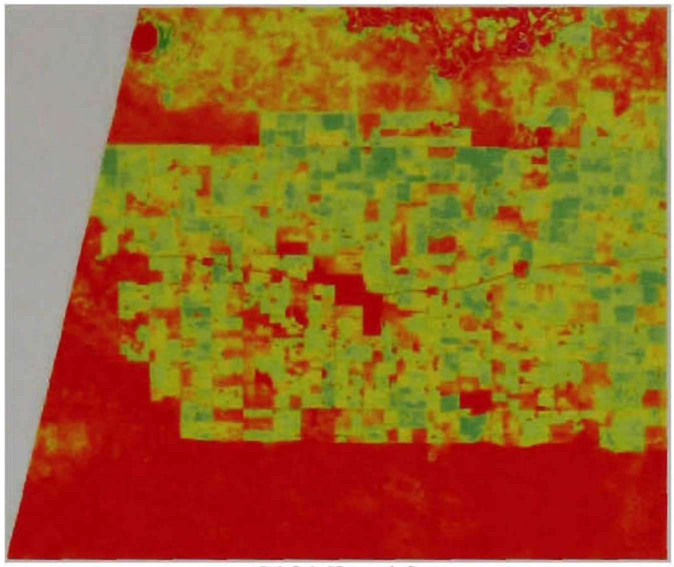
Unsupervised classification

The ISOCLUST unsupervised classification produced two clusters associated with semi-arid woodlands, clusters 2 and 19 (Figure 4.7). Whilst cluster 19 showed strong association with semi-arid woodlands, large areas of woodland were not included in this group. Considerable amounts of mallee eucalypt vegetation are observed within cluster 2 (Table 4.11).

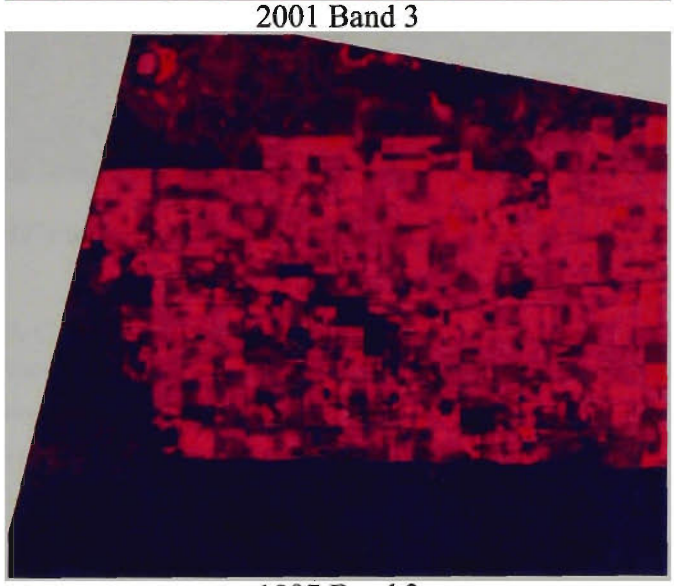
A similar result was produced with the CLUSTER algorithm, with semi-arid woodlands not adequately distinguished from mallee communities on visual inspection of the image.

Supervised classification

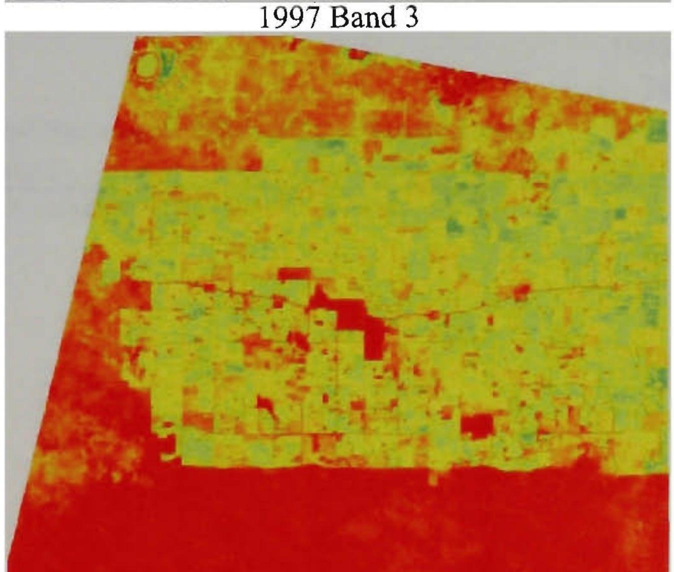
The maximum likelihood classification appeared to be a good representation of the distribution of semi-arid woodland within the MSNP. However, many areas of agricultural land known to be cleared were classified as semi-arid woodland, suggesting that sparsely vegetated semi-arid woodlands were not spectrally significantly different from sparsely vegetated agricultural land. This was confirmed by the signature comparison chart, which showed minimal separation of semi-arid woodlands from other cover classes, particularly agricultural land one (green) and mallee communities (Figure 4.8). A good correlation was found between the maximum likelihood classification of semi-arid woodland and semi-arid woodland EVCs on public land (Table 4.12).



2001 Band 3



1997 Band 3



1997 Band 3 after calibration

Figure 4.6. Band 3 of 2001 image and 1997 image before and after radiometric calibration using the CALIBRATE procedure.

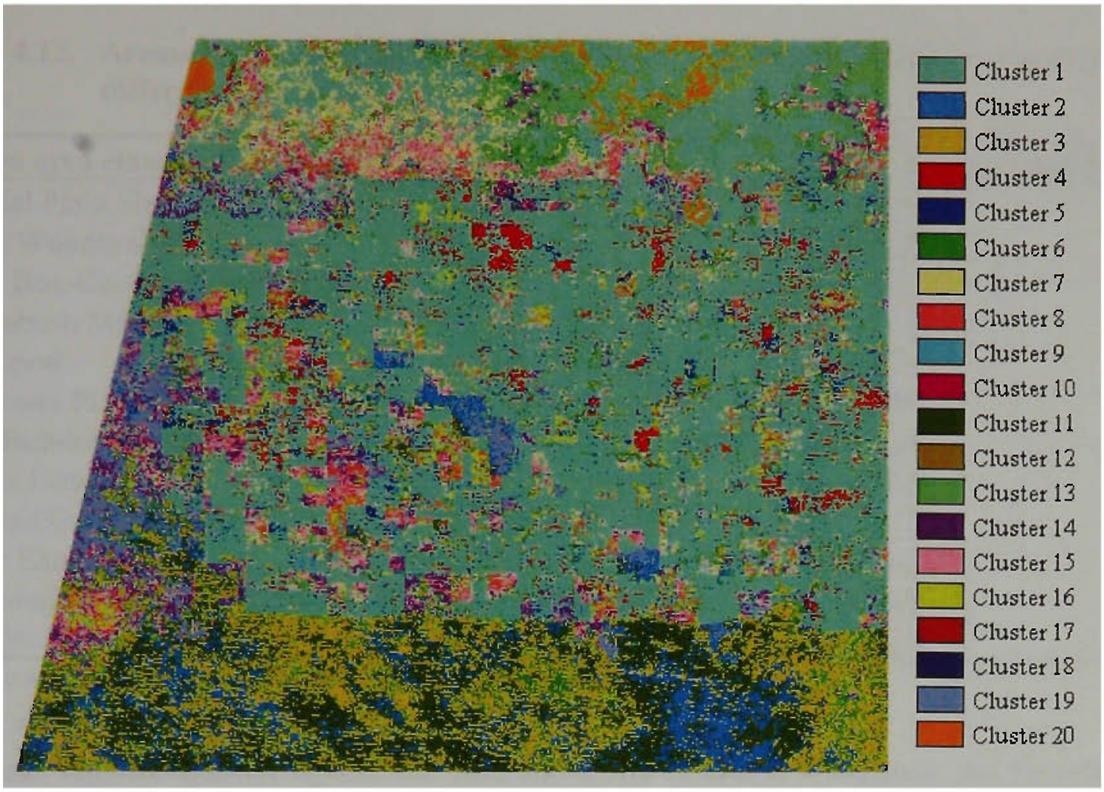


Figure 4.7. ISOCLUST unsupervised classification of land cover classes within the study area.

Table 4.11. Area of EVCs mapped as clusters 2 and 19 (semi-arid woodland) in unsupervised ISOCLUST classification of Landsat satellite image.

EVC	Area	% in study area
Belah Woodland	**986**	**28**
Chenopod	930	3
Gypseous Plain Woodland	**122**	**50**
Pine-Buloke Woodland	**5**	**12**
Private Land	1598	1
Saline Shrubland	24	7
Savannah Woodland/Savannah Mallee/Grassland Mosaic	**4,155**	**23**
Woorinen Sands Mallee	178	0

EVCs containing semi-arid woodland are presented in **bold.**

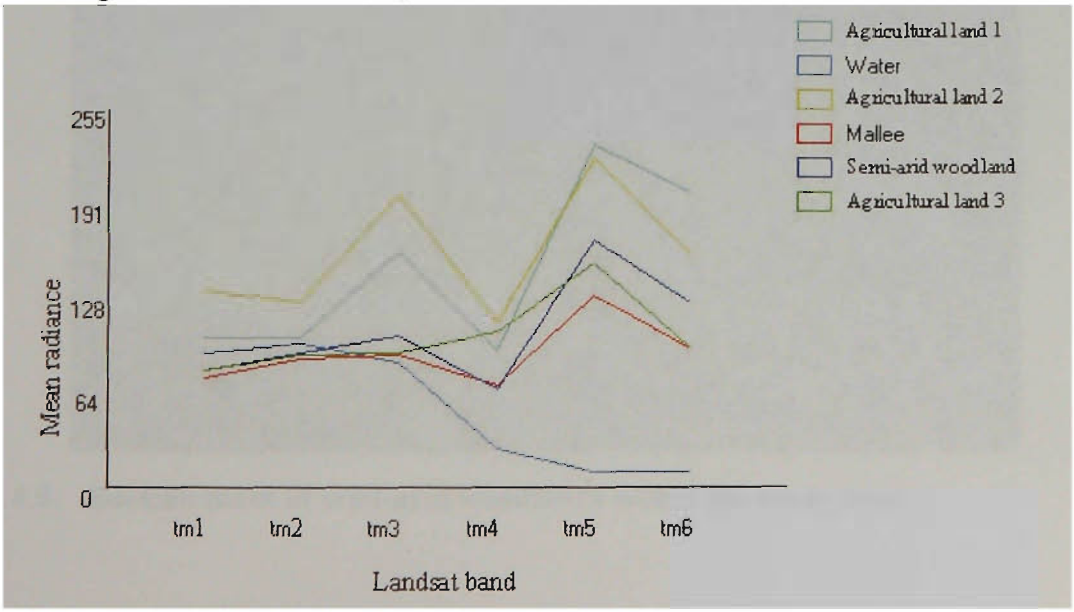


Figure 4.8. Plot of mean radiance values for signature files for land cover classes.

Table 4.12. Areas classified on 2001 Landsat image as semi-arid woodland occurring within different EVC types.

Within area classified as Belah woodland	Total area	% Total EVC
Alluvial Plain Shrubland	1,409	28
Belah Woodland	**3,035**	**87**
Black Box-Chenopod Woodland	342	11
Broombush Mallee	15	0
Chenopod	6,439	21
Gypseous Plain Woodland	**227**	**94**
Pine-Buloke Woodland	**34**	**76**
Private Land	34,485	17
Riverine Grassy Forest	12	2
Saline Shrubland	287	86
Savannah Woodland/Savannah Mallee/Grassland Mosaic	**15,819**	**87**
Woorinen Sands Mallee	1,529	3

EVCs containing semi-arid woodland are presented in **bold.**

To better separate areas of agricultural land and sparse semi-arid woodlands, the Tree100 layer was used to exclude all treeless areas from the semi-arid woodland class within private land. Areas of semi-arid woodland were then extracted from the land cover classification to produce a Boolean image of semi-arid woodland (Figure 4.9). This was then used as a mask on the image to exclude all areas that do not contain semi-arid woodlands.

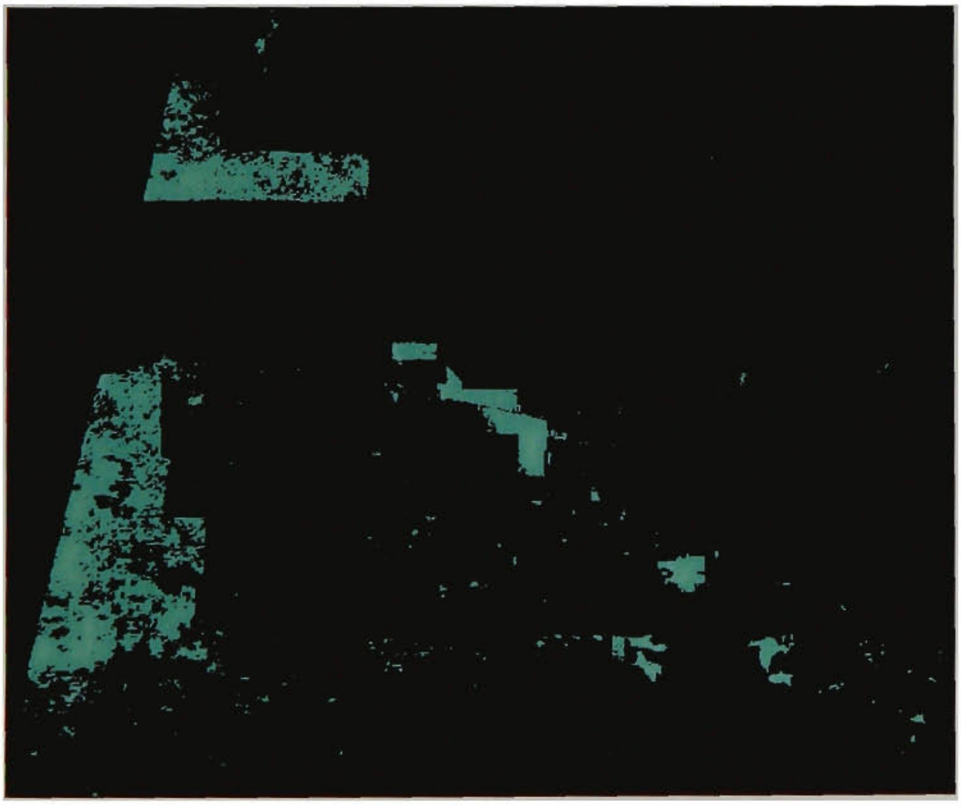


Figure 4.9. Boolean mask of semi-arid woodlands within the study area.

Classification of condition classes

To determine if semi-arid woodland could be further separated into condition classes, unsupervised and supervised classifications were performed. Figure 4.10 shows the result of an unsupervised ISOCLUST classification to three classes. The results of a supervised maximum likelihood classifier using training areas based on field condition assessment are shown in Figure 4.11.

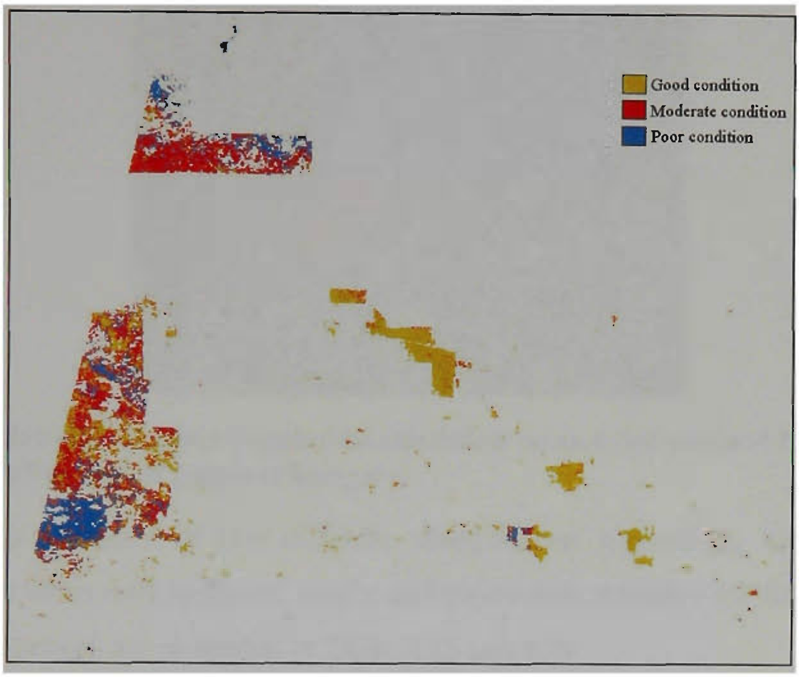


Figure 4.10. Unsupervised classification of semi-arid woodlands showing three woodland classes.

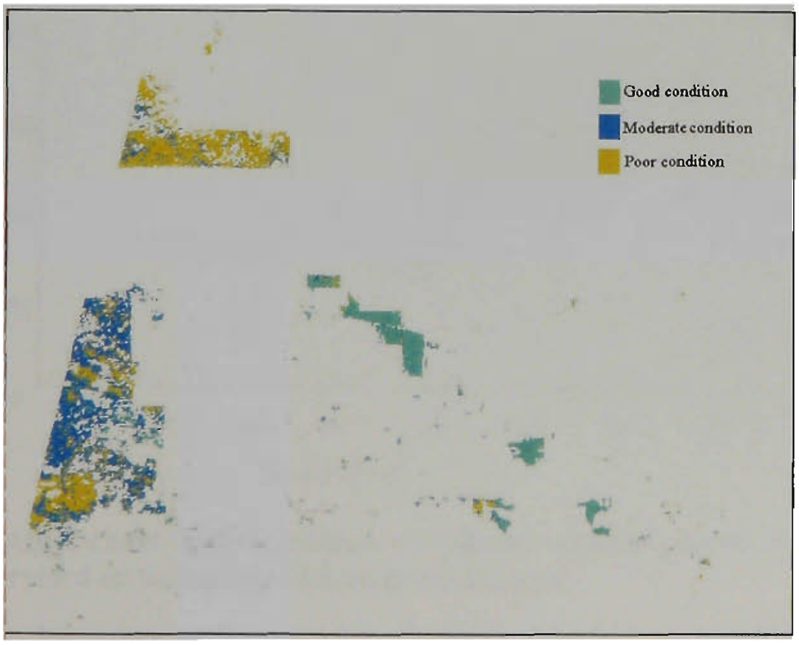


Figure 4.11. Supervised classification of three classes of semi-arid woodland vegetation condition.

To determine the separability of vegetation condition classes, the signature files were examined using scatter plots (Figure 4.12) and charts of mean reflectance for vegetation condition classes

(Figure 4.13). Figure 4.12 shows that condition classes one and three are spectrally distinct on bands 3 and 5, but both show some overlap with condition class two. Signatures for condition classes are most separable in bands 3, 5 and 6 (Figure 4.13).

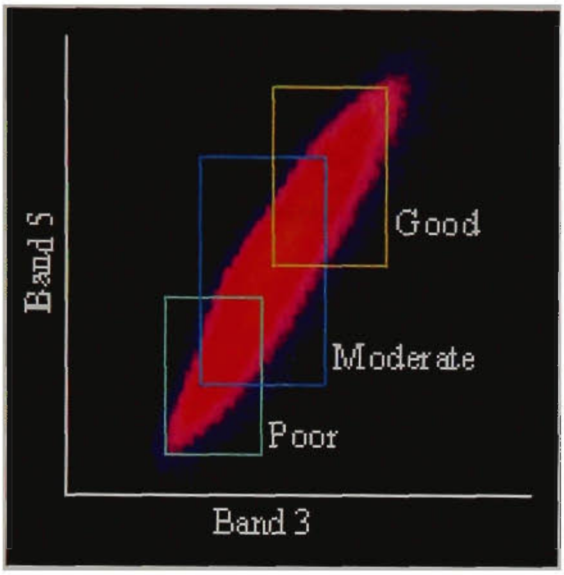


Figure 4.12. Scatter plot of three vegetation condition classes determined by supervised classification of Landsat imagery.

To determine the accuracy of the different classification algorithms, error matrices were calculated. Results of overall accuracy, user's and producer's accuracy for the unsupervised and supervised classifications are presented in Table 4.13 and 4.14.

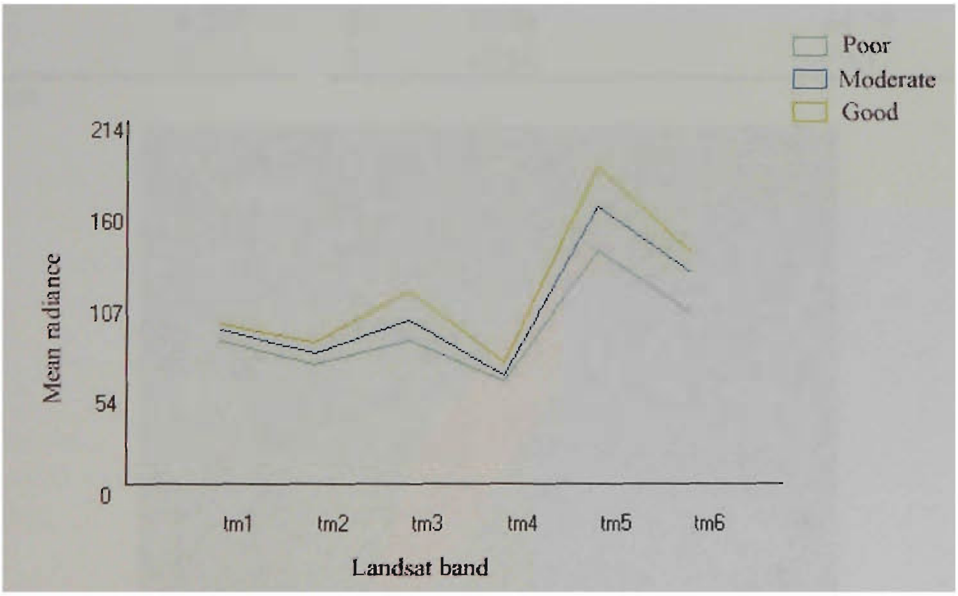


Figure 4.13. Signature chart of mean radiance of three condition classes determined by supervised classification of Landsat imagery.

As accuracies were low when three condition classes were classified from the Landsat data, the data were re-classified into two condition classes. Figure 4.14 shows that the signature files for two condition classes overlap only marginally in bands 3 and 5. Error matrices for supervised and unsupervised classification of semi-arid woodland condition are presented in Table 4.15 and 4.16 respectively.

Table 4.13. Error matrix and accuracy assessment of three condition classes of semi-arid woodlands derived from unsupervised classification of Landsat data.

Image data	Reference data 1	2	3	Total
1	15	17	4	36
2	1	14	16	31
3	0	3	8	11
Total	16	34	28	78

		Class	Producer's accuracy %	User's accuracy %
Overall accuracy	47.44			
Kappa	0.229	1	93.75	41.67
Z score	2.952	2	41.18	45.16
Result[a]	S	3	28.57	72.73

[a]At the 95 % confidence level, S = Significant.

Table 4.14. Error matrix and accuracy assessment of three condition classes produced from supervised classification of Landsat data.

Image data	Reference data 1	2	3	Total
1	15	11	0	26
2	1	18	9	28
3	0	5	19	24
Total	16	34	28	78

		Class	Producer's accuracy %	User's accuracy %
Overall accuracy	66.67			
Kappa	0.499	1	93.75	57.69
Z score	6.207	2	52.94	64.29
Result[a]	S	3	67.86	79.17

S = Significant.

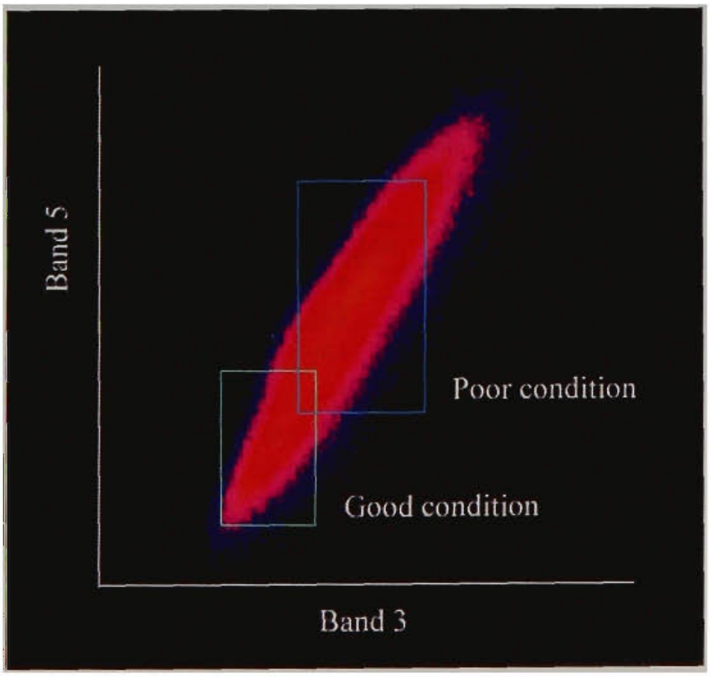


Figure 4.14. Scatter plot of two vegetation condition classes determined by supervised classification of Landsat imagery.

Table 4.15. Error matrices and accuracy assessment for two condition classes produced from ISOCLUST unsupervised classification of Landsat imagery.

Image data	Reference data 1	2	Total
1	29	8	37
2	2	39	41
Total	31	47	78

Overall accuracy	87.18
Kappa	0.741
Z score	9.805
Result	S

Class	Producer's accuracy %	User's accuracy %
1	93.55	78.38
2	82.98	95.12

To determine methods for replicable assessment of condition, vegetation indices were investigated. Table 4.17 shows the strong correlations obtained between vegetation indices and condition, as well as other parameters.

Table 4.16. Error matrix and accuracy assessment of supervised classification of semi-arid woodlands showing two condition classes.

Image data	Reference data 1	2	Total
1	23	0	23
2	8	47	55
Total	31	47	78

Overall accuracy	89.74
Kappa	0.776
Z score	10.614
Result[a]	S

Class	Producer's accuracy %	User's accuracy %
1	74.19	100.00
2	100.00	85.45

[a]At the 95 % confidence level, S = significant.

Table 4.17. Spearman's correlation of field parameters and condition index, with vegetation and brightness indices calculated from Landsat imagery.

Vegetation parameter	brightness356	tasscapgreen2	NDVI_34
% cover trees	-0.63**	0.68**	0.72**
% cover bare ground	0.74**	-0.74**	-0.69**
% cover shrubs	-0.57**	0.62**	0.68**
% cover perennial species	-0.64**	0.71**	0.78**
field Condition Index	-0.68**	0.76**	0.81**
tree sp richness	-0.46**	0.48**	0.51**
tall shrub sp richness	-0.63**	0.69**	0.65**
small shrub sp richness	-0.53**	0.57**	0.60**
regeneration shrub species	-0.42**	0.48**	0.52**

**Correlation is significant at the .01 level (2-tailed), N=78

The NDVI was selected for further investigation due to the high correlations with the field condition parameters. Figure 4.15 shows mean and standard deviation of NDVI values for quadrats of the three condition classes surveyed in April 2003. It appears that poor condition sites may be separable from good condition sites, but some overlap between moderate condition sites may occur with both good and poor condition sites.

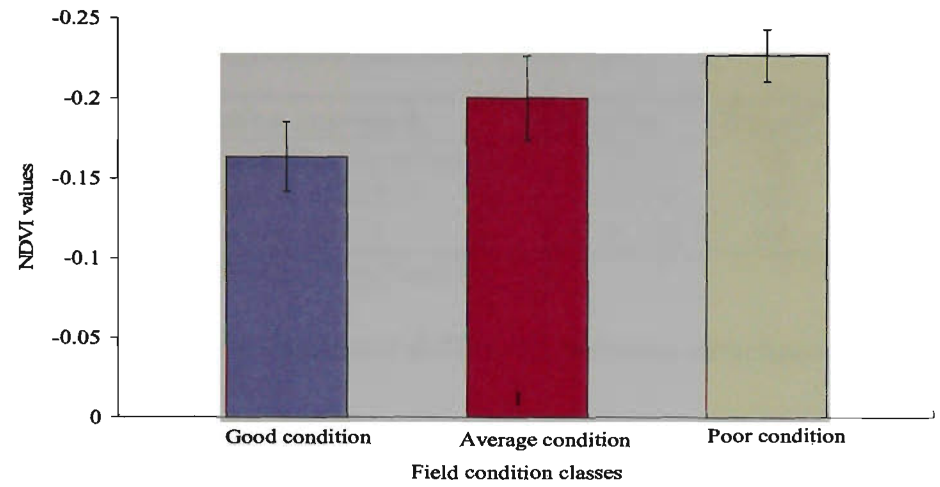


Figure 4.15. Mean (± 1 SE) NDVI values for quadrats in condition classes good, moderate and poor.

Based on this, the NDVI data were separated into two condition classes based on mean NDVI, plus or minus one standard deviation. Table 4.18 presents error matrices calculated from the NDVI classes and April 2003 ground truthing data.

Table 4.18. Error matrix and accuracy assessment for two condition classes derived from vegetation indices.

Image data	Reference data 1	2	Total
1	26	4	30
2	5	43	48
Total	31	47	78

Overall accuracy	88.46
Kappa	0.758
Z score	9.997
Result[a]	S

Class	Producer's accuracy %	User's accuracy %
1	83.87	86.67
2	91.49	89.58

[a]At the 95 % confidence level,. S = Significant.

To determine which was the most accurate of the two condition class solutions, Z scores of the supervised classification, unsupervised classification and NDVI classification were compared. Table 4.19 and 4.20 show no significant differences were found between error matrices of the different classification approaches when tested using either method.

4.3.2.3 Change detection

Image subtraction of the 2001 NDVI image from the 1997 NDVI image was applied to detect changes in NDVI over time. As expected, minimal change was detected between the two images, with 87.48 % of pixels unchanged. Figure 4.16 shows the extent of change detected in NDVI greater than one standard deviation from the mean.

Table 4.19. Z score analysis to test for significant differences between accuracy of different approaches of vegetation condition assessment.

Classification approach	Z Statistic	Result[a]
Unsupervised Vs. Supervised	0.334	NS
Unsupervised Vs. NDVI	0.158	NS
Supervised Vs. NDVI	-0.173	NS

[a]At the 95 % confidence level, NS = Non-Significant

Table 4.20. McNemar test for significant differences between techniques trialed to assess vegetation condition.

Classification approach	P value	Result
Unsupervised Vs. Supervised	0.424	NS
Unsupervised Vs. NDVI	0.453	NS
Supervised Vs. NDVI	1.000	NS

NS = Non-significant

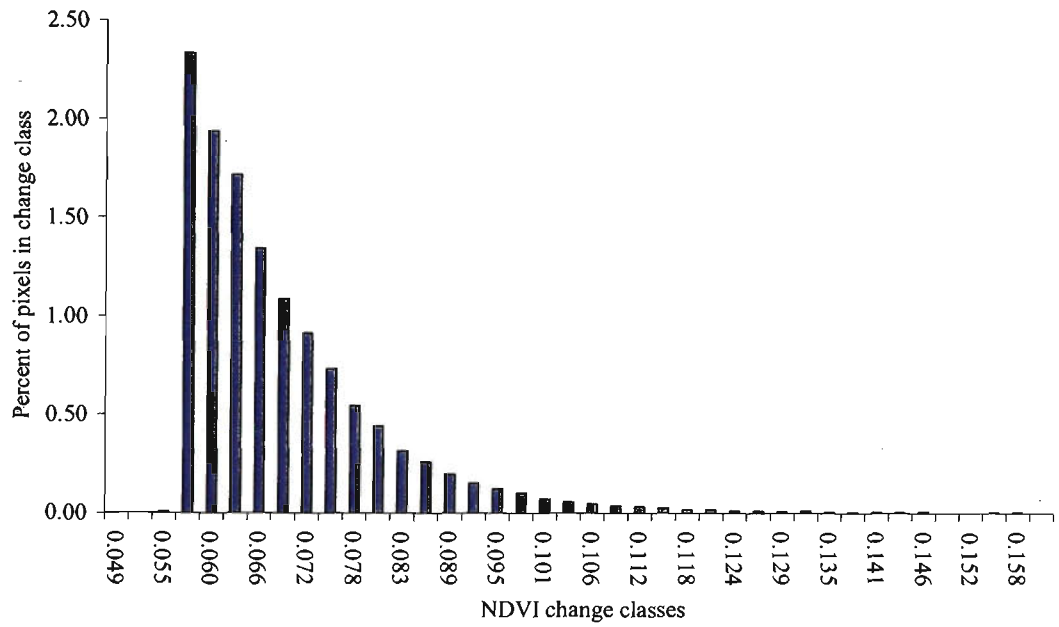


Figure 4.16. Change in NDVI determined from radiometrically normalised image subtraction of 2001 minus 1997 image.

A hybrid method of change detection (Section 4.3.1.7) was also trialed to detect change in condition classes. Figure 4.17 shows changes in condition classes detected following hybrid change detection techniques. Following application of a 5 x 5 median filter the amount of change detected decreased, with 0.27 % of pixels increasing by one condition class and 0.85 % of pixels decreasing by one condition class.

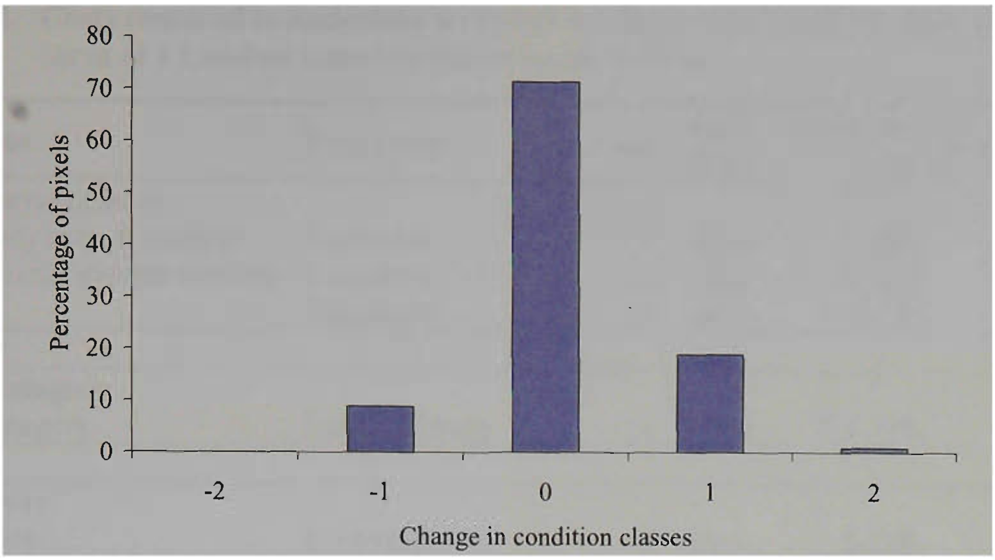


Figure 4.17. Change in condition classes calculated using hybrid change detection methods with a supervised classification approach.

4.3.2.4 Condition of semi-arid woodland as determined from satellite imagery classification

Satellite imagery classification provided an indication of vegetation condition within 3,200 km^2 of the study area. Table 4.21 shows the area of classified as good and poor condition using the three main methods investigated.

Table 4.21. Area (ha) classified as good and poor condition based on satellite imagery classification techniques to two classes (good and poor).

Condition class	Unsupervised classification	Supervised classification	NDVI
Good	7,289	4,282	6,747
Poor	11,901	14,909	12,443

4.3.2.5 Costs to undertake a remote condition assessment

Costs to undertake an assessment of vegetation condition in the MSNP were calculated based on costs associated with the current study, with time for developmental and exploratory methods excluded (Table 4.22). If other medium scale satellite data were employed, costs for sections other than for imagery acquisition would be expected to be very similar. If larger scale imagery (eg. IKONOS) were used, time for analysis may be greater due to increased pre-processing time to register, join and radiometrically balance images. One Landsat scene covers an area of 184 x 185 km (41,625 km^2), however, more than one scene may be required to cover a large area due to the "footprint" of the satellite, which may not encompass the whole study area.

Table 4.22. Costs required to undertake a remote condition assessment of up to 41,000 km² (area of 1 Landsat scene) based on Landsat data.

Task / Item	Resources	Unit	Unit Type	Cost per Unit	Total Cost
Project Development					
Define study area & method	1 scientist	2	days	$ 680	$ 1,360
Preparation for ground truthing	1 scientist	2	days	$ 680	$ 1,360
	1 assistant	1	days	$ 430	$ 430
Sub-total					**$ 3,150**
Satellite Imagery					
Acquire imagery	Landsat image	1	scene	$ 1,500	$ 1,500
Sub-total					**$ 1,500**
Field Survey					
Field Survey	1 scientist	5	days	$ 680	$ 3,400
	1 assistant	5	days	$ 430	$ 2,150
Travel	travel	2,000	km	$ 0.55	$ 1,100
	Meals/accom.	4	nights	$ 220	$ 880
Data entry	1 assistant	2	days	$ 430	$ 860
Sub-total					**$ 8,390**
Imagery Analysis					
Pre-processing	1 scientist	3	days	$ 680	$ 2,040
Condition assessment	1 scientist	12	days	$ 680	$ 8,160
Accuracy assessment	1 scientist	5	days	$ 680	$ 3,400
Sub-total					$ 13,600
Total cost					**$ 26,640**

4.3.3 Discussion

The map of semi-arid woodland extent (Figure 4.9) was derived to enable exclusion of other land cover classes from the image. The accuracy of semi-arid woodland extent was not evaluated. A study is currently underway to map EVCs of the northwest Victoria on all land tenures (EVC100), and this mapping could be used to provide a more accurate map of semi-arid woodland extent. This may impact on the extent of the area mapped as semi-arid woodland, but would not affect the methods or results obtained for the vegetation condition assessment.

Classification of semi-arid woodland vegetation condition

Results for classifications of three vegetation condition classes were not highly accurate (overall accuracy 78 % with supervised classification), however, classification accuracies were improved by decreasing to two classes (overall accuracy 89.7 % with supervised classification).

Based upon the overall accuracy scores, the classification approaches, in order of accuracy were the supervised classification, NDVI and unsupervised classification. However, no significant difference was observed in the Kappa statistic between techniques. Z scores and p values between matrices show no significant differences between the methods. This suggests that, if mapping condition classes, only two condition classes can be accurately classified with Landsat data. Of the various techniques used to classify two condition classes, there was no significant difference in accuracy between techniques, therefore the simplest method, and fastest method, either unsupervised classification or the NDVI should be used.

Strong correlations were found between the vegetation indices derived from Landsat imagery and condition parameters measured, including the field vegetation condition index (0.81 correlation with NDVI), % cover perennial species (0.78 correlation with NDVI), % cover bare ground (-0.74 correlation with tasselled cap) and % cover trees (0.72 correlation with NDVI). This shows that the NDVI explains 81% of the variability in condition index across the study area, which means that the NDVI has potential as a substitute for field condition assessment. From the NDVI, or similar vegetation indices, cover of perennial species, cover of trees, and cover of bare ground can be determined.

However, the NDVI is likely to be highly influenced by seasonal conditions. Following rainfall, growth of annual vegetation would have a large effect on the NDVI, with the potential for areas dominated by exotic annual vegetation to appear in good condition. This can be minimised by acquiring imagery in late summer and avoiding periods following large rainfall events.

Change detection

Two change detection approaches were investigated, however very little change was found between the 1997 and 2001 images. This lack of change over four years is not surprising given the episodic and slow nature of vegetation change in semi-arid regions (Stafford Smith and Morton 1990). Due to the lack of ground truthing data the accuracy of change detection approaches was not determined.

Advantages of using Landsat data for vegetation condition assessment

Some of the advantages of using Landsat satellite imagery for condition assessment include obtaining a complete data set for the area to enable production of a map of vegetation condition. The study can be reproduced in the future, and change in vegetation condition over time can be investigated. It is also possible to acquire imagery from any time to the early 1970s to enable investigation of vegetation condition change over the last 30 years. The use of satellite imagery also reduces field survey time and associated costs.

Disadvantages of using Landsat data for vegetation condition assessment

Compared with field survey studies, the resolution of data is much reduced. The type of parameters that can be measured therefore is greatly decreased. Landsat data will not provide data on individual plants, species richness, age class distribution or many other parameters recorded in the field based condition assessment. However, this study has shown that vegetation indices derived from Landsat data are highly correlated to many of these parameters.

To undertake a condition assessment based on remotely sensed data there are a number of initial set up costs including computer hardware and software. Expertise is also required to undertake the assessment.

4.4 USE OF AERIAL PHOTOGRAPHY FOR CONDITION ASSESSMENT

Aerial photography was the first available remotely sensed data source and is still commonly used in vegetation monitoring, particularly where high spatial resolution is required. Archived aerial photographs for the study area begin in 1941, with further images taken in most decades through to the 1980s. However, the spatial coverage is incomplete, and the temporal coverage is haphazard, and unpredictable (Table 4.23). Most photographs are in black and white. The scale of the photos is not consistent between years, with scales from 1:34,000 to 1:105,000. Different cameras and focal lengths have also been used for aerial photography in different years.

Table 4.23. Existing aerial photography available in the study area (Geoscience Australia 2003b).

Flight Path Name	Date	Colour or B&W	Approx Scale	Focal Length/Camera
Mildura	Oct-68	B&W	1:85,260	RC 9 373/ RC 9 629
Gol Gol area	Feb-41	B&W	unknown	EIV
Wentworth	Feb-41	B&W	unknown	+
Fletcher's Lake	Feb-41	B&W	unknown	E1M
Lindsay	Dec-53	B&W	1:40,000	E9 152.75 mm
Mildura	Oct-84	B&W	1:86,000	RC10 87.98 mm
Mildura	Dec-53	B&W	1:40,000	E9 152.75 mm
Wentworth	Dec-53	B&W	1:40,000	E9 152.75 mm
Mildura	Sep-88	B&W	1:105,000	WILD RC 10 27.42 mm
Nowingi-Kia	Dec-53	B&W	1:40,000	E9 152.75 mm
Mildura	Apr-77	B&W	1:34,000	RC9-616
Mildura	Feb-84	Colour	1:40,000	RC 10 151.5 mm
Mildura	Apr-83	B&W	1:85,000	RC 10 87.98 mm
Mildura	Dec-81	B&W	1:51,700	RC 9-373
Mildura	May-72	B&W	unknown	unknown
Carwarp-Mildura	Dec-53	B&W	1:40,000	E9 152.74 mm

The most recent aerial photography was taken in 1988, and so it was necessary to obtain up to date aerial photography of the study area. Many choices for condition assessment exist within aerial photography, with options of camera format, film, scale and measurement variables and techniques. One of the cheapest options for aerial photography is small-format photography, where a standard camera (non-metric) is used, commonly within a single engine aircraft. The camera can either be hand held, for oblique photography, or mounted in the undercarriage for vertical photographs. This also enables large-scale photographs to be taken which can provide much detail for vegetation monitoring.

4.4.1 Methods

4.4.1.1 Image acquisition

Aerial photography was taken on 28 May 2001 using a Nikon D1 digital camera mounted in the belly of a high wing Cessna 182RG aircraft. The camera was remotely controlled and slaved to a Nikon motor drive to enable photographs to be taken by the pilot in flight. The Nikon D1 is a 2.7 megapixel digital camera with a 23.7 x 15.6 mm, 12 bit RGB CCD (charge coupled device) delivering 2000 x 1312 pixel images. The lens used was a Nikkor 14 mm F2.8 with a 78-degree angle of view.

The camera was rigid mounted to a camera mount installed on the co-pilots seat rails enabling in-flight pan and tilt camera adjustments. The camera was then positioned over the aircraft's camera hatch providing a vertical view of the landscape below. A retractable undercarriage provided the opportunity for an uninterrupted view of the ground below as well as providing greater stability for the photographic platform when retracted.

A cloud free day was chosen for the aerial photography, however, by midday some cloud cover appeared resulting in shadows on some photographs. A ground speed of 100 knots (185.19 km/hr) was planned for the aerial photography flight, however, due to upper wind and drift this speed was not always achieved.

Height was determined from the aircraft's altimeter which was calibrated using area QNH determined by Flight Service in Melbourne with an estimated error of plus or minus five HPA (45.72 m). Aerial photographs were taken from three heights to determine optimal scale for condition assessment. Nominal heights were selected, based on a minimum safe flying height of 1,000 feet (304.8 m) above ground level. Two other heights were investigated; 2,000 feet (609.6 m) and 4,000 feet (1219.2 m) above ground level.

Aerial photographs were taken over four locations, chosen to cover semi-arid woodlands in a variety of conditions as determined by earlier ground-based condition assessment (Figure 4.18). Images were saved as Joint Photographic Expert Group (JPEG) files which give compression with minimal geometric or visual degradation (Lammi and Sarjakoski 1995).

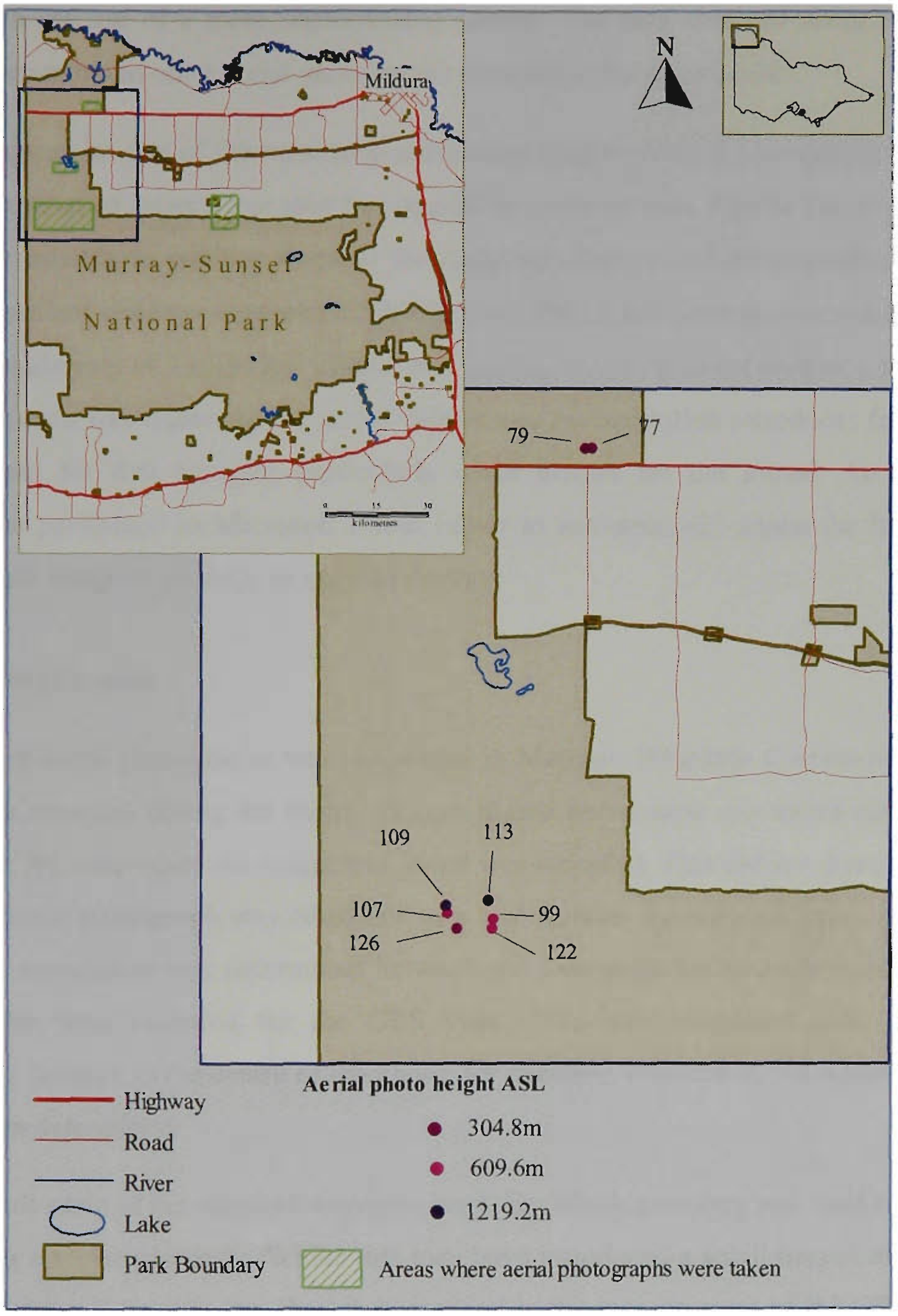


Figure 4.18. Location of aerial photographs used to investigate parameters for remote vegetation condition assessment.

An eight channel Garmin 12 XL GPS unit was carried within the aircraft with waypoints measured as part of a tracking function. An aerial attached to the inside of the rear window ensured satellite coverage throughout the flight. Under the tracking function more waypoints are recorded when the aircraft changes direction and fewer points are recorded when the aircraft is travelling in a consistent direction. Altitude may be recorded as part of the tracking function of some GPS units but was not recorded during this trial.

Many sophisticated systems of linking GPS to cameras exist, however this low-tech solution ensured low cost photo acquisition. If required, the spatial accuracy of the photos could be improved with the use of a more sophisticated system. The data obtained using this system is otherwise comparable to other aerial photographs obtained at the same scale.

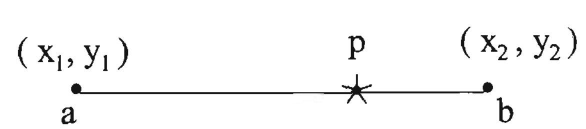
Pre-processing can be one of the most time consuming stages of aerial photography analysis due to the large number of images that may be required to cover an area. Due to the size of the study area, it is not feasible to produce maps of the study area from aerial photography at this scale. Even at the smallest scale used (approx 1:50,000) more than 5,800 photographs would be required to cover the study area of 13,930 km^2. Therefore, joining images was not seen as a priority, and a sampling approach was undertaken. Lack of radiometric normalisation introduces few limitations on the methods for data analysis, particularly when images are not joined. An auto balance procedure was performed in Microsoft Photo Editor to automatically adjust the brightness and contrast of each image to produce an optimal display.

4.4.1.2 Registration

Registration of aerial photographs was performed in MapInfo (MapInfo Corporation), based on the GPS track recorded during the flight. As each digital photograph was saved onto the camera memory card, the time when the image was saved was recorded. This did not directly respond to the time the aerial photograph was taken due to a few seconds lag between exposure and saving the image. A correlation was determined between the time recorded as each digital image was stored and the time recorded for the GPS track. This was calculated using photos with distinguishing features in the centre of the photo, for example, crossroads, for which co-ordinates could easily be determined.

Given the small-scale of the aerial photographs used, Euclidean geometry was used rather than the more complex circular geometry. Whilst this may have introduced a small degree of error, it was considered that this error was less than that involved in the measurement of the GPS waypoints. To calculate the central co-ordinate of the photo from the time the photo was taken and the GPS track, a formula was applied:

Co-ordinates of aerial photo p = $[X_1 - (X_1-X_2)]*[(a-p)/(a-b)], [y_1 - (y_1-y_2)]*[(a-p)/(a-b)]$



(GPS track from waypoint a to waypoint b, with aerial photo taken at point p.)

Rotation of each image to position north at the top of each image was performed using Microsoft Photo Editor. The direction of rotation was determined from the direction of flight at the time the photos were taken. To calculate the scale of the photographs and hence the ground coverage of each of the images, the following equation was applied:

Dimensions on the ground = (CCD dimensions x height AGL) ÷ focal length

Height above sea level was determined by the aircraft altimeter. Height of ground level was then determined using the Cont25 map layer (Appendix 1).

4.4.1.3 Selection of aerial photos for testing methods

Eight photos were selected representing at least two of the three photo heights ASL, to test the methodology. Photographs were chosen where road intersections or other identifiable features enabled location of ground control points, and registration of the images. Photos were excluded where the dominant vegetation did not consist of semi-arid woodlands.

Spatial accuracy of photo location

To assess the accuracy of the method in locating aerial photographs on the ground and determining scale, ground control points were collected using a Garmin Etrex GPS. Ten ground control points were collected for each image at distinctive features that could be identified the aerial photograph. The aerial photographs were then geo-registered using these ground data. Errors using non-differential GPS would be expected to be between seven to fifteen meters (Garmin 2003), therefore distances between identifiable points were also measured using measuring tape, or the vehicle odometer where distances were over 1 km. Scale and location of the images were determined using ground data and results of the methods were compared.

4.4.1.4 Techniques and parameters for aerial photography interpretation

To determine potential measures for vegetation condition assessment, parameters from the field survey were selected that could be assessed from aerial photography. Overstorey species were effectively weighted in the field survey by the number of measures applied. Overstorey species are also one of the more distinct features on the aerial photographs, and so five overstorey parameters were chosen to assess the ability of aerial photographs to contribute information on vegetation condition. These parameters were (i) tree species identification, (ii) tree species richness, (iii) tree condition, (iv) tree canopy size, and hence (v) number of age classes present.

Many studies have shown that vegetation cover calculated from aerial photography is a good approximation of vegetation cover on the ground (Tueller *et al.* 1988; Knapp *et al.* 1990; Soule and Knapp 1999; Fensham 2002). Three techniques of cover determination from aerial photography were compared to on-ground measures, on-screen digitising, supervised and unsupervised classification. Sample plots of 50 x 50 m were used as a means to capture variability and minimise use of edges within the aerial photographs.

On screen digitising

On screen digitising and analysis enables the interpreter to zoom in on features, but retain perspective of the entire photo. On screen digitising enables cover values of certain elements to be calculated, identification of features and counting of objects.

For interpretation of tree canopy and cover measures, on screen digitising was selected as one of the simplest methods to use with digital photographs, enabling zooming in on a feature for maximum detail, and the rescaling for better perspective across the photograph. Tree canopies were digitised on screen, usually approximated to a circular shape. Information on vegetation parameters as interpreted from the aerial photographs was added to a GIS layer.

Unsupervised classification

An unsupervised classification procedure was used to attempt to determine cover of trees, shrubs, bare ground, and shadow. An ISOCLUST procedure was applied in Idrisi32 (Clark Labs). To determine land cover categories on the aerial photographs 14 clusters were classified, and examined.

Supervised classification

Supervised classification techniques were trialled using a maximum likelihood classifier. Training areas were defined for trees, bare ground, shadow and ground cover (including cryptogams, litter and annual ground cover). The area of individual shrubs was too small to enable training areas specifically for shrub species. A maximum likelihood classifier was used with all pixels classified and with 1 % of pixels excluded. Following the first classification, training areas were modified, and a second maximum likelihood classification was undertaken (modified supervised classification).

Other parameters

Other parameters that were assessed in the field survey such as shrub regeneration and ground layer intactness were not visually distinct on the aerial photographs, and so were not assessed. Further potential parameters that were not assessed during the field surveys but appeared visually distinct on aerial photographs included bare ground and fallen timber. Some animal (stock and macropod) and vehicle tracks are also visible. Table 4.24 provides a list of potential measures of vegetation condition that may be determined from aerial photography.

Table 4.24. Potential indicators of condition that may be estimated from large-scale small-format aerial photography.

%Cover	Count	Presence/Absence	Categorical
Trees	Trees	Shrubs	Tree condition
Shrubs	Shrubs	Fallen timber	Tree species
Fallen timber	Fallen timber	Dead trees	
Dead trees	Dead trees	Roads or tracks	
Bare ground		Tree species	
Disturbed areas			
Roads or tracks			

4.4.1.5 Condition mapping from aerial photography

To complete a map of vegetation condition from aerial photography interpretation, interpolation-modelling procedures similar to those applied in the field-based assessment need to be applied. Results from the interpolation model would be limited by the same issues identified for field assessment interpolation (Section 4.2.3) and so it is unlikely that the interpolation from aerial photography would exceed the accuracy of field data. Therefore, a complete condition assessment of the study area from aerial photography was not performed.

4.4.1.6 Accuracy assessment and comparison of techniques

To determine the accuracy of aerial photograph interpretation, field measures were recorded for parameters from eight sites. The area of the eight photographs was surveyed on the ground to provide comparative data against which to measure aerial photograph interpretation results. Field data at these sites were collected using the methodology for the field assessment (Section 4.2.1). The species of up to 20 trees occurring within the photo was recorded in the field. The tree condition was recorded, and canopy width was measured at two perpendicular cross sections using a measuring tape. Canopy area was measured using the mean of the two canopy widths.

4.4.1.7 Data cost and availability

To determine the cost of obtaining small-format aerial photography, aerial photography contractors were contacted and asked to provide a quote for performing 100 aerial photos within the study area. Details including the type of camera, film (CIR / Colour / B & W) and GPS location facilities were also obtained.

4.4.2 Results

4.4.2.1 Registration and scale

Table 4.25 shows details of the test photos chosen to compare accuracy of scale and location, and test potential parameters of vegetation condition.

Table 4.25. Height of aerial photos ASL, ground elevation, and height AGL.

Photo no.	Height ASL m	Ground elevation m	Height AGL
77	304.8	70	234.8
79	304.8	70	234.8
126	304.8	35	269.8
99	609.6	25	584.6
107	609.6	40	569.6
109	1219.2	35	1184.2
113	1219.2	25	1194.2
122	1219.2	45	1174.2

To determine the accuracy of using the GPS track and focal length for determining photo scale, calculated measures of scale were compared with ground measures between identifiable points. The scale of A3 printouts shows generally small differences between calculated scale measured with a ruler and ground measures (Table 4.26). The extent of errors measured over 100 m indicates scale errors are generally low, although in one case error is up to 19.3 % (Table 4.27).

Table 4.26. Scale of A3 print-out of aerial photographs calculated from focal length and from ground measures.

Photo no.	Height ASL m	Scale from focal length	Scale from ground measures
77	304.8	1: 964.7	1: 1235.6
79	304.8)	1: 964.7	1: 1195.6
126	304.8	1: 1108.6	1: 959.9
99	609.6	1: 2402.1	1: 2303.8
107	609.6	1: 2353.4	1: 2480.0
109	1219.2	1: 4865.7	1: 4772.7
113	1219.2	1: 4906.8	1: 4902.3
122	1219.2	1: 4824.6	1: 4514.6

On ground coverage from each photo and approximate pixel size are presented in Table 4.28. This information needs to be considered when determining the type of parameters to be distinguished, for example, shrubs of 80 cm diameter are unlikely to be visible on photos at 4,000 feet, but should be visible on a photo at 304.8 m ASL.

A simple method of locating the photos on the ground using an in-flight GPS track was used. On average, this method located the central co-ordinates of photos within 261.9 m of the actual centre of the photograph (Table 4.29).

Table 4.27. Effect of error in scale of aerial photographs calculated by focal length over 100 m on the ground.

Photo no.	Focal length measurement	Error over 100 m
77	78.09	0.78
79	80.70	-19.30
126	115.48	15.48
99	104.26	4.26
107	94.90	-5.10
109	101.95	1.95
113	100.09	0.09
122	106.87	6.87
Average	97.79	6.73

Table 4.28. Mean image dimensions and pixel size for aerial photo images taken from three heights ASL.

Height above sea level	d_1 m	d_2 m	Pixel size cm
304.8 m	275	417	20
609.6 m	643	977	50
1219.2 m	1320	2005	100

Table 4.29. Centre co-ordinates of aerial photographs calculated from focal length and GPS calculations, and from on-ground GPS measures.

Photo	On-ground co-ordinates		Calculated Co-ordinates		Distance between co-ordinates (m)
77	515156.6	6208158	515410.8	6208359	324
79	514759.8	6208195.2	514941.3	6208357	243
126	506157.7	6175155.7	506141.1	6175364	209
99	508381.7	6175874.9	508623.6	6176030	287
107	505286.6	6176237.7	505477.1	6176382	239
109	505368.2	6176758.1	505349	6176938	181
113	508268.5	6177222.9	508274.1	6177297	74
122	508839.6	6175821.3	508548.3	6175370	538
Average					261.9

4.4.2.2 Potential parameters for assessing vegetation condition

To test the accuracy of a sample of potential vegetation condition measures, parameters calculated from aerial photographs were compared to ground measures. Some differences in colour, form and shadow outline were observed between different tree species, enabling differentiation of species using on screen digitising of aerial photographs. Overall accuracy calculated from the error matrix was 72.58 % with a Kappa of 0.598, which is not particularly reliable (Table 4.30).

Table 4.31 shows a good correlation between aerial photograph measures of canopy cover and ground measures at 304.8 m ASL. This relationship is illustrated in Figure 4.19. Canopy measures from greater than 304.8 m do not correlate so well with ground measures Table 4.31.

Table 4.30. Error matrix of tree species identified from aerial photography, compared with ground identification.

Aerial photo identification	Ground identification *Myoporum platycarpum*	*Casuarina pauper*	*Callitris gracilis*	*Alectryon oleifolius*
Myoporum platycarpum	7		5	2
Casuarina pauper		4	1	
Callitris gracilis			12	2
Alectryon oleifolius	3		4	22

Table 4.31. Relationship between tree canopy area calculated from aerial photography and ground measurements of tree canopy.

Height ASL m	Regression equation	R^2	P
304.8	y = 1.06x – 5.91	0.869	<0.001
609.6 & 1219.2	y = 0.54x + 11.01	0.681	<0.001

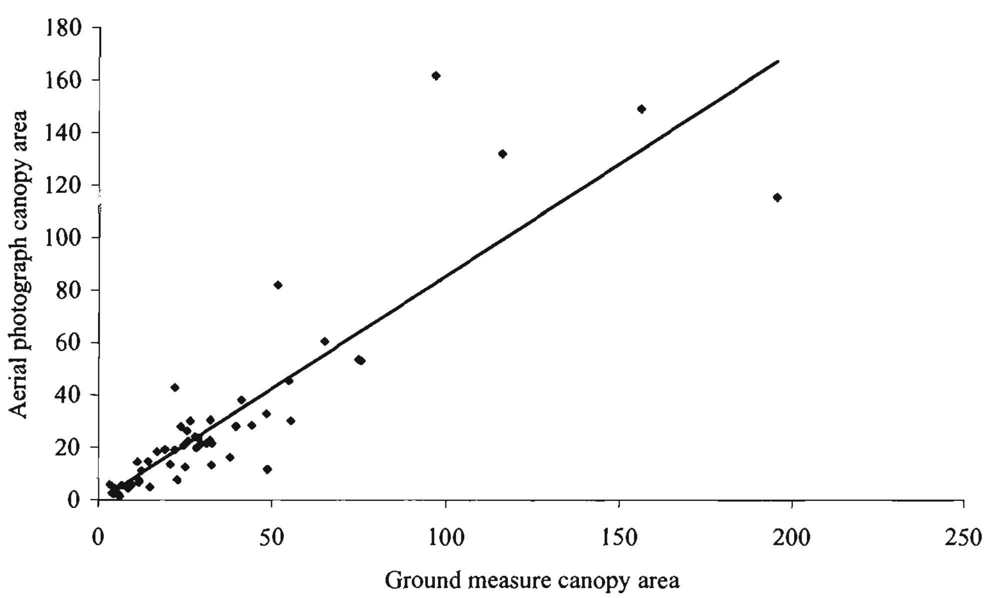


Figure 4.19. Relationship between ground and aerial photograph measure of canopy area at 3.04.8 m (1,000 ft) ASL.

Table 4.32. Error matrix of tree condition estimated from aerial photography compared with ground measures.

AP tree condition scores	Ground survey tree condition scores 1	2	3	4	Total
1	1				1
2	3	10	6	2	21
3	1	9	10	4	24
4		1			1
Total	5	20	16	6	47

Tree condition was not reliably detected from on-screen digitising of aerial photographs, with an overall accuracy 44.68 % and Kappa of 0.123 (Table 4.32). On ground measures of tree canopy area and stem diameter were compared to determine if tree canopy area could assist in determining age classes from aerial photography. Some potential for canopy measures to be used to estimate age classes was found for *A. oleifolius*, and *M. platycarpum*, however no relationship was observed between *C. gracilis* canopy area and stem diameter (Figure 4.20).

Percentage canopy cover for a 50 x 50 m plot was calculated by on screen digitising, with no more than 6 % error when compared with ground estimates (Table 4.33). Canopy cover was not reliably distinguished from shrub cover using supervised and unsupervised classification. In some photos canopy cover determined by supervised and unsupervised classification correlated well with ground estimates of canopy cover (Table 4.34), whilst for others was better correlated with total perennial cover (Table 4.35).

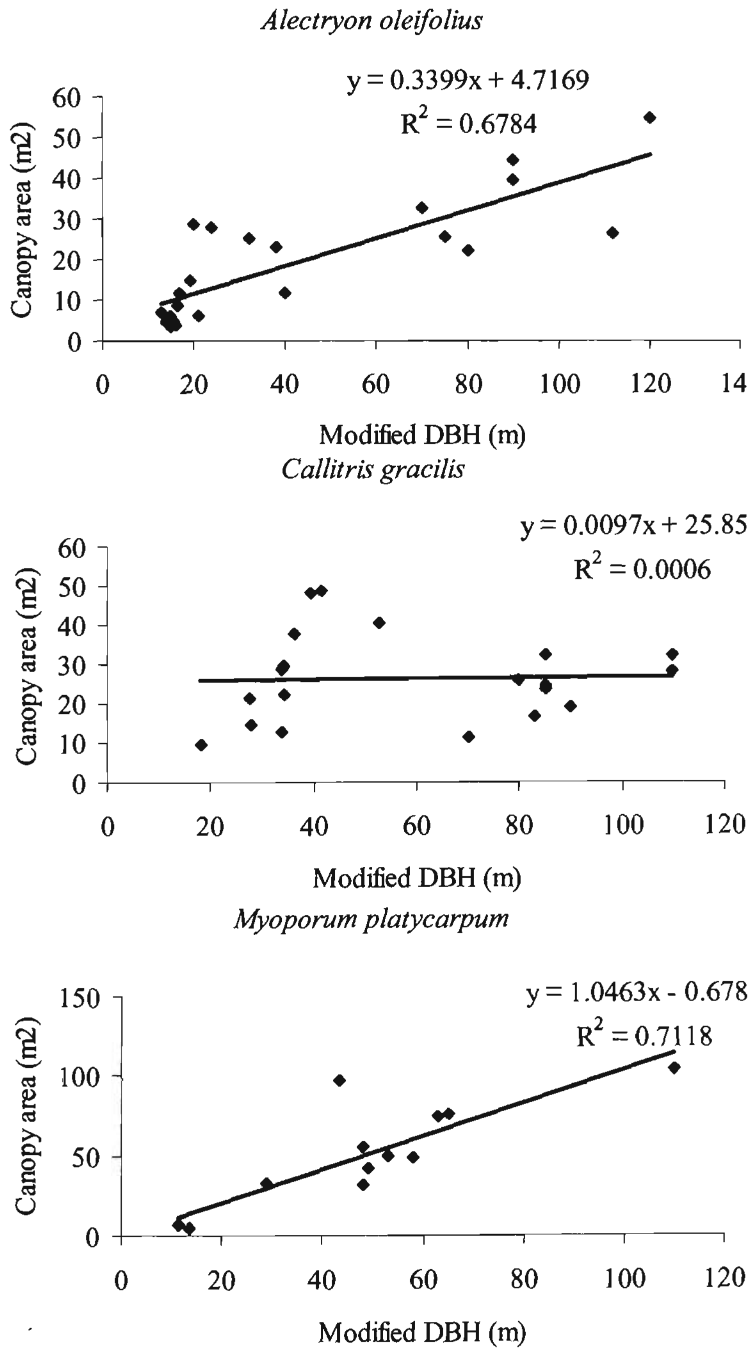


Figure 4.20. Relationship between canopy area and tree stem diameter.

Table 4.33. Canopy cover determined from on-screen digitising of aerial photography, compared with ground estimates.

Photo	Digitised canopy cover %	Ground estimate canopy %	Difference
77	2	2	0
79	4	2	2
99	7	5	-2
107	29	35	6
109	4	5	1
113	8	2	-6
122	8	5	-3
126	6	10	4

Table 4.34. Ground estimates of tree canopy cover compared with cover values calculated from supervised and unsupervised classification of aerial photographs.

Photo	Ground estimate canopy	Supervised	Unsupervised
77	2	**2.84**	**2.28**
79	2	**4.85**	21.04
99	5	**7.96**	**9.01**
107	35	48.13	**30.17**
109	5	**6.90**	**11.86**
113	2	23.87	25.83
122	5	16.33	28.82
126	10	**19.38**	**9.44**
Total within 10 %		**5**	**5**

Bold – within 10 % of cover estimate

Table 4.35. Ground estimates of total perennial cover compared with cover values calculated from supervised and unsupervised classification of aerial photographs.

Photo	Ground estimate total perennial %	Supervised	Unsupervised
77	20	2.84	2.28
79	35	4.85	21.04
99	15	**7.96**	**9.01**
107	51	**48.13**	30.17
109	6	**6.90**	**11.86**
113	16	**23.87**	**25.83**
122	21	**16.33**	**28.82**
126	15	**19.38**	**9.44**
Total within 10%		**6**	**5**

Bold – within 10% of cover estimate

Table 4.36 shows the approximate costs for acquiring and analysing 100 aerial photographs to produce an interpolated map of vegetation condition within a large study area such as the MSNP. Within the MSNP (6,363 km^2), 100 photographs at the largest photo scale would cover 11.47 km^2 or 1.79 % of the MSNP. If positional accuracy is important, then ortho rectified imagery can be purchased, although at significantly higher cost. A quote for ortho rectified imagery from a local supplier suggested costs of approximately \$520 per km^2 (Lourens, UW [Quasco] 2002 pers. comm., 4 Feb).

Vegetation condition classes derived from aerial photography were not calculated for the study area. Interpolation models based on aerial photography face the same limitations as those of field data (Section 4.2.3) As aerial photo measures of vegetation condition do not provide the same level of detail as ground measures, it would be expected that interpolation models based on aerial photography would not exceed the accuracy of field assessment models.

Table 4.36. Costs for obtaining and analysing aerial photographs to determine vegetation condition.

Task / Item	Resources	Units	Unit Type	Cost per unit	Total Cost
Project Development					
Define study area & method	1 scientist	2	days	$ 680	$ 1,360
Preparation for ground truthing	1 scientist	2	days	$ 680	$ 1,360
	1 assistant	1	days	$ 430	$ 430
Sub-total					**$ 3,150**
Aerial Photography					
	Flying time	5	hours	$ 500	$ 2,500
	Photo prints	100	photos	$ 30	$ 3,000
Sub-total					**$ 5,500**
Field Survey					
Field Survey	1 scientist	5	days	$ 680	$ 3,400
	1 assistant	5	days	$ 430	$ 2,150
Travel	travel	2,000	km	$ 0.55	$ 1,100
	Meals/accom	4	nights	$ 220	$ 880
Data entry	1 assistant	2	days	$ 430	$ 860
Sub-total					**$ 8,390**
Imagery Analysis					
Pre-processing	1 scientist	3	days	$ 680	$ 2,040
Condition assessment	1 scientist	12	days	$ 680	$ 8,160
Accuracy assessment	1 scientist	5	days	$ 680	$ 3,400
Sub-total					$ 13,600
Total cost					**$ 30,640**

4.4.3 Discussion

Large-scale aerial photography was investigated as a method for measuring vegetation condition parameters. Using small-format digital photography, and a hand held GPS unit within the aircraft it was possible to locate photographs in the field, and determine corner co-ordinates to geo-register them. Whilst inaccuracies in scale were found with the calculations from focal length, they were mostly small (on average less than 7 %). Inaccuracies in scale are not necessarily a major problem in determining vegetation condition. Commonly percentage cover values are calculated for a sample area within the photograph. Where percentage cover calculations are performed, inaccuracies of scale are irrelevant, as the percentage cover of a parameter will not change regardless of calculated scale.

Mean error in positional location of photos was 269.1 m, using a hand held GPS tracking function. To improve accuracy of positioning of small-scale aerial photography, flights over recognisable features at least at the beginning and end of each flight mission are recommended (Abd-Elrahman *et al.* 2001). If positional accuracy is important, alternative methods can be used, however, generally at greater cost. If positional accuracy is not imperative, then methods such as this provide one of the most cost effective options for obtaining aerial photography. In a method such as this where sampling of a large area is being undertaken it is unlikely that positional errors will impact on the overall result of vegetation condition within the study area.

Where vegetation is relatively homogenous, positional accuracy is less likely to impact on results. However, in more heterogenous areas, increased variability in parameters due to location errors may result in requirements for greater sample size. Further analysis is required to determine if it is more cost effective to acquire precise imagery, or to analyse more samples of less accurate data.

Many vegetation parameters could be measured with the use of large-scale, small-format aerial photography. Potential parameters for aerial photo condition assessment examined in this study were tree species richness, tree canopy cover, cover of perennial vegetation, and cover of bare ground. Whilst tree condition was not successfully determined using onscreen digitising (overall accuracy 44.68), previous studies have shown that measures of tree condition can be obtained using colour-infrared film and stereoscope (Margules & Partners Pty Ltd. *et al.* 1990).

Determining tree species richness by accurate identification of tree species was relatively successful, with onscreen identification of tree species resulting in an overall accuracy of 72.58 %. It is likely that this could be improved by using stereoscopic viewing of image pairs, as has been used in a number of other studies reporting higher accuracy in species identification (Hall and Aldred 1992; McCormick 1999). However, this would result in higher costs of image acquisition due to the need for overlapping images and an increase in analysis time.

Percentage cover calculations of bare ground, tree canopy, and perennial vegetation from aerial photography provided some of the most accurate data. Similarly, large-scale aerial photography has been shown to correlate well with on ground measures in other arid environments (Tueller *et al.* 1988; Knapp *et al.* 1990). Tueller *et al.* (1988) also found that species identification and density counts provided less reliable information than cover estimates. Greater accuracy of canopy cover and ability to distinguish tree species was found with photographs at 304.8 m ASL, compared with photographs at smaller scales.

Lack of regeneration has been identified as a major threat to semi-arid woodlands in northwest Victoria (Cheal 1993; Westbrooke 1998; Sandell *et al.* 2002). Therefore determining presence or absence of tree regeneration would be a useful assessment tool. Unfortunately, no tree seedlings were present on any aerial photograph, and so assessment of detection accuracy was unable to be investigated. In a study on forest regeneration within Mixed Boreal Forest in Canada, regeneration in cutover forestry sites was examined using 1:10,000 aerial photograph stereo pairs (Hall and Aldred 1992). Seedlings measured ranged in height from 0 to 201 cm, and as expected, greater ability to detect larger seedlings was observed. No seedlings under 15 cm were able to be detected, however, overall, seedling percent detectability for cutover sites was 62 % (Hall and Aldred 1992). Identification of tree seedlings was performed in another study of regeneration of forestry areas in Virginia, using large-scale aerial photography and colour film (Smith *et al.* 1986). *Pinus taeda* (Loblolly Pine) seedlings (mean height 44–66 cm) were not able to be accurately detected at a scale of 1:890, but were accurately detected (overall accuracy 72–75 %) at a scale of 1:297.

These previous studies suggest that tree seedlings could be identified on aerial photographs of semi-arid woodland in northwest Victoria, provided the images of the seedlings were not obscured by other vegetation such as other tree canopies, tall grasses, or shrubs. To accurately detect seedlings under half a meter tall, aerial photograph scale would have to be very large, at around 1:300.

In producing a condition index from aerial photography, compared with field survey techniques, the number of parameters identified from aerial photos would probably be reduced due to the time required to analyse many different features from aerial photography. The choice of photo scale, film types and techniques for analysis should be determined according to the parameters to be measured. For example, whilst broader scale photographs provide a cheaper coverage of large areas, the types of variables that can be calculated from broad-scale photographs is limited.

Further research is required to determine optimal sampling regimes for interpolation models if maps are to be produced from aerial photography analysis within large areas such as the MSNP.

4.4.3.1 Advantages of aerial photography

The main advantage of aerial photography over other remote data sources is the detail of potential measures. Aerial photography at an appropriate scale can be used to provide objective measures of many of the parameters used in field surveys of vegetation. The use of aerial photography can reduce travel costs whilst enabling a larger sample to be surveyed. If simple measures are used, such as dot-grid estimate of cover, set-up costs can be less than that required for other remote data sources, and staff training may be minimal (Tueller *et al.* 1988).

4.4.3.2 *Limitations with aerial photography*

One of the main limitations with aerial photography with a study area of the size of the MSNP is the need for interpolation of data. It would be far too costly to obtain complete large-scale coverage of the site, and to join images, and so sampling must be employed. The cost of obtaining ortho-rectified or colour-infra-red imagery also quickly becomes prohibitive within this large study area.

Whilst there are many methods for achieving high positional accuracy of aerial photography, the challenge is to find a method that does not greatly increase the associated costs. Factors impacting on the positional accuracy of small-format aerial photography include synchronisation between the different components (digital camera and GPS), accuracy of navigational equipment and altitude sensors, mounting platform stability, lens distortion, and weather conditions (Abd-Elrahman *et al.* 2001). The limited positional accuracy of small-format aerial photography may be problematic in a location, such as remote semi-arid woodland with few identifying ground features. However, with the use of an appropriate sampling method, positional accuracy may not be highly important in determining the vegetation condition of an area such as the MSNP, provided the photo samples are located within the correct vegetation community.

4.5 USE OF EXISTING DATA SOURCES FOR CONDITION ASSESSMENT

Vegetation data are collected throughout Victoria each year as part of state-wide monitoring schemes and numerous other projects. The opportunity exists for these data to be re-analysed to assist in determining vegetation condition. The Treeden25 map layer is a satellite imagery interpretation of the density of trees across Victoria. This is an example of data that could be analysed to provide information on vegetation condition of semi-arid woodlands in northwest Victoria.

4.5.1 Methods

4.5.1.1 *Tree density layer*

The Treeden25 dataset is held by the Catchments and Water Custodial Program L25 Master Library Group. The presence/absence of tree cover was derived from SPOT panchromatic imagery (10 m pixels) by a combination of digital classification and visual interpretation. The presence / absence dataset was then grouped into three density classes (Dense, Medium, Scattered) using neighbourhood and proximity cell based analysis (NRE 2000b). The raster dataset was then converted to vector data. The map scale is a nominal 1:25,000 and there are no limitations on access to these data (NRE 2000b). Correlations between tree density, overall diversity and the condition indices from the field condition assessment suggest that tree density may be a potential surrogate for vegetation condition (Table 4.7).

4.5.1.2 Tree density as a surrogate for condition

To extract the area of the Treeden25 layer representing semi-arid woodlands, the mask of semi-arid woodland extent produced by satellite imagery classification (Figure 4.9) was applied. In areas outside the extent of the 2001 satellite imagery, the EVCs representing semi-arid woodlands were used to produce a layer of tree density within semi-arid woodlands. Tree density classes were reclassified to three semi-arid woodland condition classes. Condition was also classified to two classes, with good condition represented by dense areas and poor condition represented by all other categories.

To investigate the relationship between tree density classes and vegetation parameters including vegetation condition, Spearman's rank correlations were performed. Error matrices were calculated to determine the accuracy of Treeden25 maps as surrogate condition maps. Overall accuracy of the Treeden25 classes as surrogate condition indices was assessed. Producer's accuracy and user's accuracy were also calculated for each category. The Kappa statistic and Z scores were calculated to assess the statistical significance of the results.

4.5.1.3 Costs for using existing data sources

Costs associated with this study were calculated based on data and time required for the current study.

4.5.2 Results

Spearman's rank correlations were performed to determine relationships between the Treeden25 classes and vegetation parameters including the condition indices. Moderate correlations were found between Treeden25 classes and a number of vegetation parameters including the field condition index, total species richness, percent cover of trees (Table 4.37). Smaller correlations were found with many other vegetation parameters.

To assess the accuracy of the map of vegetation condition defined by tree density classes, the ground truthed condition classes were compared in an error matrix to the Treeden25 classes (Table 4.38). Overall accuracy of three condition classes from Treeden25 data was 58.43 %.

Given the limited accuracy of mapping three condition classes, two condition classes were investigated. Investigation of accuracy revealed all accuracy measures were improved (Table 4.39).

Table 4.37. Correlation of Treeden25 classes with parameters measured during field condition assessments.

Parameters	Treeden25 classes
% cover cryptogams	0.245 **
% cover exotic ground layer	-0.292 **
% cover native ground layer	0.069
% cover litter	0.227 **
% cover small shrubs	0.141
% cover tall shrubs	0.075
% cover trees	0.369 *
Condition index	0.364 **
small shrub sp. richness	0.361 **
tall shrub sp. richness	0.211 **
tree sp. richness	0.316 **
total perennial sp. richness	0.453 **

N=189, **Correlation is significant at the 0.01 level (2-tailed), *Correlation is significant at the 0.05 level (2-tailed).

Table 4.38. Error matrix of Treeden25 classes compared to field vegetation condition data.

Treeden25 classes	Condition classes from field assessment			
	Good	Moderate	Poor	Total
Dense	19	20	0	39
Medium	0	9	6	15
Sparse	0	12	6	18
No trees	0	37	57	94
Total	19	78	69	166

Table 4.39. Error matrix and accuracy assessment for two condition classes from Treeden25 data.

	Condition classes from field assessment		
Treeden25 classes	1	2	Total
1	27	0	27
2	4	47	51
Total	31	47	78

Overall accuracy	94.872
Kappa	0.891
Z score	16.804
Result[a]	S

Class	Producer's accuracy %	User's accuracy %
1	87.10	100.00
2	100.00	92.16

[a]At the 9 5% confidence level, S=Significant.

4.5.2.1 *Costs with using existing data sources*

Associated costs with using the Treeden25 layer as a surrogate condition index are presented in Table 4.40. No fieldwork component has been included, as ground truthing of accuracy of the dataset has already been undertaken during creation of the map layer (NRE 2000b). For organisations affiliated with the Department of Sustainability and Environment (DSE) only costs of accessing the data are charged (as presented in Table 4.40) (NRE 2000b). Costs may be higher for other organisations.

4.5.3 Discussion

The Treeden25 layer, as a measure of tree density across Victoria has potential as a tool for determining vegetation condition within the semi-arid woodlands in northwest Victoria. Two surrogate condition classes were able to be produced using tree density classes from the Treeden25 layer.

Tree density classes showed medium correlations with many measures of vegetation condition including perennial species diversity (correlation coefficient 0.45), percentage cover of trees (correlation coefficient 0.37), and the field condition index (correlation coefficient 0.36).

Table 4.40. Costs to undertake a condition assessment from existing data such as the Treeden25 layer.

Task / Item	Resources	Unit	Unit Type	Cost per unit	Total cost
Project Development					
Define study area & method	1 scientist	1	day	$ 680	$ 680
Sub-total					**$ 680**
Data Acquisition					
Acquire data	Treeden25 layer	1	CD & access costs	$ 350	$ 350
Sub-total					**$ 350**
Data Analysis					
Production of map(s) for study area	1 scientist	2	days	$ 680	$ 1,360
Change analysis	1 scientist	1	day	$ 680	$ 680
Sub-total					**$ 2,040**
Total cost					**$ 3,070**

Another source of existing data on the vegetation of semi-arid woodlands in northwest Victoria is the Flora Information System (FIS). The FIS is a geographically-registered, relational database of observation and survey data on Victorian plants, containing more than 1.5 million plant records from over 139,000 survey sites (NRE 2000a). Data are collected from surveys carried out by botanists from government and other organisations, as well as from the botanical literature, herbarium specimens, and field naturalists. An agreement for data sharing ensures that data from most field surveys conducted in Victoria are entered into the FIS.

A further application of the FIS data is to determine vegetation condition from the parameters recorded in the FIS database and a study to investigate this has been proposed (Gullan, P [Viridans] 2000, pers. comm., 4 July).

4.5.3.1 Advantages of using existing data sources

The main benefits of using existing data sources are time and cost. It is straightforward and very quick to extract the area of interest from the tree density layer, and use this as a surrogate for tree condition classes. Image processing has already been completed, eliminating this cost. Field survey for accuracy assessment is also not necessary, as the accuracy of the data is already known, however, some ground truthing to determine how well tree density classes correlate to vegetation condition may be of benefit.

4.5.3.2 Limitations of existing data sources

There are many limitations with using existing data sources such as the Treeden25 layer. Existing data sources are the least flexible option for condition assessment. There is no ability to obtain new data or choose when further assessments will be undertaken. Often little information is available about how the data were collected and analysed which may affect its use for condition assessment and preclude replication of results in future studies. Potential errors within the data are largely unknown and cannot be easily estimated. The reliability of comparisons in time would therefore be largely unknown, and difficult to predict.

4.6 COST BENEFIT ANALYSIS

Four main techniques were compared for vegetation condition assessment: field survey data, Landsat ETM imagery, large-scale aerial photography and an existing data source, the Treeden25 layer. Data analysis shows that all of these methods have potential as valid tools for vegetation condition assessment and they each have differing advantages and disadvantages. A cost-benefit analysis was undertaken to determine the most cost-effective method for vegetation condition assessment.

4.6.1 Methods

Two main outcomes of vegetation condition assessment were considered; (i) data to contribute towards vegetation management, and (ii) production of an accurate map of vegetation condition.

Goals of vegetation management of National Parks in northwest Victoria have been outlined in Section 1.3.2, and include enhancing perennial regeneration through reducing the impact of pest animals and addressing the current imbalance of kangaroo populations. Therefore, parameters of importance for vegetation monitoring within these areas include regeneration of the overstorey, overstorey condition, and increase of woody perennial cover and species richness in the understorey.

Classification accuracy was compared for maps of two and three vegetation condition classes. To test if any condition mapping technique was significantly more accurate than another, Z scores were calculated to test for significance difference between Kappa values. As Z scores may over-estimate the standard error, resulting in a greater chance of accepting the null hypothesis (type two error) (Agresti 1990), an alternative approach, the McNemar test was also performed. Costs for each vegetation condition assessment were also compared.

It must be noted that this analysis does not attempt to assess the benefits of assessing vegetation condition over any other type of vegetation assessment, but rather to compare methods for condition assessment.

4.6.2 Results

Data for vegetation management

To contribute data towards effective management, field survey methods provide the most detailed and precise data on most vegetation parameters. Field survey data were able to provide data on regeneration of the overstorey, condition of the overstorey, and increase of woody perennial cover and species richness in the understorey.

The aerial photography analysis tested in this study was able to provide reliable data on perennial cover, particularly for larger scale photographs (scale approx 1:1,000, $R^2 = 0.87$). Whilst not reliably assessed in this study, results of previous studies suggest that canopy condition can be accurately assessed from aerial photography, and some success has been found with identifying overstorey regeneration in forestry sites. However, given the slow growth rates of many perennial overstorey species in semi-arid woodlands, it is unlikely that recruits of these species would be identifiable by aerial photography for 5 to 10 years. Detectability of recruits would also be limited by browsing by feral and native mammals.

Landsat imagery, with 30 x 30 m pixels, is unlikely to detect regeneration. Species richness, regeneration, and canopy condition cannot be directly calculated from Landsat data. However, the NDVI and other vegetation indices showed strong correlations with many vegetation parameters such as % cover of perennial species (0.78 correlation with NDVI), % cover bare ground (-0.74 correlation with tasselled cap) and % cover trees (0.72 correlation with NDVI).

The Treeden25 data were also based on satellite imagery, with SPOT providing higher spatial resolution (pixel size 10 m panchromatic) than the Landsat imagery. Information on tree density classes is valuable in assessing vegetation clearing and analysing patterns in overstorey vegetation over time. The nature of disturbances and perennial species richness has resulted in areas of more dense vegetation also containing greater species richness, with higher rates of regeneration than more sparsely vegetated areas. Therefore, measures of tree density such as the Treeden25 layer also indicate where higher perennial species richness occurs and where greater regeneration is likely to occur. Spearman's correlations indicate that the tree density classes are correlated to many vegetation parameters, but correlations were not as strong as with the vegetation indices derived from Landsat data (Table 4.37).

Accuracy of mapping

Field survey and aerial photography methods rely on interpolation modelling to produce a map of vegetation condition whereas Landsat imagery and the Treeden25 layer provide complete coverage of the study area. The accuracy of interpolation is dependent on the site heterogeneity, sampling regime and sample size. It is possible that localised disturbances resulting in vegetation change in areas not surveyed could affect the accuracy of interpolation models. This could result in areas of condition change going undetected.

Comparison of overall accuracy shows the field data interpolation model provided the most accurate classification to three condition classes, however, too few ground truthing samples within this map led to a low Kappa value. A higher Kappa value was found for the supervised classification of Landsat data (Table 4.41).

Table 4.41. Summary of accuracy of approaches to map three condition classes.

Classification approach	Overall accuracy	Kappa statistic	Z statistic	Result[a]
Field data interpolation	69.05	0.397	2.932	S
Landsat supervised classification	66.67	0.499	6.207	S
Landsat unsupervised classification	47.43	0.229	2.952	S
NDVI	65.38	0.463	8.379	S
Treeden25	61.90	0.378	6.546	S

[a]At the 95 % confidence level.

Due to the low accuracies obtained for classification of three condition classes, interpretation of data to two condition classes was undertaken. For land management purposes, classification of condition to two classes provides somewhat limited information. These types of data could be used to highlight areas of vegetation change, or target areas for more detailed survey efforts.

Good overall accuracy of classification to two condition classes was obtained for all methods, with highest overall accuracy and Kappa obtained by the Treeden25 layer (Table 4.42).

Table 4.42. Summary of accuracy of classification approaches to map two condition classes.

Classification approach	Overall accuracy	Kappa statistic	Z statistic	Result[a]
Field data interpolation	90.48	0.306	1.287	NS
Landsat supervised classification	89.74	0.776	10.614	S
Landsat unsupervised classification	87.18	0.741	9.805	S
NDVI	88.46	0.758	9.997	S
Treeden25	94.87	0.891	16.804	S

[a]At the 95% confidence level.

To determine if any significant differences were observed between the error matrices, and therefore if one technique is significantly better than another, Z scores between error matrices were calculated (Congalton and Mead 1983; Congalton 1991). No significant differences were found between error matrices for any treatment combination (Table 4.43). This indicates that none of the classification approaches significantly outperformed any other approach, and choice of technique therefore, does not centre upon accuracy of result.

Table 4.43. Z score analysis to test for significant differences between error matrices of different classification techniques.

Classification approach	Z statistic	Result[a]
NDVI Vs. field condition	-0.173	NS
NDVI Vs. supervised	-0.173	NS
NDVI Vs. Treeden25	-1.436	NS
NDVI Vs. unsupervised	0.158	NS
Supervised Vs. field interpolation	0.667	NS
Supervised Vs. unsupervised	0.334	NS
Treeden25 Vs. field interpolation	0.416	NS
Treeden25 Vs. supervised	0.515	NS
Treeden25 Vs. unsupervised	1.622	NS
Unsupervised Vs. field	0.652	NS

[a]At the 95 % confidence level.

Due to the potential for overestimated standard error with this approach, a further technique was applied to test for differences between treatments. The McNemar analysis shows a more accurate approach using the Treeden25 data than any other approach used (Table 4.44).

Table 4.44. Comparison of different vegetation condition assessment techniques using the McNemar test.

Classification approach	P value	Result[a]
NDVI Vs. supervised	1.000	NS
NDVI Vs. reeden25	0.016	S
NDVI Vs. unsupervised	0.453	NS
Supervised Vs. unsupervised	0.424	NS
Treeden25 Vs. supervised	0.031	S
Treeden25 Vs. unsupervised	0.002	S

[a]At the 95 % confidence level.

Cost of techniques for vegetation condition assessment

Two techniques were reliant on interpolation modelling for production of vegetation condition maps, large-scale aerial photography and field survey. Costs for these data types are dependent upon the number of samples. The cost to sample 0.1 % of the study area in the field using 50 x 20 m quadrats is estimated at over $123,000, compared to $30,640 to sample 1.79 % by aerial photography (Table 4.36). However, costs to sample 100 field survey quadrats are reduced to $29,250 (Table 4.8).

The cheapest technique is the Treeden25 data (costs to produce vegetation condition map estimated at $3,070 (Table 4.40)). However, if change detection is required, then Landsat imagery provides the most flexible option for image acquisition, enabling investigation of both past vegetation condition and assessments into the future. Whilst costs are higher for Landsat data acquisition and analysis (approximately $26,640, Table 4.22), costs for Landsat data are comparable to aerial photography or field survey of approximately 100 quadrats, but provide complete coverage of the area.

4.6.3 Discussion

Medium-scale satellite imagery such as Landsat ETM cannot provide the same detailed data as large-scale aerial photography or field survey data. Individual trees cannot be detected, and analyses are limited to classifications of land cover types, or indices, such as vegetation or brightness indices. However, vegetation indices have been shown in this study and previous research to correlate well with many measures of vegetation (Justice *et al.* 1985; Kogan 1990; Liu and Kogan 1996; Coops *et al.* 1997; Schmidt and Karnieli 2002).

Particularly where accurate data on regeneration of perennial species are required, field survey data are irreplaceable. Large-scale aerial photography provides the next most detailed measures of vegetation parameters and similar scale data may be provided with videography or hyperspectral imagery such as CASI (Lewis 2000; Pickup *et al.* 2000). However, aerial photography is unlikely to provide information on overstorey regeneration for at least five to ten years given the slow growth rates of these species.

Whilst the Treeden25 data provide a very cost-effective method for condition assessment, there are a number of limitations in flexibility and interpretation of these data.

4.6.3.1 Change detection

The ability to detect change in condition over time is crucial to the relevance of vegetation condition assessment and of particular interest to land managers such as Parks Victoria (Parks

Victoria 1998). Whilst most methods explored in this study enable change detection, the Treeden25 layer, or any other existing data source has limitations in flexibility in acquisition of new data for repeat studies.

Little data exist to assess the accuracy of comparative studies of vegetation change over time. The first vegetation condition assessment of a national park in northwest Victoria, a study at Wyperfeld National Park conducted in 1998, is proposed to be repeated in 2004 (CEM 1998). With the first condition assessment of semi-arid woodlands in the study area conducted in 2000 (Westbrooke *et al.* 2001), it is unlikely that there will be significant changes detectable from medium scale remote data sources for five to ten years, depending upon the occurrence of rainfall events. This limits the ability to fully test change detection methods at this time. Further research is required to determine the accuracy of change detection studies based on remote data sources.

4.6.3.2 Condition of semi-arid woodlands in northwest Victoria

The results from all methods of condition assessment highlight the extent of vegetation in poor condition throughout the study area. When reduced to two condition classes, based on supervised classification of Landsat, good condition areas occupy only 42.81 km^2 (22.3 %) of semi-arid woodlands in the Landsat image selection. Similar results were obtained through all methods used.

The results of the field survey show that compared to benchmark sites, the majority of semi-arid woodlands within the MSNP support significantly fewer native perennial species. Compared with benchmark sites, they also have significantly lower native shrub cover, and the proportion of regenerating shrubs is less than 0.27. On average, approximately one intact stratum was found across all semi-arid woodland sites surveyed, compared with three intact strata within benchmark sites and data from overstorey age classes indicates very limited tree regeneration since European settlement.

4.6.3.3 Limitations

It should be noted that condition is a subjective measure, based on a comparison of the actual state of a site with a desired state. One or more desirable parameters are chosen to determine condition of a site relative to a goal or benchmark site. The condition classes, formed from condition indices are also largely subjective, and the divisions between classes were subjectively placed to produce categories that appeared ecologically reasonable. The condition classes therefore, differ from the usual land cover classes determined from classification of remote imagery, in that the divisions between classes are subjectively defined. This has implications for accuracy assessment, where greater accuracy or correlation with ground data may be obtained by manipulating the divisions between classes.

Ideally, for accuracy assessment and comparison between methods, all data should be collected at precisely the same time. Unfortunately, this is rarely possible, as scheduling of flights and remote field trips is a significant logistical exercise. A lag between data acquisition and ground truthing may have affected accuracy assessment for some methods within this study, however, this does not appear to have been a major issue. Significant changes in perennial vegetation were not observed during this study, possibly due to the low rainfall experienced. The high correlations observed between ground and remote measures also suggest that errors due to time lag were minimal.

Benchmarks used within this study were chosen from more densely vegetated and species rich areas. Locations chosen for benchmarks sites have been subject to lower grazing pressure, compared to the majority of semi-arid woodlands in the area. In chapter four, historical maps showed some semi-arid woodland areas were denser and some were more open at the time of settlement. Therefore, to compare all areas of semi-arid woodlands to examples of the densest vegetation may not be valid. However, the goal of maximising diversity is valid, particularly within the framework of management for conservation. If these dense, species rich areas are to be used as benchmarks, it must be considered that not all areas may be capable of supporting the same density. Furthermore, uniformity across the landscape is probably not desirable.

Annual and herbaceous plant diversity was not assessed during this study, due to drought conditions that led to few herbaceous species being present during much of the study period. Measures of annual vegetation were also excluded from the field survey, as fluctuations in species richness and abundance are highly dependent upon rainfall and seasonal conditions. Some open sites may have supported higher annual species diversity prior to European settlement. However, these open sites have been heavily used for grazing, and no examples of undisturbed open remnants were observed during this study. Evidence of a lack of annual diversity was seen by the extensive cover of dried annual weeds that were observed on most sites.

Vegetation condition correlated well with perennial vegetation cover within the study area. However, remote vegetation condition techniques cannot distinguish between native and exotic perennials. There are very few shrubby perennial weeds present within the study area, and therefore this did not present a problem in the current study. This method would be unlikely to be of use in areas where woody weeds pose a significant threat to vegetation diversity.

Remote vegetation methods were also unable to directly identify many other components of vegetation condition such as perennial regeneration, or weed abundance. However, the nature of disturbances in the study area has led to a high correlation between the cover of perennial vegetation and many other measures of vegetation condition. Therefore, vegetation indices and tree density measures provided an indirect measure of other aspects of vegetation condition.

4.7 CONCLUSION

The Australian and New Zealand Environment and Conservation Council (ANZECC) National Framework for the Management and Monitoring of Australia's Native Vegetation states "Condition is a very hard quality to measure and monitor and more work is needed on how it would best be done" (ANZECC 2000).

The ANZECC guidelines also provide us with some insight to the desirability of condition assessment stating "Monitoring change in cover and condition is an essential component of improving and adapting best practice in native vegetation management" (ANZECC 2000). Despite the list of potential problems with condition assessment, it remains an important tool for assessing vegetation change.

Five fundamental components of any condition assessment have been outlined in an earlier chapter (Section 1.3). This chapter has addressed component (iii) vegetation measures or indicators able to measure the current state of the vegetation, detect change and determine direction of change. Component (v) has been considered in the selection of vegetation measures and in determining sampling procedures to minimise the effect of short-term fluctuations on the data.

To determine the most cost-effective method for vegetation condition assessment, four different methods were compared; field assessment, aerial photography, Landsat satellite imagery and an existing dataset – the Treeden25 map layer. All techniques assessed provided vegetation measures or indicators that were reliably able to measure the current state of vegetation, and when evaluated as two condition classes (good and poor), little significant difference in map accuracy was observed.

Change detection was recognised as a significant component of condition assessment together with the ability to distinguish between short-term fluctuations and long-term change. Change detection methods for satellite imagery classification were explored, however, lack of ground truthing data for earlier satellite images limited the ability to determine accuracy of these methods. Techniques to minimise short-term fluctuation included acquiring imagery in mid-summer and sampling only perennial vegetation. Further research is required to fully evaluate the ability of methods used to reliably detect change.

As little significant difference was observed in the ability to map vegetation condition classes, choice of an appropriate assessment technique is dependent upon the data provided for land managers to assess the effectiveness of management actions, and cost of assessment. It is not surprising that with higher cost options (eg. field survey), more data can be provided on a wide range of parameters of importance to land managers. Lower cost options (eg. Treeden25), however, can achieve good results in mapping broad vegetation condition categories.

The vegetation indices calculated from Landsat data showed very high correlations with the field condition index and many other parameters assessed. Vegetation indices have often been used as a reliable method for detecting change in vegetation and show excellent potential for remote vegetation condition assessment in northwest Victoria.

Remote survey techniques are unlikely to completely replace ground survey methods, but are more likely to be used in conjunction with ground survey techniques to provide a broad overview of vegetation (Hobbs *et al.* 1989). Often, more precise and detailed information is required than can easily be obtained from current remote methods. In addition, the need to check for error and verify classification results will ensure a continuing need for ground survey. However, when used in conjunction with field survey, remotely sensed data provides a useful tool to assess change in vegetation condition over time. Remote survey techniques can be valuable in detecting and analysing vegetation change across the landscape, and determining areas where more intensive field survey may be required.

5. REGENERATION OF PERENNIAL SPECIES OF *CASUARINA PAUPER* WOODLAND

5.1 CHARACTERISATION OF PERENNIAL VEGETATION REGENERATION IN THE MURRAY-SUNSET NATIONAL PARK

The Murray-Sunset National Park (MSNP) was declared in 1991, and there is evidence of some vegetation recovery (Sandell *et al.* 2002). Studying natural recovery processes has the potential to give insight for vegetation management, such as revealing the conditions for regeneration of different species, or potential barriers to natural recovery processes. Barriers or thresholds to recovery have been described in state and transition models, where disturbances exceed the ability of natural recovery mechanisms. Thresholds to natural recovery have been identified for heavily grazed systems, where ceasing grazing is insufficient to enable natural recovery to occur, despite sufficient rainfall. By identifying sites where natural recovery is slower or less likely to occur, priorities for management interventions such as direct seeding or tree planting can be better determined.

To investigate the extent of vegetation recovery across the MSNP, determine potential predictors of recovery, and assist in prioritising management of vegetation recovery in the MSNP, the following objectives were addressed:

(i) to quantify the extent of perennial regeneration in Belah and Pine-Buloke woodland in the study area;

(ii) to determine which perennial species are regenerating, and which species are not regenerating;

(iii) to determine if sites where regeneration is occurring are different in terms of site characteristics such as species richness and percentage cover, from sites where regeneration is not occurring; and

(iv) to characterise sites where tree regeneration is occurring and hence determine potential predictors of resilience.

5.1.1 Methods

To investigate regeneration of trees and shrubs of semi-arid woodland species across the study area, 119 floristic quadrats were investigated. Quadrat data were collected from randomly selected sites within the MSNP, and good condition sites selected from nearby vegetation reserves (see Section 4.2.1 for details on data collection). From these data, 20 vegetation variables were derived (Table 5.1).

Table 5.1. Vegetation variables used to assess differences in regeneration response.

Percentage cover	Number of mature trees	Species richness
Trees	All species	Trees
Tall shrubs	*Allocasuarina luehmannii*	Tall shrubs
Small shrubs	*Alectryon oleifolius*	Small shrubs
Native ground species	*Callitris gracilis*	Native ground species
Exotic ground species	*Casuarina pauper*	
Total ground species	*Myoporum platycarpum*	
Cryptogams	**Other**	
Bare ground	Number of regenerating shrub species	
	Mature tree condition	

Mature trees were all those considered to have established more than 30 years ago. From tree stem diameter data, size class distributions were calculated for the dominant woodland trees in the study area (*Allocasuarina luehmannii*, *Alectryon oleifolius*, *Casuarina pauper*, *Callitris gracilis* and *Myoporum platycarpum*). To determine how size classes relate to age class, data from Westbrooke (1998) on size and age classes of *C. pauper* woodland species in south-eastern Australia were consulted, as well as observations from within the MSNP.

From measures of stem diameter in NSW and Victoria, *M. platycarpum* trees established at any time in the last 100 years were estimated to have a stem diameter of less than 40 cm (Westbrooke 1998). *M. platycarpum* established during the 1973/4/5 event were estimated to have a stem diameter of less than 25 cm, and greater than 5 cm, and trees with a stem diameter less than 5 cm are estimated to have established from the late 1980s to early 2000s (Westbrooke 1998).

Less data were available for *Casuarina pauper* and *Callitris gracilis* as very few seedlings have been observed, and the date of most *C. pauper* sucker regeneration events is unknown. Data suggest that the stem diameter for recruits less than 30 years old would be no more than 5 cm (Westbrooke 1998).

Few recruits of known age have been measured for *A. oleifolius* (Westbrooke 1998). Sucker regeneration resulted from three cut stumps of *A. oleifolius* in the MSNP following erection of a grazing exclosure in 1991. In 2001, the stem diameter of suckers (N=55) ranged from 0.2 to 3 cm (mean 1.01). It is estimated that recruits less than 30 years of age, would have a stem diameter of approximately 5 cm or less.

Very slow growth rates have been reported for *A. luehmannii* (Morcom 2000). Data on *A. luehmannii* in the Wimmera suggest that trees aged up to 30 years would have a diameter at breast height (DBH) less than 12 cm and trees aged up to 100 years may have a DBH up to 30 cm. Lower DBH measures were found for drier sites in the Wimmera, and with significantly lower rainfall in the study area, it is expected that stem diameters would be significantly lower again. The same stem diameters as determined for age classes of *C. pauper* have been applied for *A. luehmannii*. Whilst these data are uncertain, there are only very small numbers of trees in small size classes of *A. luehmannii* (Table 5.2).

Table 5.2. Range of stem diameter (cm) of tree species cohorts, and date of estimated establishment (data from Westbrooke 1998; Morcom 2000).

Tree species	All regeneration since European settlement ~125 yrs	1973/4/5, late 1980s, early 1990s
Alectryon oleifolius	<10	≤5
Allocasuarina luehmannii	<17	≤5
Callitris gracilis	<10	≤5
Casuarina pauper	<17	≤5
Myoporum platycarpum	<30	<25

Size classes are only estimates of age class, as little data are available on actual establishment date for trees in northwest Victoria. In addition, site-specific factors may lead to small differences in stem diameter growth rate.

Whilst estimates of recent regeneration used have included all recruits over the last 30 years, the majority of this regeneration is believed to have occurred since the establishment of the MSNP in 1991. During a study of *M. platycarpum* in 1988, no regeneration was found of this less palatable species in the northern area of the MSNP (Westbrooke *et al.* 1988).

The frequency of shrub species occurrence and percentage of sites where shrub regeneration had occurred was calculated.

Statistical analysis

Spearman's correlations were performed between presence of regeneration of the four most frequently recorded tree species (*A. oleifolius*, *Callitris gracilis*, *Casuarina pauper* and *M. platycarpum*) and the quadrat variables measured. The condition index was also included in the variables correlated to determine how well overall vegetation condition correlated with regeneration of these tree species.

Canonical discriminant functions (CDF) were performed to determine if sites where tree regeneration had occurred could be separated from sites where regeneration had not occurred. Canonical discriminant functions (CDF) is a descriptive technique to enable a more interpretable summary of the data. CDF allocates sites to a category using a linear combination of variables measured at the site (Shaw 2003). Categories were defined as sites with no regeneration, sites with low regeneration (1–2 individuals), and sites with moderate regeneration (greater than two individuals). Tree species were explored separately, as Spearmans correlations suggested a weak negative relationship between regeneration of some tree species. *A. luehmannii* and *C. pauper* were not examined, as too few regeneration sites were available for these species.

As classification is less statistically demanding than inference, the assumptions are less rigorous (Tabachnick and Fidell 2001). The data were screened for outliers, and homogeneity of variance-covariance matrices were tested by examination of the plots of the first two discriminant functions (Appendix 7). As the spread of cases was relatively equal for each group, no further test of homogeneity of covariance matrices was required (Tabachnick and Fidell 2001).

Significance tests of the two orthogonal axes created were performed, however, these inferential tests assume multivariate normality (Tabachnick and Fidell 2001). Univariate normality was tested using Q-Q plots, and heterogeneity of variance was tested with Levenes test. Variables were largely non-normal, and heterogeneity of variance was found for nine variables; combined with unequal sample sizes, this may affect significance tests for the discriminant functions, which should be interpreted with caution (Tabachnick and Fidell 2001).

5.1.2 Results

Most trees species have few individuals in the smallest size class, and few sites supporting the smallest size class (Figure 5.1 & Table 5.3). *A. oleifolius* has the highest percentage of sites with trees in the smallest size class, with 42.86 % of sites showing recruitment. No individuals were observed in the smallest size class for *Eremophila longifolia* (Table 5.3). Very low numbers of *A. luehmannii* and *C. pauper* are observed in the smallest size classes (Figure 5.1).

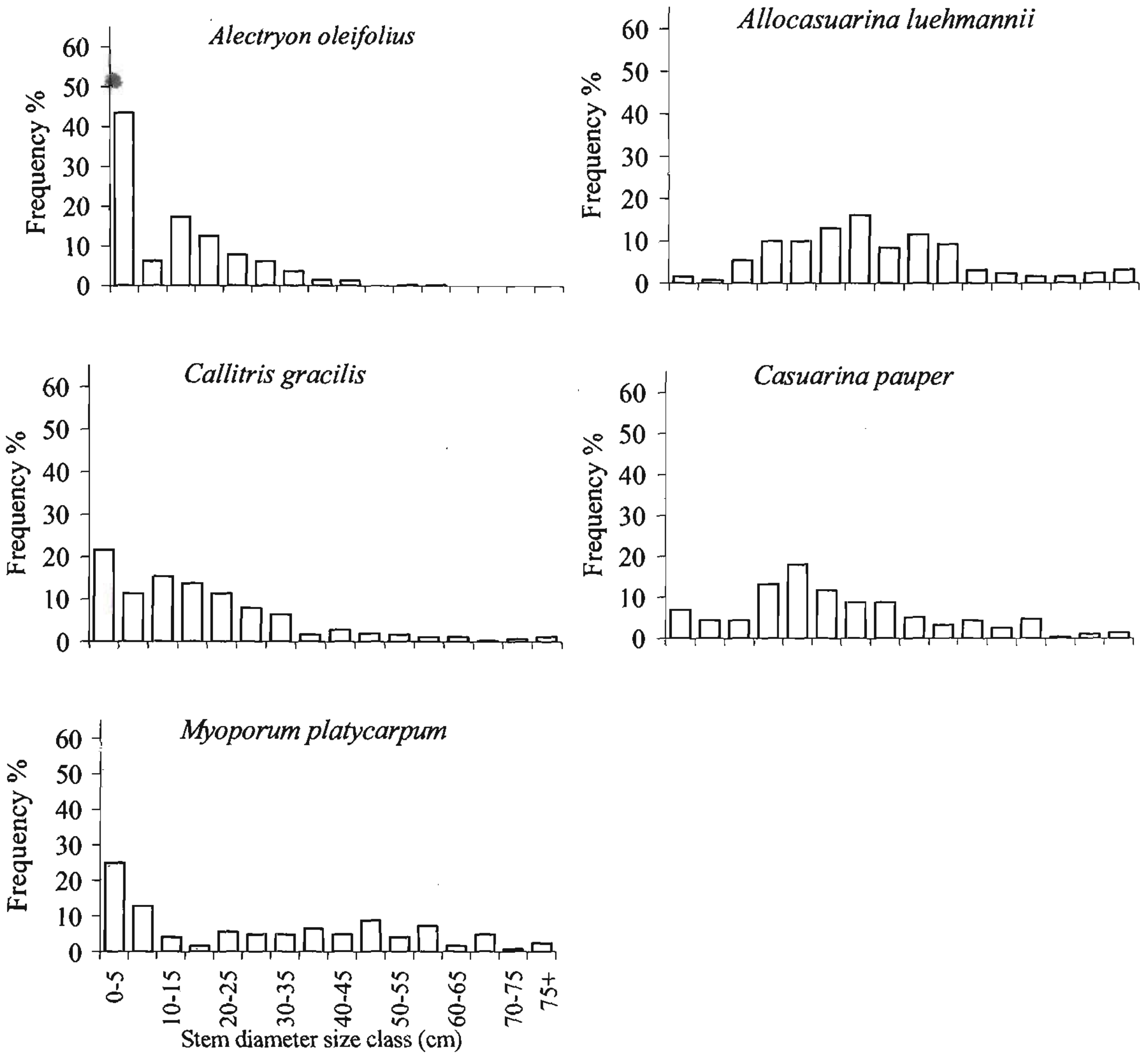


Figure 5.1. Stem diameter size classes of woodland trees measured in the MSNP, and nearby reserves.

Very little regeneration appears to have occurred for *A. luehmannii* or *C. pauper* in the 125 years since European settlement (Table 5.4). However, there has been a significant increase in regeneration in the last 30 years for *A. oleifolius, C. gracilis* and *M. platycarpum.*

Shrub regeneration was measured for all shrubs recorded at more than one quadrat (Table 5.5). The 11 most frequently occurring shrub species were found to be regenerating at more than 30 % of sites. The four most frequently occurring shrubs *Enchylaena tomentosa, Sclerolaena diacantha, Atriplex vesicaria* and *Sclerolaena obliquicuspis* were all regenerating at more than 60 % of sites (Table 5.5). No spatial distribution of regeneration was found, with recruits of trees and shrubs spread over much of the MSNP.

Table 5.3. Trees measured from 118 randomly selected 20 x 50 m quadrats in the MSNP and surrounding reserves.

Tree species	No. individuals	No. sites	No. individuals DBH ≤ 5 cm	No. sites stem ≤ 5 cm	% sites stem ≤ 5
Acacia oswaldii	8	5	2	1	20.00
Alectryon oleifolius	553	63	240	27	42.86
Allocasuarina luehmannii	130	19	2	2	10.53
Callitris gracilis	425	56	92	9	16.07
Casuarina pauper	270	29	19	5	17.24
Eremophila longifolia	3	2	0	0	0
Hakea leucoptera ssp. *leucoptera*	90	13	55	5	38.46
Hakea tephrosperma	37	3	20	1	33.33
Myoporum platycarpum	124	35	31	13	37.14

Table 5.4. Number of trees recruited in the past 125 and 30 years and number of sites where recruits were measured.

Species	No. individuals	% individuals	No. sites	% sites
Recruitment past 125 years				
Alectryon oleifolius	264	47.74	30	47.62
Allocasuarina luehmannii	15	11.54	8	42.11
Callitris gracilis	125	29.41	12	21.43
Casuarina pauper	49	18.15	11	37.93
Myoporum platycarpum	64	51.61	19	54.29
Recruitment past 30 years				
Alectryon oleifolius	240	43.40	27	47.62
Allocasuarina luehmannii	2	1.54	2	10.53
Callitris gracilis	92	21.65	9	16.07
Casuarina pauper	19	7.04	5	17.24
Myoporum platycarpum	59	47.58	17	37.14

5.1.2.1 Relationship between occurrence of regeneration and site parameters

Analyses were performed to determine if any relationship between the presence of recruits, and site variables was present. For most species, one of the highest correlations with the presence of juveniles was the number of mature trees (Table 5.6).

Canonical discriminant functions were calculated to see if sites with low or moderate regeneration could be distinguished from sites with no regeneration, based on floristic variables. Predicted group memberships are presented in Appendix 7 for all canonical discriminant functions. The first two canonical discriminant functions for *C. gracilis* regeneration groups were significant (Function 1: Wilks Lambda =0.256, $p<0.001$, Function 2: Wilks Lambda = 0.685, $P<0.01$), explaining 78.5 % and 21.5 % of the variance respectively. Figure 5.2 shows the separation of group centroids along the two functions.

Table 5.5. Shrub frequency, and number and percentage of sites where shrub regeneration is occurring (N=118).

Species	No. sites present	% frequency	No. sites regeneration	% regeneration
Tall shrubs				
Acacia colletioides	7	5.93	3	42.86
Acacia wilhelmiana	6	5.08	0	0
Beyeria opaca	6	5.08	3	50.00
Dodonaea viscosa	23	19.49	12	52.17
Eremophila glabra	7	5.93	1	14.29
Eremophila oppositifolia	5	4.24	5	100.00
Exocarpos aphyllus	9	7.63	2	22.22
Maireana brevifolia	49	41.53	15	30.61
Maireana turbinata	5	4.24	1	20.00
Olearia muelleri	7	5.93	4	57.14
Pimelea microcephala	7	5.93	5	71.43
Pittosporum phylliraeoides	18	15.25	9	50.00
Scaevola spinescens	7	5.93	5	71.43
Senna artemisioides ssp. *coriacea*	7	5.93	3	42.86
Senna artemisioides ssp. *petiolaris*	9	7.63	4	44.44
Small shrubs				
Atriplex stipitata	31	26.27	9	29.03
Atriplex vesicaria	67	56.78	42	62.69
Chenopodium curvispicatum	43	36.44	14	32.56
Chenopodium desertorum	41	34.75	22	53.66
Dissocarpus paradoxus	3	2.54	1	33.33
Einadia nutans	13	11.02	2	15.38
Enchylaena tomentosa	112	94.92	71	63.39
Eriochiton sclerolaenoides	10	8.47	5	50.00
Maireana appressa	6	5.08	1	16.67
Maireana pentatropis	25	21.19	8	32.00
Maireana pyramidata	7	5.93	1	14.29
Olearia pimeleoides	12	10.17	10	83.33
Osteocarpum salsuginosum	2	1.69	0	0
Rhagodia spinescens	2	1.69	1	50.00
Sclerolaena diacantha	72	61.02	48	66.67
Sclerolaena obliquicuspis	66	55.93	48	72.73
Westringia rigida	3	2.54	1	33.33
Zygophyllum apiculatum	11	9.32	4	36.36
Zygophyllum aurantiacum	14	11.86	2	14.29

Table 5.6. Spearmans correlation of regeneration of tree species in the last 30 years, with site variables (N=118).

Site variables	Recruitment of *Alectryon oleifolius*	*Callitris gracilis*	*Casuarina pauper*	*Myoporum platycarpum*	Recruitment abundance
%cover trees	0.143	0.249**	0.263**	-0.164	0.155
%cover tall shrubs	-0.004	0.262**	0.183*	0.086	0.240**
%cover small shrubs	0.062	-0.017	0.144	-0.068	0.012
%cover total ground layer	0.104	-0.042	-0.245**	0.235*	0.175
%cover native ground layer	-0.135	0.049	-0.139	-0.034	-0.136
%cover exotic ground layer	0.182*	-0.127	-0.257**	0.258**	0.211*
native ground sp. richness	-0.057	0.027	-0.023	-0.211*	-0.122
small shrub sp. richness	0.176	-0.011	0.243**	0.099	0.240**
tall shrub sp. richness	0.049	0.289**	0.240**	-0.054	0.200*
tree sp. richness	0.094	-0.046	0.216*	0.089	0.056
No. mature *Allocasuarina luehmannii*	-0.137	-0.032	-0.092	-0.178	-0.257**
No. mature *Alectryon oleifolius*	0.528**	-0.155	0.182*	-0.158	0.266**
No. mature *Callitris gracilis*	-0.172	0.352**	0.077	-0.180	-0.078
No. mature *Casuarina pauper*	0.134	0.009	0.434**	0.082	0.183*
No. mature *Myoporum platycarpum*	-0.113	-0.176	-0.129	0.544**	0.086
No. mature trees	0.280**	0.168	0.295**	-0.231**	0.145
No. shrubs regenerating	0.164	0.114	0.210*	0.052	0.247**
Av mature tree condition	0.021	0.124	0.146	-0.208*	-0.089

**Correlation is significant at the .01 level (2-tailed).
*Correlation is significant at the .05 level (2-tailed).

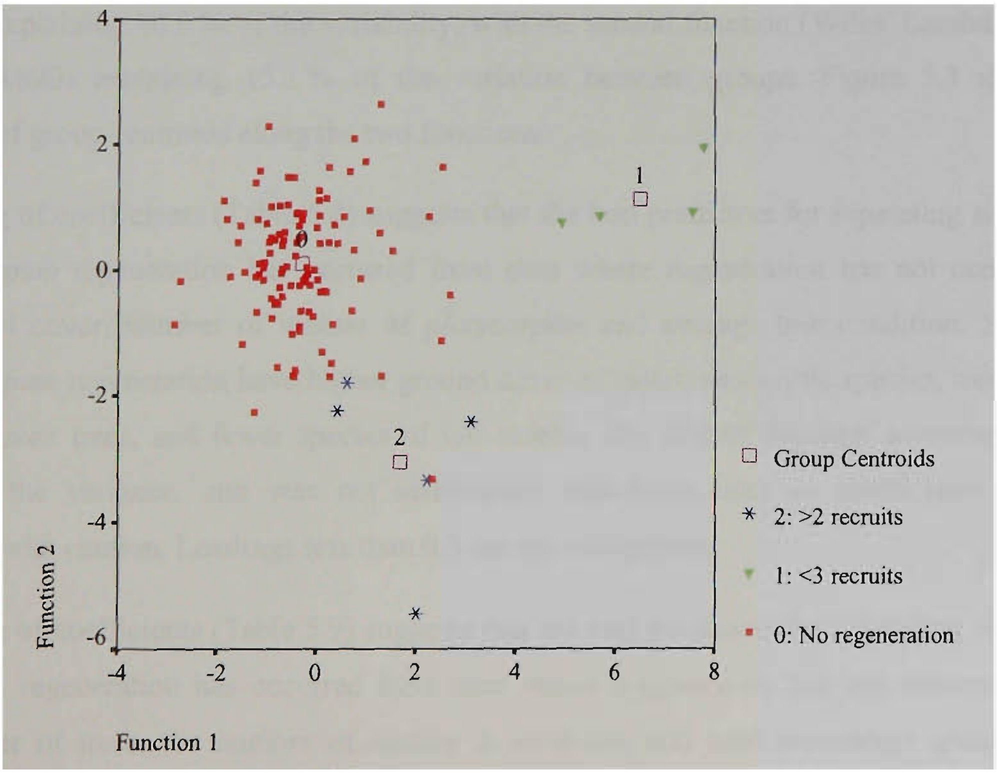


Figure 5.2. Plot of group centroid and sites of *Callitris gracilis* recruitment on two discriminant functions derived from 20 site variables.

The loading of coefficients (Table 5.7) suggests that the best predictors for separating sites where regeneration has occurred, from sites where regeneration is absent, are the number of mature *C. gracilis*, cover of tall shrubs, and cryptogam cover. Sites with *C. gracilis* recruits have higher numbers of mature *C. gracilis* trees, higher percentage cover of tall shrubs and less cryptogam cover than sites without recruits (Table 5.7). Higher percentage tree cover and more mature *C. pauper* were also associated with *C. gracilis* regeneration. The second discriminant function accounts for only 21.5 % of the variance, and separates the low regeneration sites from other sites, which is less ecologically relevant and has not been interpreted. Loadings less than 0.5 are not presented.

Table 5.7. Canonical discriminant function coefficients on function 1 and function 2 for groups based on recruitment of *Callitris gracilis.*

Variables	Function 1	Function 2
No. mature *Callitris gracilis*	0.726	
Tall shrub % cover	0.647	0.610
Cryptogam % cover	-0.529	0.743
Tree % cover	0.512	0.529
No. mature *Casuarina pauper*	0.501	0.769
No. shrubs regenerating		0.605
No. mature trees		-0.682

Eight sites were identified with low *M. platycarpum* regeneration, nine sites with moderate regeneration, and 91 sites without any regeneration. For *M. platycarpum* regeneration groups, only the first discriminant function was statistically significant (Wilks' Lambda=0.519, df 38, $p<0.001$), explaining 86.9 % of the variability, with the second function (Wilks' Lambda =0.901, df 18, $p=0.890$) explaining 13.1 % of the variation between groups. Figure 5.3 shows the separation of group centroids along the two functions.

The loading of coefficients (Table 5.8) suggests that the best predictors for separating sites where *M. platycarpum* regeneration has occurred from sites where regeneration has not occurred are total ground cover, number of mature *M. platycarpum* and average tree condition. Sites with *M. platycarpum* regeneration have higher ground cover of native and exotic species, more mature *M. platycarpum* trees, and fewer species of tall shrubs. The second function accounts for only 13.1 % of the variance, and was not statistically significant, and so coefficients must be interpreted with caution. Loadings less than 0.5 are not interpreted.

The loading of coefficients (Table 5.9) suggests that the best predictors for separating sites where *A. oleifolius* regeneration has occurred from sites where regeneration has not occurred are the total number of trees, the number of mature *A. oleifolius* and total percentage ground cover. Function 1 describes 85.9 % of the variance (Wilks' Lambda 0.428, df 38, $p<0.001$), and function 2 (Wilks' Lambda=0.858, df 18, $p=0.578$) 14.1 % of the variance.

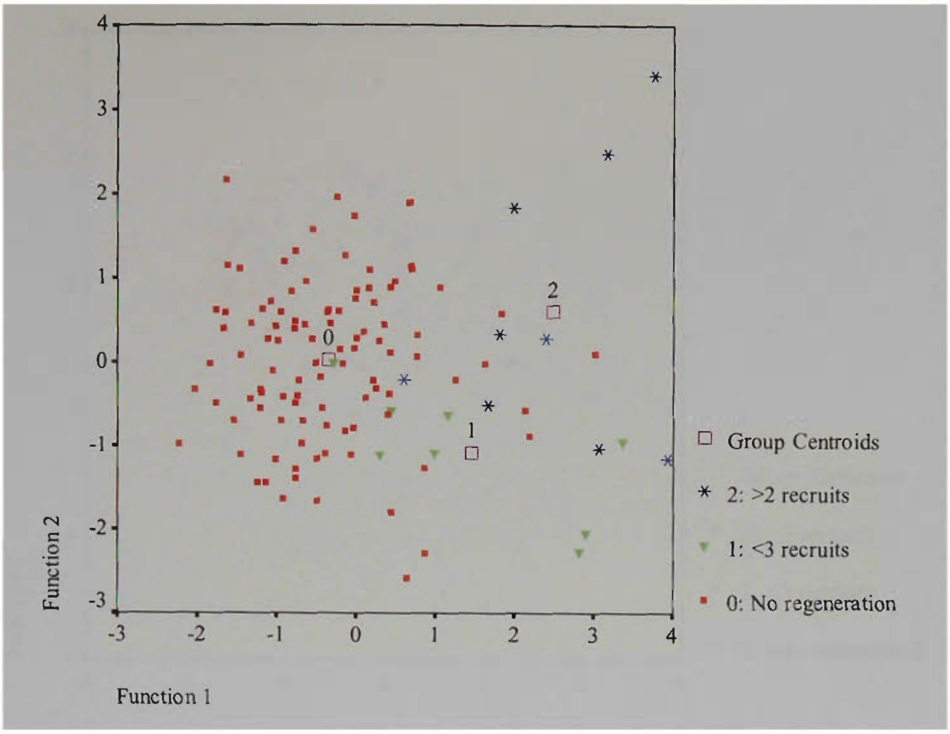


Figure 5.3. Plots of group centroid and sites of three groups of *Myoporum platycarpum* regeneration on two discriminant functions derived from 20 site variables.

Table 5.8. Canonical discriminant function coefficients on functions 1 and 2 for groups based on recruitment of *Myoporum platycarpum*.

Variables	Function 1	Function 2
Total ground % cover	0.842	1.365
No. mature *Myoporum platycarpum*	0.819	
Tall shrub richness	-0.598	
Av. tree condition	0.591	
Bare ground % cover		1.187
Cryptogam % cover		0.918
Tree % cover		0.565
Tree sp richness		-0.503

Figure 5.4 shows the separation of group centroids along the two functions. Sites with *A. oleifolius* regeneration have higher ground cover of native and exotic species, and more mature *A. oleifolius* trees, with fewer species of tall shrubs. The second function accounts for only 13.1 % of the variance, and was not statistically significant, and so coefficients must be interpreted with caution. As the second function separates the low regeneration sites from other sites, it has not been interpreted. Loadings less than 0.5 are not interpreted.

Table 5.9. Canonical discriminant function coefficients on functions 1 and 2 for groups based on recruitment of *Alectryon oleifolius*.

Variables	Function 1	Function 2
No. mature trees	0.984	0.702
No. mature *Alectryon oleifolius*	0.733	
Total ground % cover	0.683	
Bare ground % cover	0.539	
No. mature *Casuarina pauper*		-1.080

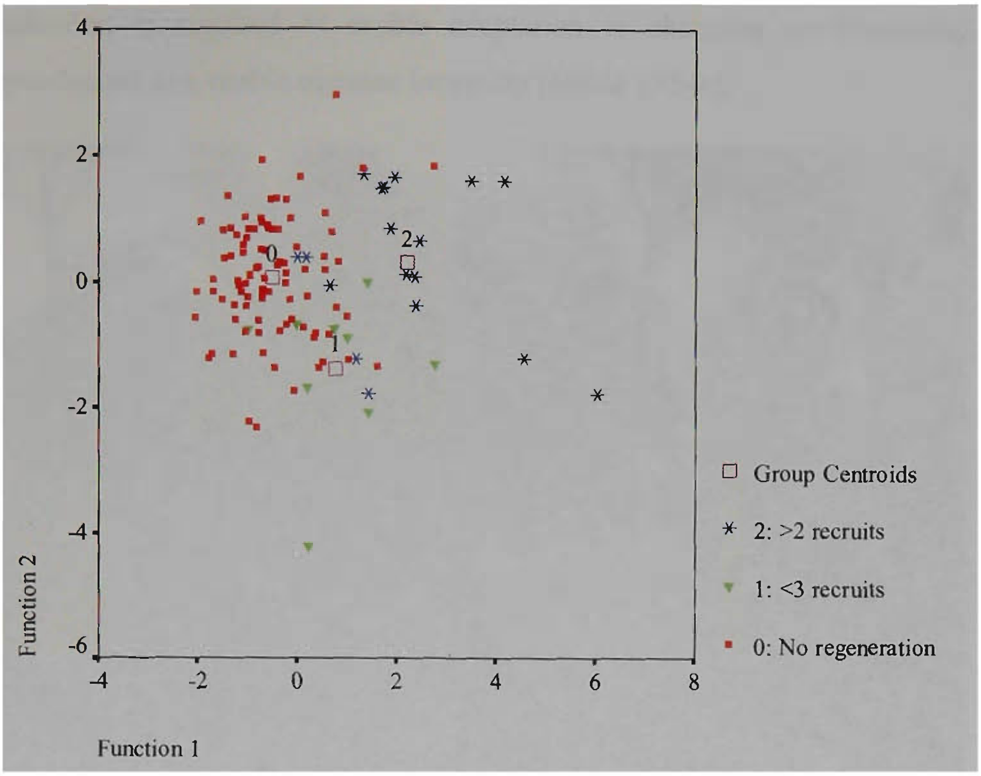


Figure 5.4. Plots of group centroid and groups of *Alectryon oleifolius* recruitment on two discriminant functions derived from 20 site variables.

5.1.3 Discussion

Regeneration of perennial species in the MSNP

Relatively few trees were observed in the smaller size classes of most species in the current study, with the most recruitment found for *A. oleifolius* (Figure 5.1). Some regeneration of *M. platycarpum*, *C. gracilis*, *Hakea leucoptera* and *H. tephrosperma* has also occurred. Very few recruits of *C. pauper*, *A. luehmannii*, *Acacia oswaldii* or *Eremophila longifolia* were observed (Table 5.3).

It is important to note that much of the observed regeneration of *A. oleifolius*, *C. pauper*, *H. leucoptera*, *H. tephrosperma* and *A. luehmannii* appears to have occurred largely from root suckering. Whilst no recruits were excavated, the proximity to mature individuals and presence of soil disturbance in most cases suggests these are likely to be suckers. For most suckers of *C. pauper* and *A. luehmannii*, the exposed lateral roots from the parent tree were clearly visible or a line of trees along a lateral root was evident (Plate 5.1).

Root suckering typically occurs where seedling regeneration is inhibited by severe environmental conditions, with the extension of life cycle of the plant enabling seedling regeneration to occur in rare opportunities of more favourable conditions (Lacey and Johnston 1990). Adventitious buds can lead to clonal growth from roots that have been stimulated by root damage, exposure to light, change in temperature, or loss of apical dominance of the parent tree (Lacey and Johnston 1990).

Sexual reproduction is required to enable adaptation to changing environments, however, vegetative reproduction can enable extreme longevity (Noble 1984a).

Plate 5.1. ***Casuarina pauper*** **root suckers occurring along a lateral root in an eroded site within the grazing exclosure, MSNP.**

Spatial distribution of regeneration

Recruits of most tree species appear to be scattered throughout the MSNP, suggesting thresholds for natural recovery of tree species have not been exceeded at the broad scale within the study area.

Around the Taparoo area of the MSNP, occurrences of *A. oleifolius, C. gracilis* and *M. platycarpum* regeneration have been observed (personal observation). The vegetation in this area is in relatively poor condition following heavy grazing pressure prior to the declaration of the MSNP. Regeneration at Taparoo of some tree species was first observed in the early 1990s following extensive rabbit control work (Sandell *et al.* 2002). The reasons regeneration occurred at this site, and not elsewhere across the MSNP are unclear, but may be related to rainfall, soil disturbance following ripping, or successful rabbit control.

Above average rainfall occurred across the study area in 1992/3, coinciding with extensive soil disturbance from the rabbit control works, which is likely to have contributed to favourable conditions for establishment of these species. Alternatively, a higher rainfall event may have occurred in the Taparoo area that was not received elsewhere in the region, with patchy rainfall distribution common in semi-arid regions.

Shrub regeneration

Recruits of most shrub species were identified at more than 50 % of sites (Table 5.5). As episodic regeneration occurs for most perennial species in semi-arid areas, it is not expected that regeneration will be observed for all species at any one time. However, there is cause for concern when failure to regenerate is observed for a number of species at large temporal and spatial scales.

The most extensive shrub recruitment is observed for the most frequently occurring shrub species, *E. tomentosa*, *Atriplex vesicaria*, *Sclerolaena diacantha*, and *S obliquicuspis*. These shrubs were observed to be only minimally grazed in the MSNP (personal observation), although varying reports exist on the relative palatability of these species.

Predictors of tree regeneration

The need for mature trees as a source of propagules has frequently been identified as a threat to future recovery, with most mature trees pre-dating European settlement and many reaching senescence (Westbrooke *et al.* 1988; Cheal 1993; Westbrooke 1998; Westbrooke *et al.* 2001). The presence of recruits of all species was positively correlated with the number of mature trees of the species, and for *A. oleifolius* and *C. pauper*, regeneration was positively correlated with the total number of trees present (Table 5.6). This was also identified in the factor analysis, which found the presence of mature trees was a predictor of regeneration for all species analysed.

Variables showing a positive correlation to total abundance of tree species regeneration included percentage cover of tall shrubs, small shrub species richness, tall shrub species richness and number of shrubs regenerating. Many studies have found a decrease in species richness with grazing, and so sites with higher species richness would be expected to have experienced less grazing pressure. Similarly, cover of tall shrubs and regeneration of shrub species are found only in less disturbed sites, with decreased overall grazing pressure.

A positive correlation was also observed with exotic ground species cover and abundance of tree regeneration. This is probably related to the effects of disturbance, which is known to promote suckering. Many other affects of soil disturbance may improve chances of regeneration such as increased moisture retention, increased soil nutrients and creation of safe sites for germination (Cunningham *et al.* 1976; Hobbs and Atkins 1988; Stoneman *et al.* 1994; Eldridge and Robson 1997; Kotanen 1997; Rokich *et al.* 2001).

Recruitment of *A. oleifolius* was positively correlated with the number of mature *A. oleifolius*, the total number of mature trees and the percentage cover of exotic ground layer species (Table 5.6), suggesting a requirement for propagule source and disturbance. Similar variables were identified as predictors of recruitment in the discriminant analysis with number of mature *A. oleifolius* and number of mature trees loading most heavily on the first function. Total percentage ground cover and percentage bare ground were also identified in the discriminant analysis. Total percentage ground cover and percentage exotic covers were highly correlated variables (Table 4.8) due to the generally high exotic ground layer at most sites. Percentage bare ground may also suggest a requirement for soil disturbance for suckering.

C. gracilis regeneration was correlated with variables indicating sites of better condition, or lower grazing pressure, such as percentage cover and species richness of tall shrubs, and percentage cover of trees (Table 5.6). This was supported by the discriminant analysis, which found *C. gracilis* recruitment occurring at sites with more mature *C. gracilis* and *Casuarina pauper* trees and higher percentage cover of tall shrubs. Percentage cover of cryptogams was a predictor of sites not supporting recruitment. *C. gracilis* recruitment was commonly observed on sites with high soil disturbance, such as mobilised dunes. Zimmer (1944) also found that removal of the cryptogamic layer was essential for regeneration of *C. gracilis*.

C. pauper regeneration was positively correlated with variables suggesting regeneration is occurring at more densely vegetated sites in good condition. Strong positive correlations were found with variables such as tree cover, tall shrub cover, species richness, and shrub regeneration (Table 5.6).

Correlations suggest recruitment of *M. platycarpum* is occurring in more open sites, with negative correlations with total percentage tree cover, native ground species richness, and a positive correlation with the percentage cover of exotic ground layer (Table 5.6). This is supported by findings of the discriminant analysis, which suggests recruitment is more likely to occur on sites with high total ground cover, but low tall shrub species richness. A high number of mature *M. platycarpum* in good condition, were also important predictors of *M. platycarpum* regeneration.

5.2 EFFECTS OF WATER ADDITION, SOIL DISTURBANCE AND FIRE ON REGENERATION OF PERENNIAL SPECIES OF *CASUARINA PAUPER* WOODLAND

There has been much research on the impacts of grazing on regeneration of semi-arid perennial species (eg. Barker 1972; Crisp 1978; Lange and Willcocks 1980; Auld 1995a, 1995b; Tiver and Andrew 1997), however little attention has been given to establishing whether perennial species such as *C. pauper* will regenerate successfully in the absence of grazing, and under what conditions germination and early seedling establishment may take place (Westbrooke 1998).

There is a view that well above average rainfall over two to three years is required for regeneration of perennial species in semi-arid environments, and this has been supported by comparison of rainfall records with dates of known seedling establishment (Wood 1936; Woodell 1990; Westbrooke 1998). However, germination and early seedling survival have not been experimentally tested under varying water conditions in the field.

C. pauper and other perennial *C. pauper* woodland species are easily grown in the glasshouse, but the problems limiting field regeneration have not been elucidated in the laboratory. Glasshouse and laboratory studies are limited in the extent to which they can mimic field conditions and in the ability to translate results into the field situation. Glasshouse studies have not been able to shed light on the complex interaction of factors necessary for regeneration in semi-arid woodland.

However, there are also many limitations with field regeneration studies. Studies of field regeneration are highly reliant on episodic events, and are therefore not well suited to short-term studies. Watson *et al* (1997b) note the problems in regeneration studies in the arid zone "For arid systems where change can be slow and incremental but can also occur rapidly and unpredictably, the data required … would have to be collected over decades unless (by chance) a major episode happened to occur during a shorter study".

Even long-term studies conducted within grazing exclosures have frequently failed to show regeneration of dominant perennial species (Hall *et al.* 1964; Ludwig *et al.* 1997). Lack of observed regeneration may result from infrequent and irregular monitoring of long-term studies (Hall *et al.* 1964). Another inherent problem with this type of research is the spatial scale of regeneration compared to the exclosure size, whereby low levels of recruitment may fail to be monitored due to the small plot or exclosure size.

The majority of data on regeneration in semi-arid trees and shrubs has been modelled using the size structure of populations as a surrogate for age structure (eg. Crisp and Lange 1976; Auld 1990; Batty and Parsons 1992; Westbrooke 1998). Similarly, much of the evidence for regeneration of perennial species within *C. pauper* woodland is based on size structure of populations, and anecdotal evidence (Chesterfield and Parsons 1985; Westbrooke 1998)). Limitations with this type of study include difficulties in determining if germination occurred during the event, or if increased growth of seedlings during the event leads to similarly sized cohorts (Johnson *et al.* 1994; Watson *et al.* 1997b).

A research design that has commonly been used in field regeneration studies involves manipulation of relevant variables (fire, soil, seed application etc.) to determine triggers and requirements for regeneration. This type of manipulative regeneration experiment has been undertaken in many ecosystem types, and has provided useful data on recruitment for many species (Parsons 1968; Tadmore *et al.* 1968; Castellanos and Molina 1990; Adams *et al.* 1992; Battaglia 1996). However, this type of experiment has not commonly been used in arid and semi-arid zones due to the potentially long period between rainfall events.

In this trial, a watering treatment was included to attempt to simulate two years of well above average rainfall. If successful, replicating episodic rainfall has the potential to enable investigation of the regeneration niche of semi-arid trees and shrubs that are rarely observed as juveniles in the field.

Four elements were identified from the literature as essential to the regeneration of semi-arid woodland (i) protection from grazing (ii) well above average rainfall, (iii) sufficient viable seed, and (iv) disturbance. Well above average rainfall and protection from grazing both appear to be an important requirement, but not necessarily sufficient to lead to regeneration of perennial species. Fire and soil disturbance were identified as two types of disturbance important for woodland regeneration, however, the extent to which these disturbances may influence regeneration in *C. pauper* woodlands is largely unknown. Therefore, the objectives of this research were to:

(i) determine if water addition in quantities recorded in "event" years can replicate conditions of soil moisture to trigger episodic regeneration of *C. pauper* woodland perennial species;
(ii) determine the effects of well above average rainfall on germination and seedling survival of perennial species within a *C. pauper* woodland remnant under low grazing conditions;
(iii) determine if soil disturbance may influence germination and subsequent seedling survival of perennial *C. pauper* woodland species;
(iv) determine the effects of fire on perennial seedling germination and seedling survival under low grazing conditions; and
(v) determine the effects of seed application on perennial regeneration under low grazing conditions.

5.2.1 Methods

5.2.1.1 Site description and experimental design

Site description

The study was undertaken within *C. pauper* woodland in the MSNP (Plate 5.2). *C. pauper* dominated the site, with a total tree density of approximately 100 trees ha^{-1} (Appendix 11). A total of 16 species of tall and low shrubs were recorded at the site (See Appendix 8 for a complete species list for the site).

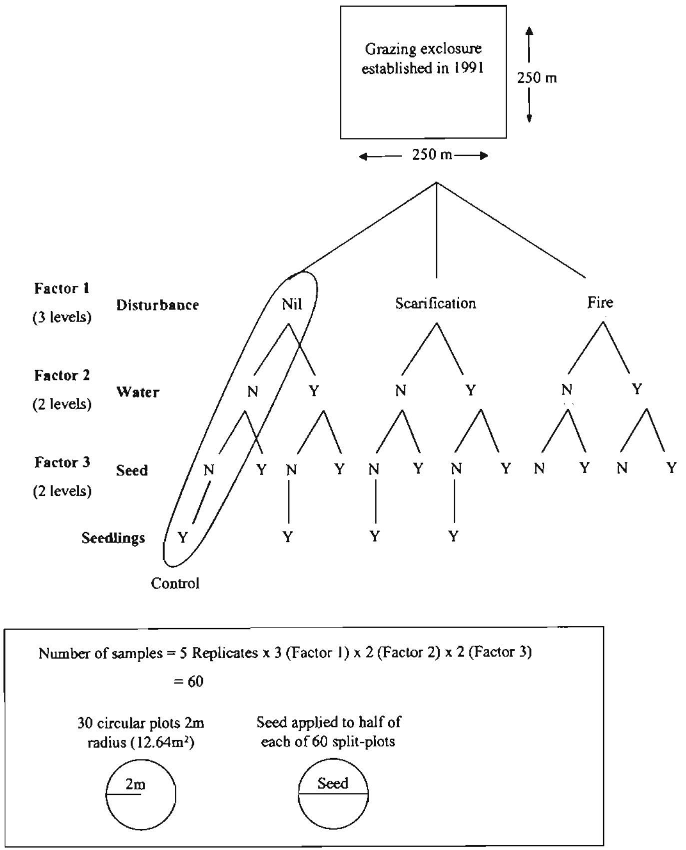
Plate 5.2. Grazing exclosure (34°32', 141°06') in the MSNP showing an overstorey dominated by *Casuarina pauper*, and extensive *Austrostipa* spp. cover in 2000.

Experimental design

A randomised split plot design was adopted with three factors, (i) disturbance (three levels, no disturbance, soil disturbance and fire) (ii) water addition (two levels, watered and unwatered) and, (iii) seed application (two levels) (Figure 5.5). A randomised approach was chosen over a blocked design due to the possibility of water addition affecting other neighbouring treatments, and to assist in control of extraneous variables.

Potential extraneous variables identified included canopy cover and shading, variability in soil, ground cover and landscape function variables such as cryptogamic cover, and soil water infiltration rates. Within a relatively homogenous tree canopy of a woodland patch, the majority of differences in landscape function would be at the intermediate and small-scale (Ludwig *et al.* 1997). Sloping areas and fallen trees within the exclosure were excluded to remove much of the intermediate scale variability in resource capture. Randomisation and a plot size of 12.5 m^2 were used to capture small-scale variability in resource capture throughout the exclosure.

The study was undertaken within a large grazing exclosure (6.25 ha) as the effect of grazing on regeneration has been identified as the major cause for failure of regeneration in *C. pauper* woodland since European settlement (Cheal 1993; Westbrooke 1998). The effects of grazing on regeneration have been investigated in many previous studies (eg. Auld 1995a; Auld 1995c; Tiver and Andrew 1997). If a study of the effects of grazing on regeneration were to be undertaken, multiple exclosures would be required for independent grazing treatments to avoid pseudoreplication (Hurlbert 1984).



Y = Yes

N = No

	2000		2001						2002					
	Sep	Nov	Jan	Feb	Apr	Jun	Sep	Nov	Jan	Feb	Mar	Apr	Jun	Sep
Treatment														
Scarification	●				●									
Fire					●									
Water	●	●	●	●	●	●	●	●	●	●			●	
Weekly water											●	●		
Seed					●		●				●			
Seedlings planted							●				●			
Measurement	★		★		★	★	★		★			★	★	★

Figure 5.5. **Schematic diagram of plot treatments and time frames for field regeneration experiment.**

Ten plots were also established outside the exclosure to provide comparative data on water and seed application in the presence of grazing. The ten plots in the grazed area are true randomised replicates for the treatments in each plot (water addition and seed application). However, the plots represent pseudoreplicates if grazed treatments were compared to ungrazed treatments. Hence the design allows for the valid determination of the effect of treatments within a grazed site and the effect of treatments within an ungrazed site but does not allow a statistical analysis of the differences between grazed and ungrazed sites.

A case-study approach was selected to trial this new methodology for investigating episodic regeneration in semi-arid regions. This approach was chosen to determine the effectiveness of the methodology prior to landscape scale investigation due to the logistics of water addition in an environment characterised by limited water resources. As few similar methodologies have been tested previously, it was difficult to determine the probable outcomes.

A case-study approach where one large grazing exclosure is used for all replication, limits the ability to generalise the results to all *C. pauper* woodland. This was partially addressed by the vegetation condition surveys undertaken across the MSNP, which suggested that the exclosure site was typical of many moderate condition sites in northwest Victoria (Section 4.2.2). Underwood (1997) suggests use of such supportive evidence can increase confidence in the generality of localised studies.

This research design has greatest potential to assist in understanding regeneration requirements in semi-arid *C. pauper* woodland if implemented in other studies within *C. pauper* woodland in other locations and other semi-arid woodland types.

Exclosure selection

Grazing exclosures (250 x 250 m, 6.25 ha) were erected in the MSNP between 1991 and 1995 for long-term monitoring of grazing pressure and its effects on vegetation (Sandell *et al.* 2002).

Four of these exclosures contained *C. pauper* woodland, and an appropriate location for the current research was selected from a survey of the exclosures conducted in March 2000. Requirements for an appropriate exclosure included (i) an overstorey of *C. pauper* woodland species, including *C. pauper*, *A. oleifolius*, *C. gracilis* and *M. platycarpum*, (ii) a relatively homogenous vegetation composition and structure to minimise variability within the exclosure, (iii) a relatively flat area to minimise runoff during water addition, and (iv) a degree of resilience, as evidenced by a largely native and diverse understorey and a relatively intact overstorey. Logistical requirements dictated a water source in reasonable proximity. Only one exclosure met all these requirements.

Number of replicates

The number of replicates was determined from an *a priori* power analysis using Gpower (Erdfelder *et al.* 1996). Little relevant data were available from similar studies, and so an estimated data set was produced based on expert opinion (Westbrooke, M [University of Ballarat] 2000 pers. comm., 21 June). Assumptions were that in an average rainfall year no regeneration of most perennial shrub and tree species would be observed and minimal regeneration of some small shrub species. It was expected that any treatment that led to successful regeneration would have a large positive effect. The nature of episodic regeneration is such that when regeneration requirements are met, extensive germination and subsequent seedling survival of perennial species occurs (Stafford Smith and Morton 1990). The mean number of seedlings that may establish per plot was estimated to be between 0–12, with standard deviation ranging from 0–7.

A very large effect size of 4.10 was calculated using Gpower, with an ANOVA Fixed Effects Multi-Factor Design (Buchner *et al.* 1997). A more modest effect size of 0.50 was used to calculate the required sample size for an alpha value of 0.05 and a power level of 0.90. To calculate the main effects, two degrees of freedom were used. A total of 55 samples were required to determine main effects with the requisite power level. To ensure an equal number of samples per treatment, 60 plots were investigated (5 replicates of each treatment).

Plot allocation and design

It was decided to investigate open spaces beyond the canopy of mature trees as: (i) effective natural regeneration of degraded woodland requires germinants to colonise gaps beyond the canopy of existing trees; (ii) juveniles close to existing mature trees may have to compete for water and other resources; and, (iii) to minimise impacts of treatments on mature trees in the exclosure.

To assist in random selection of quadrats, all overstorey vegetation was mapped throughout the 6.25 ha exclosure and a 50 x 250 m belt adjoining the exclosure (Appendix 11). The exclosure was divided into 25 50 x 50 m quadrats using baling twine and wooden stakes to delineate quadrats. The location of all trees was mapped within each 50 m quadrat. Each tree was then entered into a GIS database, with species name, and status as mature or juvenile.

Two 50 x 50 m quadrats on a slope (>5°), and a 200 x 25 m permanent belt transect used by Parks Victoria for monitoring within the exclosure were excluded from the available area for plot allocation, leaving 5.25 ha for random allocation of plots. Using MapInfo (MapInfo Corporation), a 7 m buffer was created around each tree to exclude these areas from the random selection. A 7 m buffer ensured that all areas of a plot of 2 m radius would be at least 5 m from any existing tree. A randomisation program then selected 40 points from the map of potential sites. Treatments were allocated in the order that the plots were randomly selected, with the first five points allocated to water and scarification, the second to scarification, then water and fire, fire, water, and control. Ten extra sites were selected to allow substitution of any plots that were allocated to unsuitable areas such as unmapped fallen trees. Three plots were substituted in the field due to proximity to the permanent belt transect, and fallen trees.

Plot size was determined from the expected seedling density that would allow sufficient seedling numbers for a reliable statistical analysis. The logistics of water addition and coverage of various sprinkler types were also taken into consideration. An area of 12.57 m^2 (circle radius 2 m) was determined to be an appropriate size to fulfil these criteria. A circular plot enabled easy watering by a standard sprinkler.

Sample size, and number of replicates are comparable to those used in similar manipulative experiments studies of seedling establishment (Castellanos and Molina 1990; Battaglia 1996; Setterfield 2002).

5.2.1.2 Plot treatments

Water addition

The aim of water addition was to apply a quantity of water which, combined with actual rainfall in the field, would equal rainfall during "event" years of 1973/4/5 and 1992/3. The quantity of water applied was determined by analysis of rainfall records of known regeneration events (1973/4/5 and 1992/3).

During the years 1973/4/5, well above average rainfall was recorded across much of south-eastern Australia (Clewett *et al.* 2003). This event was one of the largest consecutive years rainfall since records began, for much of semi-arid south-eastern Australia (Clewett *et al.* 2003). Regeneration of a number of semi-arid and arid perennial species was noted during this period, however, few seedlings survived, which has been largely attributed to high grazing pressure (Chesterfield and Parsons 1985; Woodell 1990; Batty and Parsons 1992). Little evidence remains of regeneration of perennial species at the MSNP during this event (Westbrooke 1998), despite the heaviest consecutive years rainfall on record at nearby weather stations (Clewett *et al.* 2003). Limited regeneration of perennial species in the MSNP has been attributed to the above-average rainfall years of 1992/3 (Sandell *et al.* 2002).

An average of the monthly rainfall from these event years was compiled from four rainfall stations near the northwest MSNP (Figure 5.6); Lindsay Point, Meringur Post Office, Werrimull and Morkalla North (see Figure 2.7. for location of rainfall stations). This "event average" represents five of the highest consecutive year's rainfall in the 20th century.

Watering was undertaken to "top up" natural rainfall to the quantities observed in these months. Watering was initially performed six times per year in early spring (September), late spring (November), early summer (early January), late summer (end of February), autumn (April) and Winter (June). A feature of these event years is well above average rainfall in spring and summer (Figure 5.6) therefore, water additions were planned at least every two months during these times. Watering was performed to top up natural rainfall to the levels expected to have been received by that date. For example, 78 mm fell in January and February during the "average event" year, if by the end of February only 20 mm of rainfall were recorded, 58 mm of additional watering would be required (Figure 5.7).

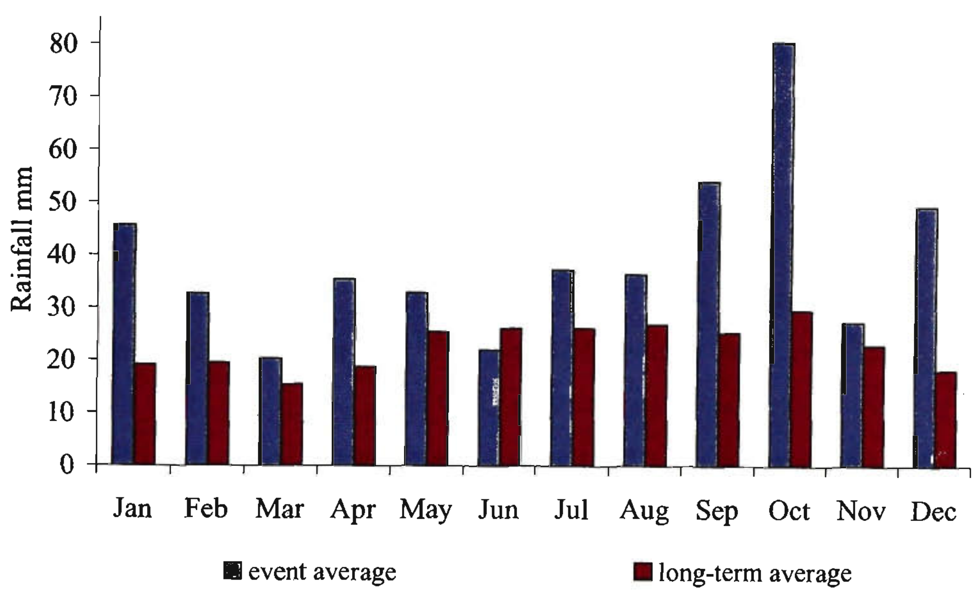


Figure 5.6. Average monthly rainfall received during the "event" years of 1973/4/5 and 1992/3, compared with long-term average data at four rainfall stations in northwest Victoria; Lindsay Point, Meringur, Werrimull and Morkalla North.

A total area of 392.7 m^2 was watered over 40 split plots with a radius of 2 m, plus buffer of 50 cm radius. Two 1,000-litre tanks delivered the equivalent of 5.1 mm of rainfall to this area. This was confirmed by placing small sample jars on each plot to measure the amount of water received.

Field precipitation was measured in a rain gauge within the exclosure, which was measured every one to two months, or following high rainfall events.

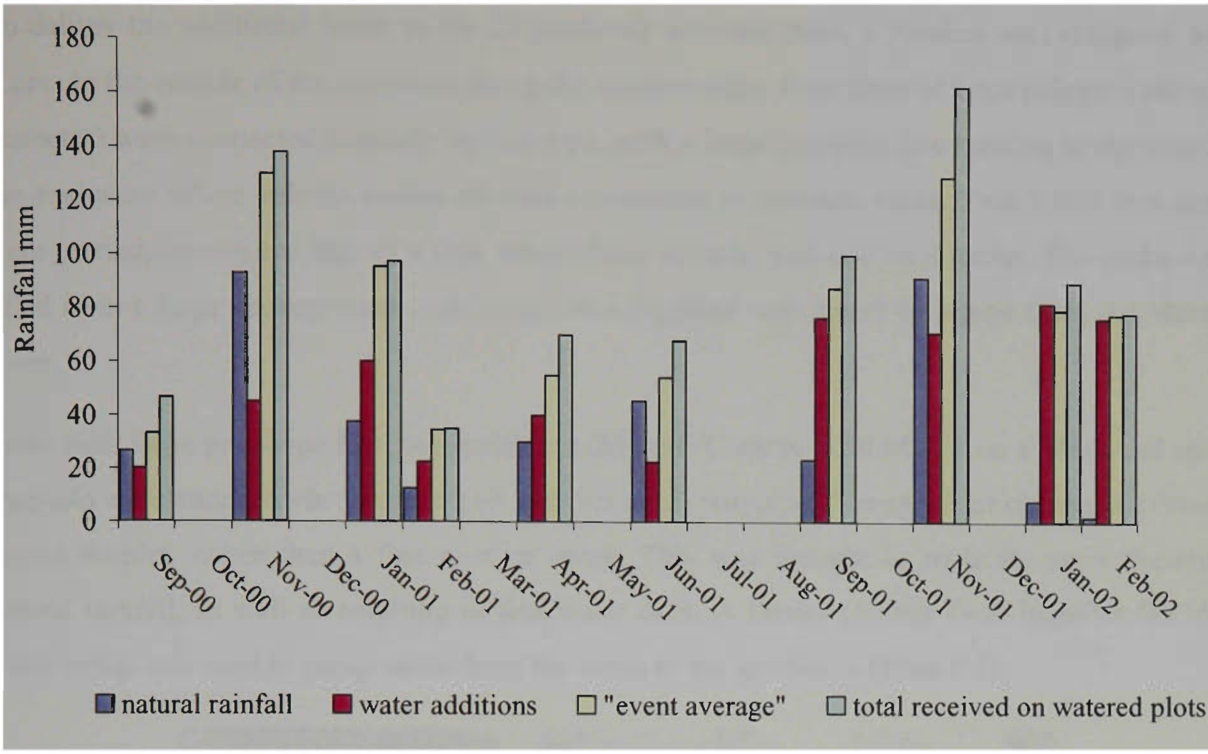


Figure 5.7. Water regime during the experiment showing total received on watered plots and unwatered plots (natural rainfall) compared to the calculated "event average".

Late summer or early autumn has previously been identified as a likely time for regeneration of *A. luehmannii*, a closely related Casuarinaceae in northwest Victoria (Morcom 2000). In autumn 2002, a second watering regime was trialed. Daily rainfall records from Morkalla Post Office, the closest rainfall station to the study site, were examined to determine the wettest summer–autumn period during the 1973/4/5 event. It was found that in 1974, 188 mm rainfall was recorded in a six-week period between March and April. Weekly watering to replicate this was applied over a six-week period from the 19th of March to the 28th of April. The total rainfall quantity was divided into equal weekly amounts of 31 mm, rounded up to 35 mm per week, which was applied over 2–3 days each week.

Method of water addition

A pilot study was conducted during June 2000 to test the logistics of the water addition and determine the rate of soil infiltration and time to runoff. Water was applied to a plot of 2 m radius using a pump and garden sprinkler. During the initial application 150 litres was applied to the 12.57 m^2 area over eight minutes. Runoff commenced following eight minutes watering and water addition was ceased. Half an hour later 120 litres was applied over six minutes before runoff started again. A further 120 litres was applied over six minutes 30 minutes later. This equated to approximately 30 mm of rainfall delivered to the 12.57 m^2 plot. This established that a water quantity similar to that received during well above average rainfall years could be delivered to plots over a period of a few days.

To deliver the additional water to the 20 randomly selected plots, a pipeline was designed with access to the outside of the exclosure along the western edge. Four lines of large polypipe (40 mm diameter) were connected centrally to four taps, with a large polypipe line running to the front of the exclosure where vehicle access allowed connection to portable tanks. Two 1,000 litre tanks were carried, one on the tray of a four wheel drive vehicle, and one on a trailer. The tanks were filled from a large aboveground tank, which was supplied with water by a pipe from the Murray River.

From each large poly-pipe line, ten sprinklers (Moss F/C spray head MG51 on a Moss end spike snap-on) were attached via narrower (19 mm diameter) polypipe. The sprinkler chosen delivered a coarse droplet, rather than a fine mist or spray. This was thought to replicate more closely a natural rainfall, as well as resulting in less water drift. A Davey (5.5Hp Twin impeller GX160) water pump was used to pump water from the tanks to the sprinklers (Plate 5.3).

Plate 5.3. Sprinklers watering experimental plots within the grazing exclosure at the MSNP.

High wind conditions could greatly affect the delivery of water, blowing the water from the sprinkler outside of the experimental plot, and increasing evaporation rates. Therefore watering was always done during low wind conditions, and where possible, during early morning and late evening to minimise evaporation.

Large numbers of *Apis mellifera* (Honey Bees) were attracted to the water used for irrigation. Thousands of feral Honey Bees drowned in the concrete and portable water tanks, subsequently blocking the sprinklers. This problem was rectified by installing a water filter (ARKAL 40 mm diameter super filter No. 1152-0 Mesh 120-130 micron) into the sprinkler system.

Rate of water addition

Soil infiltration rate was the largest limiting factor for water addition. Watering was monitored to minimise runoff from the plot sites. Watering was ceased once run-off was observed. Occasionally small quantities of runoff occurred from some plots, particularly those on a slope and plots with lower vegetation cover.

Water quality

Prior to the initial rainfall simulation, water from the water storage tank to be used for the water addition treatment was tested for electrical conductivity using a Model 103 CHK electrical conductivity instrument. On initial testing in June 2000 electrical conductivity was found to be 740 µS/cm. Subsequent readings between June 2000 and March 2002 ranged between 540–790 µS/cm (mean 710 µS/cm, N=8). Water quality guidelines for agricultural irrigation suggest water with less than 1300 µS/cm is appropriate for all crops except those with extremely low salt tolerance (DNR 1997). As levels were found to be significantly below this measure water salinity was unlikely to pose a problem for irrigation. Germination trials using water collected from the tank also found no significant difference in germination for any seed species tested (Section 5.5.2)

Fire

On 19th April 2001, a fire treatment was carried out on five watered and five non-watered plots within the grazing exclosure. Autumn was chosen for the fire treatment, as this was the first available time for safe burning, after summer fire-bans were lifted. Very limited data are available on natural fire regimes of *C. pauper* woodland, but fires may have occurred in late summer as *Austrostipa* spp. dried off. An LPG gas flame-thrower constructed by Department of Sustainability and Environment (DSE) was used to heat soil and burn all above ground biomass to simulate a low intensity fire (Plate 5.4). The use of gas prevented residue formation on the plots that may result from burning liquid fuels, and potentially could effect germination.

A variable heat was deemed acceptable, as a range of temperatures is required for optimal germination of different species and also, a naturally occurring bushfire is likely to produce variable heating effects according to the extent and distribution of fuel, season of fire, wind (Hobbs and Atkins 1988).

Only in exceptional years does *C. pauper* woodland support an understorey sufficient to carry a fire. At the time of this experiment, most plots supported some cover of *Austrostipa scabra* ssp. *falcata* and other grasses, however, ground cover remained relatively low, and so approximately 1 kg dried biomass of straw was added to each plot. Addition of straw produced some smoke and ash that would not otherwise have resulted from direct heating of the soil.

Plate 5.4. Fire treatment being applied within the grazing exclosure at the MSNP April 2001.

Maximum temperature at the soil surface and 1 cm below the surface was recorded using temperature sensitive crayons, which change colour on exceeding a temperature threshold. Lines of six different crayons, sensitive to temperatures of 120, 150, 175, 200, 280 and 320°C respectively, were drawn on 10 thin sheets of aluminium, which were then folded in half to protect the crayon from direct exposure to flame, and soil. Five aluminium sheets were then placed on the soil surface, and five sheets 1 cm below the soil surface.

Aluminium sheets were retrieved after the fire had passed. At the soil surface, three sheets showed a change in colour of the first crayon line, indicating temperatures greater than 120°C, but less than 150°C. The remaining two sheets recorded colour changes in the first two lines, indicating temperatures greater than 150°C, but less than 175°C.

At 1 cm below the soil surface the first crayon line changed colour on one sheet indicating temperatures exceeding 120°C, and no change was recorded on the remaining sheets. At two plots (34 and 16) small logs on the plot burnt for over two hours, producing more ash and smoke than on other plots. Higher temperatures are expected to have occurred at these sites.

Soil disturbance

Soil disturbance was undertaken on ten plots within the exclosure. All ground cover vegetation was uprooted, and litter was turned over, but left in situ in September 2000 using a rake. Soil was scarified to a depth of 5 to 10 cm. In April 2001, before sowing seed, the soil was scarified again using a MINI-CULTI rotary hoe attached to a Stihl brush cutter to ensure a soil seedbed with minimal cryptogamic crust and no living plant cover at the time of seeding (Plate 5.5). Cryptogamic cover has been reported to prevent germination of *C. gracilis* and may form an inappropriate seedbed for other species (Zimmer 1944).

Plate 5.5. Scarification of soil using a brush cutter with hoe attachment within the grazing exclosure at the MSNP April 2001.

Seed application

To ensure regeneration was not limited by lack of a viable seed source, a replicated seeding treatment was included within the experiment. Seed of perennial species common to *C. pauper* woodland was collected by Greening Australia between October 2000 and April 2001, and between October 2001 to February 2002 (For seed collection, storage and viability see Section 5.5.1.).

As the goal of the experiment was to understand regeneration requirements, calculating desirable numbers of germinants per unit area was not necessary. Seed viability was not determined until after the first seed application, and so seed quantity was determined by the amount of seed available. Most seed samples contained a proportion of leaf litter and chaff that was not removed by sieving and so weights do not directly equate to seed weight. For tree species where sufficient seed allowed, approximately 200 seeds were applied to each plot. Fifty to 100 seeds of all other species were applied to each plot where seed was available (Table 5.10).

Table 5.10. Species and quantity of seed applied to each experimental plot.

Species for which seed was applied:	April 2001		September 2001		March 2002	
	seeds	(g)	seeds	(g)	seeds	(g)
Acacia colletioides	100	2.2	100	2.2	50	1.1
Acacia oswaldii	50	4.5	*		50	4.5
Alectryon oleifolius	200	12	100	6	*	
Callitris gracilis	100	1.8	50	0.9	50	0.9
Casuarina pauper	200	0.6	200	0.6	200	0.6
Dodonaea viscosa ssp. *angustissima*	200	2.0	200	2.0	200	2.0
Myoporum platycarpum	200	0.6	100	0.3	*	
Olearia pimeleoides	200	4.4	200	4.4	200	4.4
Pittosporum phylliraeoides	100	2.2	100	2.2	50	1.1
Senna artemisioides	200	1.2	200	1.2	200	1.2
Enchylaena tomentosa var. *tomentosa*	200	3.2	200	3.2	200	3.2
Total seeds per plot	1750	34.7	1450	23	1200	19

*No seed available

Seed was applied to one half of each circular plot. Prior to seeding, a pine plank of 25 cm height and 4 m length was placed approximately 1 cm into the soil down the centre of each plot, to prevent seed from being washed or blown onto the non-seeded side. Pre-treatments to break dormancy were applied to species as recommended by Greening Australia (Walters D [Greening Australia] 2001 pers. comm., 14 April) (Table 5.11). Pre-treatments were applied to increase the chance of treatments leading to germination.

Table 5.11. Seed treatments applied to seeds prior to sowing.

Species	Treatment
Callitris gracilis	Seed was stored near zero degrees for three days prior to sowing.
Acacia colletioides *Acacia oswaldii* *Dodonaea viscosa* ssp. *angustissima* *Senna artemisioides*	Seed was placed in near boiling water for five minutes, and then rinsed in cold water. Seed was placed onto plots less than 24 hours following treatment.

Seed was hand broadcast over one half of each plot, with the seeded side randomly allocated before seed was sown. To ensure even spread of seed, the seed bag was visually divided into quarters, which were spread over quarters of each plot. Large seed size of some species enabled a visual estimate of evenness of seed distribution.

Seed was sown during a light rain on 22nd April 2001 during which 5 mm rainfall was recorded over 24 hours. Seed on watered treatments received a further 40 mm of additional water over 72 hours such that the soil was kept near field capacity for three days. A further 8.5 mm natural rainfall was received on 29th May 2001 at the exclosure, and 37 mm had been received by the time of next additional watering on the 24th June. Plots were re-seeded in September 2001, and March 2002 using the same methods.

5.2.1.3 Seedling survival of Casuarina pauper and Myoporum platycarpum

Seedling establishment trial

In summer 2002, *M. platycarpum* and *C. pauper* seedlings were established in two seedling trays at a nursery in Mildura in similar climatic conditions to the field site. Seeds were sown on 14th January 2002 and seedlings trays were kept outdoors under shade cloth and kept moist with overhead sprinklers.

On 20th and 21st March 2003, at just over nine weeks (66 days) after sowing, (approximately six weeks post germination) seedlings were transplanted into the ground within the exclosure. Late March was chosen for seedling transplants as it was thought that *C. pauper* might naturally germinate during late summer rainfall events following seed fall to minimise the effects of post dispersal seed predation, and give seedlings a greater chance to develop before encountering summer conditions of heat and low soil moisture.

Ten seedlings of each species were planted into the non-seeded side of 20 plots of four treatment combinations (i) watered, (ii) unwatered, (iii) soil disturbed, and (iv) soil disturbed and watered treatments. Seedling height and number of leaves of seedlings were measured before planting (Table 5.12).

Table 5.12. Mean (± 1 SE) seedling height and number of leaves of *Casuarina pauper* and *Myoporum platycarpum* at the time of transplanting (nine weeks of age) (N=20).

	Casuarina pauper	***Myoporum platycarpum***	
	Height mm	Height mm	No. leaves
Mean (SE)	42 (4.47)	11.7 (1.51)	2.35 (0.25)

Seedlings were transplanted directly into the ground, into local soil at 10 cm intervals along a 1 m transect. Seedlings were watered in immediately and were marked with a small stick to enable re-location. Sufficient water was applied when planting seedlings to wet the soil to at least 10 cm, locally. Watering was limited to 20 cm each side of the seedlings to minimise impact to the rest of the plot. Seedlings were watered daily for one week with an equivalent of 10 mm rainfall and then alternate days for one week. This water was applied above the calculated water regime. Seedlings that did not survive the first five days were replaced on 26th March. Seedling survival was then monitored on alternate days for 10 days, and at 27 days, 34 days, 100 days, 189 days and 372 days. No further measurements were taken as no seedlings were surviving at the 372 day measurement.

Whilst seedlings were not measured daily, seedling survival was analysed using a conservative measure of the half way point between the last day the seedlings were observed alive, and the day they were found to be dead.

5.2.1.4 Data collection

Soil moisture

Soil samples to assess soil moisture content were collected prior to each water treatment, approximately four hours following completion of the watering treatment. Two replicate samples were taken within the A1 horizon, to approximately 5 cm, and in the B1 horizon to a depth of 10 cm. Four different plots were used at each water treatment to minimise disturbance to the plots and to investigate differences in soil moisture between watered plots, watered plots with soil disturbance, and non-watered plots with and without soil disturbance.

Soil samples of 10–50 g were weighed into a drying tin of known weight. Samples were oven dried at 105°C to a constant weight. Samples were cooled in a desiccator and then re-weighed (Rayment and Higginson 1992). Gravimetric soil water was then calculated.

Plot assessment

Plots were assessed seasonally from prior to commencement of treatments in September 2000, until two months after watering was ceased in October 2002 (Figure 5.5).

All species present on the plot were identified to species level (Walsh and Entwisle 1994, 1996, 1999) and estimates were made of the projected foliage cover for each species. In most cases, the foliage cover was very low, with cover values rarely exceeding 10 %, limiting the ability to perform rapid quantitative measures of foliage cover. Where cover was estimated at less than 10 %, cover estimates were made to the nearest percentage point. Where cover was greater than 10 %, estimates were made to the nearest 5 %.

Percentage cover of bare ground, leaf litter, wood (diameter greater than 2 cm), cryptogams and basal area of vegetation were measured using a point frame, with 10 vertical points (Greig-Smith 1983). The point frame was placed at five sites on each plot at evenly spaced radii from the centre of each plot. Measures were made of the surface feature, for example, if litter were recorded, it was not determined if soil or cryptogamic crust was underlying the litter.

The number of mature individuals and the number of juveniles of each species present on the plot was also counted. Mature plants were defined as those with evidence of reproductive capability, such as flowers, fruits, or seeds.

Biomass measures may have been useful to determine effects of watering, and biomass burnt during fires, however, destructive sampling to determine dried biomass was not possible on experimental plots, or within other areas of the grazing exclosure due to a requirement to minimise disturbance within the grazing exclosure. Sampling of similar areas outside of the exclosure was also not possible, due to grazing effects.

Germinants

All germinants of species for which seed was applied were marked with flags to enable re-location and determination of survival period and cause of death.

Germinants of other perennial species were not individually marked, but numbers of juveniles were counted. For disturbed plots (fire and scarification) all juveniles represent new recruits as all standing vegetation was removed from the plots. For undisturbed plots, the number of new recruits of species for which seed was not applied is unknown.

Soil profile

To describe the soil profile, a soil pit within 30 m of the exclosure site, was dug to a depth of 72 cm where consolidated material was encountered. A hard layer of calcrete limited manual excavation beyond this depth. It was not possible to remove any of this calcrete layer for further testing with the available tools, and so results for horizons above this layer have been presented. Five 75 mm soil cores were sampled within the A1/B21 horizon, B22(k) horizon and B23(k) horizon to determine bulk density and soil moisture content.

Soils were initially examined in the field, and further testing was undertaken in the laboratory at the University of Ballarat. Soil texture was determined by hand manipulation of a soil bolus (McDonald and Isbell 1990). The abundance and distribution of roots was observed. Soil colour was measured using Munsell soil colour charts with moist soils within the pit. Field pH was tested using the Raupach (universal indicator) method. Effervescence of carbonate was tested using two to three drops of 1-molar HCl was used to determine the presence and characteristics of calcium carbonate ($CaCO_3$) (McDonald and Isbell 1990). Soils were classified according to *The Australian Soil Classification* (Isbell 1996), and *A Factual Key For The Recognition Of Australian Soils* (Northcote 1979).

The pH of a 1:5 soil/water suspension was measured using a Hach Mini 17200 pH meter (method 4A1 Rayment and Higginson 1992). The pH of the soil/water suspension was also determined using a soil/0.01 M calcium chloride ($CaCl_2$) extract (method 4B1 Rayment and Higginson 1992).

Electrical conductivity of the A1, B21 and B22 horizons were determined based on a 1:5 soil/water extract measured at 25°C, using a hand-held WTW LF 340 conductivity meter (method 3A1 Rayment and Higginson 1992).

Gravimetric soil water content and soil bulk density were calculated from five 75 mm cores taken from each horizon. Cores were weighed, and a known weight sub-sample was oven dried at 105°C for 48 hours to determine bulk density and gravimetric water content (Rayment and Higginson 1992).

Field capacity is defined as the amount of water held in the soil after excess gravitational water has drained, and the rate of downward movement of water has largely ceased (Cassel and Nielsen 1986). Field capacity was determined by wetting an area of approximately 50 cm radius with 40 litres of water. A dike was formed using some aluminium sheeting pressed into the ground to stop runoff and minimise lateral movement of water at the soil surface. The area was then covered with plastic to minimise evaporation, and left for 48 hours (Cassel and Nielsen 1986). Five 75 mm cores were sampled, weighed and a sample was oven dried at 105°C for 48 hours to determine gravimetric water content (Rayment and Higginson 1992).

5.2.1.5 Data analysis

Perennial regeneration

Low numbers of germinants limited the analysis of perennial regeneration data. Germinants of individual species except for *E. tomentosa* were too low for reliable analysis of the response of individual species, and so germinants of all perennial shrub species were combined to investigate the overall response of perennial species to the treatments. The limitation here is that different species may have different regeneration niches. However, to obtain a picture of the overall response of perennial shrub species to disturbance, water addition and seed application, four perennial regeneration response variables were calculated. These were:

(i) the total number of germinants of species for which seed was applied, that emerged over the entire experimental period;

(ii) presence/absence of germinants of species for which seed was applied;

(iii) the total number of juveniles of perennial shrubs and subshrubs recorded in April 2002 (April 2002 was chosen as this represented a small peak in perennial juveniles, and was 12 months after all treatments were commenced, allowing for seasonal regeneration); and

(iv) presence/absence of germinants of juvenile shrubs and subshrubs.

Differentiating between germinants and juveniles gave additional data as germinants of species for which seed was applied were marked seasonally, enabling the total number of germinants that emerged during the study to be analysed. For perennial shrub species for which seed was not applied, juveniles were not marked, and so measures of juveniles do not take into account when the juveniles emerged.

Two types of analyses were performed on the perennial regeneration response variables; logistic regressions and three-way ANOVAs. Results for both the logistic regression and ANOVA have been presented as they test different response variables (presence and absence of juveniles, versus number of juveniles present). The logistic regression is better suited to the distribution of the data, and therefore more reliable, but less sensitive to differences and unable to assess treatment interactions (Tabachnick and Fidell 2001).

A logistic regression approach was chosen due to the low numbers of germinants and juveniles observed, which resulted in many zero values, and non-normally distributed data. Logistic regression enabled investigation of factors associated with the presence or absence of regeneration at a plot. Binary data such as these are rarely normally distributed, and are better suited to a logistic function (Tabachnick and Fidell 2001). The logistic regression makes no assumptions about the distribution of predictor variables, and enables prediction of group membership based on a number of variables that may help define the group (Tabachnick and Fidell 2001). The logistic regression tests if there is a relationship between the outcome and the set of predictors, and then simplifies the model by excluding non-significant variables. The equation can then be used to predict group membership (Tabachnick and Fidell 2001).

Logistic models were estimated using two methods, (i) a block entry of all predictor variables, and (ii) a forward stepwise entry approach, with the same factors identified as significant in both models. The block entry results have been presented. Goodness-of-fit of the logistic regression model was evaluated using the Hosmer-Lemeshow statistic, which compares the observed number of subjects in a group to the number predicted by the model. A non-significant Chi-square indicates a good fit for the model (Tabachnick and Fidell 2001).

A three-way ANOVA was performed to investigate the differences in numbers of perennial juveniles between treatments (disturbance, water, and seed). For the measurement of all juvenile species, a log transformation ($\log_{10}$ (perennial juveniles+1)) stabilised the variance as tested by Levenes tests, although did not result in a normal distribution. However, the ANOVA is relatively robust to departures from normality, particularly with balanced samples (Underwood 1997). No transformation was able to stabilise the variance or normalise the distribution of the data for germinants of species for which seed was applied, therefore only the logistic regression was performed for these data.

All logistic models and ANOVAs were analysed using SPSS Version 10.

Seedling survival

To determine if significant differences occurred in seedling survival between seedling species, water addition and soil disturbance, a three-way ANOVA was performed. Conformation to the assumptions for ANOVA was tested using Levenes test and Q-Q plots. Seedling survival (days) data were log-transformed to stabilise the variance. All mean and standard error values presented represent untransformed values.

Climate during the study period

To determine the impact of seasonal climatic effects on regeneration, temperature, rainfall and evaporation data from previous "event" years (1973/4/5 and 1992/3) were compared to the study period.

5.2.2 Results

5.2.2.1 Germinants of species for which seed was applied

Of those species for which seed was applied, five species germinated on experimental plots; *E. tomentosa*, *Olearia pimeleoides*, *Acacia oswaldii*, *Dodonaea viscosa* ssp. *angustissima* and *Acacia* sp.. Only one *M. platycarpum* individual was found, on a grazed and watered plot in January 2001 (prior to seed application). No other tree species were found to germinate under any treatment combination. Few germinants were found for most species for which seed was applied, with the exception of *E. tomentosa* (Table 5.13). Of all germinants for which seed was applied, 88 % were of *E. tomentosa.* All germinants except *E. tomentosa* were only observed on seeded plots.

Table 5.13. Number of germinants of perennial species for which seed was applied, and the fate of germinants.

Species	Total no. germinants	dead		% mortality
		disappeared	water stress	
Acacia oswaldii	1	0	0	0.0
Acacia sp.	3	1	0	33.3
Dodonaea viscosa	5	0	2	40.0
Enchylaena tomentosa	199	43	26	34.7
Olearia pimeleoides	17	0	2	11.8
Myoporum platycarpum	1	0	1	100.0

Of the species for which seed was applied, more seedlings germinated on watered and scarified plots than any other treatment combination, and more germinants were observed where seed was applied (Figure 5.8). This pattern was consistent for most species of germinants (Figure 5.9).

Seedling mortalities ranged from 0 to 100 %, with little difference in the proportion of mortalities between different treatment combinations (Table 5.13).

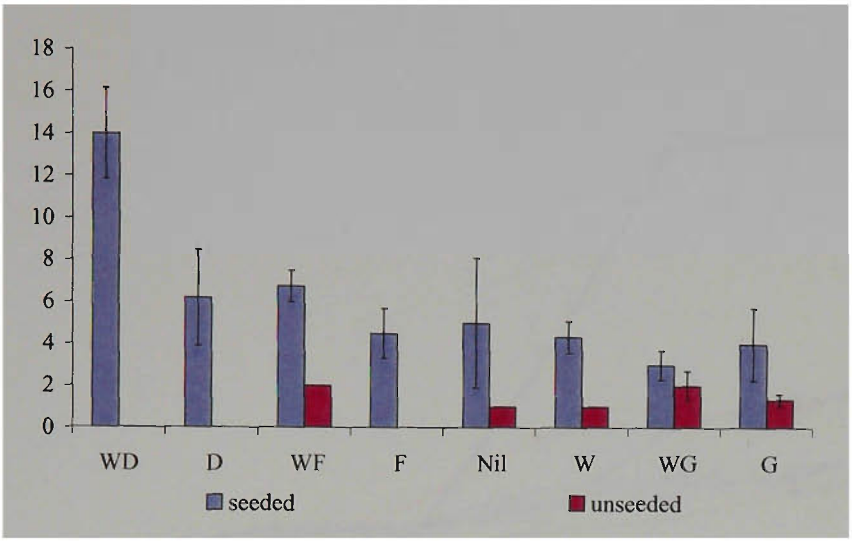


Figure 5.8. Mean (± 1 SE) germinants of species for which seed was applied across all treatment combinations (W watered, D soil disturbance, F fire, G grazed).

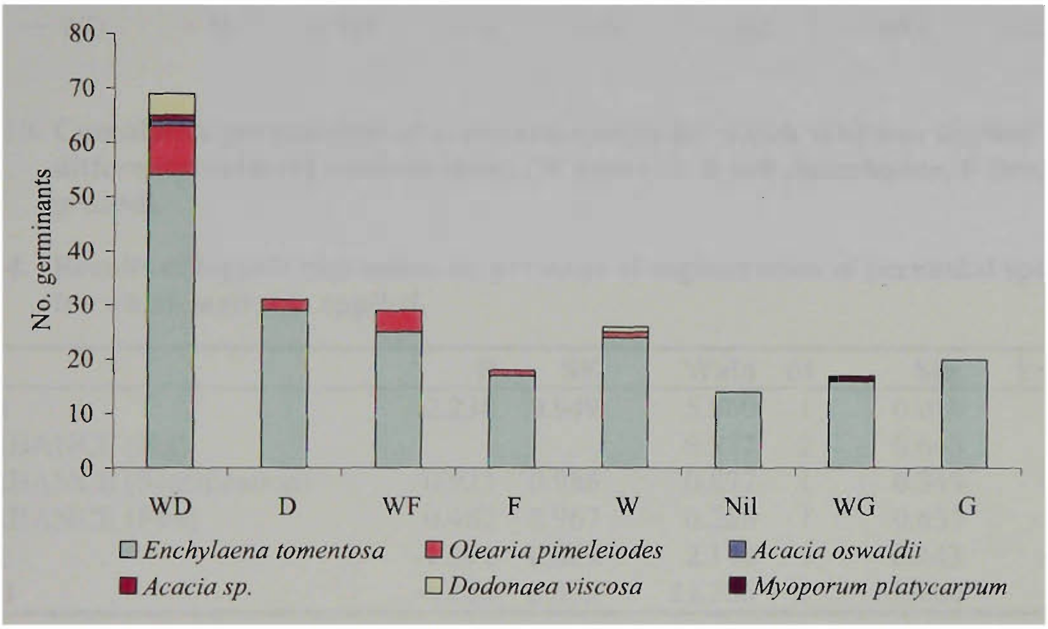


Figure 5.9. Total number of germinants of six species for which seed was applied across different treatment combinations, (W watered, D soil disturbance, F fire, G grazed).

The majority of germinants of seeded species emerged between September 2001 and April 2002 with smaller numbers of germinants observed between April and September in both 2001 and 2002 (Figure 5.10). Increases in germinants for the watered and scarified plots, and watered plots between January and April 2002 coincided with the period of increased water addition (Figure 5.10).

The logistic regression on the presence of germinants shows a significant effect for seed application (Table 5.14). The odds of germinants occurring were significantly improved by seed application. No other treatment resulted in a significant change in the odds of regeneration (Table 5.14).

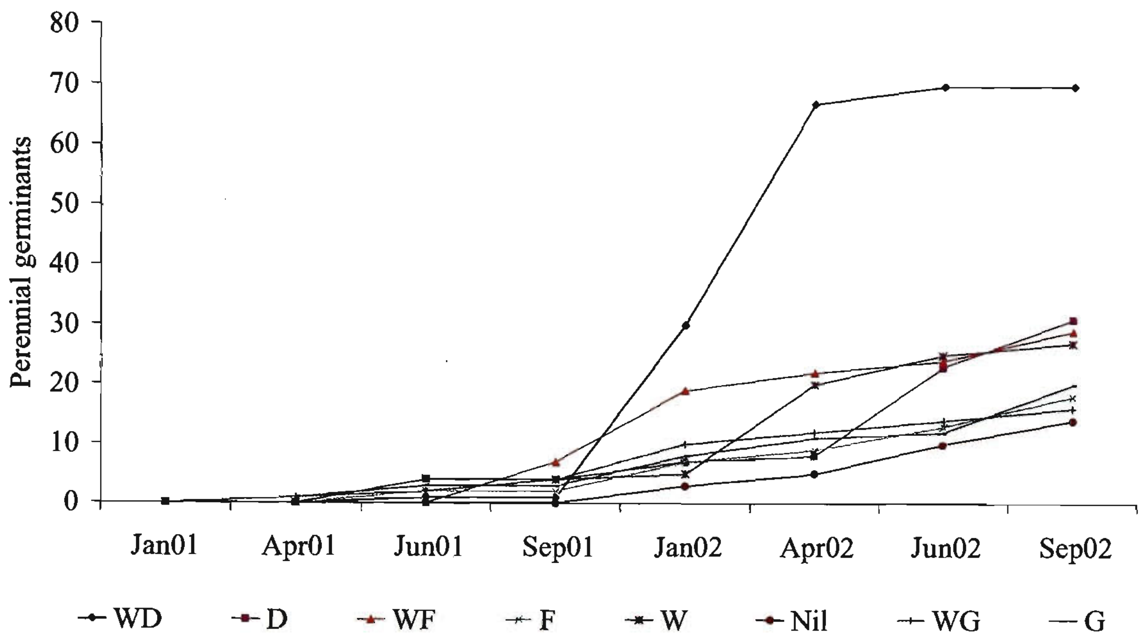


Figure 5.10. Cumulative germination of perennial species for which seed was applied under different treatment combinations, (W watered, D soil disturbance, F fire, G grazed).

Table 5.14. Results of logistic regression on presence of regeneration of perennial species for which seed was applied.

	B	SE	Wald	df	Sig.	Exp (B)
Constant	2.238	0.949	5.560	1	0.018	9.373
DISTURBANCE (Nil)			0.877	2	0.645	
DISTURBANCE (Scarification)	0.923	0.986	0.877	1	0.349	2.517
DISTURBANCE (Fire)	0.462	0.967	0.228	1	0.633	1.587
WATER	-1.271	0.869	2.140	1	0.143	0.281
SEEDED	-4.128	0.895	21.286	1	0.000	0.016

The classification matrix shows that 86.67 % of plots are correctly classified by the logistic model. The Hosmer and Lemeshow test for goodness-of-fit suggests a good fit for the logistic regression model (Chi-square = 7.47, df 8, p=0.49).

Estimated marginal means of germinants show an increase in germinants for seed application under scarified conditions both with and without water addition (Figure 5.11). However, this effect of scarification was not significant in the logistic model.

5.2.2.2 All perennial germinants

Perennial shrubs for which seed was not applied also germinated during the experiment. These were *Atriplex stipitata*, *A. vesicaria*, *Chenopodium desertorum*, *Sclerolaena diacantha*, *S. obliquicuspis, S. parviflora, Maireana pentatropis, M. radiata* and *Zygophyllum apiculatum.*

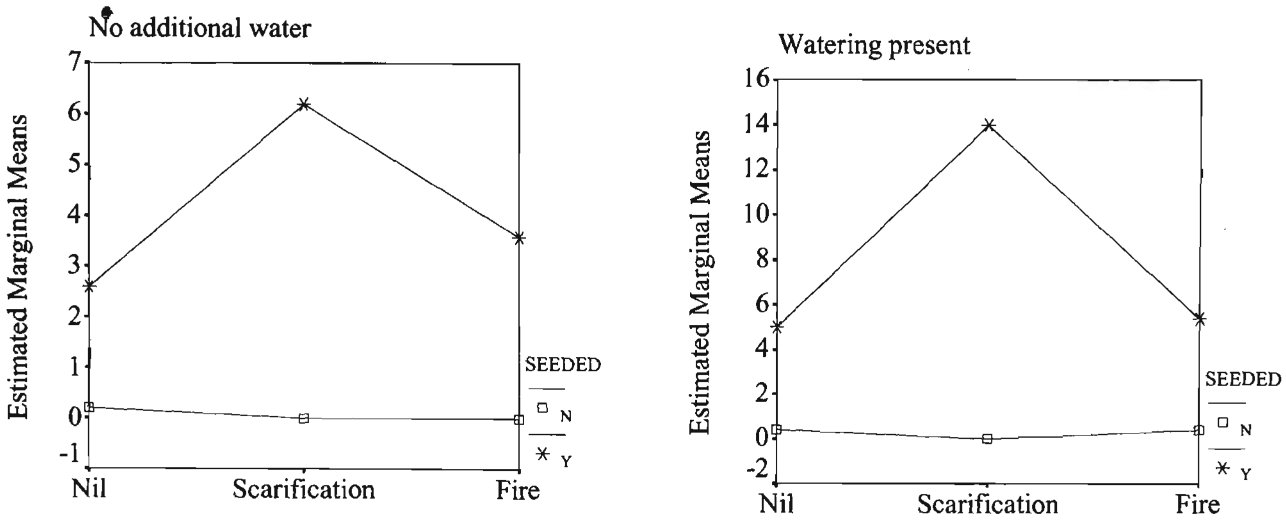


Figure 5.11. Mean number of germinants of perennial species for which seed was applied on watered and unwatered plots under different disturbances.

Juveniles of perennial shrubs were observed throughout the experiment, across all treatment combinations. Peaks in the number of juveniles varied between treatments, with an increase in the number of juveniles for most treatments in June 2001, dominated by undisturbed plots within the exclosure (Figure 5.12).

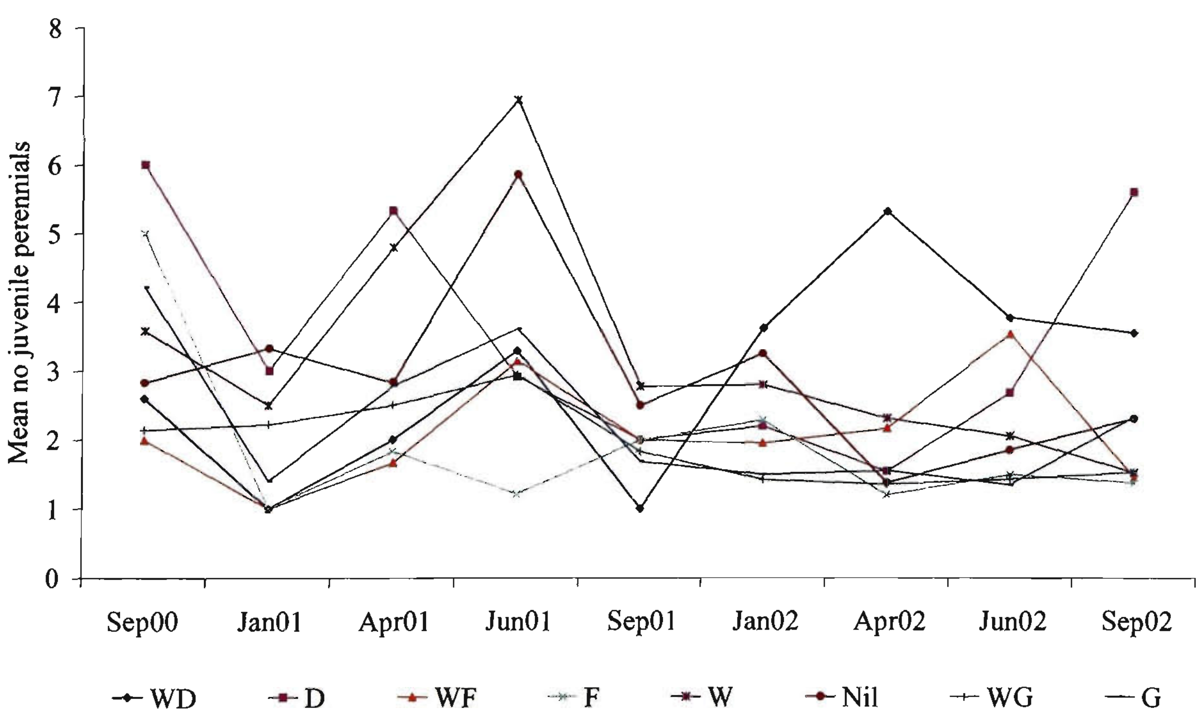


Figure 5.12. Mean number of perennial juveniles observed on experimental plots (W watered, D soil disturbance, F fire, G grazed).

Greater numbers of juveniles were found on watered and scarified treatments in April 2002, than other treatment combinations (Figure 5.13). More juveniles of most species were observed on watered plots, than unwatered plots. *E. tomentosa* and *S. diacantha* were the most frequently occurring juvenile species (Figure 5.13).

The logistic regression on the presence / absence of juveniles on plots shows a significant effect for seed application (Table 5.15). The odds of juveniles occurring on a plot were significantly improved by seed application, however no other treatment resulted in a significant change in the odds of regeneration (Table 5.15).

Table 5.15. Results of logistic regression on presence of regeneration of perennial species for which seed was applied.

	B	SE	Wald	df	Sig.	Exp (B)
Constant	0.471	0.662	0.508	1	0.476	1.602
DISTURBANCE (Nil)			3.359	2	0.187	
DISTURBANCE (Scarification)	0.690	0.845	0.666	1	0.414	1.994
DISTURBANCE (Fire)	-0.783	0.735	1.135	1	0.287	0.457
WATER	0.000	0.634	0.000	1	1.000	1.000
SEEDED	1.562	0.680	5.284	1	0.022	4.768

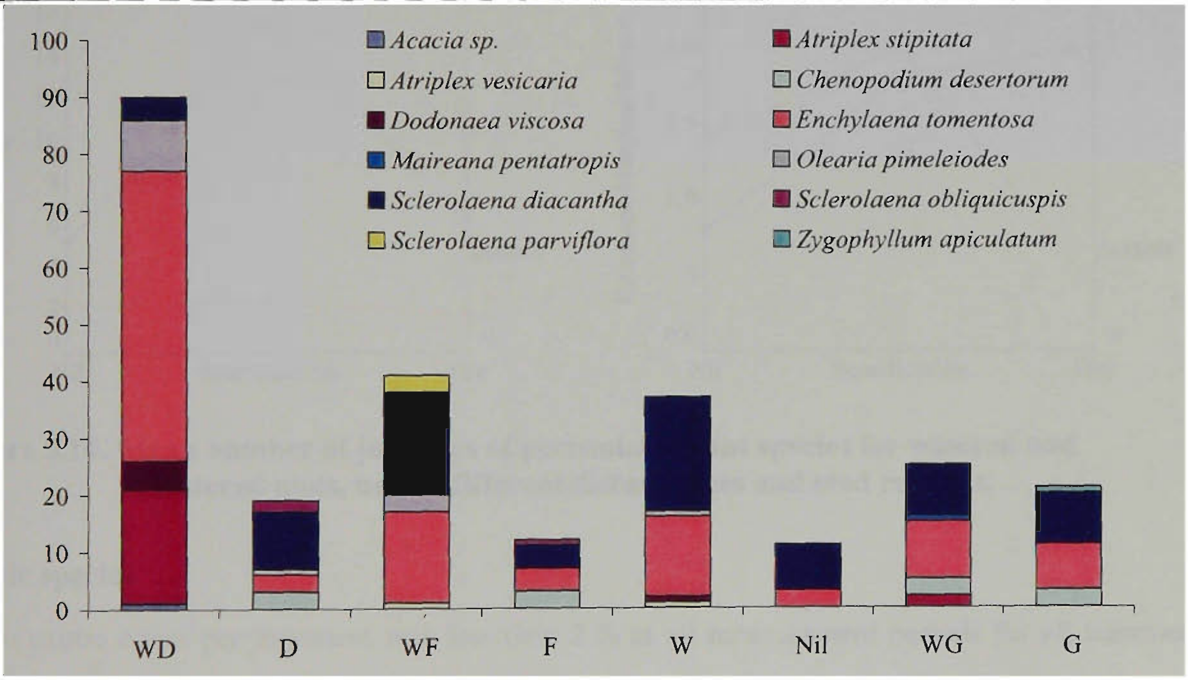


Figure 5.13. Total number of juvenile shrubs and subshrubs across different treatments in April 2002.

The classification matrix shows that 76.25 % of plots are correctly classified by the logistic model (Table 5.16). The Hosmer and Lemeshow test for goodness-of-fit suggests a good fit for the logistic regression model (Chi-square=1.91, df 7, p=0.965).

To determine the effects of treatments on the number of juveniles for which seed was applied, a three-way ANOVA was performed. Significant main effects were found for watering and seed application on the regeneration of perennial species (Table 5.17). A significant interaction was found between water and seed application (Table 5.17). Figure 5.14 shows an increase in the number of juveniles with seed application in the presence of water.

Table 5.16. Error matrix showing percentage of observed and predicted juveniles by the logistic model.

		Predicted juveniles		
		0	1	% Correct
Observed	0	6	10	37.50
juveniles	1	4	40	90.91
			Overall % correct	76.67

In the absence of additional water and seed, very limited regeneration occurred under all treatments. Regeneration under the fire treatment, however, was marginally improved by the application of seed. Scarification improved regeneration over no disturbance, with little improvement resulting from application of seed. In the presence of watering, the effect of seeding has a much greater positive effect on regeneration under the scarified treatment (Figure 5.14).

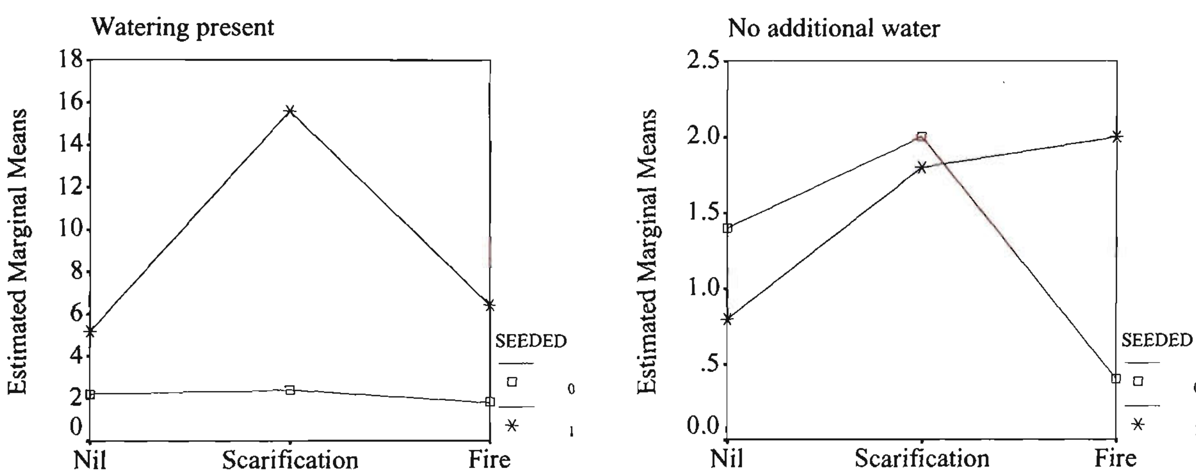


Figure 5.14. Mean number of juveniles of perennial shrubs species for watered and unwatered plots, under different disturbances and seed regimes.

Exotic species

Mean exotic cover per treatment was less than 2 % at all measurement periods for all treatment combinations. The number of individuals fluctuated seasonally, and was generally higher on watered and scarified treatments within the exclosure.

Species richness

The number of species present on each plot fluctuated seasonally, with a peak of annual species observed in spring 2000 and 2001. The effects of intensive watering are observed in April 2002, with more species present on watered treatments. The number of spring annual species was reduced for all treatments in September 2002, but less so for the scarified plots (Figure 5.15).

Table 5.17. Results of three-way ANOVA for all perennial regeneration on experimental plots.

Source	*Type III Sum of Squares*	*df*	*Mean Square*	*F*	*Sig.*	*Eta Squared*	*Noncent. Parameter*	*Observed Power*
Corrected Model	4.888	11	0.444	5.092	0.000	0.539	56.009	0.999
Intercept	12.941	1	12.941	148.295	0.000	0.755	148.295	1.000
WATER	1.187	1	1.187	13.604	0.001	0.221	13.604	0.951
SEEDED	1.583	1	1.583	18.141	0.000	0.274	18.141	0.987
DISTURBANCE	0.445	2	0.222	2.549	0.089	0.096	5.098	0.486
WATER * SEEDED	0.960	1	0.960	10.996	0.002	0.186	10.996	0.901
WATER * DISTURBANCE	0.021	2	0.011	0.122	0.885	0.005	0.244	0.068
SEEDED * DISTURBANCE	0.305	2	0.152	1.747	0.185	0.068	3.494	0.348
WATER * SEEDED * DISTURBANCE	0.387	2	0.193	2.216	0.120	0.085	4.432	0.430

Table 5.18. Results of three-way ANOVA for survival of transplanted seedlings on experimental plots.

Source	*Type III Sum of Squares*	*df*	*Mean Square*	*F*	*Sig.*	*Eta Squared*	*Noncent. Parameter*	*Observed Power*
Corrected Model	2.965	7	0.424	5.249	0.000	0.534	36.740	0.991
Intercept	57.698	1	57.698	714.973	0.000	0.957	714.973	1.000
WATER	0.559	1	0.559	6.933	0.013	0.178	6.933	0.724
SCARIFICATION	0.027	1	0.027	0.336	0.566	0.010	0.336	0.087
SEEDLING SP.	2.318	1	2.318	28.718	0.000	0.473	28.718	0.999
WATER * SCARIFICATION	0.001	1	0.001	0.011	0.918	0.000	0.011	0.051
WATER * SEEDLING	0.031	1	0.031	0.390	0.537	0.012	0.390	0.093
SCARIFICATION * SEEDLING SP.	0.019	1	0.019	0.235	0.631	0.007	0.235	0.076
WATER * SCARIFI * SEEDLING SP.	0.009	1	0.009	0.117	0.735	0.004	0.117	0.063
Error	2.582	32	0.081					
Total	63.245	40						
Corrected Total	5.547	39						

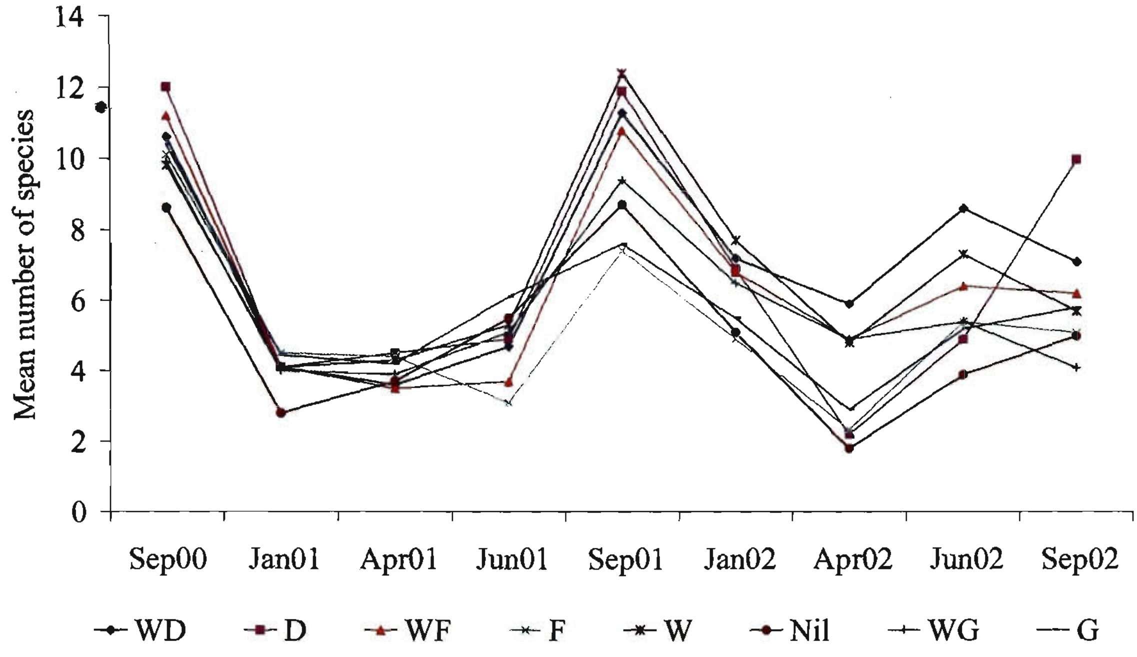


Figure 5.15. Mean species richness for each treatment (pooled for seed application) (W watered, D soil disturbance, F fire, G grazed).

5.2.2.3 Seedling survival

Initial mortality of transplanted seedlings was high despite daily watering for one week, and alternate day watering for a further week. Only three *M. platycarpum* seedlings survived beyond 27 days. Survivorship was slightly higher in *C. pauper* seedlings (Figure 5.16). Watering treatments ceased on 6th July 2002, despite this, 66.7 % of seedlings surviving in July survived over winter to the 30th September 2002. No further measurements were taken until April 2003, when all seedlings were found to be dead.

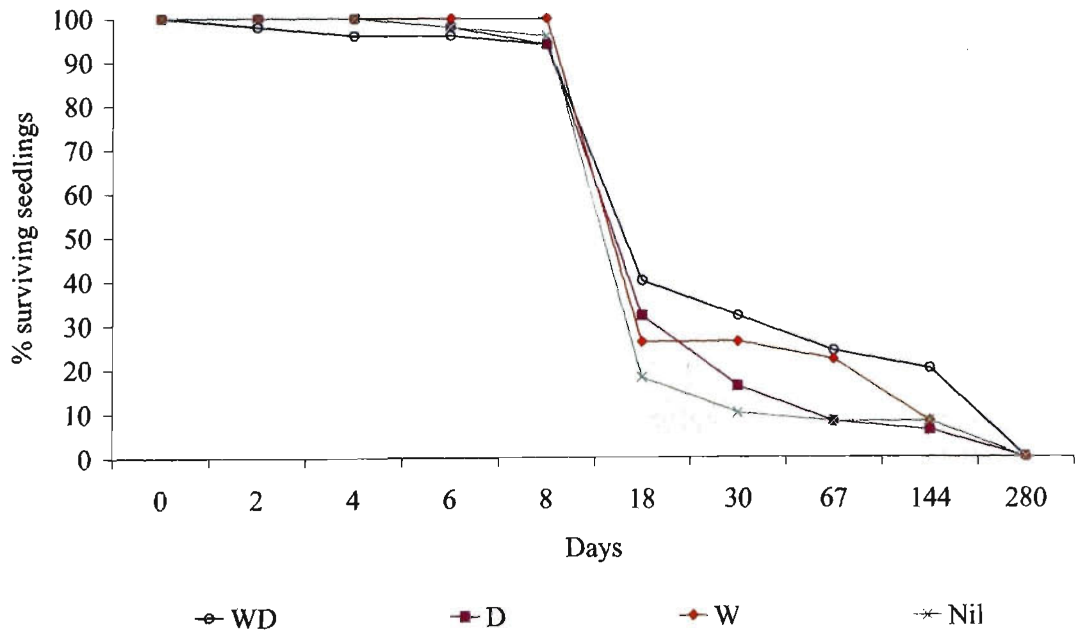
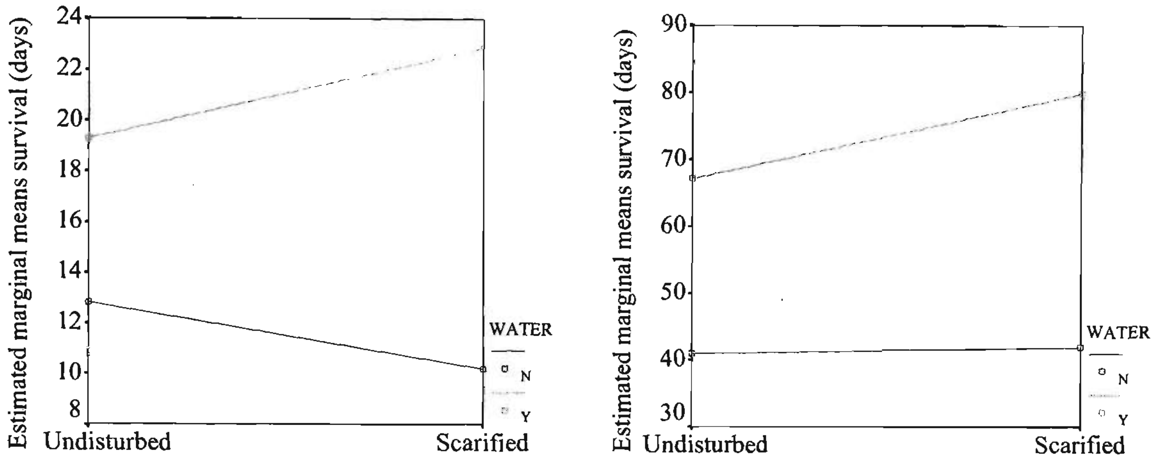


Figure 5.16. Percentage survival of *Casuarina pauper* seedlings under four treatment combinations (WD watered and soil disturbance, W watered, D soil disturbance and Nil no treatment).

Results of the three-way ANOVA showed significant main effects for seedling species and water (Table 5.18). No significant effect was found for soil disturbance, or for any treatment interactions (Table 5.18). Watering led to a significant increase in seedling survival for both *C. pauper* and *M. platycarpum*, with *C. pauper* seedlings surviving significantly longer than *M. platycarpum* seedlings (Figure 5.17). Whilst no significant interaction was obtained, mean seedling survival was greater in scarified soil in the presence of water (Figure 5.17).



Myoporum platycarpum *Casuarina pauper*

Figure 5.17. Mean survival (days) of *Casuarina pauper* seedlings in undisturbed or scarified soil with and without additional water.

Soil description

Soils supporting *C. pauper* woodlands at the grazing exclosure were found to be gradational and calcareous throughout, and were classified as Calcareous earths (Gc 1.12) (Northcote 1979). These soils are characterised by a gradational texture profile, with a very gradual increase in clay with depth, and calcareous throughout (Northcote *et al.* 1975; Badawy 1982).

According to the more recent Australian soil classification method (Isbell 1996), soils at the exclosure were classified as 4 CA CQ DZ YY B E M M V, listed by Order, Suborder, Great Group, Subgroup, and Family. This classification denotes a Calcarosol, with a Hypercalcic Suborder, and Petrocalcic Great Group. Subgroup was indeterminable with the available information.

The soil depth above the Bk horizon was medium (0.1 – <0.3 m), with no surface gravel (Figure 5.18). The surface soil texture was sandy clay loam (20–35 % clay based on field texture), with a slight increase in clay with increasing depth (Table 5.19). Total soil depth was moderate (0.5 – <1 m). A confidence level of four was used (Isbell 1996), indicating a provisional classification only, as collection of all analytical data was beyond the scope of this study.

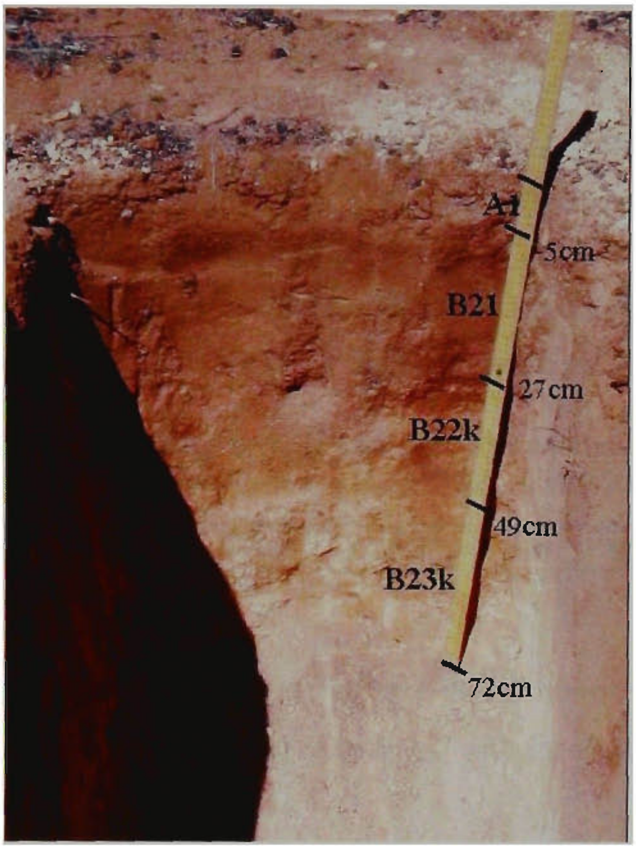


Figure 5.18. Soil pit adjacent to exclosure in MSNP with annotated 1 m ruler showing soil horizons and depth of horizon.

Table 5.19. Soil profile description of horizons measured from a soil pit adjacent to the grazing exclosure.

Depth (cm)	Horizon	Description
0–0.5	O1	a minimal, discontinuous layer of undecomposed organic debris consisting largely of *Casuarina pauper* litter.
0–5	A1	reddish brown (5YR 3/4 moist) sandy clay loam. Apedal. Few medium to coarse roots (3–5 mm diameter). pH (Raupach) 8 ½. Effervescence audible and visible.
	Gradual boundary to	
5–27	B21	dark red (2.5YR 3/6 moist) sandy clay loam. Apedal. Few medium to coarse roots (3–5 mm diameter). pH (Raupach) 8 ½. Effervescence audible and visible.
	Gradual boundary to	
27–49	B22 (k)	yellowish red (5YR 5/6 moist) sandy clay loam. Apedal. Few fine to very fine roots (diameter <2 mm). pH (Raupach) 8 ½–9. Effervescence audible and visible, bubbles 2–8 mm.
	Gradual boundary to	
49–72	B23 (k)	yellowish red mottled (5YR 5/6 moist) sandy clay loam. Apedal. Soft and hard carbonate aggregations. Few fine to very fine roots (diameter <2 mm). pH (Raupach) 9½. Effervescence audible and visible, bubbles 6–8 mm.
	Clear boundary to	
72+	Calcrete	hard calcrete pan.

Features of these soils for agriculture are high soil permeability, with low resistance to root penetration, except for horizons of maximum calcium carbonate. Soil moisture availability is dependent on the clay content of horizons. Soil fertility is described as low to moderate (Badawy 1982).

Results from the laboratory tests (Table 5.20) confirm the description of the Gc1.12 soils, with low salt content and alkaline surface soils, increasing to strongly alkaline and moderate to high soluble salts in subsoils (Northcote *et al.* 1975). Results from the calcium chloride solution were 0.5 to 1 pH unit lower than those obtained using the soil/water suspension method, as is commonly found (Rayment and Higginson 1992).

Table 5.20. Laboratory results for soil pH and electrical conductivity of samples taken from soil pit adjacent to grazing exclosure within MSNP.

Soil properties	A1/B21 horizon	B22 (k) horizon	B23 (k) horizon
pH-H_2O	8.02	9.04	9.20
pH-$CaCl_2$	7.52	8.00	8.15
EC (mS cm^{-1})	0.15	0.47	1.05

Increasing gravimetric soil water content was found with increasing depth, with little difference in soil bulk density (Table 5.21).

Table 5.21. Mean (± 1 SE) gravimetric soil water content and bulk density determined from five 75 mm cores from each soil horizon.

Soil properties	A1/ B21 horizon	B22(k) horizon	B23(k) horizon
Gravimetric water %	4.64 (0.18)	8.49 (0.51)	11.03 (0.22)
Bulk density (gcm^{-3})	1.19 (0.07)	1.24 (0.04)	1.21 (0.02)

Field capacity as measured at the exclosure site was 5.8 % (SE 0.29). It is possible the field method used may not be ideal under the severe drought conditions, which could have resulted in a low field capacity value.

Soil gravimetric water content

On unwatered plots, soil gravimetric water content for the A1 horizon fluctuated between 0.83 to 4.92 %, with little difference between scarified and undisturbed plots. Soil moisture was marginally higher at the B21 horizon for both scarified and undisturbed areas (Appendix 9).

For plots where additional water was applied, soil gravimetric water content reached 6.46 to 14.78 % approximately four hours following watering, with water content frequently higher on disturbed plots. Soil gravimetric water content did not remain elevated between watering events. Prior to watering, there was no consistent difference between soil water content on watered and unwatered plots (Appendix 9).

5.2.2.4 *Climate during the study period*

Serious rainfall deficiencies (rainfalls in the lowest 10 % of historical totals, but not in the lowest 5 %) to severe rainfall deficiencies (rainfalls in the lowest 5 % of historical totals) were experienced across much of Australia, during an El Nino event from mid 2001 to early 2003 (BOM 2003). Northwest Victoria also experienced serious to severe rainfall deficiencies from September 2001 to March 2003 (Clewett *et al.* 2003). Although some relief was experienced to the east and south of the study area in December 2002 with a large rainfall event, low rainfall continued through early 2003. Severe rainfall deficiencies were greater to the northwest of the study area (Clewett *et al.* 2003). Annual rainfall figures for four stations across the study area are presented in Figure 5.19. Departures from mean monthly rainfall in Mildura show the rainfall deficiencies experienced from late 2000, through to 2002 (Figure 5.20).

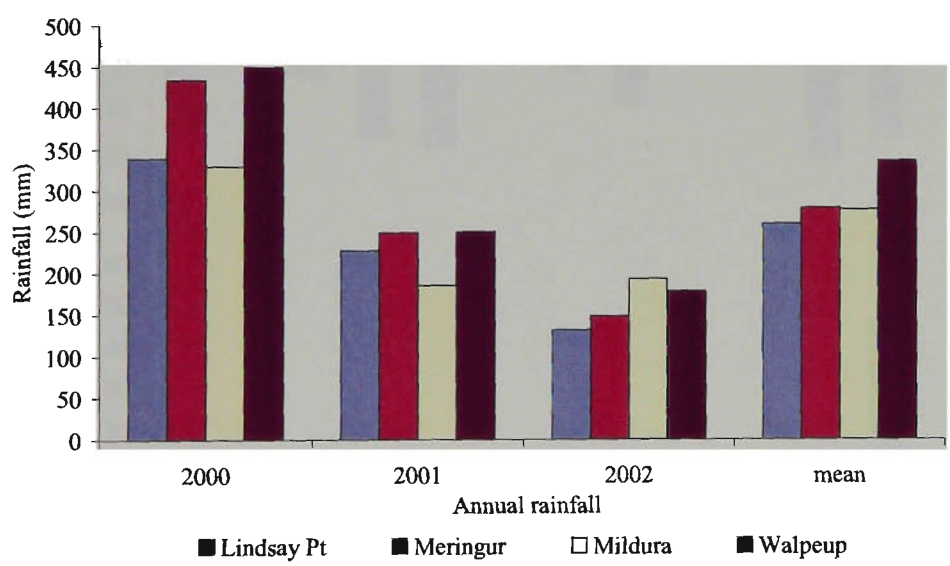


Figure 5.19. Annual rainfall recorded from stations across the study area for 2000, 2001 and 2002, compared with long-term means (Clewett *et al.* 2003).

Temperatures during the study period were above average in January and February 2001, and maximum temperatures were above average in April and May 2002 (BOM 2003). During the "event" years of the early 1970s, temperatures fluctuated around the long-term average, but were more frequently below average, particularly daily maximum temperatures in spring (BOM 2003).

Mean daily evaporation during the study period was above average for most of 2002, but otherwise closely followed the long-term average (Figure 5.21). During the early 1970s, mean daily evaporation was more frequently below average (BOM 2003). During 1974, mean daily evaporation was more than 2 mm lower per day than the long-term average over many months of the year (Figure 5.22).

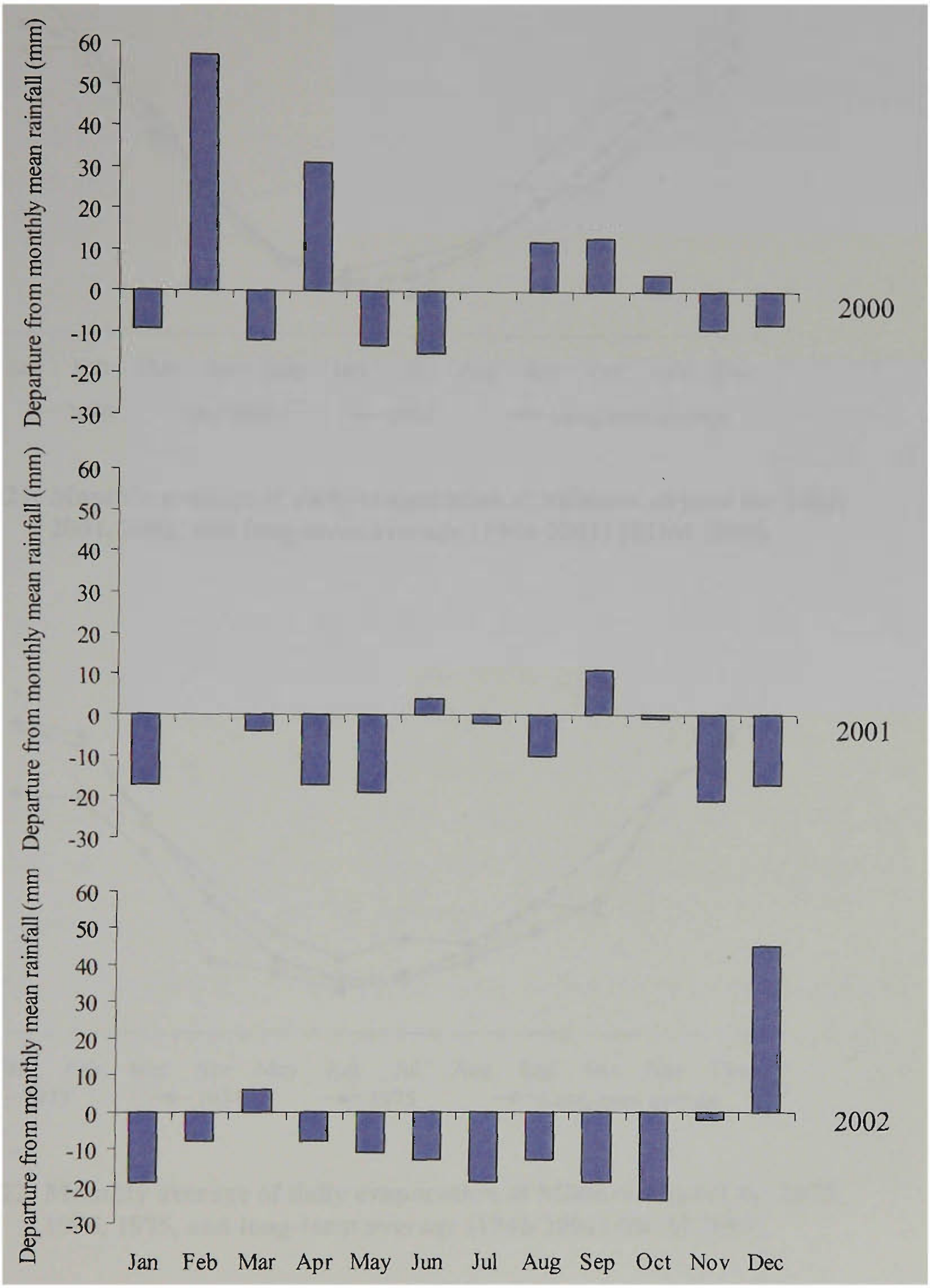


Figure 5.20. Departures in monthly rainfall during 2000, 2001 and 2002 from long-term averages at Mildura (Clewett *et al.* 2003).

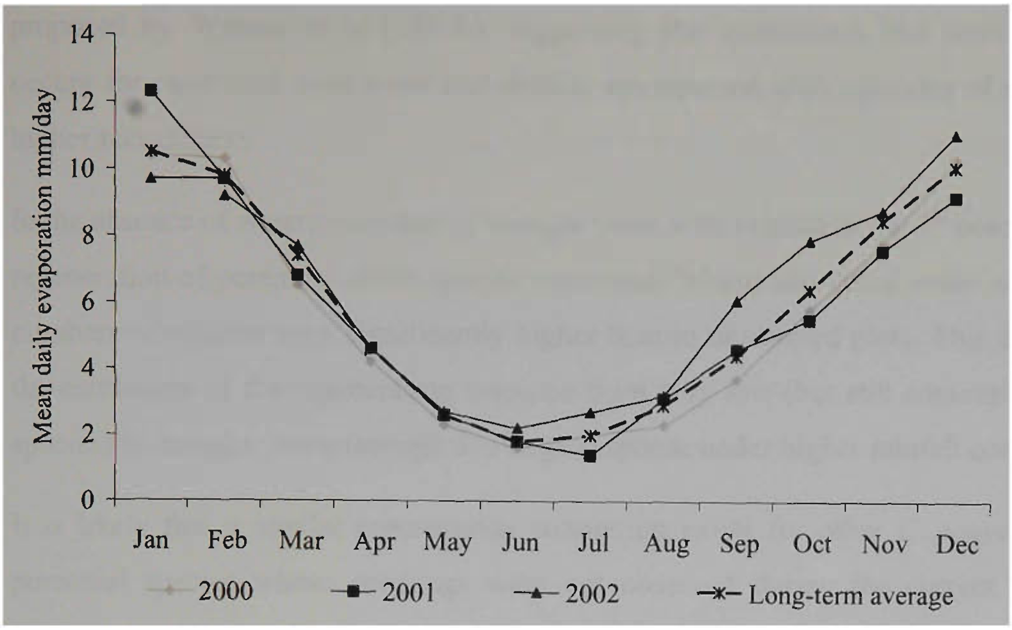


Figure 5.21. Monthly average of daily evaporation at Mildura airport for 2000, 2001, 2002, and long-term average (1946-2001) (BOM 2003).

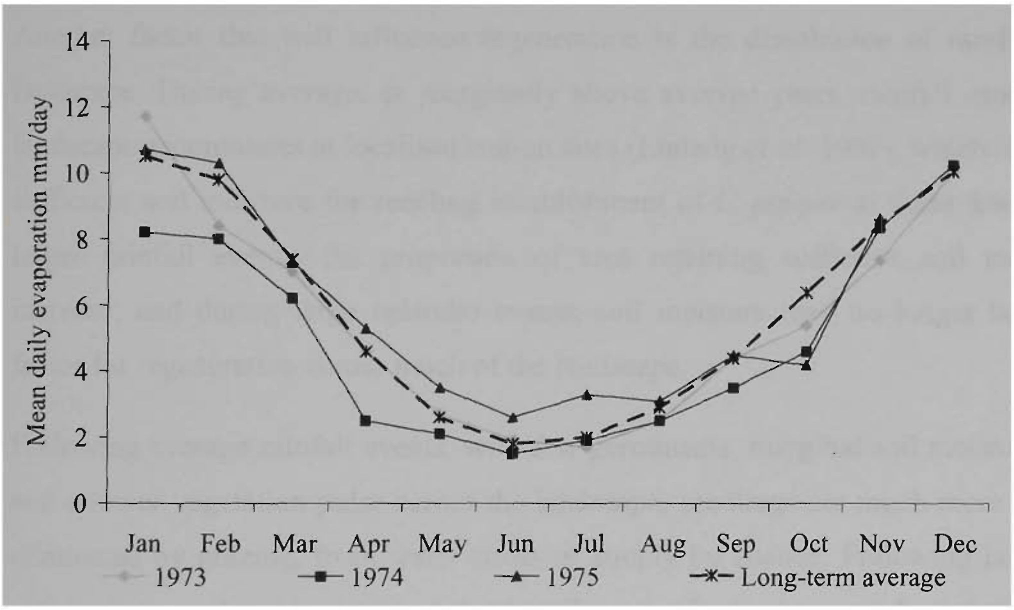


Figure 5.22. Monthly average of daily evaporation at Mildura airport for 1973, 1974, 1975, and long-term average (1946-2001) (BOM 2003).

5.2.3 Discussion

Whilst regeneration for many perennial *C. pauper* woodland species was not triggered by any treatment combination in the experiment, low levels of regeneration were observed for a number of perennial shrub species.

Water addition did not significantly improve the likelihood of perennial regeneration occurring at a site (Table 5.14 & 5.15), however, water addition had a significant effect on the total number of juvenile perennials at a site (Table 5.17). This supports the model proposed by Watson *et al* (1997b), suggesting that continuous low level recruitment occurs for most arid zone trees and shrubs, interspersed with episodes of substantially higher recruitment.

In the absence of water, even during drought years with rainfall in the 5th percentile, some regeneration of perennial shrub species continued. Where additional water was received, numbers of recruits were significantly higher than in unwatered plots. This demonstrates the continuum of the regeneration response from very low (but still continuing for some species) in drought years, through to a large response under higher rainfall conditions.

It is likely that a similar regeneration continuum exists for other *C. pauper* woodland perennial species whose seedlings were not observed during the current experiment. However, it appears that higher or more prolonged soil moisture levels are required than what was provided in the current study for seedling establishment of *C. pauper*, and other *C. pauper* woodland overstorey species.

Another factor that will influence regeneration is the distribution of rainfall over the landscape. During average, or marginally above average years, rainfall runoff over the landscape accumulates at localised run-on sites (Ludwig *et al.* 1997), which may result in sufficient soil moisture for seedling establishment of *C. pauper* at these few sites. With larger rainfall events, the proportion of area retaining sufficient soil moisture will increase, and during large episodic events, soil moisture may no longer be a limiting factor for regeneration across much of the landscape.

Following average rainfall events, with few germinants, marginal soil moisture reserves, and a lower vegetation pulse across the landscape, seedlings are much more likely to be eliminated by grazing, from water stress or simply by chance. Following larger rainfall events, increased soil moisture at depth will persist for longer periods giving seedlings a better chance of establishment. Greater vegetation cover will physically hide seedlings from grazing animals, and provide alternative feed.

Only the addition of seed significantly increased the likelihood of regeneration occurring at a site (Table 5.14 & 5.16). Seed addition also significantly increased the number of juveniles observed, particularly in the presence of watering (Table 5.17). This indicates that seed abundance is limiting the chances of regeneration occurring at a site, as well as the quantity of regeneration. Whilst the majority of individuals that this analysis are based upon are *E. tomentosa*, it is interesting that seed availability actually had a greater impact on the odds of regeneration occurring than soil moisture or soil seedbed. *E. tomentosa* is widely distributed through the exclosure, as through most *C. pauper* woodlands and was commonly observed with abundant fruit (personal observation). This apparent lack of seed, however, may be partly due to the plot size. As plot size is increased, the probability of regeneration of *E. tomentosa* (or other species) being observed will increase.

Soil disturbance had no significant impact of either the probability of regeneration occurring, or the number of recruits observed (Table 5.14 & Table 5.17). However, a non-significant increase in the number of recruits on scarified plots was observed in the presence of seed application and watering (Figure 5.14). Lack of a significant response may be due to the low numbers of recruits measured during the experiment. It is also likely that the response to soil disturbance is species specific, and that the combined measure for all species were therefore not able to distinguish this effect. Further data on individual species are required to better elucidate the effect of scarification.

No significant effect of fire on regeneration response was found. Fire did not appear to either increase or reduce regeneration response. This is not surprising in a community where fire is a rare event, occurring only following years of well above average rainfall (Leigh and Noble 1981).

No regeneration was found for many species for which seed was applied. These included *Alectryon oleifolius*, *Acacia colletioides*, *Callitris gracilis*, *Casuarina pauper*, *Pittosporum phylliraeoides*, and *Senna artemisioides.* It is possible that limited *A. colletioides* regeneration may have occurred, with two unidentified *Acacia* juveniles observed.

There are many potential causes for lack of regeneration, and some of these have been investigated in following sections. Some potential causes include:

(i) lack of germination;

(ii) death of germinants before they were identified. *C. pauper* cotyledons begin to develop a central spike within the first week of emergence (Section 5.6.2.2), but before this are indistinguishable from many other cotyledons which were measured on the plots. Germinants less than one week old therefore, could have been overlooked;

(iii) insufficient soil moisture for germination, or insufficient soil moisture for seedling survival;

(iv) lack of viable seed, (due to low seed viability of distributed seed, no seed in soil seedbank, seed harvesting, seed losses through wind or water dispersal); and

(v) inappropriate soil seed bed conditions.

Seedling survival

Results for the seedling survival analysis showed a significant improvement in seedling survival with additional water, and significantly greater survival for *C. pauper* seedlings than *M. platycarpum* seedlings (Table 5.18).

High seedling mortalities are likely to represent the natural situation due to dehydration, competition and grazing, from both vertebrates and invertebrates. Survival rates of less than 1 % have been found for seedlings of perennial species in arid environments (Castellanos and Molina 1990; Watson *et al.* 1997b).

It is likely that only those seedlings germinating in "safe sites" are likely to survive long in harsh semi-arid conditions. The chance of survival is significantly enhanced by increased soil moisture. A non-significant response of improved seedling survival where soil was scarified could be due to decreased competition, decreased resistance to root elongation, or improved water retention.

C. pauper seedlings were taller than *M. platycarpum* seedlings at the same stage, and root systems, whilst not measured, appeared significantly longer. However, as these seedlings were grown in potting mix under shade cloth, it is not possible to determine if this pattern of more rapid development of *C. pauper* seedlings also occurs in the field situation. It is possible that the increased development of *C. pauper* seedlings led to longer survival in the field.

High mortalities of seedlings limit the ability to determine effects of treatments on seedling survival. However, it is interesting to note that after watering ceased, more than half the remaining seedlings survived over winter. No seedlings survived the ensuing dry summer. This is in accordance with previous research that suggests that the first summer is a critical time for survival of juvenile perennials (Zimmer 1944; Lacey 1972).

Method

Overall, the method for triggering an episodic regeneration response was unsuccessful. Many difficulties were encountered in topping up rainfall to match that of years of well above rainfall. The largest problem was to maintain soil moisture levels at levels likely to be found during event years, with high evaporation rates leading to rapid drying. Data for event years shows significantly reduced daily evaporation rates during most of 1973 and 1974, compared to the long-term average. In addition, daily maximum temperatures were also less during event years, contributing to reduced evaporation rates. Additional limitations with the methodology include:

(i) water runoff over the landscape contributes considerably to soil moistures in resource rich patches. Adding only the quantity of water received during event years is probably insufficient for the types of soil moistures found during a major regeneration event;

(ii) high soil water levels below the surface are likely to occur in event years, as consistent rainfall exceeds evaporation rates. This is les likely to be achieved by regular smaller water additions; and

(iii) extended drought conditions during the study period may also have depleted soil water from deeper soils that often retain higher soil moisture.

5.3 SOIL SEEDBANK OF *CASUARINA PAUPER* WOODLAND IN NORTHWEST VICTORIA

To assist in understanding regeneration requirements, soil samples were taken to determine the extent of viable seed in the soil seedbank. Determining the presence or absence of viable seed in the soil contributes to understanding the likelihood of regeneration when favourable conditions occur. In the absence of a soil seedbank, regeneration can only occur following seed fall or seed distribution from other sources. Species may be absent from the extant vegetation but present within the soil seedbank due to grazing, lack of appropriate germination triggers, or seasonal effects.

Therefore, the objectives of this study were to:

(i) determine the abundance and distribution of *C. pauper* seed in the soil seedbank; and

(ii) characterise the soil seedbank within a *C. pauper* remnant woodland beneath and outside the canopy of mature *C. pauper* trees.

5.3.1 Methods

Site selection

The soil seedbank was investigated in *C. pauper* woodland adjoining the exclosure used for regeneration trials as outlined in section 5.2 (condition index 0.37, 34°32', 141°06'). This was compared to two good condition *C. pauper* remnants; (i) *C. pauper* woodland south of Settlement Road (condition index 0.70, 34°32', 141°31'), and (ii) Mallanbool Flora and Fauna Reserve (FFR) (condition index 0.71, 34°30', 141°33') (Plate 5.6). The good condition sites were selected from quadrats sampled in the vegetation condition survey in 2000.

Perennial vegetation and dominant annual vegetation were initially surveyed in December 2000. Flora surveys at the time of soil collection on 3rd May 2002, and in April 2003 failed to locate any additional annual or perennial species. All vascular plants were monitored seasonally between spring 2000 and winter 2002 within experimental plots adjoining the exclosure.

The soil seedbank was sampled from within a 100 x 100 m quadrat at each site. Soil samples were taken from beneath and beyond the canopy of four trees within each quadrat. Trees were selected with retained fruit capsules indicating that they had produced seed in the last season and to enable positioning of two transects from each tree that did not overlap other canopies.

Samples were collected at 1 m intervals along two 12 m transects radiating from the base of each tree. Samples beneath the canopy (0–4 m) were bulked, and samples beyond the canopy (4–12 m) were also bulked.

Soil samples were collected on 3rd May 2002, after *C. pauper* had finished fruiting and releasing seed. The majority of *C. pauper* seed release in 2002 was observed in late summer, by March few cones retained on the tree contained any seed, but seed was widely visible on the ground until April. *C. pauper* has been shown not to maintain a persistent soil seedbank, and so it is unlikely that viable seeds will be deeply buried (Auld 1995b). Therefore, whilst soil samples are commonly sampled to at least 5 cm (Read *et al.* 2000; Tekle and Bekele 2000; Wills and Read 2002), only the first 3 cm was sampled in this study.

Ninety-six soil samples [24 (samples) x 4 (trees)] were collected at each site, with a total of approximately 12.6 kg soil per site. Soil samples were collected using a hand trowel within a 6 x 6 cm area to a depth of 3 cm. Beneath the canopy, litter samples were collected separately from soil samples. Beyond the canopy, litter and soil were combined, as only sparse litter was found.

(a)

(b)

(c)

Plate 5.6. ***Casuarina pauper*** **woodland sites where soil seedbank was investigated (a) adjoining the exclosure, (b) South of Settlement Road, and (c) Mallanbool Flora and Fauna Reserve.**

Soil samples were placed into labelled clear polyethylene bags. Soil samples were close to air-dry when collected (gravimetric water content 2.2 %) and were stored in a dark cupboard in a laboratory at room temperature at the University of Ballarat for two weeks until used.

Prior to potting soil samples, coarse litter was removed by sieving samples through a 4.75 mm sieve. *C. pauper* seeds, at less than 4 mm wide passed through the sieve, as did all *C. pauper* stem fragments. Each bulked soil sample and litter sample was weighed and then spread into a plastic seedling tray (34 x 28 x 5 cm) to a depth of approximately 1 cm (mean soil weight 1048 g, litter on top of 3 cm washed sand. Trays were lined with paper towel to prevent the sand from washing out of the tray but allow free drainage. One tray of washed river sand and one tray of potting mix on top of washed river sand were monitored to identify any existing seeds in the washed river sand, or any seeds that may have blown into the glasshouse from the local Ballarat flora.

The samples were kept in a heated glasshouse, with the minimum temperature set to the average monthly minimum experienced in the study area (Figure 2.4). Temperature and relative humidity were measured two hourly for the duration of the experiment using a T-TEWC data logger. Mean temperatures ranged from 12.1°C (min 5.1°C to max 29.1°C) in June 2002 to 17.7°C (min 8.6°C to max 35.9°C) in February 2003. Mean relative humidity during the experiment was 79.8 % (min 22.3 % to max 101.3 %). Seedling trays were watered three times daily for two minutes by an automatic misting sprinkler system.

Seedling trays were randomly allocated in the glasshouse, and were re-allocated every three weeks to minimise variability in lighting or watering. Seedling emergence was monitored weekly for the first six weeks, and then every three weeks for 42 weeks until no further emergents were noted over a three-week period. Seedlings were removed once positively identified. Seedlings that could not be identified were pricked out into pots and grown on until identifiable.

One individual of *Hypochoeris radicata* germinated from the sand only tray. No other germinants were recorded on either of the control trays, however, eleven individuals of five species from the Ballarat flora were found in seedling trays. All individuals of *Polygonum aviculare*, *Leptospermum* sp., *Stellaria media*, *Cyperus tenellus*, and *Epilobium billardierianum* ssp. *cinereum* were not included in the results as they have not previously been recorded in the study area (NRE 2000a).

To characterise the seedbank, seed density (m^{-2}), status as native or exotic, and proportion of emergents in life form categories (exotic herbs, native herbs, exotic grasses, native grasses, shrubs and sub shrubs, and trees) were calculated for each site. The seed density of *C. pauper* seeds at each site was calculated and cumulative emergence of *C. pauper* was plotted. Total values for sites are presented for all analyses, rather than means, as all replicates have been aggregated due to high variability in individual samples.

5.3.2 Results

Seedbank characterisation

A total of 8,701 individuals, of 64 species belonging to 19 families germinated from the soil seedbank samples (Appendix 10). Twenty-five species germinated at all three sites, 12 species at two sites, and 27 at only one site (Appendix 10). A further 1.1 % (103 individuals) died before identification was possible, with some seedling deaths occurring in seedling trays, and some in transplanting. These were not *C. pauper*, or grasses, but were preliminarily identified as herb species.

Annual herbs comprised the major portion (between 88.2 – 92.6 %) of the soil seedbank at all sites, with few (4.8 % to 7.6 %) perennial species represented (Figure 5.23). Two annual herbs *Crassula sieberiana* and *C. colorata* dominated the soil seedbank across all sites, comprising between 69–78 % of all herbs observed.

Weeds represented only a small proportion (5.1−19.4 %) of the total seedbank. The number of native herb seeds in the soil seedbank greatly outnumbered exotic herbs. The percentage of exotic weeds was higher at Mallanbool than the two sites within the MSNP (Figure 5.23). No exotic shrubs were recorded.

At the exclosure, 13 species were recorded in the vegetation survey, with 46.2 % (1 semi-perennial and five perennials) germinating in the soil seedbank. At the Settlement Road site, 23 species were recorded in the extant vegetation, with 26.1 % (*Austrostipa* sp. and *Austrodanthonia* sp. plus four perennials) found in the soil seedbank. At the Mallanbool site, 27 species were recorded in the extant vegetation and 18.5 % (1 semi-perennial and four perennials) in the soil seedbank. The majority of species located in the soil seedbank were annuals and were not observed during vegetation surveys at these sites.

Total seed density was greatest beyond the canopy, largely due to the contribution by herbs (Table 5.22 & Figure 5.24). The density of exotic seeds was greatest within litter and beneath the tree canopy (Table 5.22).

(a)

(b)

(c)

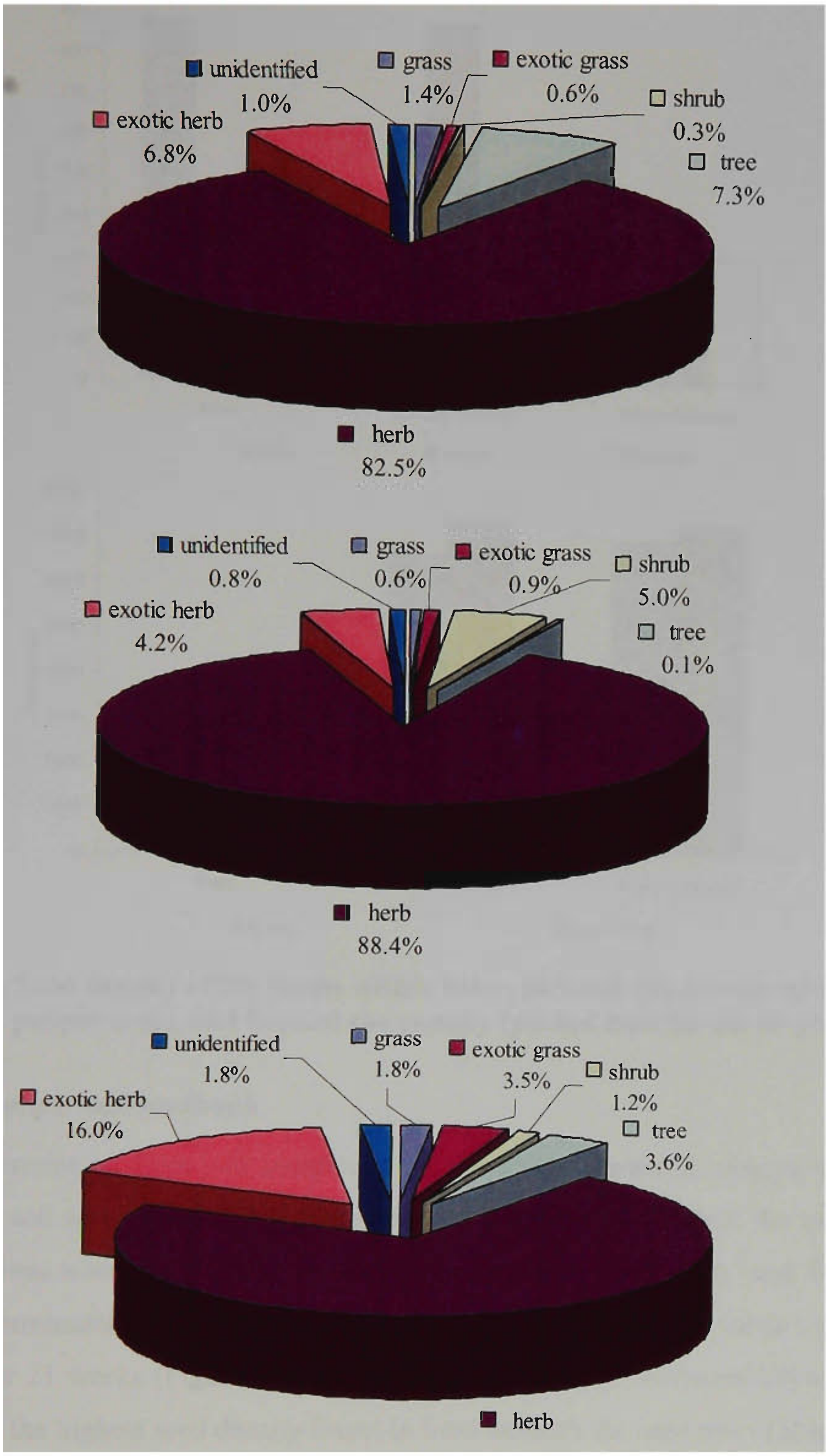


Figure 5.23. Proportion of soil seedbank occurring in each life form at (a) Exclosure, (b) Settlement Road, and (c) Mallanbool.

Table 5.22. Abundance of *Casuarina pauper* and exotic species seed germinated within litter, beneath the tree canopy and beyond the canopy.

Seed parameters	Within litter	Beneath canopy	Beyond canopy
Exotic seed m^{-2}	960.65	818.87	461.75
Casuarina pauper m^{-2}	434.03	112.85	93.56

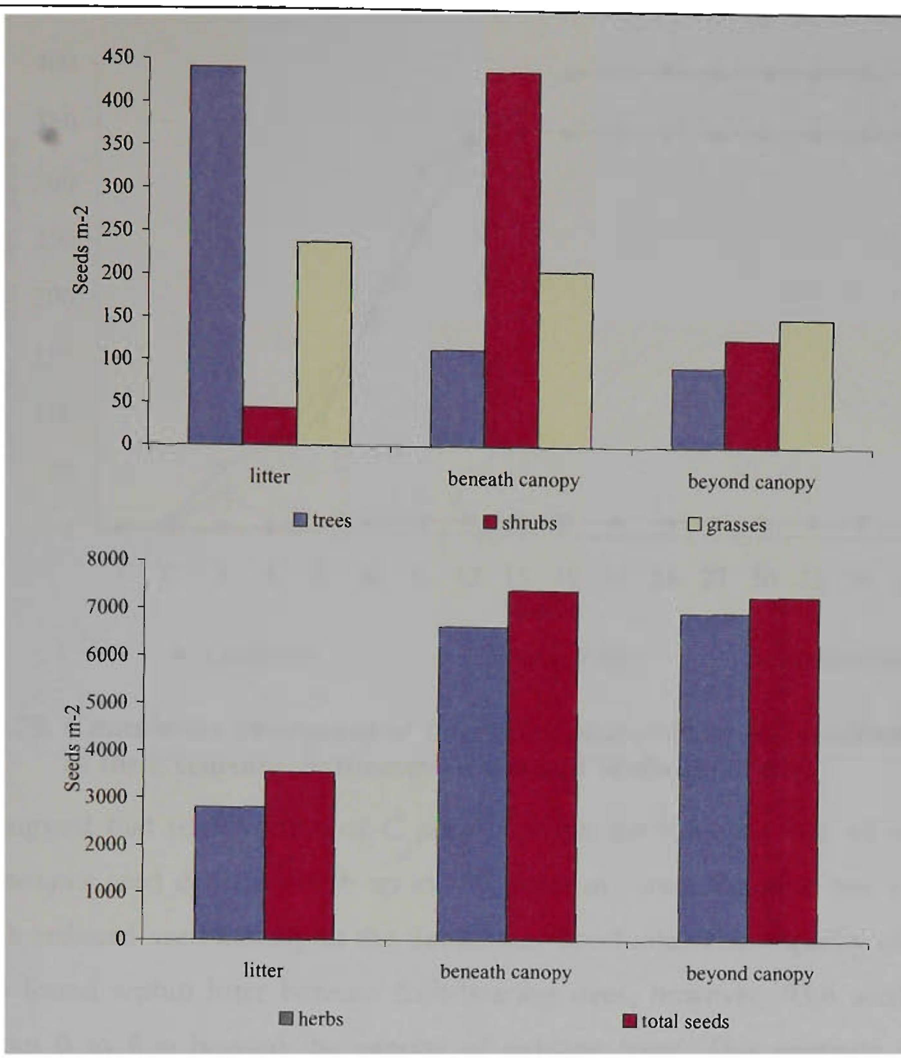


Figure 5.24. Seed density of life forms within litter, beneath the canopy of *Casuarina pauper* trees, and beyond the canopy (pooled data for three sites).

***Casuarina pauper* soil seedbank**

C. pauper germinated from soil samples at all three sites. Fewer *C. pauper* seeds were found in the soil seedbank at the Settlement Road site (11.57 m^{-2}), than the exclosure or Mallanbool sites where similar seed densities were observed (399.31 m^{-2} and 347.22 m^{-2}). *C. pauper* germinants were first observed after two weeks, and no further germinants emerged after 21 weeks (Figure 5.25). *C. pauper* seed density decreased beyond the tree canopy, with the highest seed density found in litter beneath the seed tree (Table 5.22).

5.3.3 Discussion

***Casuarina pauper* seedbank**

It has been suggested that lack of a suitable soil seedbank could be a contributing factor to lack of regeneration of perennial species within grazing exclosures in the MSNP (Sandell *et al.* 2002). There have been few studies of soil seedbanks in northwest Victoria (Carrol and Ashton 1965; Hayes 1993).

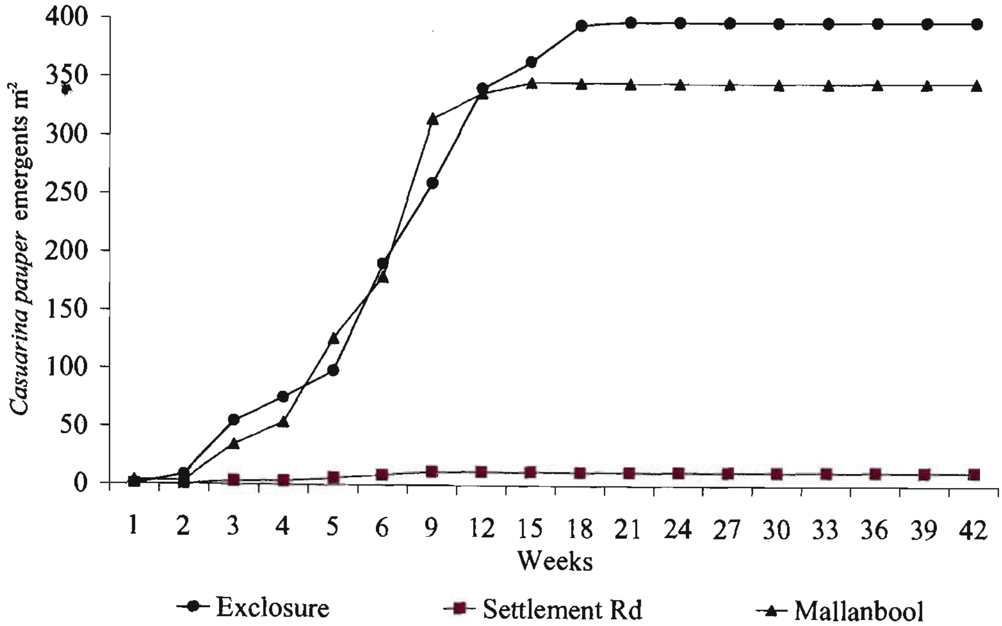


Figure 5.25. Cumulative emergence of *Casuarina pauper* from soil seedbank samples at the Exclosure, Settlement Road and Mallanbool sites.

Results suggest that regeneration of *C. pauper* is not limited by a lack of viable seed. High *C. pauper* seed densities with up to 399 seeds m^{-2} were found at two out of three sites, with reduced seed density at the Settlement Road site. The majority of *C. pauper* seed was found within litter beneath fruit-bearing trees, however, 93.6 seeds m^{-2} were found from 0 to 8 m beyond the canopy of existing trees. This supports findings of Chesterfield and Parsons, who also found high abundance of *C. pauper* seed (1.7×10^5 ha^1) in soil and litter. This demonstrates seed density is unlikely to be a limiting factor for regeneration within remnants with a relatively intact *C. pauper* canopy.

The soil samples were taken in May, more than one month after *C. pauper* seed had finished dispersal. It is likely that seed density was higher during February and early March when abundant seed was visible on the ground and seed was being dispersed. Seed harvesting by parrots was often observed in late summer and early autumn with parrots feeding on *C. pauper* seed both in the canopy and on the ground.

Characterisation of soil seedbank

The soil seedbank of *C. pauper* woodland remnants was characterised by high densities of native herbs, and low densities of perennial tree and shrubs, with the exception of *C. pauper* (Figure 5.23).

Absence or low densities of seed of perennial species may be due to many reasons including:

(i) seed losses from the soil seedbank – seed may be lost from the soil seed store through deep burial by ants, or washed down through the soil profile during rainfall events (Kemp 1989). Seeds may also be lost to granivores, fungal attack or seed death under unfavourable environmental conditions (Kemp 1989);

(ii) viable seed may have failed to germinate in the glasshouse if dormancy requirements were not met;

(iii) viable seed did not enter the soil seedbank due to failure of seed set, or pre-dispersal granivory;

(iv) transient seedbanks may also have been missed by the single sampling event; or

(v) inadequate sampling failed to locate seed present in low seed densities.

Whilst it is difficult to determine the reason for low seed density of perennial species, these results are consistent with findings of low seed density of perennial species, particularly of trees across many ecosystem types (Donelan and Thompson 1980; Vlahos and Bell 1986; Tekle and Bekele 2000). Low seed density of perennial species is also typical for arid regions where shrubs and long-lived perennials show little reliance on soil seedbanks, instead producing seed most years (Kemp 1989). Low representation of perennial plants in the soil seedbank has been found in other studies within northwest Victoria (Carrol and Ashton 1965; Hayes 1993).

It has been shown that early successional species, or disturbance favouring species commonly create a larger store of dormant seeds, compared to later successional species such as tree species (Donelan and Thompson 1980). Disadvantages of creating a large soil seed store for higher perennial plants include problems of predation of relatively large seeds, and reliance on disturbance for seed burial and germination (Donelan and Thompson 1980).

Along with low seed density, low seed distribution has been found in previous studies in arid ecosystems, with the distribution of seeds determined largely from the spatial distribution of extant vegetation (Kemp 1989). Therefore, unless soil samples were taken from the vicinity of individuals they may appear as absent in the soil seedbank. In much *C. pauper* woodland, shrub densities are low, with patchy distributions, which may limit the ability to reliably quantify the soil seedbank using a randomised or systematic sampling strategy. This may also explain why perennial species appear absent from the soil seedbank when total percentage cover of the species is low.

The majority of seeds of perennial species were found within 4 m from established trees, suggesting facilitation effects might occur. Benefits to plants beneath mature trees include increased soil moisture, and increased soil nutrients (Pugnaire and Haase 1996). Trapping of wind and water dispersed seed within litter is also likely to contribute to greater concentrations of larger perennial seeds beneath trees.

Fewer numbers of herbs germinated from litter samples, which may be due to lower seed abundance underneath tree canopies, competition effects, or allelopathic compounds in litter. Leachates from *C. pauper* litter have been shown to reduce germination rates of a native grass *Austrodanthonia caespitosa*, but had no effect on an exotic annual herb *Carrichtera annua* (Barritt and Facelli 2001). It is possible that decreased herb abundance has resulted from allelopathy, but is more likely to be due to competition effects reducing herb survival and seed production directly beneath *C. pauper* canopies.

This study has measured the soil seedbank within remnants of moderate to good condition. High *C. pauper* seed densities were found, but otherwise, low abundance of most perennial species was found. Further research is required to determine seed densities at very open or degraded sites, and possible effects on regeneration. These results suggest that transient soil seedbanks or absence from the soil seedbank may be a contributing factor in widespread regeneration failure for many perennial species within *C. pauper* woodlands.

5.4 SEED PRODUCTION AND ANT SEED HARVESTING

5.4.1 Estimated seed counts

Seed production of *C. pauper*, *M. platycarpum* and *A. oleifolius* were estimated in summer 2001, 2002 and 2003. Estimated seed counts were performed by counting seed on an individual, representative branchlet. The number of branchlets per branch, and the number of branches per tree were counted for ten trees to provide an estimate of seed production.

High seed production was found for *C. pauper* in all years (average estimated 50,000 seeds per tree). Seed production was also high for *M. platycarpum* in 2001 and 2002, but decreased in 2003 (average 9,000 seeds per tree). Seed production of *A. oleifolius* was more erratic, with good seed production in 2001 (average 9,000 seeds per tree estimated), but minimal seed production in 2002 and 2003.

5.4.2 Ant seed harvesting

Ants are significant seed dispersers and predators, and particularly in arid zones have been shown to have a large impact on seed availability (Berg 1975; Andersen 1982). Indeed, ant seed harvesting has been shown to remove up to 100 % of newly fallen seed, thus having a major impact on regeneration (Wellington and Noble 1985; Ireland and Andrew 1995).

The extent of seed harvesting by ants or other small granivores was examined in the area surrounding the grazing exclosure for four tree species; *A. oleifolius, C. gracilis, C. pauper* and *M. platycarpum*. It was hypothesised that the relatively high seed weight and hard seed coats of *A. oleifolius* and *C. gracilis* would limit the harvesting of these seeds by ants. High rates of ant harvesting have been found for *Allocasuarina pusilla* and *A. luehmannii* (Andersen 1987; Castle 1989), and it was hypothesised that high rates of seed harvesting of *C. pauper* may also occur. Chesterfield and Parsons found up to 90 % of *C. pauper* seed was removed from tulle bags *M. platycarpum* also produces a relatively small seed and again it was hypothesised that high rates of ant seed harvesting would occur.

Seed was collected by Greening Australia in spring/summer 2000–2001. To measure seed removal rates, seed was placed on the soil surface beneath a small round, white opaque plastic container (diameter 10 cm, depth 4 cm), and secured by a large galvanised nail piercing the centre of the container and pushed into the ground. Five holes, approximately 10 x 10 mm were cut into the base of each container to allow access to the seed by ants, but to minimise seed loss by wind and predation by birds or other mammals. Similar bait stations have been used in other studies, often using Petri dishes (Andersen and Ashton 1985; Andersen 1988). Plastic containers were chosen over Petri dishes for greater ease of creating holes. Minimal seed losses to wind have been shown for similar designs in other studies (Andersen and Ashton 1985).

Ten seeds were placed at each bait station. Bait stations were spaced 25 m apart along two 250 m transects. The number of seeds under each container was monitored every 24 hours in the morning, and seed was added, where necessary to return each bait to 10 seeds. Seed removal was monitored for between three to six consecutive days at each of six monitoring periods between spring 2001 and autumn 2002 (Table 5.23).

Table 5.23. Species investigated for ant seed harvesting, number of days investigated, and number of bait stations per species.

Date	Species	Days	Bait stations per species
Nov 2001	*Alectryon oleifolius, Callitris gracilis, Casuarina pauper*, and *Myoporum platycarpum*	4	5
Jan 2002	*Alectryon oleifolius, Callitris gracilis, Casuarina pauper*, and *Myoporum platycarpum*	5	5
Mar 2002	*Casuarina pauper*	5	10
Apr 2002	*Casuarina pauper*	6	20
May 2002	*Casuarina pauper*	3	20
Jun 2002	*Casuarina pauper*	4	20

Results

No seed was removed from bait traps containing *A. oleifolius* or *C. gracilis*. For bait stations containing *C. pauper* and *M. platycarpum* seed, high spatial and temporal variability was found, with removal rates for individual bait stations ranging from 0 to 100 %. Mean seed harvesting rates for *C. pauper* were observed to decrease each month surveyed from peak harvesting in November to minimal activity in May and June (Table 5.24).

Table 5.24. Mean (SE) percentage seeds harvested from ant baits at the experimental site in *Casuarina pauper* woodland in the MSNP between spring 2001 and autumn 2002.

Date	*Casuarina pauper*		*Myoporum platycarpum*		*Callitris gracilis*	*Alectryon oleifolius*
Nov 2001	28.0	(9.69)	4.5	(1.74)	0	0
Jan 2002	26.0	(8.97)	11.0	(4.58)	0	0
Mar 2002	18.6	(4.84)				
Apr 2002	8.89	(3.03)	No further monitoring			
May 2002	0.17	(0.02)				
Jun 2002	0.00	(0.00)				

In late summer, small brown ants (*Phedole* sp.) were observed removing *C. pauper* seed from the ground, and creating piles of seed around the entrance to ant nests. Ants were also observed to remove seed of *A. colletioides*, *M. platycarpum* and *S. artemisioides* from plots following seed application. Monospecific piles of *Austrostipa* and *Atriplex stipitata* seed were also found to be accumulated around ants' nests.

Discussion

The hypotheses that ant seed harvesting is unlikely to affect seed availability of *C. gracilis* or *A. oleifolius* was supported, with no seed of these species removed from baits. *M. platycarpum* seed reserves appear to be impacted by ant seed harvesting, however, the extent of this impact appears to be relatively low. It appears that ant seed harvesting does not significantly impact upon seed reserves of *C. gracilis*, *A. oleifolius* or *M. platycarpum*. Further research may be required to confirm that harvesting rates remain at relatively low levels throughout the year.

Higher rates of *C. pauper* seed harvesting were observed, however, it appears unlikely that this will be a factor limiting recruitment. In November, seed of *C. pauper* has not yet begun to fall, and so appears to be in higher demand by seed harvesters. Interestingly, during summer and early autumn whilst much seed is available on the ground, relatively high seed removal rates continue. By late autumn and early winter, seed harvesting rates were observed to decline to very low levels suggesting a decrease in ant seed harvesting of *C. pauper* with cooler weather. Andersen (1983) found decreased ant abundance, diversity and activity levels in winter than in summer in mallee vegetation in the southern Mallee Victoria. It is possible that low ant harvesting rates would continue over winter.

Implications for field trial

Seed harvesting and removal by other processes such as wind and water dispersal could have resulted in loss of seed applied to experimental plots, however results of ant seed harvesting studies suggest no removal of *C. gracilis* and *A. oleifolius* by ants or other small seed harvesters (Table 5.24). *M. platycarpum* seed harvesting rates were relatively low (4–11 %). Seed removal rates were higher for *C. pauper* seeds between November 2001 and March 2002 (18.6–28 %), however, temporal and spatial variability were high. These results suggest that a significant loss of seed of these four tree species was unlikely to have occurred due to ant seed harvesting.

5.5 GERMINATION CHARACTERISTICS OF PERENNIAL SPECIES FROM SEMI-ARID WOODLAND

To determine if germination is a limiting factor in regeneration of *C. pauper* woodland of northwest Victoria, germination characteristics of twelve perennial species were investigated. The major ongoing threat to these woodlands is lack of perennial species regeneration. Determining seed germination characteristics of these perennial species will assist in understanding regeneration mechanisms and potential methods to promote germination, which may assist in sustainable woodland management.

Dormancy mechanisms are employed by many species to ensure that seed germinates under favourable condition for establishment. Many different dormancy mechanisms have been identified in the literature. A number of potential dormancy mechanisms have been tested and a selection of commonly used treatments was applied to seed to investigate dormancy mechanisms of seed of semi-arid *C. pauper* woodland.

Many arid and semi-arid species employ dormancy mechanisms enabling germination following suitable rainfall events (Hartman *et al.* 1997). Leaching of allelopathic compounds is a frequently cited example of a dormancy mechanism enabling germination following rainfall. One example of this is *Eremophila maculata* and *E. racemosa* where leaching of water-soluble compounds from the fruit walls enables germination to take place following intense rainfall (Richmond and Ghisalberti 1994).

Some hard seeded species may require fire to overcome dormancy, enabling germination to occur when decreased competition, or increased soil nutrients may be found. However, the extent to which fire facilitates regeneration in semi-arid woodland is uncertain (Leigh and Noble 1981). Extensive fires in the *C. pauper* woodland of south-eastern Australia occur only following periods of prolonged above average rainfall (such as occurred in 1973/4/5), when is there sufficient fuel to carry a fire (Leigh and Noble 1981). Despite this, given the prevalence of fire related dormancy mechanisms in Australian woodland seed, understanding the effects of fire on these species is critical.

Fire related effects that influence seed germination include direct and indirect factors. Direct effects of fire that have been shown to influence germination include chemical effects of smoke and charcoal, and physical effects of heat that may assist in cracking the seed coat of hard seeded species (Hodgkinson 1991; Dixon *et al.* 1995; Cocks and Stock 1997; Enright *et al.* 1997; Lloyd *et al.* 2000; Tieu *et al.* 2001; Wills and Read 2002). Indirect effects of fire that may promote germination include decreased canopy cover and increased light availability (Wilson and Mulham 1979), and reduced levels of allelopathic chemicals (Keeley 1987).

Another common germination trigger is cold stratification or chilling. Stratification is commonly required in cool or temperate climate species enabling germination to occur in a favourable season (Hartman *et al.* 1997). Cold stratification has been recommended for species of *Callitris* (Bonney 1994), however, cold stratification has been associated with decreased viability in some woodland species (Lush *et al.* 1984; Willis and Groves 1991; Clarke *et al.* 2000).

Scarification, or softening of the seed coat is required for germination of many species with a hard testa. Hard seed coats can preserve seed for extended periods by preventing imbibition, and in some cases, gas exchange. To enable germination to occur, the testa must be softened or broken down, and this be may achieved by exposure to fire, gradual abrasion as sand is blown across seeds, passage through the gut, or soil micro-organisms (Rogers *et al.* 1993; Cocks and Stock 1997; Hartman *et al.* 1997). To mimic this activity, and soften an impervious seed coat, scarification can be performed by nicking the seed coat with a razor blade, abrading with sand paper, or other similar activities (Lynch *et al.* 1999; Baes *et al.* 2002).

Within this study, the effect of the tank water used to water all plots within the exclosure was also examined. It was hypothesised that increased nutrients, salts or altered pH may affect the germination of seeds. Therefore, the effect of this water on germination was also investigated.

Therefore, to investigate germination characteristics, six seed pre-treatments were employed on nine species of semi-arid *C. pauper* woodland, and compared to a control, a further treatment testing the tank water was also investigated. This resulted in eight treatments; (i) control (ii) dry heat (iii) wet heat (iv) soaked in smoky water solution for 24 hours (v) soaked in water for 24 hours (vi) cold stratification (vii) scarification. (viii) tank water (no pre-treatment). These treatments have been applied to nine perennial species of semi-arid woodland firstly to determine the germination percentage and secondly to determine what dormancy mechanisms are utilised by these species. Germinability of a further three perennial semi-arid woodland species was also examined without investigation of germination characteristics.

5.5.1 Methods

Germinability and dormancy characteristics of nine perennial species common to semi-arid *C. pauper* woodland were examined. Dormancy characteristics were not investigated for two species due to insufficient seed of *A. oswaldii* and for *S. artemisioides* as the seed contained a mix of three subspecies, and it is possible that dormancy characteristics will vary between species (Table 5.25).

Hard seed coats were observed for *Acacia colletioides*, *A. oswaldii*, *Alectryon oleifolius*, *D. viscosa*, *P. phylliraeoides* and *S. artemisioides*. *C. gracilis* seed also is encased by a woody coat that may impose physical dormancy. Seed was collected by Greening Australia in the Murray-Sunset and Hattah-Kulkyne National Parks and surrounding reserves in northwest Victoria between October 2000 and April 2001. Seed for each species was collected from a minimum of ten individual plants. Air-dried seeds were placed in a separated labelled airtight zip lock bags and kept in a cupboard until use. Seed viability was tested prior to commencement of the experiment by examination of the embryo of 50 seeds under a dissecting microscope. Seeds containing a firm, white embryo were determined to be viable (Baskin and Baskin 1998).

Table 5.25. *Casuarina pauper* woodland perennial species tested for dormancy characteristics and germinability.

Family	Scientific name	Life form	Parameters
Fabaceae	*Acacia colletioides*	Tall shrub	Dormancy
Fabaceae	*Acacia oswaldii*	Tree	Germinability
Sapindaceae	*Alectryon oleifolius*	Tree	Dormancy
Cupressaceae	*Callitris gracilis*	Tree	Dormancy
Casuarinaceae	*Casuarina pauper*	Tree	Dormancy
Sapindaceae	*Dodonaea viscosa* ssp. *angustissima*	Tall shrub	Dormancy
Chenopodiaceae	*Enchylaena tomentosa* var. *tomentosa*	Small shrub	Dormancy
Myoporaceae	*Myoporum platycarpum*	Tree	Dormancy
Asteraceae	*Olearia pimeleoides*	Small shrub	Dormancy
Pittosporaceae	*Pittosporum phylliraeoides*	Tree	Dormancy
Caesalpiniaceae	*Senna artemisioides* ssp. *filifolia*, *petiolaris*, and *coriacea*	Tall shrub	Germinability

Seed weight of each species was determined by weighing seeds from a random sub sample of 50 seeds on a precision electronic balance. Seeds were classified according to mean weight, with seeds <0.01 g classified as small, medium seeds 0.02< >0.01 g, and large seeds >0.02 g.

Seeds were mixed thoroughly before taking a random subsample of 25 seeds of each species. Seeds were treated with one of seven treatments and compared to a control (no seed pre-treatment); (i) control (ii) placed in an 80°C oven for 15 minutes in a pre-heated metal drying tin. Temperatures of between 80 to 100°C have been shown to produce optimal germination for a number of woodland species and are commonly used to simulate heating effects of fire (Auld 1986; Auld and O'Connell 1991; Bradstock *et al.* 1992; Bradstock and Auld 1995; Cocks and Stock 1997; Tieu *et al.* 2001) (iii) placed into water just ceased boiling (initial water temperature measured at 85°C), (iv) soaked in Regen2000™ a commercially available smoky water solution for 24 hours, (v) soaked in distilled water for 24 hours, to mimic the effect of soaking in rainwater puddles (vi) cold stratification – seed stored at 0°C for 48 hours, (vii) shaken with sharp sand for 15 minutes (scarification) using a mechanical rotating device (Stihl Mini-Culti BT 510 'Q' 325 attached to a Stihl FS66), and (vii) watered with tank water collected from the concrete storage tank from which water was accessed for the field study. Tank water was collected in June 2001 and stored in a polyethylene container. To investigate the effect of using water from the Murray River, which had been stored in an open concrete tank, seeds were kept moist using the tank water.

Seeds from four replicates (N = 25) of each treatment were placed in sterilised Petri dishes (9 cm diameter) on a Whatman® No. 1 filter paper and moistened with distilled water. Benlate fungicide was applied to each Petri dish on commencement of the treatment and then as required to treat any fungal damage. Testing of each replicate in the germination chamber was staggered in time to minimise pseudoreplication (Morrison and Morris 2000). The experiment was run for 172 days, with replicates commenced at 43 day intervals. Seeds of the shrub species were removed at this time due to space limitations and seeds of tree species continued in the germination chamber for a further 19 days (62 days in total for trees).

Germination trials were conducted between August 2001 and December 2002. Seeds were placed in a germination chamber with 12 hour alternating light and dark and alternating temperatures of 25°C and 12°C. Petri dishes were checked for germinants three times a week and seeds kept moist with distilled water at these times. Germination was considered to have occurred when emergence of the radicle from the testa was observed (Hartman *et al.* 1997).

From the seed germination data, total percentage germination, time in days to first germination, time in days to 50 % germination and germination rate were calculated where:

$$\text{percentage germination} = \frac{\sum n_x}{25} \times 100$$

time to first germination is defined as the time from first exposure of the seed to moisture to emergence of the radicle through the testa, and

$$\text{germination rate} = \frac{\sum (n_x \times t_x)}{\sum n_x}$$

where n is the number of germinants at time t (days). It must be noted that seeds were not monitored daily, and so the daily rate is a conservative measure. It is possible that time to first germination, time to 50 % germination and germination rate may be slightly changed if measured daily.

Analysis

Analysis was performed using SPSS (V10). To stabilise inequality of variance, percentage data were transformed using an arcsine transformation. One-way ANOVAs were used to examine significant differences between treatments for each species. Tukeys HSD post-hoc tests were performed to determine where differences between treatments occurred. To test for conformation to assumptions of the ANOVA, normality was checked using Q-Q plots and equality of variance using Levenes test. Untransformed values have been presented for all mean and standard error values.

5.5.2 Results

Total seed viability as observed under the microscope and seed weight for each species are listed in Table 5.26. Consistent with the low seed viability observed under microscope, no germinants were observed for *A. oleifolius* under any treatment. Therefore, no further results have been presented for *A. oleifolius*. Mean germination for *A. oswaldii* and *S. artemisioides* are presented in Table 5.27. Results of a one-way ANOVA, testing for significant differences between seed pre-treatments are presented in Table 5.28.

Table 5.26. Seed viability and mean (± 1 SE) of seed weight of semi-arid woodland perennial species (N = 50).

Species	Viability	Seed weight		Relative seed size
	%	Mean	SE	
Acacia colletioides	24	0.0144	0.0008	medium
Acacia oswaldii	60	0.1060	0.0145	large
Alectryon oleifolius	2	0.0269	0.0007	large
Callitris gracilis	20	0.0121	0.0007	medium
Casuarina pauper	72	0.0026	0.0001	small
Dodonaea viscosa	84	0.0126	0.0024	medium
Enchylaena tomentosa	92	0.0230	0.0007	large
Myoporum platycarpum	64	0.0021	0.0001	small
Olearia pimeleoides	76	0.0007	0.0001	small
Pittosporum phylliraeoides	100	0.0225	0.0009	large
Senna artemisioides	56	0.0161	0.0008	medium

Table 5.27. Mean germination ± 1 SE of *Acacia oswaldii* and *Senna artemisioides*.

	Percentage germination		Days to first germination		Days to 50 % germination		Germination rate	
Species	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Acacia oswaldii	17.00	2.52	16.50	4.91	19.00	4.24	20.23	3.66
Senna artemisioides	60.00	6.73	9.25	2.29	14.00	3.59	14.98	6.62

Percentage germination

Highest mean percentage germination was observed for *D. viscosa* under the wet heat treatment (67 %, SE 3.42). *M. platycarpum* and *C. gracilis* showed consistently low percentage germination, with maximum mean germination reaching only 14 % (SE 7.02) and 20 % (SE 5.16) respectively (Figure 5.26).

Germination rate

Mean germination rate ranged from 9.04 (SE 1.14) for *E. tomentosa* under the smoky water treatment to 50.63 (SE 3.25) for *P. phylliraeoides* under the tank water treatment (Table 5.29).

Days to first germination

The first seeds to germinate were *E. tomentosa*, with a mean of 3.75 ($\pm$ 0.48) days under the smoky water treatment. *P. phylliraeoides* seeds took the longest time (43.75 days, SE 5.23 when watered with tank water) to commence germination under most treatments except wet heat, closely followed by *C. gracilis* (Table 5.30).

Days to 50 % germination

Results for mean number of days to 50 % germination closely followed those for days to first germination, with *P. phylliraeoides* and *C. gracilis* taking the longest time to reach 50 % germination. *E. tomentosa* and *C. pauper* recorded the least time to reach 50 % germination (Table 5.31).

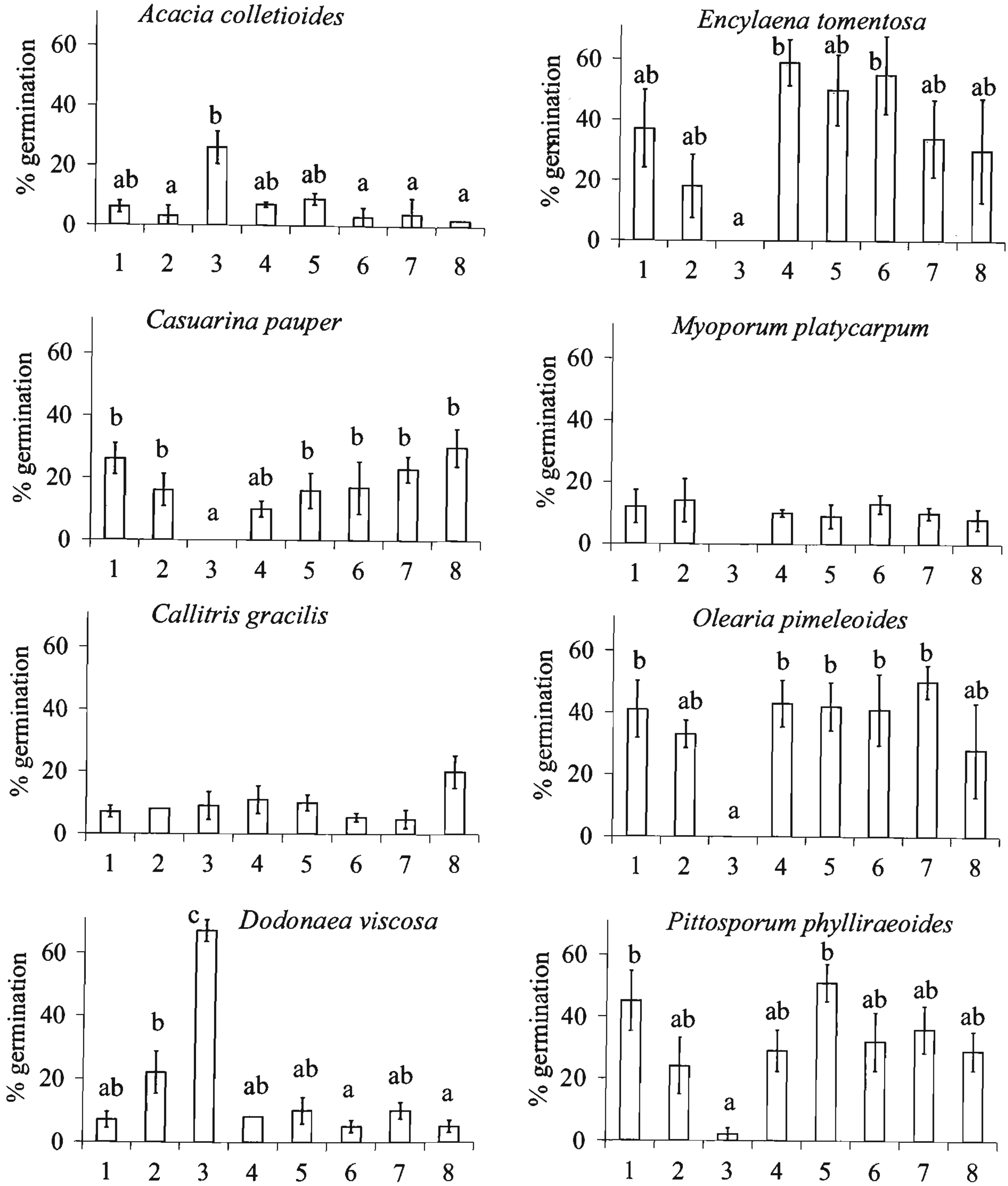
Dry heat

No significant difference was found between the untreated control and the dry heat treatment for any of the germination characteristics measured for any of the semi-arid *C. pauper* woodland species tested. Dry heat was associated with non-significant improvements of percentage germination in *D. viscosa, M. platycarpum* and *C. gracilis.* A non-significant decrease in germination rate, and time to first germination and 50 % germination in response to dry heat was found for *E. tomentosa, M. platycarpum* and *P. phylliraeoides.*

Wet heat

Wet heat was detrimental to four of the semi-arid *C. pauper* woodland species: *C. pauper, E. tomentosa, M. platycarpum* and *O. pimeleoides* with no germinants resulting. However, wet heat treatment was favourable for the two hard seeded species

A. colletioides and *D. viscosa*. For *D. viscosa* wet heat led to significantly more germinants than for all other treatments, however, for *A. colletioides*, wet heat was not significantly better than the control.



(1) control, (2) dry heat, (3) wet heat, (4) smoky water, (5) 24 hr soak, (6) frozen, (7) scarified, and (8) tank water.

Figure 5.26. Mean (error bars ± 1 SE) percentage germination for each of eight semi-arid woodland species under seven treatments compared with a control.

Wet heat treatment did not result in any significant decrease in germination rate or the number of days to first germination for any species. A significant decrease in the number of days to 50 % germination was observed for *P. phylliraeoides* treated with wet heat, however, only 2 % germination was observed.

Smoky water

The smoky water treatment was not found to be significantly different to the control for any measure of germination in any species. Smoky water resulted in a non-significant increase in germination percentage for many semi-arid woodland species with the exception of *C. pauper*, *M. platycarpum* and *P. phylliraeoides.* A decrease in germination rate was also observed for all species except *D. viscosa*, *M. platycarpum* and *O. pimeleoides.*

Most species showed a decreased time to first germination after soaking in a smoky water solution (*A. colletioides*, *C. gracilis*, *C. pauper*, *E. tomentosa*, and *P. phylliraeoides*). However the increased initial rate of germination was not sustained for many species, with only *C. pauper*, *E. tomentosa*, and *P. phylliraeoides* also recording reduced time to 50 % germination.

24 hour soak

The 24-hour soak treatment resulted in a significantly greater number of days to 50 % germination. No other significant results between the 24-hour soak treatment and the control were observed. The 24-hour soak treatment led to an increase in percentage germination for all species except *C. pauper*, and *M. platycarpum.*

Germination rate was decreased following 24-hour soak for *C. gracilis*, *E. tomentosa* and *P. phylliraeoides.* Initial germination was more rapid following the 24-hour soak, with time to first germination decreased for most species except *C. pauper*, *M. platycarpum* and *O. pimeleoides.* This result was not sustained to the 50 % germination with only *A. colletioides*, *E. tomentosa* and *P. phylliraeoides* showing a decrease in the time to 50 % germination.

Cold stratification

No significant differences were observed between the cold stratification treatment and the control. Stratification resulted in an increase in percentage germination for *M. platycarpum* and *E. tomentosa.* Cold stratification resulted in a decrease in germination rate for *A. colletioides*, *C. pauper D. viscosa* and *E. tomentosa.*

Table 5.28. Results of one-way ANOVA for seed germination testing for significant differences between seed pre-treatments.

Species	Percentage Germination		Days to first germination		Days to 50 % germination		Germination rate	
	F	p	F	p	F	p	F	p
Acacia colletioides	4.43	<0.01	0.72		1.68	0.162	1.78	0.137
Callitris gracilis	0.57		0.51		0.53	0.802	0.63	0.727
Casuarina pauper	5.44	<0.01	1.08		2.13	0.080	2.84	<0.05
Dodonaea viscosa	16.44	<0.001	0.28		0.44	0.866	0.37	0.913
Enchylaena tomentosa	6.25	<0.01	6.15	<0.01	4.13	<0.01	4.26	<0.01
Myoporum platycarpum	3.52	<0.05	4.92	<0.05	4.27	<0.01	3.99	<0.01
Olearia pimeleoides	11.80	<0.001	4.92	<0.01	5.84	<0.001	15.09	<0.001
Pittosporum phylliraeoides	5.85	<0.01	2.48		3.26	<0.05	3.47	<0.05

Table 5.29. Effects of eight treatments on seed germination rate of eight species of semi-arid woodland in northwest Victoria, showing average of four replicates, and significant difference between treatments.

	nil	dry heat	wet heat	smoky water	24hr-soak	frozen	scarified	tank water
Acacia colletioides	13.75 a	17.00 a	24.30 a	12.42 a	15.04 a	8.63 a	12.50 a	2.50 a
Callitris gracilis	34.75 a	41.13 a	40.55 a	34.30 a	34.56 a	43.17 a	21.00 a	31.98 a
Casuarina pauper	21.68 ab	29.65 b	0.00 a	17.90 ab	28.50 b	14.11 ab	30.31 b	23.97 ab
Dodonea viscosa	15.50 a	15.77 a	22.15 a	16.38 a	16.00 a	13.75 a	22.15 a	14.88 a
Enchylaena tomentosa	26.32 b	11.99 ab	0.00 a	9.04 ab	11.64 ab	24.86 b	22.59 b	16.08 ab
Myoporum platycarpum	22.20 ab	17.15 ab	0.00 b	24.00 ab	38.98 a	38.18 a	25.63 ab	26.56 ab
Olearia pimeleoides	18.86 a	26.45 a	0.00 b	21.54 a	19.85 a	25.82 a	25.62 a	27.10 a
Pittosporum phylliraeoides	47.71 b	38.70 ab	11.75 a	44.96 b	45.00 b	48.55 b	50.02 b	50.63 b

Different letters indicate significant differences between soil water contents (p<0.05).

Table 5.30. Effects of treatments on the number of days to first germination of eight species of semi-arid woodland in northwest Victoria, showing average of four replicates, and significant difference between treatments.

	nil	dry heat	wet heat	smoky water	24hr-soak	frozen	scarified	tank water
Acacia colletioides	13.75 ^{a}	17.00 ^{a}	19.50 ^{a}	9.75 ^{a}	10.00 ^{a}	7.75 ^{a}	12.50 ^{a}	2.50 ^{a}
Callitris gracilis	29.00 ^{a}	36.75 ^{a}	38.50 ^{a}	31.25 ^{a}	27.50 ^{a}	42.00 ^{a}	20.80 ^{a}	19.25 ^{a}
Casuarina pauper	14.00 ^{a}	22.00 ^{a}	0.00 ^{a}	13.50 ^{a}	14.25 ^{a}	10.50 ^{a}	20.00 ^{a}	12.00 ^{a}
Dodonaea viscosa	12.25 ^{a}	12.25 ^{a}	15.75 ^{a}	14.25 ^{a}	9.50 ^{a}	12.50 ^{a}	16.25 ^{a}	13.50 ^{a}
Enchylaena tomentosa	18.75 ^{b}	7.25 ^{ab}	0.00 ^{a}	3.75 ^{ab}	5.00 ^{ab}	18.50 ^{b}	19.25 ^{b}	11.25 ^{ab}
Myoporum platycarpum	17.00 ^{ab}	11.25 ^{ab}	0.00 ^{a}	21.00 ^{ab}	35.75 ^{b}	30.50 ^{b}	21.00 ^{ab}	15.00 ^{ab}
Olearia pimeleoides	10.00 ^{ab}	19.50 ^{b}	0.00 ^{a}	11.75 ^{ab}	10.75 ^{ab}	17.00 ^{b}	16.25 ^{b}	19.25 ^{b}
Pittosporum phylliraeoides	36.75 ^{ab}	28.00 ^{ab}	10.50 ^{a}	36.00 ^{ab}	34.75 ^{ab}	40.25 ^{ab}	42.25 ^{b}	43.75 ^{b}

Different letters indicate significant differences between soil water contents ($p<0.05$).

Table 5.31. Effects of treatments on the number of days to 50 % germination of eight species of semi-arid woodland in northwest Victoria, showing average of four replicates, and significant difference between treatments.

	nil	dry heat	wet heat	smoky water	24hr soak	frozen	scarified	tank water
Acacia colletioides	13.75 ^{a}	17.00 ^{a}	23.00 ^{a}	13.75 ^{a}	10.00 ^{a}	7.75 ^{a}	12.50 ^{a}	2.50 ^{a}
Callitris gracilis	30.75 ^{a}	36.75 ^{a}	40.25 ^{a}	35.75 ^{a}	31.00 ^{a}	42.00 ^{a}	20.80 ^{a}	26.75 ^{a}
Casuarina pauper	17.50 ^{a}	26.50 ^{a}	0.00 ^{a}	8.75 ^{a}	20.00 ^{a}	12.25 ^{a}	26.25 ^{a}	22.75 ^{a}
Dodonea viscosa	13.50 ^{a}	15.25 ^{a}	21.75 ^{a}	14.25 ^{a}	15.25 ^{a}	12.50 ^{a}	20.25 ^{a}	13.50 ^{a}
Enchylaena tomentosa	26.25 ^{b}	12.75 ^{ab}	0.00 ^{a}	8.25 ^{ab}	8.00 ^{ab}	24.00 ^{b}	23.00 ^{b}	13.75 ^{ab}
Myoporum platycarpum	19.75 ^{ab}	14.50 ^{ab}	0.00 ^{a}	21.25 ^{ab}	38.50 ^{b}	35.00 ^{b}	21.00 ^{ab}	15.00 ^{ab}
Olearia pimeleoides	17.50 ^{b}	28.00 ^{b}	0.00 ^{b}	18.50 ^{b}	19.00 ^{a}	24.00 ^{b}	26.00 ^{b}	20.75 ^{b}
Pittosporum phylliraeoides	46.25 ^{b}	41.25 ^{ab}	10.50 ^{a}	43.25 ^{ab}	43.25 ^{ab}	47.50 ^{b}	49.00 ^{b}	49.25 ^{b}

Different letters indicate significant differences between soil water contents ($p<0.05$).

Cold stratification led to a decrease in time to first germination for *A. colletioides*, *C. pauper* and *E. tomentosa*. Cold stratification also reduced time to 50 % germination for *A. colletioides*, *C. pauper* and *D. viscosa* (Figure 5.26, Table 5.29, Table 5.30 and Table 5.31).

Scarification

Scarification resulted in the greatest percentage germination for *O. pimeleoides* and an increase in germination percentage for *D. viscosa*. No statistically significant differences were observed between scarification and the control treatment.

A decrease in overall germination rate was observed for *A. colletioides*, *C. gracilis* and *E. tomentosa*. A reduction in the number of days to first germination was found for *A. colletioides* and *C. gracilis*, and a decrease in the time to 50 % germination was also observed for *A. colletioides*, *C. gracilis* and *E. tomentosa*

Tank water

No significant results were obtained between the use of tank water and distilled water on any measure of germination for any species. The highest percentage germination was reached for *Casuarina pauper* and *Callitris gracilis* under the tank water treatment and the lowest percentage germination was measured for *A. colletioides*, *M. platycarpum* and *O. pimeleoides*.

A decrease in germination rate and days to first and 50 % germination resulted from the tank water treatment for *A. colletioides* and *C. gracilis, E. tomentosa* and *M. platycarpum*.

Comparison of germination characteristics across all treatments showed higher percentage germination for large seeds, and an increase in time to first and 50 % germination for larger seeds (Table 5.32).

Table 5.32. Germination characteristics of large, medium and small sized seed from semi-arid woodland in northwest Victoria.

Germination	large seed		medium seed		small seed	
	Mean	SE	Mean	SE	Mean	SE
% germination	32.06	2.69	13.16	1.71	20.50	1.87
germination rate	27.84	2.26	21.34	1.57	21.83	1.38
days to 1st germination	21.36	1.98	17.92	1.53	15.09	1.30
days to 50 % germination	27.03	2.29	20.22	1.63	18.86	1.37

5.5.3 Discussion

Viable seed was found for all species except *A. oleifolius*, and whilst germination percentages were relatively low for some species, it appears unlikely that seed viability will be a significant factor limiting regeneration for species other than *A. oleifolius*.

Quiescent seed were observed for all species, suggesting that germination will occur for all species (except *A. oleifolius*) once sufficient soil moisture occurs. Exogenous dormancy was observed for the hard seeded species *A. colletioides* and *D. viscosa*, with increased germination following treatments to increase permeability of the seed coat. No other species investigated showed significant dormancy release under the treatments tested.

No germinants were observed for *A. oleifolius* under any treatment, and only one seed of the 50 examined under the microscope was found to contain an intact embryo prior to commencing the experiment. It has been found that viability in *A. oleifolius* deteriorates rapidly with age, and Westbrooke (1998) found no viable seed following 12 months storage. Germination occurred with fresh seed in 1993, under a range of treatments including no treatment (12 %), 24-hour soak (17 %), seven day soak (7 %), acid soak (20 %) and scarification (25 %) (Westbrooke 1998). Seed viability may vary between years, with greater viability occurring under more optimal conditions. Seed used in this experiment was collected in late summer and stored for between seven to ten months from commencement to completion of the experiment.

Dry and wet heat

Dry heat to 80°C had no significant effect on any germination characteristics for any species tested. However, non-significant increases in percentage germination and a reduction in time to first germination were observed for *C. gracilis, D. viscosa, E. tomentosa, M. platycarpum,* and *P. phylliraeoides*. These included seeds with and without hard testa.

Wet heat proved successful in promoting germination of *A. colletioides* and *D. viscosa*. However, wet heat had a detrimental effect on most other species tested, with only a few germinants resulting from seeds of *C. gracilis* and *P. phylliraeoides* and none from seeds without hard testa (*C. pauper*, *E. tomentosa*, *M. platycarpum* and *O. pimeleoides*). This suggests these species may not survive an intense fire, particularly if seeds are lying on the soil surface where temperatures are greater, or within the top layer of moist soils.

Heat effects from fire have been shown to promote germination of seeds from two plant families tested in this study, Fabaceae and Sapindaceae (Hodgkinson and Oxley 1990). The response of *Dodonaea viscosa* was similar to that described by Hodgkinson and Oxley (1990), with a small increase in germinability from exposure to temperatures up to 80°C. It is likely that *D. viscosa* would show increased germination in response to fire.

Contrary to other studies on Fabaceae in eastern Australia, *A. colletioides* did not show improved germination following exposure to temperatures of 80°C degrees (Auld and O'Connell 1991). However, heating of moist soils has been shown to lead to an increase of germination in some

Acacia species where no response was found from heating in dry soils (Hodgkinson and Oxley 1990). The observed positive effects of wet heat treatment on *A. colletioides* suggest that heating within moist soils would be more likely to stimulate germination. It is unlikely that seed of many other perennial species would germinate following fire with high soil moisture conditions.

Smoky water

Whilst smoky water appeared to result in an early acceleration of germination in some species, no significant difference was measured in the number of days to first germination or to 50 % germination. The overall percentage germination of *E. tomentosa* was greatest under the smoky water treatment, but this result was not significant.

Smoke has been shown in many studies to be successful in promoting germination of many woodland species in different environments (Adkins and Peters 2000; Clarke *et al.* 2000; Read *et al.* 2000; Tieu *et al.* 2001; Wills and Read 2002). However, dormancy mechanisms overcome by smoke may not necessarily signify a need for fire in the natural environment (Bell 1999). Many of the species that showed a decrease in the number of days to first germination in response to smoky water solution were adversely affected by either the dry heat or wet heat treatments, suggesting that fire is likely to reduce germination.

The same suite of species that showed an increase in germination to 24-hour soak in smoky water showed an increase in germination after soaking in distilled water, suggesting water imbibition during the 24-hour soak in smoky water may have contributed to the increased germination, rather than exposure to the smoke chemicals.

24-hour soak

Pre-soaking for 24 hours in distilled water may have enabled seeds to imbibe water more quickly and expel leachates. However, a negative effect of pre-soaking was observed for *O. pimeleoides*, with an increase in time to 50 % germination found. No other significant effects of pre-soaking were found for any germination characteristics for any species.

Scarification

Scarification was associated with higher germination rates in the hard seeded species *A. colletioides* and *D. viscosa*. It is likely that abrasion of the hard testa seed case in these species will have facilitated imbibition (Hartman *et al.* 1997). Scarification also led to the greatest percentage germination of *O. pimeleoides*, which is unexpected in a soft seeded species. It was observed that the scarification resulted in removal of most of the attached pappus bristles. The pappus bristles and fine hairs on the seed can trap air when the seed is placed in water, and it is possible that this could decrease the contact of the seed coat with water.

Cold stratification

Stratification has generally shown a positive effect on temperate to cool climate species and with average minimum winter temperatures of 4.4°C, and the potential for between 13 to 24 frosts per year (Clewett *et al.* 2003), it is possible that cold stratification may be required for germination of species in the northern Victorian mallee. However, stratification did not have a significant effect on germination of any species, but led to non-significant increases in germination percentage of two species, *E. tomentosa* and *M. platycarpum*. This supports the findings of Westbrooke (1998) who also found an increase in percentage germination for *M. platycarpum* following 24-hour freezing.

Stratification has been used to improve germination of semi-arid *Callitris* species including *C. gracilis* (Bonney 1994), but appeared to reduce germinability of *C. gracilis* in the current study. This could be due to prior seed storage, temperature of stratification or length of stratification period. Time of stratification is likely to be important, and only a short stratification treatment was used in the current study.

Tank water

The tank water used for water addition in the field regeneration trial exhibited no significant effects on germination, when compared to using distilled water. This indicates that the water used in the field experiment is unlikely to have affected seed germination.

Implications for regeneration of semi-arid woodland and field trial

Of the ten species tested, seed viability only seems to be a limiting factor for regeneration of *A. oleifolius.*

Results suggest that fire is not a requirement for dormancy release of most perennial species tested, although the soil seedbank of most species would be expected to survive a low intensity grass fire where soil temperatures did not exceed 80°C. One possible exception to this may be *C. pauper*, which does not form a persistent soil seedbank (Auld 1995b), and so recently fallen seed may be close to the surface, potentially leading to exposure to lethal temperatures.

Physical dormancy of *D. viscosa* and *A. colletioides* may be released in a low intensity fire resulting in a higher percentage germination of these species post fire. Seed dormancy was not found for any other species investigated. These results provide no evidence of requirement for cooler temperatures, scarification, or other environmental conditions to release dormancy and promote germination.

5.6 GERMINATION AND EARLY SEEDLING SURVIVAL OF *CASUARINA PAUPER*: EFFECTS OF SOIL WATER CONTENT AND TEMPERATURE

Germination and seedling establishment are two critical steps in the life cycle of most plant species. In arid and semi-arid zones, soil water potential and temperature are two of the key factors affecting germination and subsequent seedling survival (Hartman *et al.* 1997). To determine the effects of temperature and soil moisture on seed germination and subsequent survival of *C. pauper* seedlings a series of experiments was performed under controlled conditions to test the following hypotheses:

(i) total percentage germination will increase with higher soil water content and higher temperatures;
(ii) germination will occur more rapidly with increasing soil water content and increasing temperature;
(iii) increased seedling growth will occur with higher soil moisture content; and
(iv) increased seedling survival will result from higher soil moisture content.

5.6.1 Methods

5.6.1.1 Seed germination trial

Soil (brownish sand) was collected from the top 10 cm of a *C. pauper* woodland site (lat 141.75; long 34.37) in December 2002. The soil was passed through a 2 mm sieve to remove any seed and bring to a uniform consistency and oven dried at 105°C for 48 hours to sterilise and bring the soil to uniform water content (Tan 1996). The oven-dried soil was returned to air-dry moisture levels over two days at room temperature within the laboratory.

C. pauper seed was collected in summer and autumn 2002 from at least 10 trees to the north of the MSNP. Seed was stored at room temperature in airtight zip lock bags in a dark cupboard. Seed viability was tested prior to commencement of the experiment by examination of the embryo of 50 seeds under a dissecting microscope. Seeds containing a firm, white embryo were determined to be viable (Baskin and Baskin 1998).

Field capacity (FC) of the disturbed soil was determined in the laboratory by measuring the amount of soil wetted by 10 ml of water in a 24-hour period in 100 ml cylinder filled with soil (Tan 1996). The quantity of water was calculated to bring the air-dried soil to four gravimetric soil water contents representing different environmental conditions (Table 5.33).

Experiment 1

One kilogram of air-dried soil was placed in each of four thick, clear polyethylene bags. Water was added to the soils to bring each to one of the four FC conditions (Table 5.33). The bags were tied off with an elastic band and mixed frequently for five days in the laboratory to enable samples to reach equilibrium (Etherington and Evans 1986; Schütz *et al.* 2002).

Table 5.33. Four soil moisture treatments representing environmental conditions showing % field capacity, available water, and gravimetric water.

Field capacity %	Available water	Gravimetric water %	Water (ml) per 100 g air-dry soil	Environmental conditions
12.5	Below WP[a]	3.2	0.8	Dry surface soil below wilting point
35	WP[a]	9	6.6	Dry surface soils at wilting point
50	Available water	13	10.6	Soils drying from larger event or light rainfall conditions
100	Field capacity (unlimited)	26	23.6	Following large rainfall event

[a]WP= estimated wilting point for sandy clay soils (Salter and Williams 1967, 1969).

Two samples were then taken from each and the gravimetric water content calculated by oven drying at 105°C for 48 hours. Further water or soil was then added where required to bring soils to the desired water content.

Soil of known water content was then placed into Petri dishes (90 mm diameter, sterilised with a 5 % sodium hypochlorite solution) to fill to approximately 3 mm below the rim (approximately 60 g of air-dry soil plus water). Following the commonly used method described by Etherington and Evans (1986), fifty seeds were then placed on the soil surface, a polyethylene (PE) sheet was placed on top of the soil to minimise condensation and the lid was sealed with parafilm.

Four replicates of each soil water treatment were placed into a germination chamber with 12 hours alternating light/dark at one of three temperature treatments representing average day/night temperatures for winter (16/5°C), spring/autumn (23/10°C) and summer (31/16°C).

On the third day of the experiment, thermoregulation of the germination chamber under spring/autumn conditions failed, and the chamber heated to above 50°C. The spring/autumn treatment was ceased, but summer and winter treatments were continued. As germination response could not be reliably observed underneath the PE sheet, parafilm and Petri dish lid, germination was examined at the end of 35 days. Germination was considered to have occurred when emergence of the radicle from the testa was observed (Hartman *et al.* 1997). Total percentage germination was calculated. Observation of the Petri dishes after 35 days revealed significant drying of the soils under the high temperatures of the summer treatment.

A pilot study of three different methods for sealing Petri dishes was undertaken prior to the second experiment. Five Petri dishes containing moist soil were sealed with (i) parafilm; (ii) Vaseline and parafilm; and, (iii) Vaseline. Petri dishes were stored in a germination chamber under the summer temperature regime and weighed after one week and after three weeks. Based on the results of the pilot study, the experiment was undertaken a second time with a modified methodology.

Experiment 2

Petri dishes were sterilised with 5 % sodium hypochlorite and seeds were sterilised with 1 % sodium hypochlorite solution for 10 minutes (Mortlock 1999). Fifty-four grams of air-dry soil was placed into each Petri dish filled to approximately 3 mm below the rim. Fifty seeds were then placed on the soil surface and the required amount of water was added to bring the air-dry soil to 100 %, 50 %, 35 % or 12.5 % of FC (Table 5.33). A PE sheet was placed on top of the soil to minimise condensation on the lid. The lid was sealed with Vaseline and the entire unit was weighed.

Four replicates of each soil water treatment were placed in a germination chamber with 12 hours light/dark at one of three temperature treatments as above. Three times a week the Petri dish was weighed, and opened to count and remove germinants. Additional water was added when weight dropped by one gram or more. When water requirements allowed, fungal outbreaks were controlled with Benlate fungicide wettable powder applied as a solution of one gram powder to one litre of water (Turnbull and Doran 1987).

The experiment was ceased after 35 days. Four germination variables were calculated; (i) total germination percentage, (ii) germination rate, (iii) time (days) to first germination, and (iv) time (days) to 50 % germination where;

$$\text{percentage germination} = \frac{\sum n_x}{50} \times 100$$

$$\text{germination rate} = \frac{\sum (n_x \times t_x)}{\sum n_x}$$

Time to first germination is defined as the time from first exposure of the seed to moisture to emergence of the radicle through the testa, where n is the number of germinants at time t (days). It must be noted that seeds were not monitored daily, and so the daily rate is a conservative measure. It is possible that time to first germination, time to 50 % germination and germination rate may be slightly reduced if measured daily.

5.6.1.2 Early seedling growth and survival

One hundred and fifty plastic pots (50 x 50 x 120 mm) were filled with approximately 2 cm of perlite (exploded silicon). The pots were then filled with 220 g air-dry soil sieved, and oven dried as above (Section 5.6.1.1).

Three seeds were buried at a depth of 2 mm, approximately twice the seed diameter in the centre of each pot. Pots were placed within a glasshouse heated to maintain a minimum temperature of six degrees at the University of Ballarat in January 2003 and automatically watered three times daily for two minutes under a misting spray to allow seedlings to establish under conditions simulating those of a large rainfall event. Temperature and relative humidity were measured every two hours using a T-TEWC data logger. Mean temperature during the experiment was 17.9°C (7.5 to 37.7°C). Mean relative humidity was 79.7 % (23.7 to 101.3 %).

Germinants were monitored weekly, and after three weeks, seedlings had germinated in 60 pots. Where more than one seed germinated in a pot, the second or smaller seedling was cut off at the base. Seedlings were ordered by height and divided into two classes based on height (height class one 29–38 mm, class two 12–29 mm). Seedlings from each group were randomly allocated into one of four water treatments with a total of 15 replicates per treatment. The four treatments consisted of (i) pots maintained with unlimited water, (ii) 50 % FC, (iii) 35 % FC, and (iv) receiving no additional water.

The unlimited water content treatment continued to be watered under the automatic sprinkler system three times a day, seven days a week for the duration of the experiment. Soil water content was checked every two days by weighing pots with an electronic balance and adding the required amount of water to return pots to the desired weight (Table 5.33). Seedlings were monitored weekly for survival, shoot height, and branch count. The experiment was ceased after ten weeks after all seedlings receiving no further water had died. Determining seedling death, without re-hydrating seedlings proved difficult, and so seedlings were presumed dead when both the spike and cotyledons had turned brown and appeared desiccated.

Shoot height, number of branches, root length, relative growth rate, above, and below ground dry biomass were measured. Relative growth rate was calculated as:

$$\text{Relative growth rate} = \frac{\log(h_2) - \log(h_1)}{T_2 - T_1}$$

Where T = time and h =height

Analysis

Analysis was performed using SPSS (V10). Two-way ANOVAs were performed on all four seed variables to examine significant differences between temperature and soil water content treatments and investigate interaction between soil water content and temperature. One-way ANOVAs were used to examine significant differences between seedling variables under different soil water content treatments. Tukeys HSD post-hoc tests were performed to determine where differences between groups occurred.

To test for conformation to assumptions of the ANOVA, normality was checked using Kolmogorov-Smironv and equality of variance using Levenes test. To normalise data and standardise variance, germination rate, seedling height and above ground biomass data were transformed using $\log_{10}$ transformations. Inequality of variance for seedling relative growth rates was stabilised using arcsine transformations (=Arsine(sqrt(x/100)) where x is the variable to be transformed. Untransformed values are presented for all mean and standard error values. Data for percent germination, days to first germination, 50 % germination and number of branches were not normally distributed and were not able to be transformed using log, square root or arcsine transformations. Analyses of variance are relatively robust to inequality of variance and are quite robust to violations of the assumptions of normality, even when sample sizes are small and so were performed (Underwood 1997).

To investigate potential errors caused by non-normality, non-parametric Kruskal-Wallis analyses were performed to check results of the ANOVA where inequality of variance or non-normality was found. As similar results were obtained for both parametric and non-parametric methods the ANOVA, enabling investigation of interactions between water and temperature was chosen.

5.6.2 Results

5.6.2.1 Seed germination

Total seed viability of *C. pauper* seed, as determined under dissecting microscope was 64 %. Percentage seed germination for the first experiment is presented in Table 5.34. As water evaporated throughout the experiment, soil water values represent only initial soil water, with final soil water content approaching that of air-dry soil.

Parafilm exposed to Vaseline perished within 24 hours when placed in the germination chamber representing summer conditions. Mean water loss over 22 days for Petri dishes sealed with Vaseline was 0.81 g (SE 0.17), and for those sealed with Parafilm 5.35 g (SE 2.33).

Table 5.34. Mean of *Casuarina pauper* percentage seed germination (Field capacity represents levels at commencement of the experiment.

Field capacity (%)	Summer %	Winter %
100.0	25.5	18.0
50.0	0.5	18.0
35.0	0.0	1.5
12.5	0.0	0.0

In the second seed germination experiment, seeds germinated in all temperature treatments and at all soil water levels except for the 12.5 % FC treatment where no germinants were recorded under any temperature treatment.

Percentage germination

A two-way between-groups analysis of variance was conducted to explore the effects of temperature and soil water content on total percentage germination. A significant main effect was found for soil water content (F=10.342, df 2, $p<0.001$) with a large effect size (eta squared = 0.434). A similar result was obtained with a Kruskal-Wallis non-parametric test (Chi-Square=13.572, df 2, $p<0.001$).

Post hoc tests revealed significantly lower total germination percentage at 35 % FC than either 100 % or 50 % FC ($p<0.01$). No significant difference was observed in percentage germination between 50 % and 100 % FC ($p=0.903$) (Figure 5.27).

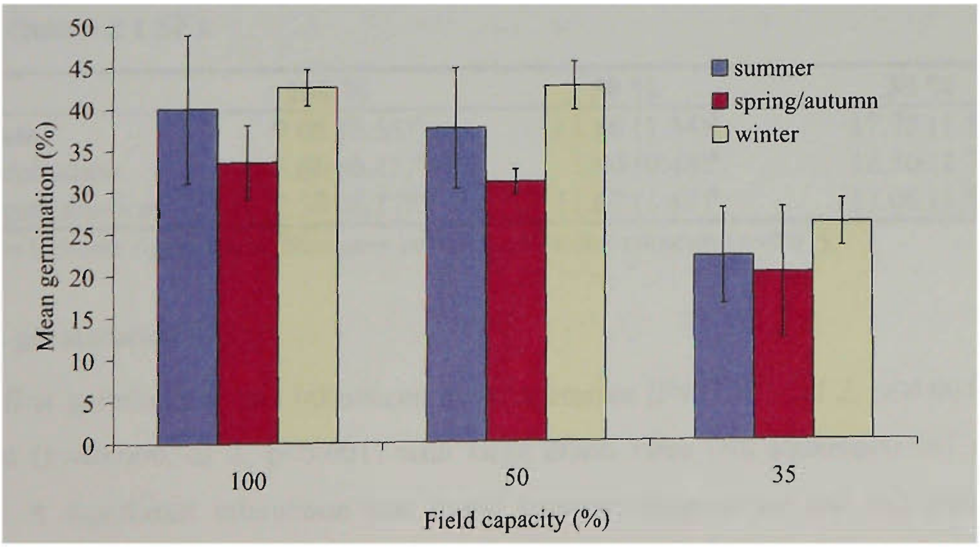


Figure 5.27. Mean (error bars show ± 1 SE) percentage germination of *Casuarina pauper* seed for summer, spring/autumn and winter temperatures at different soil water contents.

No significant differences were observed in overall percentage germination at any temperature ($F=2.623$, df 2, $p=0.091$). No significant interaction between temperature and water was found ($F=0.118$, df 4, $p=0.975$). A similar result was obtained with a Kruskal-Wallis non-parametric test (Chi-Square=3.502, df 2, $p=0.174$).

Germination rate

A two-way ANOVA was conducted to explore the effects of temperature and soil water content on germination rate. Significant main effects were found for temperature ($F=29.29$, df 2, $p<0.001$) and water content ($F=38.79$, df 2, $p<0.001$) with large effect sizes (eta squared = 0.68, and 0.74 respectively). No interaction between temperature and water was found ($F=2.27$, df 4, $p=0.08$).

Tukey HSD post hoc tests revealed a significantly greater germination rate under summer than under winter or spring/autumn temperatures ($p<0.001$). No significant difference in germination rate was observed between spring/autumn and winter temperatures ($p=0.435$) (Table 5.35). Germination rate increased as soil water content increased with significant differences between all treatment levels (Table 5.36).

Table 5.35. Effects of temperature on days to first germination, days to 50 % germination and germination rate of *Casuarina pauper* seed (mean ± 1 SE).

Germination	Summer	Spring/autumn	Winter
Germination rate	9.10 (0.79)[a]	16.01 (1.72)[b]	14.39 (1.55)[b]
Days to 1st germination	5.33 (0.47)[a]	9.75 (1.68)[b]	10.08 (1.28)[b]
Days to 50 % germination	8.58 (0.85)[a]	14.83 (1.73)[b]	12.83 (1.84)[b]

Different letters indicate significant differences between temperatures ($p<0.01$).

Table 5.36. Effects of soil water content (% of field capacity) on days to first germination, days to 50 % germination and germination rate of *Casuarina pauper* seed (mean ± 1 SE).

Germination	100 %	50 %	35 %
Germination rate	9.05 (0.65)[a]	12.66 (1.34)[b]	17.78 (1.58)[c]
Days to 1st germination	5.66 (0.43)[a]	7.00 (0.48)[a]	12.50 (1.75)[b]
Days to 50 % germination	7.50 (0.27)[a]	11.67 (1.42)[b]	17.08 (1.58)[c]

Different letters indicate significant differences between soil water contents ($p<0.01$).

Days to first germination

The time to first germination was influenced by temperature ($F=17.675$, df 2, $p<0.001$) and soil water content ($F=40.060$, df 2, $p<0.001$) with large effect sizes (eta squared=0.567 and 0.748 respectively). A significant interaction was found between temperature and soil water content ($F=4.386$, df 4, $p<0.01$).

Post hoc tests revealed significantly faster germination response ($p<0.001$) under summer temperatures, compared with both spring/autumn and winter temperatures (Table 5.35). No significant difference was found in time to first germination between spring/autumn and winter temperatures ($p=0.169$). More rapid germination ($p<0.01$) was observed under increasing soil water content (Table 5.36).

To verify results of the two-way ANOVA, Kruskal Wallis tests were performed on temperature (Chi-Square=9.778, df 2, $p<0.01$), and soil water content data (Chi-Square=18.284, df 2, $p<0.001$). The significant result with alpha levels less than 0.01 corresponds to the results of the two-way ANOVA, suggesting that violations of the assumptions of normality and equality of variance have not influenced results.

Days to 50 % germination

Significant main effects were observed for temperature ($F=11.533$, df 2, $p<0.001$) and soil water content ($F=21.524$, df 2, $p<0.001$) on time to 50 % germination with large effect sizes (Eta squared =0.461 and 0.615 respectively). No significant interaction between water and temperature was found ($F=1.89$, df 4, $P=0.140$).

Post hoc tests revealed 50 % germination was achieved significantly faster under summer temperatures than spring/autumn or winter temperatures ($p<0.001$). No significant differences were observed between spring/autumn and winter temperatures ($p=0.951$) (Table 5.35). Soil water content of 35 % FC was found to result in a significantly longer time to 50 % germination than either 50 % or 100 % FC ($p<0.001$) (Table 5.36).

To verify results of the two-way ANOVA, Kruskal Wallis tests were performed on temperature (Chi-Square=14.801, df=2, $p<0.001$), and soil water content data (Chi-Square=15.322, df 2, $p<0.001$). The significant result with alpha levels less than 0.001 corresponds closely to the results of the two-way ANOVA, suggesting that violations of the assumptions have not influenced results.

5.6.2.2 Early seedling growth and survival

By week three, seedlings had emerged in 60 pots. Germination rate was most rapid in the first two weeks, and seeds continued to germinate through to week seven in all treatments despite reducing soil water levels (Figure 5.28). Within one to two weeks of germination, emergence of a central spikelet was observed between the two cotyledons. From two weeks, branching was observed along the central spikelet. Water treatments were commenced at week three, but under the humid conditions of the glasshouse, soil water content did not reach the desired levels until four weeks after treatments commenced (Figure 5.29).

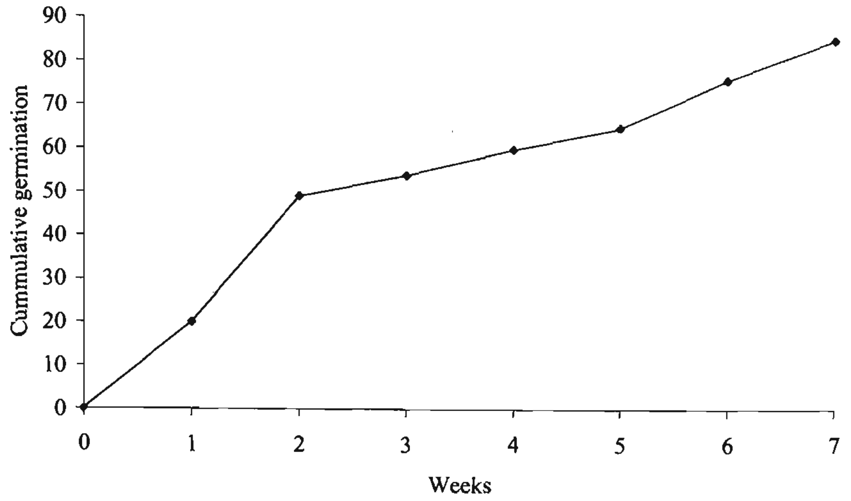


Figure 5.28. Cumulative germination of *Casuarina pauper* seeds in the glasshouse from week one to seven.

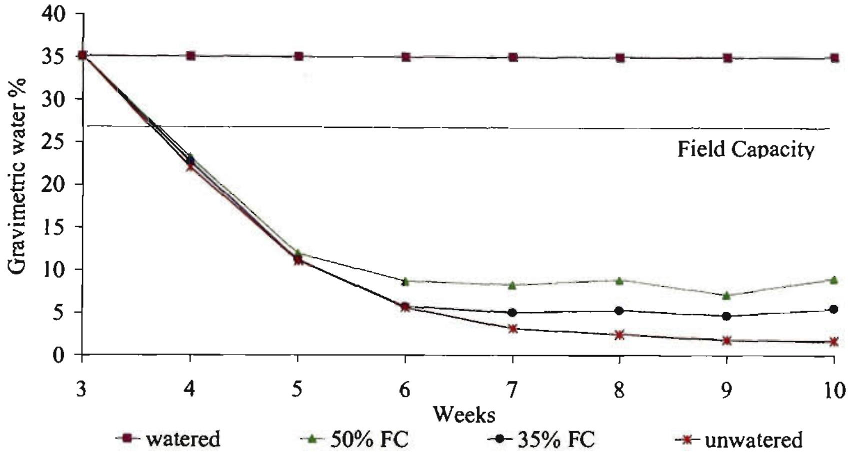


Figure 5.29. Minimum weekly soil water content of each treatment, prior to the addition of water.

Relative growth rate

Figure 5.30 shows the reduction in relative growth rate as soil water treatments are implemented. A rapid reduction in growth rate is observed in all treatments after week four, with growth rate in unwatered treatments continuing to drop rapidly from weeks four to seven. A levelling out of decline in relative growth rate is observed at around week seven.

No significant differences were observed in relative growth rates in weeks three to four (F=0.21, df 3, p=0.889), four to five (F=1.352, df 3, p=0.267) or five to six (F=0.995, df 3, p=0.402). Relative growth rate between weeks six and seven was significantly different between treatments (F=5.721, df 3, p<0.01). Tukeys post hoc tests revealed that the unwatered treatment had a significantly lower growth rate than all other treatments (p<0.01). A continuation of this pattern was found in weeks seven to eight (F=14.729, df 3, p<0.001) and eight to nine (F=6.075, df 3, p<0.001). By weeks nine to 10, relative growth rate was comparably low in all treatments (F=2.133, df 3, p=0.107).

Seedling characteristics at experiment termination (10 weeks)

By nine weeks, unwatered seedlings were beginning to wilt, and by the end of 10 weeks, all unwatered seedlings were significantly desiccated or dead (Figure 5.31).

Significant differences in final seedling height (F=5.798, df 3, p<0.01) and above ground biomass (F=3.752, df 3, p<0.05) were observed. Unwatered seedlings were significantly shorter than seedlings maintained at 50 % FC (p<0.01) (Figure 5.32) and had significantly lower above ground biomass (Table 5.37) than those receiving daily watering or maintained at 50 % FC (p<0.05).

No significant differences were detected in below ground biomass (F=0.767, df 3, p=0.518), and all roots had grown to the bottom of the pot, suggesting root length was limited by pot size (Table 5.37).

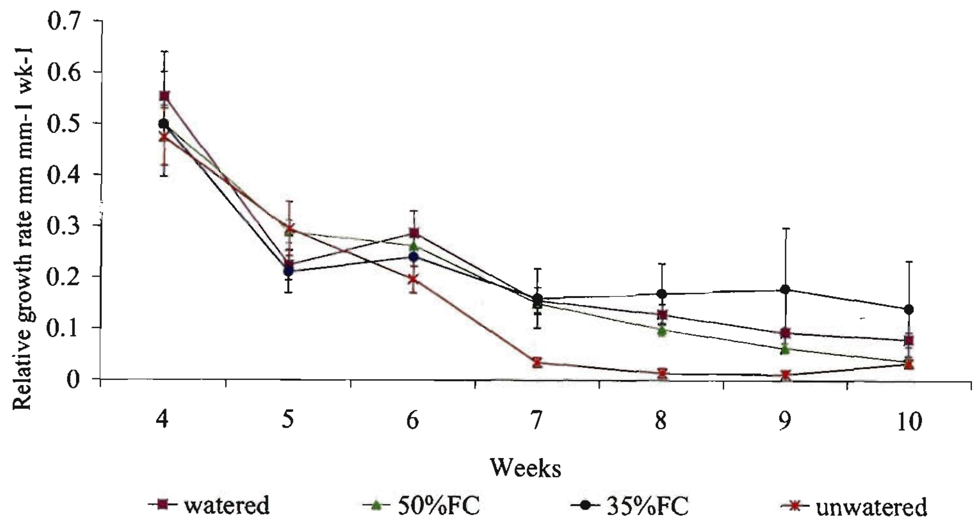


Figure 5.30. Relative weekly growth rate of *Casuarina pauper* seedlings under different water regimes from weeks 3–10 (error bars show ± 1 SE).

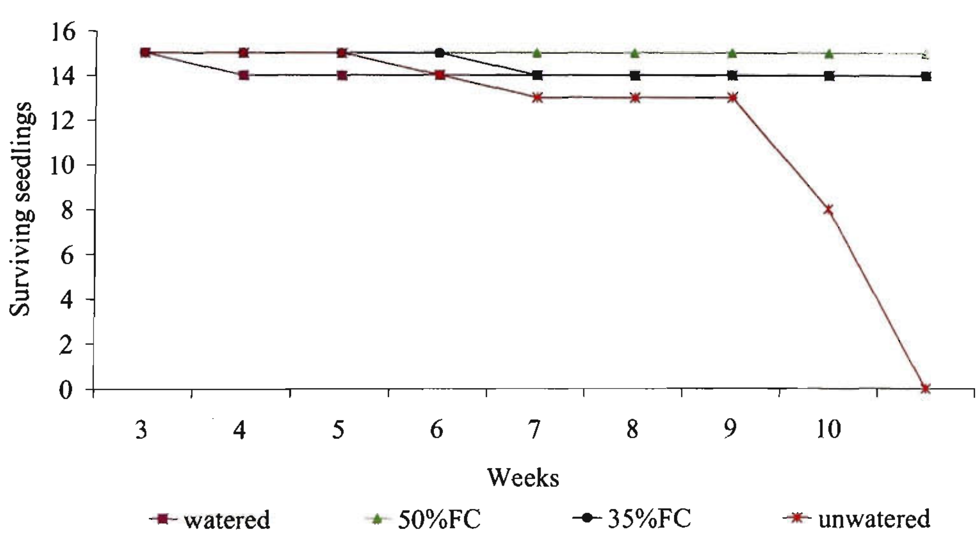


Figure 5.31. Survivorship of *Casuarina pauper* seedlings under different water regimes from weeks 3–10.

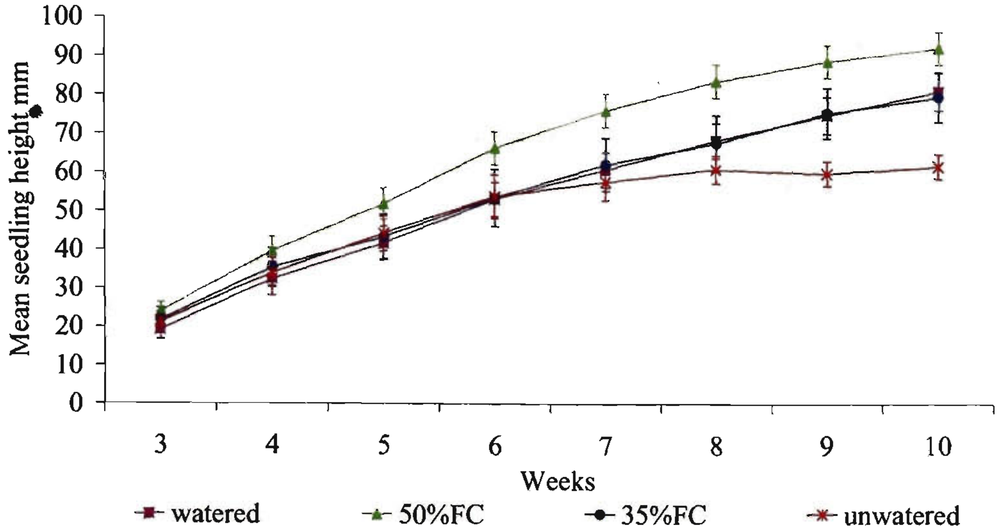


Figure 5.32. Mean *Casuarina pauper* seedling height under different water regimes from weeks 3–10 (error bars show ± 1 SE).

Table 5.37. Mean (± 1 SE) of above and below ground seedling dry biomass and the number of branches of *Casuarina pauper* seedlings.

	Above ground biomass (mg)		Below ground biomass (mg)		No. branches	
	Mean	**SE**	**Mean**	**SE**	**Mean**	**SE**
Watered	11.89	1.367	8.179	1.695	3.000	0.574
50 % FC	12.68	1.337	8.180	1.275	2.200	0.222
35 % FC	10.19	1.319	5.750	0.799	1.929	0.355
Unwatered	6.24	0.649	7.492	1.096	1.000	0.363

Branching had occurred in all individuals from the 50 % FC treatment by the end of ten weeks. In 23 % of individuals from all other soil water treatments, branching did not occur during the ten weeks of the experiment. By the end of the experiment, the mean number of branches was 2.04, with a maximum of six branches. A significant difference in the number of branches at the end of 10 weeks was detected ($F=3.279$, df 3, $p<0.05$), and this result was confirmed with a Kruskal Wallis non-parametric test (Chi-square=7.908, df 3, $p<0.05$). Tukeys post-hoc tests reveal significantly fewer branches in the unwatered treatment, compared with the continuously watered treatment ($p<0.01$) (Table 5.37).

5.6.3 Discussion

5.6.3.1 Seed germination

Results from the germination experiment show that germination of *C. pauper* seed occurs readily under temperatures in the range of mean seasonal temperatures for the study area. This indicates that germination could occur at any time that sufficient moisture was available. Total germination percentage was reduced by low soil water (FC 35 % or less), supporting the original hypothesis that greater percentage germination will result from increased soil water content.

Whilst low water potential of the dry seed will enable imbibition at relatively low soil matric potentials (Hartman *et al.* 1997), at 12.5 % FC, imbibition is not sufficient to initiate radicle emergence. At 35 % FC germination percentage was reduced, but all levels above this resulted in similar percentage germination. A similar response of high germinability over a range of soil water contents has been reported for a number of studies on of arid and semi-arid perennial species (Wilson and Witowski 1998; Flores and Briones 2001; Villalobos and Peláez 2001).

The main response to temperature was speed of germination, with significantly faster time to first germination and to 50 % germination under mean summer temperatures. Reduced time to first germination and to 50 % germination was also found with increasing soil water content. This was also reflected in an increase in germination rate for summer temperatures with increased soil water content.

Throughout its distribution, *C. pauper* occurs largely in semi-arid regions where rainfall is highly variable, and only weakly seasonal (Colls and Whitaker 1990). Spatial and temporal variability in rainfall require adaptations to enable germination under a range of temperatures whenever sufficient rainfall occurs. A faster response to summer temperatures would enable *C. pauper* greater chance of success during summer rainfall events. The effect of summer rains is more quickly negated by the high summer evaporation rates, exacerbated by poor surface water holding capabilities of sandy soils (Charman and Murphy 2000). Rapid germination response, followed by root elongation into the deeper soil horizons may enable successful survival under summer conditions.

5.6.3.2 Early seedling growth and subsequent survival

All seedlings died when gravimetric soil water content dropped below 2 % (near air-dry). It is possible that small *C. pauper* seedlings may survive brief reductions in soil water to these levels, but it appears unlikely that seedlings under two months will tolerate extended periods of near air-dry soils. Soil moisture levels measured in the study area were commonly at near air-dry levels (Appendix 9), therefore, for seedlings to successfully establish, it is likely that an extended period of rainfall is required.

Humidity in northwest Victoria is much lower than that recorded in the glasshouse, however, potential factors mitigating seedling survival in the field include dew, and the ability for roots to elongate to reach deeper moister soils. Root elongation is likely to be a critical factor in seedling survival. Average gravimetric soil water in the B21 horizon in November 2000 was found to be 3.82 % higher than in the A1 horizon, and 2.54 % higher in the B22 horizon than the B21 horizon (Table 5.21). These relatively small differences in soil water are likely to have a major effect on seedling survival.

Relative growth rates were significantly lower for the unwatered treatment after soil gravimetric water reached approximately 3.2 %. However, no significant differences in relative growth rates were found between any of the watered treatments over the seven weeks. Growth rates in the watered treatments declined substantially over the course of the experiment, for all treatments. This may have been due to pot depth limiting root elongation or lack of soil nutrients after initial seed reserves were exhausted. Growth rates in the daily watered treatment may also have been limited by water logging. When the soil becomes waterlogged, oxygen in the soil is replaced by water. When oxygen availability becomes a limiting factor in the soil, physiological activity will be reduced placing the plants under severe water stress (Kramer and Boyer 1995).

Root length was limited by pot size, and roots of all seedlings extended to 100 mm (full length of pot) over 10 weeks. No significant differences were detected in below-ground biomass, possibly due to limitation in pot size.

Seedling height and above ground biomass was significantly reduced in seedlings that did not receive additional water. No significant differences were observed between any of the watered treatments, however, seedlings watered daily were found to have a slightly decreased height, and relative growth rate compared with seedlings grown under 50 % FC conditions. Seedling growth and survival would be expected to decrease as soil water content increases above 100 % FC due to water logging (Figure 5.33).

Limitations

Inability to replicate field conditions is a universal problem of laboratory studies and a few issues were encountered during this study. In particular, root growth was limited by pot size, possibly influencing overall seedling growth, with relative growth rates approaching zero by the end of the experiment. High humidity in the glasshouse contrasted with the very low humidity of the study area.

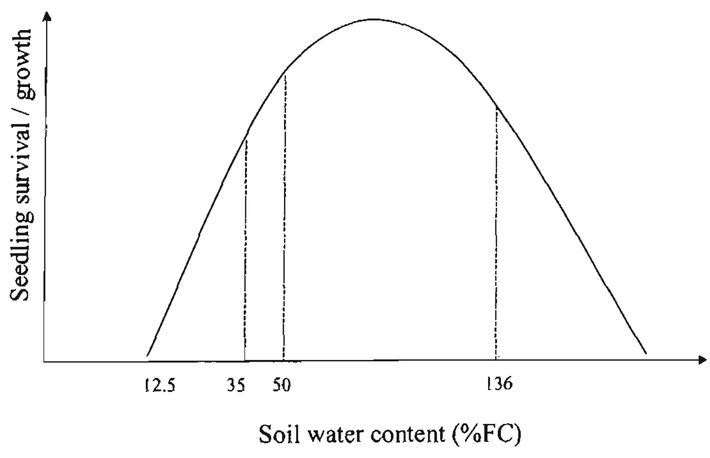


Figure 5.33. Predicted *Casuarina pauper* seedling survival (or growth) under increasing soil water content.

Implications for field study

Results of *C. pauper* germination under different soil moisture conditions suggest that the watering treatment raised soil moisture levels sufficiently to allow germination. However, the time of watering in the field (three to seven days), would have been sufficient time for some germination of *C. pauper* seed under summer conditions, but was probably not sufficient for germination at other times (Table 5.35).

These results also suggest that if any seed germinated, low soil moisture between watering treatments (Appendix 9) would have led to death of any *C. pauper* seedlings that germinated, except perhaps in winter. Seedling survival was not tested at 4 %, but little mortality occurred in the glasshouse when soil moisture content reached 4 % (Figure 5.29), suggesting that seedlings are likely to survive soil water conditions recorded during winter. Increasing soil moisture content at the B21 horizon may mean a greater chance of survival as roots penetrate deeper into the soil.

5.7 SUMMARY OF REGENERATION CHARACTERISTICS OF OVERSTOREY SPECIES

ALECTRYON OLEIFOLIUS

Extent of regeneration

A. oleifolius has exhibited the most extensive regeneration response of all woodland trees to decreased grazing pressure following creation of the MSNP and decreased rabbit numbers (Table 5.4). However, regeneration appears to be occurring almost entirely by suckering from root disturbance with no juveniles observed believed to be seedlings observed in the current survey. It appears that most regeneration since European settlement in the study area has occurred in the last 30 years, and much of this is likely to have occurred since the early 1990s.

Suckering

Suckering of *A. oleifolius* in the MSNP has been observed following fire, road grading, creation of firebreaks, ripping of rabbit warrens, and regeneration from cut stumps (Westbrooke 1998 and personal observations). This is consistent with previous studies on regeneration of *A. oleifolius* across south-eastern Australia (Hall *et al.* 1964; Chesterfield and Parsons 1985; Wisniewski and Parsons 1986).

The rainfall requirements for sucker regeneration are unknown, but are likely to depend upon the suckering mechanism. It is likely that suckering from distal detached root fragments will have high water requirements, but those still attached to the parent plant may be able to draw on resources from the parent plant .

Predictors of sites where regeneration may occur

Presence of *A. oleifolius* regeneration is correlated to the number of mature *A. oleifolius* trees, highlighting the need for a supply of propagules for regeneration to occur (Table 5.6). A weak negative relationship was found between regeneration of *A. oleifolius* and cryptogamic cover, which is thought to be due to the requirement for soil disturbance to promote root suckering. A weak positive correlation was found between exotic ground cover and *A. oleifolius* regeneration, which may also be related to the effects of soil disturbance, which commonly favours establishment of exotic weedy species. It must be noted that this refers to regeneration by suckering, as no evidence of seedling recruitment was observed.

Seed availability and viability

Whilst seed production was observed in all years (Section 5.4.1), no viable seeds were found in the soil seedbank in May 2002. Westbrooke (1998) similarly found no germination occurred for 16 month old seed collected from litter.

Very low viability was found for seeds collected in summer 2001 with only one seed found to contain an intact embryo (Table 5.26). The embryos of all other *A. oleifolius* seeds appeared dried and shrunken, and it was not possible to determine if complete development of the embryo had occurred. Similarly, Wisniewski and Parsons (1986) found few filled seed in 1985 with less than 10 % filled seed at most sites. They found higher percentage of filled seeds in recently opened fruits in 1986 with 88 to 93 % germination found in nicked seed germinated under continuous light and continuous darkness respectively (Wisniewski and Parsons 1986). Germination trials by Westbrooke (1998) confirmed the hard seededness of *A. oleifolius* seed, with 12 % of seed germinating without seed treatment, increasing to 25 % following scarification.

Optimal germination rates have been found at high temperatures from 30–35°C, and Burbidge (1960) suggested that this provides some evidence that *A. oleifolius* may have originated in, or developed from elements from tropical climates, explaining the reliance on vegetative reproduction in the more southerly areas of its distribution.

Spasmodic flowering, incomplete seed development, rapid decline of viability and seed damage by insects may also be responsible for variable *A. oleifolius* seed viability (Wisniewski and Parsons 1986). Seed set is highly variable with no viable seeds in some years, and heavy seed set observed at times (Burbidge 1960; Hall *et al.* 1964; Westbrooke 1998). Low dispersal is observed for *A. oleifolius* seed (Chesterfield and Parsons 1985).

In some species, seed production continues despite vegetative spread being the major form of reproduction. Extensive seed production in certain years can enable the benefits of sexual reproduction during periods of ameliorated conditions. Alternatively, vegetative reproduction may be the only possible method of species survival for relict communities where the persistence of a species has continued despite climatic changes which prevent successful sexual reproduction (Lacey and Johnston 1990). Given the extreme rarity of *A. oleifolius* seedlings, despite investigation by a number of researchers, it appears probable that *A. oleifolius* may represent a relict species where conditions for seedling recruitment are no longer met due to climatic changes.

CALLITRIS GRACILIS REGENERATION

Extent of regeneration

C. gracilis regeneration (any trees estimated to have established in the last 30 years) was observed at 9 % of sites where mature *C. gracilis* were present. Regeneration of *C. gracilis* was observed at a few sites in the Taparoo area, although it has been reported that this cohort has been severely impacted by grazing pressure (Sandell *et al.* 2002). Regeneration of *C. gracilis* has occurred at Yarrara Flora and Fauna reserve, particularly along constructed water channels where the combination of soil disturbance and flooding in a low grazing pressure environment has led to establishment of multiple aged stands (Plate 5.7.). *C. gracilis* regeneration was also observed along some road reserves in the study area (personal observation).

Plate 5.7. Four size classes of *Callitris gracilis* regeneration along an irrigation channel at Yarrara Flora and Fauna reserve (March 2003).

Regeneration requirements

Zimmer (1944) examined *C. gracilis* regeneration at Yarrara between 1933 to 1938 within a one acre harrowed (surface soil broken into a fine condition) grazing exclosure, and found successful germination resulted in 1933, 1934, 1935 and 1937. All seedlings survived until final observation in April 1938, except those germinated in 1933. Based on the years when successful germination occurred, three rainfall criteria were derived; (1) summer rainfall greater than 36 mm, (2) rainfall event greater than 50 mm in any month between May to November, (3) annual rainfall greater than 205 mm. These were the most stringent criteria that still included all years that Zimmer (1944) reported successful *C. gracilis* regeneration.

When these rules were applied to the Werrimull rainfall data (nearest rainfall station to Yarrara), 24 years out of the last 75 years between 1927–2002 were identified where regeneration could have occurred: 1930, 1931, 1934, 1936, 1937, 1951, 1954, 1958, 1959, 1961, 1963, 1970, 1973, 1975, 1978, 1983, 1985, 1986, 1987, 1992, 1993, 1995, 1996, 1997. This includes known regeneration events such as 1951, 1973, and 1992/3 (Onans and Parsons 1980).

Obviously, recruitment has not occurred at this frequency, and few germinants have been observed. Grazing is thought to be the major factor limiting recruitment, potentially eliminating all recruits before they are observed. It is also possible that recruitment will only occur at this frequency in certain sites, such as within existing woodland with a significantly different microclimate to open degraded sites. Another possible explanation is that many years of grazing pressure and associated degradation and erosion of soils and plant cover may have substantially altered sites such that higher rainfall conditions are required for regeneration to occur.

The Taparoo cohort was observed to regenerate during above average rainfall in 1992/3. The nearest rainfall station is at Meringur, with monthly rainfall figures indicating well above average falls from late winter to early summer for both 1992 and 1993 (Figure 5.34). High January rainfall in 1993 may have assisted summer survival of seedlings as good summer rainfall has been shown to be crucial for *C. gracilis* and *C. glaucophylla* regeneration (Lacey 1972).

Competition

Whilst competition effects were not investigated in the current study, it has been found that competition with existing mature *C. glaucophylla* prevents successful establishment of germinants up to 12 m from mature trees (Zimmer 1944; Read 1995).

For survival of *Callitris* seedlings, evidence suggests that competition from an herbaceous, or grassy understorey does not effect seedling survival, and in fact, increased grass cover may improve seedling survival by ameliorating conditions such as soil surface temperature and soil water loss (Zimmer 1944; Johnston 1968).

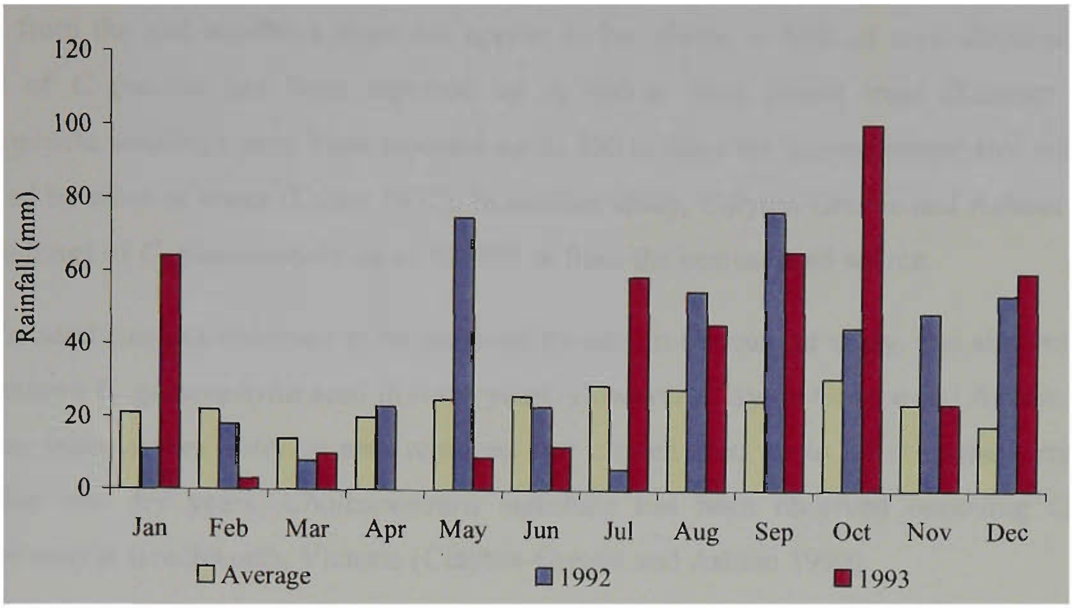


Figure 5.34. Monthly rainfall at Meringur in 1992 and 1993 compared with the long-term average.

Predictors of sites where regeneration may occur

C. gracilis regeneration was found to occur on higher condition sites indicated by a higher percentage cover and species richness of tall shrubs, and percentage cover of trees (Table 5.6). The discriminant analysis also found *C. gracilis* recruitment occurring at sites with reduced cover of cryptogams, supporting findings of previous research for *C. gracilis* and *C. glaucophylla* suggesting removal of the cryptogamic layer is essential for regeneration (Zimmer 1944; Johnston 1968).

Seed availability and viability

Seed viability of *C. gracilis* seed collected in summer 2000 was 20 %, which supports the findings of much earlier research by Zimmer, who found average viability of was approximately one third (1942). Viability of the closely related *C. glaucophylla* is also around this figure, with viability figures from previous studies including: 26 % (Clayton-Greene and Ashton 1990), 44–72 % (Lacey 1972) and 35 % (Westbrooke 1998).

High temperature has been observed to inhibit germination of *C. gracilis* seed, with a greater percentage of germinants emerging as weather conditions become cooler (Langdon T [Mildura Native Nursery] 2002 pers. com., 5 February). High temperature has also been reported to have an inhibitory effect on germination for *C. glaucophylla*, with percentage germination reduced from 95 % at 20°C to 9 % at 25°C (Lacey 1972).

No *C. gracilis* seed geminated from the soil seedbank, which supports finding of previous studies that *C. gracilis* does not form a soil seedbank (Lacey 1972; Clayton-Greene and Ashton 1990).

Absence from the soil seedbank does not appear to be related to lack of seed dispersal. Seed dispersal of *C. gracilis* has been reported up to 480 m from parent trees (Zimmer 1942). *C. glaucophylla* seedlings have been reported up to 350 m from the nearest parent tree with seed transported by wind or water (Lacey 1972). In another study, Clayton-Greene and Ashton (1990) found seedlings of *C. glaucophylla* up to 50–100 m from the nearest seed source.

C. gracilis seed was not observed to be removed by ants in the current study, and similarly, ants did not remove *C. glaucophylla* seed in some years. However, Clayton-Greene and Ashton (1990) found that *Iridomyrmex detectus* ants removed and carried seed up to 20 m in the summer of 1980, after two dry years. *Chalcopeneura metallica* has been observed removing *Callitris endlicheri* seed at Beechworth, Victoria (Clayton-Greene and Ashton 1990).

Despite the almost complete absence of regeneration of *C. gracilis* in northwest Victoria, and *C. glaucophylla* in many areas of western NSW, in higher rainfall areas (rainfall greater than 500 mm) of NSW such as Forbes, Dubbo, Baradine, Pilliga and Narrabri there has been extensive regeneration. Regeneration has been so great in some regions that *C. glaucophylla* has been described as a woody weed, and numerous strategies to thin or eliminate regeneration have been trialed .

CASUARINA PAUPER REGENERATION

Extent of regeneration

Less than 8 % of *C. pauper* individuals recorded across the MSNP were estimated to represent regeneration in the last 30 years. Of the juvenile *C. pauper* found, all were believed to be of sucker origin due to close proximity to existing trees, and frequently obvious exposed root stock (Plate 5.1). Infrequent *C. pauper* regeneration was noted in good condition remnants across the study area.

Predictors of sites where regeneration may occur

Sites where regeneration was found had a higher number of mature *C. pauper* and other tree species present. These sites also had high species richness of trees, tall shrubs and small shrubs. *C. pauper* regeneration was observed mainly in good condition sites, indicative of a history of low grazing pressure and other disturbances. There were insufficient sites of *C. pauper* regeneration to enable further analysis of regeneration requirements.

Seed availability and viability

High *C. pauper* seed production in most years results in abundant seed in soil and litter within *C. pauper* remnants. Whilst the soil seed store is transient due to lack of seed dormancy, it is replenished by high seed production most years (Section 5.4.1). Seed was observed to be released from the cones in late summer and early autumn where it is vulnerable to predation by ants, birds, and other granivores (Section 5.4.2). Ant seed harvesting rates are high at times , but were found to reduce over winter (Table 5.24), and did not appear to significantly reduce soil seed stores. These results support the findings of previous research of *C. pauper* seed production (Chesterfield and Parsons 1985; Westbrooke 1998), and soil seedbanks (Chesterfield and Parsons 1985; Auld 1995b).

Total seed viability determined by the cut test was 72 %, but lower germination rates (max 30 %) were found in the germination chamber under all treatments (Figure 5.26). In an earlier study of *C. pauper* seed viability, Westbrooke (1998) found 53 % germination for four month old seed. Given high seed production most years, good seed viability, and lack of dormancy mechanisms, seed is unlikely to be a limiting factor in *C. pauper* regeneration.

Regeneration requirements

Seed was found to germinate readily under all temperature regimes tested, suggesting that germination will occur at any time of year, provided sufficient soil moisture is available (Figure 5.27). Time to first germination was significantly faster under summer temperatures, which may assist in more rapid seedling establishment following large summer rainfall events. It is probable that germination of *C. pauper* seed will occur most years.

Seedlings did not survive very low soil moisture conditions (>2 % gravimetric soil water), and as near air-dry soils were recorded at most times except winter (Appendix 9), this is likely to pose a significant limitation to seedling establishment. No seedlings planted in the field survived the first summer, and it is likely that this is a common occurrence. Research has found *Callitris glaucophylla* seedlings (Zimmer 1944; Lacey 1972; Read 1995), require above average rainfall over summer to enable seedlings to survive the first summer, and this is likely to be similar in *C. pauper*.

Regeneration is more commonly observed to be of sucker origin, stimulated by damage to exposed lateral roots. Further research is required to determine the processes of sucker establishment and growth.

MYOPORUM PLATYCARPUM REGENERATION

Extent of regeneration

M. platycarpum is the only tree species for which seed regeneration appears to be occurring across the landscape. However, almost half the sites surveyed did not show any evidence of recruitment over the last 125 years. More recent regeneration (within the last 30 years) is estimated to have occurred at 37 % of sites, but represents nearly 48 % of trees surveyed, as commonly, multiple recruits occur at a site. Despite more regeneration occurring for *M. platycarpum* than many other semi-arid tree species, there is still a need for further regeneration events. Many trees are now reaching senescence, and for an adequate seed source to be present, it is crucial that a successful regeneration event occur within the next 15 to 20 years (Westbrooke 1998).

Predictors of sites where regeneration may occur

M. platycarpum juveniles were found at sites with higher numbers of mature *M. platycarpum*. Sites where juveniles were found were generally less diverse, and somewhat degraded with low ground species richness and high exotic ground cover. Similarly, high total ground cover and low tall shrub species richness were isolated as predictors of sites where regeneration of *M. platycarpum* is likely to occur (Table 5.8).

There is no evidence that soil disturbance, fire or other requirements beyond sufficient rainfall and seed source are necessary for *M. platycarpum* regeneration. It is likely, however, that soil disturbance, creating a greater number of safe sites may be required for mass regeneration events.

Seed availability and viability

Previous research has found *M. platycarpum* fruits frequently do not contain viable seed (Chesterfield and Parsons 1985), however, in the current study, 64 % of seeds appeared viable under the cut test. This was significantly reduced when seed was tested in the germination chamber, with only 14 % of seed germinating (Figure 5.26). There was no evidence of seed dormancy, with little difference found between seed pre-treatments.

Harvesting of *M. platycarpum* seed by ants was measured at relatively low rates (4.5–11 %), however, only very low soil seed stores of *M. platycarpum* (5.8 m^{-2}) were observed at one out of three sites sampled in July 2002 (Appendix 10). Insufficient evidence exists to determine the impact of this seed harvesting on abundance of *M. platycarpum* in the soil seedbank. However, it would appear that *M. platycarpum* has only a transient soil seed-store, replenished by heavy seed production in some years.

5.8 CONCLUSION

To gain a greater understanding of the factors limiting vegetation condition, regeneration requirements were investigated for a number of perennial species commonly occurring within *C. pauper* woodlands. A field study of regeneration was carried out within a large (6 ha) grazing exclosure, and a number of laboratory experiments were also performed.

It was found that the watering treatment within the field experiment was unable to replicate the soil moisture conditions of event years, and therefore was largely unsuccessful in triggering a "regeneration event". There may be many reasons why the watering treatment failed to replicate conditions of event years, however, the major problem is thought to be related to landscape scale re-distribution of water.

Following large rainfall events, water run-off and run-on zones result in unequal distribution of water, and hence soil moisture (Noy-Meir 1985; Noble *et al.* 1998). Therefore, only some sites will receive effective rainfall, and provide suitable sites for seed germination and seedling establishment. This could not be replicated within the watering treatment. Additional problems included high evaporation, short periods of watering and long periods without watering.

However, some perennial species did germinate following the watering treatment, and results from these species suggested that watering and seed application had a significant effect on regeneration. No significant effects of soil disturbance or fire were found, however, a non-significant increase in germination following soil disturbance was found, suggesting that further investigation of the effects of soil disturbance may be valuable.

Across the study area, relatively low levels of regeneration were found for most tree species, however, there is evidence of increased regeneration rates following the creation of the MSNP. Regeneration was found to be distributed across the study area, suggesting that thresholds for regeneration have not been exceeded at the broad-scale. Regeneration of woodland trees was positively correlated with the number of mature trees of the species for all species investigated highlighting the importance of a propagule source. Regeneration of *Casuarina pauper* and *Callitris gracilis* was found to be occurring at the less disturbed sites, while evidence of soil disturbance was found to be correlated with regeneration of *A. oleifolius* and *M. platycarpum*.

Few germinants of most perennial species resulted from soil seedbank samples, with the exception of *C. pauper* where abundant seed was found within both soil and litter samples. Data suggest that soil seedbanks of most perennial species are transient, and low abundance of seed is likely in remnants where low shrub abundance has resulted from past grazing.

Given the abundance of *C. pauper* seed, and relatively low soil moisture requirements for germination, it is likely that germination will occur most years. However, seedlings did not survive low soil moisture conditions, which are likely to occur under field conditions over most summers. It is suggested that seedling survival of *C. pauper* will only occur in years of well above average rainfall, particularly, wet summer conditions.

Implications for restoration

This research has implications for restoration of *C. pauper* woodlands. The importance of above average rainfall events highlights the unpredictable nature of regeneration in the study area. This has definite disadvantages for resource intensive techniques such as tree planting and direct seeding where successful results rely on adequate natural rainfall.

Tree planting is likely only to be beneficial for aesthetics in areas of high visitor numbers such as picnic grounds. Tree planting success is contingent on above-average rainfall, or regular watering. Direct seeding is also likely only to be effective in above-average rainfall years, and only where the natural seed source is limited. Direct seeding may have potential in some highly degraded remnants to re-establish native perennial cover. Species such as *A. oswaldii*, *Dodonaea viscosa*, *E. tomentosa* and *O. pimeleoides* showed some potential to establish from direct seeding. In *C. pauper* remnants, high seed production of *C. pauper* suggests that direct seeding is unlikely to enhance regeneration.

In areas where *C. pauper* trees are unable to produce sufficient seed, *C. pauper* litter could be investigated as a potential treatment to introduce seed, and retain soil moisture. No evidence of allelopathic effects of *C. pauper* litter on *C. pauper* seedlings was observed, and many other native perennials germinated from within litter samples.

Previous research has highlighted the importance of landscape function to trap and retain resources such as seed, litter and rainfall. There have been methods detailed to test the landscape function (Tongway 1994), and techniques outlined to restore function (Ludwig *et al.* 1997). Investigation of the landscape function within *C. pauper* woodlands in the study area may assist in determining areas where these restoration techniques may be effective. Techniques to improve resource trapping such as soil pitting or placement of branches on degraded areas could be investigated within highly degraded areas of *C. pauper* woodlands.

6. CONCLUSIONS

Casuarina pauper (Belah) and *Callitris gracilis–Allocasuarina luehmannii* (Pine-Buloke) woodlands in northwest Victoria have been greatly impacted by clearing for cereal cropping, timber harvesting and thinning, and by prolonged heavy grazing pressure, resulting in almost complete inhibition of recruitment of perennial species.

Following conservation of a significant proportion of remaining semi-arid woodlands in Victoria within the Murray-Sunset National Park (MSNP) (1991), priorities for gathering baseline data on the vegetation condition and establishing a monitoring program to determine the effects of management intervention over time were identified (NRE 1996). This research has investigated three main components of the ecology and condition assessment of semi-arid woodlands in northwest Victoria:

1. historical survey plans and other documents were investigated to provide information on the original distribution of semi-arid woodlands, pre-European structure and composition and to assist in developing appropriate benchmarks for vegetation condition assessment.

2. methods for vegetation condition assessment to investigate the opportunities for remote vegetation condition assessment, and determine the most cost-effective condition assessment techniques.

3. regeneration requirements for perennial trees, tall and small shrubs, including seed availability and viability, effects of water application, fire and soil disturbance.

Pre-European distribution, structure and composition

Historical survey plans and other data sources were investigated to determine the likely pre-European distribution, structure and composition of semi-arid woodlands within the study area. A Geographic Information System (GIS) database was compiled from information on the survey plans. This enabled production of maps showing woodland distribution, woodland density and enabled calculation of the extent of clearing of semi-arid woodlands. The spatial database also provides a base to which any additional historical data could be added.

The survey plans revealed high variability in woodland structure at the time of survey (1860s–1930s). The historical data suggest that some woodland areas were open at the time of settlement, whilst others were densely treed, and where some woodlands supported a grassy understorey others supported a number of shrub species.

The high variability in vegetation structure and composition suggests the need for caution when applying benchmarks. Benchmarks for vegetation condition were originally chosen from the more densely treed and shrubby examples of woodland (Westbrooke *et al.* 2001). This research suggests that goals for regeneration of all woodlands to the shrubbiest remnants are unlikely to be realistic. To achieve patchiness across the landscape and conformity to the structure of woodlands observed by the early surveyors, a range of sites from open to dense and shrubby to grassy is required.

The historical survey plans indicated *C. gracilis* was previously more widely distributed and frequently occurring than is the case today. Pine was the most frequently recorded woodland species on the historical survey plans, but was greatly impacted by clearing, timber harvesting and by dieback in the late 1930s (Zimmer 1944). Another species that was mentioned on the survey plans, 'Myall', is thought to refer to either *Acacia loderi* or *A. melvillei.* Myall was recorded frequently around the Nowingi area to the east of the MSNP, however, no *Acacia* species or uncommon tree or shrubs were able to be located at these sites. Myall may have been lost either through harvesting for firewood along the Nowingi railway (*A. melvillei* is listed as a good firewood and was used for fence posts), through grazing pressure limiting regeneration (Cunningham *et al.* 1981), or a combination of both. These apparent changes in individual species distribution and abundance provide further evidence of the change in vegetation composition following European settlement.

Accuracy assessment techniques were performed to assess the accuracy of maps produced from historical survey plans. The overall accuracy was not high, largely due to errors of omission – that is, areas of semi-arid woodland not recorded on survey plans. However, the extent of correlation with current day vegetation suggests that interpretation of the survey plans is a valid technique for obtaining historical information on these communities.

Analysis of the extent of vegetation clearance shows that more than 96 % of areas previously supporting semi-arid woodlands have been cleared on private land surrounding the MSNP. This highlights the conservation significance of remnants and the need to manage these remnants to promote regeneration to ensure their continued survival.

There has been little previous investigation of the pre-European vegetation structure and composition across much of Australia, despite the relative recency of major vegetation change resulting from pastoralism. However, the benefits of integrating historical understanding into current ecological investigations have been highlighted in a number of recent studies (eg. Pickard 1994; Fensham and Holman 1998; Morcom and Westbrooke 1998). Historical data can be of great benefit in determining appropriate benchmarks; goals for vegetation restoration; in understanding trajectories of vegetation change, and therefore assisting in better prediction of future vegetation change.

This study of pre-European vegetation structure, composition and distribution covered an area of more than 13,000 km^2, representing a significant portion of the semi-arid woodlands within Victoria. This is the first extensive study of historical vegetation information within northwest Victoria and provides a format and spatial database that can be used as the basis for further historical investigation within the region.

Vegetation condition assessment

Condition assessments of various forms are increasingly being undertaken as the need for objective measures on the quality of a site are required for multiple purposes such as:

- documenting outcomes of management;
- determining the state of national parks (Parks Victoria 2000);
- calculating offsets for vegetation clearance (DNRE 2002; Parkes *et al.* 2003); and,
- determining the quality and health of streams and rivers (NRE *et al.* 1997).

Comparison of condition assessment techniques showed that only two vegetation condition classes could be reliably mapped using any of the field or remote techniques investigated. Good condition areas of semi-arid woodland were reliably distinguished from poor condition areas using interpolation modelling of field data, supervised and unsupervised classification of Landsat imagery, vegetation indices calculated from Landsat imagery, and the Treeden25 layer. However, maps of two vegetation condition classes provide limited information on the outcomes of vegetation management, and are unlikely to be sensitive to vegetation change.

The NDVI, calculated from Landsat imagery was found to be highly correlated with the field vegetation condition index, and many of the field condition parameters. This is thought to be due to the nature of vegetation clearance and disturbance within the study area. High grazing pressure, tree clearing and thinning have all resulted in a reduction of perennial vegetation cover, along with a decrease in native species richness, inhibition of perennial recruitment, and increase in cover and abundance of exotic annual vegetation. In the absence of exotic perennials, or woody weed problems in the study area, perennial vegetation cover is strongly correlated with good vegetation condition.

By obtaining satellite imagery in late summer, most annual vegetation had dried off, and did not contribute to the vegetation index. Therefore, under these conditions the NDVI functions largely as a measure of perennial vegetation cover. The NDVI shows potential as a monitoring tool that could be used in conjunction with field survey to assist in detecting change in vegetation condition of semi-arid woodlands within the MSNP and surrounding remnants.

Interestingly, the costs for the remote condition assessments were very similar to that of the field based survey. This is in part due to the relatively low sample size for the field survey. However, the need for ground truthing of remote data to assess the accuracy of the analysis, and costs of data acquisition, pre-processing and analysis all contributed to costs of remote condition assessment. The main advantages of remote vegetation condition assessment over field assessment are greater ability to detect changes across the entire study area, and greater flexibility in assessment timing. For example, Landsat imagery is available from the 1970s, which could enable assessment of the condition of woodlands under grazing. Change detection techniques could then detect changes in vegetation condition under different management regimes.

Use of an existing data set, the Treeden25 layer, was the lowest cost option for mapping vegetation condition, however, there are a number of limitations with using existing data sets. Replication of the assessment is reliant on other agencies to repeat their analysis, therefore repeat assessments cannot be undertaken when desired. As the data have been analysed on a state-wide basis, parameters have been developed that are not necessarily sensitive to the needs of assessment in semi-arid woodlands.

There has been very little research into the use of remotely sensed data for vegetation condition assessment within areas managed for conservation prior to this study. Previous research into condition assessment has focused largely on agricultural production, such as the ability of pastures to respond to rainfall, or crop productivity (eg. Pickup 1996; Reeves *et al.* 2001). The majority of remote vegetation assessment has investigated broad land cover classes, and changes in land cover, however, there has been little research investigating change in vegetation condition.

Whilst the study area for this project was northwest Victoria, *C. pauper* woodlands and other semi-arid woodlands with lack of recruitment and high grazing pressure are widely distributed throughout south-eastern Australia, and therefore these techniques could have a much wider application. Further research is required to determine the applicability of these methods to other semi-arid woodland communities.

Regeneration requirements

To gain a greater understanding of the main factors limiting vegetation condition, regeneration requirements were investigated for a number of perennial species commonly occurring within *C. pauper* woodlands. A more detailed study of regeneration requirements for *C. pauper* was also undertaken.

Regeneration requirements for *Casuarina pauper*

Investigation of the requirements for regeneration of *C. pauper* revealed that abundant viable seed is produced most years. Whilst ants and birds harvest *C. pauper* seed, large quantities (>340 m^{-2}) of seed were still present in two out of three sites sampled in May 2002 when seed harvesting by ants appeared to reduce to very low rates over winter. This showed that ample viable seed was available from late summer through to early winter in 2002. Low seed harvesting by ants during May and June suggests that seed harvesting may be reduced over winter. Further research with a longer monitoring period over more sites is required to confirm these initial trends.

C. pauper seed was found to germinate under relatively low soil moisture contents (35 % FC) and therefore is likely to germinate following minor rainfall events most years. As regeneration does not appear to be limited by seed viability or germination, it is probable that regeneration of *C. pauper* is limited at the stage of seedling establishment.

Field and laboratory studies confirmed that *C. pauper* seedlings did not survive very low soil moisture conditions (<2 % FC). During the study period, soil moisture of the A and B1 horizons rarely exceeded this value and high evaporation rates resulted in rapid return to low soil moisture following rainfall events.

Of the seedlings that were planted in the field study, more than half that survived to July continued to survive over winter until the end of September. However, no seedlings survived the hot dry conditions over the subsequent summer. These results reflect the findings of researchers working on regeneration of *Callitris* species in semi-arid regions, where it has been found that seedlings frequently do not survive hot dry conditions over summer (Zimmer 1944; Lacey 1972; Read 1995). Rare occurrences of wet and possibly cooler summers may be crucial for the establishment of *C. pauper* seedlings. These findings are also in keeping with the view that rare episodes of well above average rainfall are critical in shaping the structure of arid zone vegetation (Griffin and Friedel 1985; Stafford Smith and Morton 1990).

There are a number of implications resulting from this research for land managers wanting to encourage regeneration of *C. pauper*. Firstly, direct seeding of *C. pauper* is unlikely to enhance regeneration due to adequate production of seed within moderate to good condition remnants. Direct seeding could be trialed however, in highly degraded areas where natural seed sources are limited. It must be noted, however, that seeding is only going to be successful in rare years of well above average rainfall.

Tree planting of *C. pauper* is also unlikely to be successful in most years without additional watering, as seedlings are unlikely to survive harsh summer conditions without extensive root systems to access deeper soil moisture. Tree planting may have a limited role for aesthetic purposes in areas of high visitor access such as campgrounds and picnic areas where regular watering may improve survival rates.

The high number of *C. pauper* seeds collecting in litter beneath trees may represent a useful resource for revegetation. Seeds germinated and grew readily in litter samples in the glasshouse, suggesting that allelopathic compounds were not inhibiting *C. pauper* regeneration. Further research into techniques to re-establish *C. pauper* into highly degraded sites could investigate spreading *C. pauper* litter as both a seed source, and a method of limiting soil moisture evaporation. Techniques to trap resources such as litter, seeds and water may also be required on highly degraded sites.

Regeneration requirements for other perennial species

Only limited regeneration of perennial species resulted from the field trial replicating rainfall of known regeneration events. Species that appeared to benefit from additional water included *Enchylaena tomentosa*, *Olearia pimeleoides*, and to a small extent, *Dodonaea viscosa* and *Acacia oswaldii*. Regeneration of these species also appeared to be enhanced by the addition of seed.

It is likely that the lack of regeneration of other perennial species, particularly those for which considerable quantities of seed were added, was due to the inability to replicate soil moisture conditions of event years by the addition of water. During the drought conditions experienced during the current study, very low soil moisture conditions prevailed in both the A and B horizons. The watering treatment was also unable to replicate the effects of water moving across the landscape. It has been suggested that water run-on could contribute to high soil moisture in localised resource traps where seed, litter, water and soil nutrients also collect (Ludwig *et al.* 1997).

Seed appears to be produced most years for *A. oleifolius*, and *Myoporum platycarpum*. However, only very low viability (<1%) and frequently low seed production was found for *A. oleifolius* during the current study, supporting the findings of previous research (Hall *et al.* 1964; Wisniewski and Parsons 1986)

Soil seedbank studies found only low numbers of germinants for most perennial species (excepting *C. pauper*) suggesting that regeneration may be limited by seed abundance. This is not surprising, given the relatively low abundance of many perennial species.

This suggests that direct seedling could be of benefit for establishment of some perennial species, particularly in highly degraded areas where few seed sources remain. However, lack of sufficient rainfall remains the major hurdle to success of direct seeding efforts in most years. Predicting the appropriate year to sow seed is impossible, but to reduce costs associated with unsuccessful efforts, rainfall prediction models could be investigated.

Whilst this study has highlighted the importance of episodic rainfall events, these events are outside the control of land mangers. It is important, nevertheless, that land managers recognise the importance of these events, and are prepared to act to maximise regeneration success following such events. The main management action required is grazing control.

Many studies have shown that seedlings will not survive unless total grazing pressure is reduced to very low levels (Hall *et al.* 1964; Johnston 1968; Lange and Willcocks 1980; Tiver and Andrew 1997). Despite total removal of stock across National Parks in northwest Victoria, grazing pressure has remained high for many years due to the combined effects of kangaroos, goats and rabbits (Sandell *et al.* 2002). Control of grazing animals is critical following significant rainfall events to ensure recruitment success, and land managers must be ready to act at these times. Given the slow growth rate of many trees and shrubs, a high level of grazing protection may be required for a number of years to allow growth to a stage where plants are less susceptible to grazing (Westbrooke 1999).

6.1 RECOMMENDATIONS FOR FUTURE RESEARCH

Many additional questions have arisen during the course of this research. Recommendations for further research have been divided into the main elements of the current study and include:

1. Vegetation condition assessment

- Further research into change detection methodologies for remote vegetation condition assessment could be undertaken once relevant field data become available.
- A large number of vegetation indices have been developed for use in semi-arid conditions. Comparison of other vegetation indices with field condition data could assist in determining the most appropriate vegetation index for use in vegetation condition assessment.
- Vegetation indices may have potential for monitoring in other semi-arid woodland communities. The methods developed here could also be tested in other semi-arid woodland communities.

2. Historical distribution, structure and composition

- As any additional data on the historical distribution, structure and composition of semi-arid woodlands in northwest Victoria become available, this should be added to the spatial database developed.

- It would be interesting to conduct a study of *Callitris* sp. density across the study region, as many stumps and dead trees remain visible across the landscape. These stumps have provided a resource for determining tree density in previous studies and can assist in answering questions on pre-European tree densities (Parker and Lunt 2000).

3. Regeneration requirements

- Further research is required to better elucidate the regeneration requirements of perennial species of *C. pauper* woodlands in the field situation. Given the limited results from the watering treatment during the current study, it is likely that long-term monitoring and close observation during periods of above-average rainfall will be necessary.
- In highly degraded woodland areas, resource traps created by depressions in the earth's surface, branches or fallen trees show some potential as a method to trap seed and capture water leading to a greater chance of natural regeneration. Further research into methods to trap resources to promote regeneration is recommended.
- It may be possible to limit sowing to years with a greater probability of success by only seeding during La Nina events. Further research to determine relationships between regeneration events and long-reach rainfall forecasts may be valuable in increasing success rates of direct seeding and tree planting.
- Preliminary results from this research have suggested that soil disturbance may promote regeneration of some perennial species. Further research into soil disturbance or other disturbance regimes to promote regeneration is required

Appendix 1. Geographical Information Systems (GIS) map layers

Map name	Source	Description	Scale
Road25	Land Information Group - Land Victoria	Roads	1:25,000
Hydro25	Catchment and Water Department of Natural Resources and Environment	Linear hydrological features	1:25,000
Cont25	Land Information Group - Land Victoria	Topographical features	1:25,000
Tree100	Catchment and Water Department of Natural Resources and Environment	Tree cover	1:100,000
Lsys250	Department of Natural Resources and Environment	Land systems of Victoria	1:250,000
EVC100	Parks, Flora and Fauna	Ecological vegetation classes	1:100,000
BVT250_1750	Parks, Flora and Fauna	Pre 1750 Broad Vegetation Types based on 1:250 000 land systems	1:250,000
Geol250		Geological rock types and rock type boundaries	1:250,000
Treeden25	Catchment and Water Department of Natural Resources and Environment	Tree cover density	1:25,000

All layers were supplied by and used with the permission of the Department of Natural Resources and Environment Corporate Geospatial Data Library (NRE 2000b).

Appendix 2. Historical maps and survey plans

Maps of semi-arid woodlands located at State Library of Victoria

Date	Map title
1864	[Mallee district, showing agricultural potential
1887	Mallee block surveys, counties of Millewa and Weeah surveyed by T. H. Turner, contract surveyor, Nhill, 9th May 1887
1896	Mallee lands lithographed at the Department of Lands and Survey
1898	Plan shewing classification of Mallee lands
1904	Plan of Mallee blocks and allotments, counties of Tatchera and Karkarooc
1921	Northern Mallee comprising portions of the counties of Millewa, Weeah, Karkarooc & Tatchera
1921	Nowingi-Mildura to South Australian border plan showing (approximately) classification of land in Victorian territory west of Ouyen-Mildura Railway & Nth. of settlement along Ouyen-Pinnaroo Railway

Maps consulted at Land Victoria office of NRE

Approx. date	Survey Title	Reference
1923	Plan of subdivision survey of Parish of Benetook County of Millewa	PA B783B
1910	Subdivisional survey Parish of Boinka County of Weeah	PA B774
1926	Boorlee County of Millewa	PA B791
1930	Carool county of Karkarooc	PA C490
1911	Subdivision of eastern portion of Parish of Danyo county of Weeah	PA D194B
1912	Part of Parishes of Duddo & Tyalla County of Weeah	PA T251B1
1911	Plan of allotments Parish of Duddo county of Weeah	PA D218
1912	Subdivision of portion of parish of Ginquam county of Karkarooc	PA G2441
1912	Mallee allotments parish of Gnarr county of Weeah	PA G243I
1911	Parish of Karawinna county of Millewa	PA K213I
1912	Mallee allotments parish of Kattyoong county of Weeah	PA K202I
?	Kia	PA 201A2
1926	Parish of Koleya county of Millewa	PA K216
1912	Mallee Allotments Parish of Koondah County of Weeah	PA K203
1926	Parish of Kurnwill County of Millewa	PA K216
1925?	Parish of Mallanbool county of Millewa	PA M596A
1926	Parish of Malloren county of Milewa	PA M598
1912	Part of Parishes of Mamengoroock and Purnya county of Weeah	PA M578A
1921	Parish of Mamengoroock county of Weah	PA M579
1921	Plan of Allotments 137 to 173 (Merbein West) Merbein Parish of Merbein County of Karkarooc	PA M572_N1
1910	Plan of homestead allotments White Cliffs Irrigation Settlement parish of Merbein county of Karkarooc	PA M572C2
1915	Parish of Meringur county of Millewa	PA M595
1922	Parish of Merrinee county of Millewa	PA M591
1929	Survey of allotments Parish of Mildura	PA M593Z
1917	"Birdwoodton" Allotments Mildura-Merbein Parish of Mildura County of Karkarooc	PA M556J1
1924	Parish of Murrnroong	PA M592
1929	Parish of Nowingi county of Karkarooc	PA N173B
1922	Plan portion of parish of Nulkwyne county of Karkarooc	PA N120c
1929	Parishes of Nulkwyne and Walpunda	PA N120E
1929	Allotments parish of Nurnurnemal	PA N178E

Approx. date	Survey Title	Reference
1924	Olney county of Millewa	PA O28A
1935	Plan of allotment 3 Parish of Olney County of Millewa	PA O28C
1906	Subdivision of Mallee Lands parishes of Ouyen & Booring county of Karkarooc	PA O22
1906	Township of Ouyen Parish of Ouyen county of Karkarooc	PA O22B
1930	Allotments parish of Raak	PA R86C
1926	Parish of Taparoo county of Millewa	PA T310
1925	Parish of Tarrango county of Millewa	PA T306
1909	Parish of Tiega county of Karkarooc	PA T222B
1911	Subdivision of Mallee Lands portion of parish of Tiega County of Karkarooc	PA T222E
1923	Parish of Tullilah county of Millewa	PA T304A
1923	Parish of Tullilah county of Millewa	PA T305
1925	Parish of Tunart county of Millewa	PA T308
1912	Part of Parishes of Tutye & Bunnurouk	PA T246B
1928	Tyagook county of Millewa	PA T312
1913	Mallee allotments Parish of Tyalla county of Weeah	PA T251A2
1911	Plan of Parish of Tyalla county of Weeah	PA T251C2
1911	Plan of parish of Underbool county of Weeah	PA U66
1923	Parish of Walpolla county of Millewa	PA W422A
1912	Parish of Walpa county of Weeah	PA W417
1909	Subdivision of allots. Parish of Walpeup county of Karkarooc	PA W406I
1922	Plan of subdivision survey of part of Parish of Wargan County of Millewa	PA W417A1
1926	Warrimoo County of Weeah	PA W226
1912	Plan of allotments county of Weah [Parish of Tutye]	PA T246C
1924	Parish of Werrimull County of Millewa	PA W423A
1930	Woolwoola County of Millewa	PA W337B
1912	Parish of Worooa County of Weeah	PA W411I
1925	Wymlet County of Karkarooc	PA W425
1925	Parish of Yarrara county of Millewa	PA Y132
1912	Subdivision of portion of Parish of Yatpool County of Karkarooc	PA Y126
1910	Plan of farm allotments White Cliffs Irrigation Settlement Parishes of Yelta, Merbein, Wargun and Tullilah. Counties of Millewa and Karkarooc	PA Y102E
1910	Plan of farm allotments White Cliffs Irrigation Settlement Parishes of Yelta, Merbein, Wargun and Tullilah. Counties of Millewa and Karkarooc	PA Y102C
1927	Feature survey Raak country Karkarooc	PA F107N2
1905	Feature surveys on Mallee Blocks 29A & 29B Parishes of Tiega and Nulkwyne County of Karkarooc	PA T222A
1908	Feature survey Ouyen	PA O22C
1905	Feature survey Walpeup	PA W406B

Appendix 3. Municipalities, Counties and Parishes searched for survey plans

Municipal	County	Parish
Mildura	Karkarooc	Bitterang
Mildura	Karkarooc	Carool
Mildura	Karkarooc	Carwarp West
Mildura	Karkarooc	Ginquam
Mildura	Karkarooc	Merbein
Mildura	Karkarooc	Mildura
Mildura	Karkarooc	Mounpoul
Mildura	Karkarooc	Nowingi
Mildura	Karkarooc	Nurnurnemal
Mildura	Karkarooc	Raak
Mildura	Karkarooc	Walpamunda
Mildura	Karkarooc	Yatpool
Mildura	Millewa	Barchan
Mildura	Millewa	Benetook
Mildura	Millewa	Boorlee
Mildura	Millewa	Copi plains
Mildura	Millewa	Galick
Mildura	Millewa	Galpunga
Mildura	Millewa	Gimpa
Mildura	Millewa	Goonegul
Mildura	Millewa	Karawinna
Mildura	Millewa	Karween
Mildura	Millewa	Kernwill
Mildura	Millewa	Koleya
Mildura	Millewa	Mallanbool
Mildura	Millewa	Malloren
Mildura	Millewa	Merick
Mildura	Millewa	Meringur
Mildura	Millewa	Merrinee
Mildura	Millewa	Minook
Mildura	Millewa	Morkalla
Mildura	Millewa	Mullroo
Mildura	Millewa	Murlong
Mildura	Millewa	Murrnroong
Mildura	Millewa	Olney
Mildura	Millewa	Sunset
Mildura	Millewa	Taparoo
Mildura	Millewa	Tarranoo
Mildura	Millewa	Tulillah
Mildura	Millewa	Tunart
Mildura	Millewa	Tyagook
Mildura	Millewa	Wallpolla
Mildura	Millewa	Wargan
Mildura	Millewa	Warina
Mildura	Millewa	Warrimoo
Mildura	Millewa	Werrimul
Mildura	Millewa	Willah
Mildura	Millewa	Woolwoola
Mildura	Millewa	Yaramba
Mildura	Millewa	Yarrara
Mildura	Millewa	Yelta
Walpeup	Karkarooc	Kia
Walpeup	Karkarooc	Nulkwyne
Walpeup	Karkarooc	Ouyen
Walpeup	Karkarooc	Paignie
Walpeup	Karkarooc	Tiega
Walpeup	Karkarooc	Walpeup
Walpeup	Karkarooc	Wymlet
Walpeup	Millewa	Berrook
Walpeup	Weeah	Boinka
Walpeup	Weeah	Carina
Walpeup	Weeah	Danyo
Walpeup	Weeah	Duddo
Walpeup	Weeah	Gnarr
Walpeup	Weeah	Goongee
Walpeup	Weeah	Kattyoong
Walpeup	Weeah	Koonda
Walpeup	Weeah	Mamengoroock
Walpeup	Weeah	Manya
Walpeup	Weeah	Mulcra
Walpeup	Weeah	Nyang
Walpeup	Weeah	Pallarang
Walpeup	Weeah	Purnya
Walpeup	Weeah	Tutye
Walpeup	Weeah	Tyalla
Walpeup	Weeah	Underbool
Walpeup	Weeah	Walpa
Walpeup	Weeah	Woatwoara
Walpeup	Weeah	Worooa

Appendix 4. Tree and shrub species frequencies

Comparison*of tree and shrub species frequencies recorded in this study with those recorded by Zimmer (1937).

Species	Current study %freq	Current study Status	Zimmer Status
Acacia acinacea	0.6	R	
Acacia colletioides	7.6	O	F
Acacia hakeoides	1.3	O	F
Acacia ligulata	1.3	O	F
Acacia montana	0.6	R	
Acacia oswaldii	7.0	O	F
Acacia wilhelmiana	3.2	O	
Alectryon oleifolius	51.6	P	P
Allocasuarina luehmannii	14.6	F	O
Arabidella trisecta	0.6	R	
Atriplex limbata	0.0	R	O
Atriplex stipitata	26.1	F	
Atriplex vesicaria	63.1	P	
Beyeria lechenaultii	0.0	R	P
Beyeria opaca	7.0	O	
Callitris gracilis	48.4	F	P
Cassinia arcuata	0.0	R	R
Casuarina pauper	26.8	F	P
Chenopodium curvispicatum	31.2	F	
Chenopodium desertorum	33.8	F	
Dissocarpus paradoxus	1.3	O	
Dodonaea bursariifolia	0.6	R	
Dodonea viscosa ssp. *angustissima*	18.5	F	P
Enchylaena tomentosa	93.6	P	
Eremophila glabra	7.0	O	F
Eremophila longifolia	1.3	O	O
Eremophila maculata	0.0	R	O
Eremophila oppositifolia	3.8	O	F
Eriochiton sclerolaenoides	8.9	O	O
Exocarpos aphyllus	8.3	O	F
Frankenia foliosa	0.6	R	
Grevillea huegelii	0.6	R	F
Hakea leucoptera	9.6	O	P
Hakea tephrosperma	2.5	O	O
Jasminum didymum	1.3	O	R
Lawrencia squamata	0.6	R	
Lepidium leptopetalum	0.0	R	O
Lycium australe	1.3	O	
Maireana appressa	3.8	O	
Maireana brevifolia	43.9	F	
Maireana georgei	3.2	O	
Maireana pentatropis	22.9	F	O
Maireana pyramidata	7.6	O	O

Species	**Current study**		**Zimmer**
	%freq	**Status**	**Status**
Maireana sedifolia	1.3	O	
Maireana trichoptera	3.8	O	
Marsdenia australis	0.0	R	R
Myoporum platycarpum	35.0	F	P
Nitraria billardierei	0.0	R	O
Olearia decurrens	0.0	R	O
Olearia magniflora	1.3	O	
Olearia muelleri	8.9	O	F
Olearia pimeleoides	14.0	F	O
Osteocarpum salsuginosum	1.9	O	
Pimelea microcephala	8.3	O	
Pittosporum phylliraeoides	14.6	F	
Rhagodia spinescens	7.0	O	
Santalum acuminatum	1.9	O	O
Santalum murrayanum	0.6	R	
Scaevola spinescens	8.3	O	O
Sclerolaena diacantha	54.8	P	
Sclerolaena obliquicuspis	56.1	P	
Sclerolaena parviflora	0.0	R	O
Senna artemisioides ssp. *coriacea*	8.3	O	F
Senna artemisioides ssp. *filifolia*	5.7	O	
Senna artemisioides ssp. *petiolaris*	8.3	O	F
Templetonia egena	0.6	R	O
Westringia rigida	2.5	O	
Zygophyllum apiculatum	7.6	O	
Zygophyllum aurantiacum ssp. *aurantiacum*	8.9	O	
Zygophyllum eremaeum	1.3	O	

Current study:

R Rare less than 1% species frequency

O Occasional 1-14% frequency

F Frequent 15-49% frequency

P Prominent greater than 50% frequency

Appendix 5. Scientific and common plant names from survey plans

Showing current scientific and common names of species referred to on survey plans and other historical documents.

Names used on early descriptions and survey plans		**Current Names (Walsh and Entwisle 1994, 1996, 1999)**	
Scientific Name	Common Name	Scientific name	Common Name
Casuarina lepidophloia	Belar, Belah	*Casuarina pauper*	Belah
	Blue Bush, Bluebush	*Maireana* spp.	Bluebush
Acacia Oswaldii	Boree	*Acacia oswaldii*	Umbrella Wattle
	Box	*Eucalyptus largiflorens*	Black Box
	Boxthorn	*Lycium australe / Lycium ferocissimum*	
	Broom, Broom Bush	*Melaleuca uncinata / Templetonia egena /Baeckea behrii*	Broombush / Desert Broombush / Broom Baeckea
	Brown bush	Unknown	
	Bulloak, Oak	*Allocasuarina luehmannii*	Buloke
	Cabbage, Cabbage Bush, Cabbagetree	*Alectryon oleifolius*	Cattle Bush
	Cherry Tree	*Exocarpos* spp. / *Santalum acuminatum*	
Exocarpos aphylla	Currant	*Exocarpos aphyllus* / *Exocarpos* spp.	Leafless Ballarat
	Dead Finish	*Acacia colletioides / Acacia nyssophylla*	Wait-a while (Spine-bush)
	Dillon Bush	*Nitraria billardierei*	Nitre-bush, Dillon Bush
	Emu Bush	*Eremophila* spp.	Emu-bush
	Green Box	Unknown	
	Hop, Hop Bush	*Dodonaea viscosa* spp.	Sticky Hop-bush
	Mallee	*Eucalyptus* spp.	Mallee
	Myall	*Acacia melvillei / Acacia loderi*	Yarran / Nealie
	Needle Bush, Needle Wood, N. Wood	*Hakea leucoptera* ssp. *leucoptera*	Needle Bush
	Pine	*Callitris gracilis*	Slender Cypress-pine
	Quandong	*Santalum acuminatum / Santalum murrayanum*	Quandong, Sweet Quandong, Bitter Quandong
	Resinbush	*Cressa cretica*?	Rosinweed
	Rottenweed	Unknown	
	Salt Bush, Saltbush	*Atriplex* spp.	Saltbush

Names used on early descriptions and survey plans	Current Names (Walsh and Entwisle 1994, 1996, 1999)	
Sandal Wood (S. Wood)	*Myoporum platycarpum*	Sugarwood
Scrub Pine	*Callitris verrucosa*	Scrub Cypress-pine
Sourweed	*Oxalis corniculata / Oxalis spp.*	Yellow Wood-sorrel
Spear Grass	*Austrostipa spp.*	Spear-grass
Spinifex, Porcupine Grass	*Triodia scariosa* ssp. *scariosa*	Spinifex, Porcupine-grass
Thornbush	*Acacia colletioides / Acacia nyssophylla / Hakea leucoptera* ssp. *leucoptera*	Wait-a while (Spine-bush)/ Needle Bush
Ti-Tree / Tea Tree	*Leptospermum* sp. / *Melaleuca* sp.	
Tobacco / Tobacco bush	*Nicotiana glauca*	
Turpentine	*Eremophila sturtii*	
Willow	*Pittosporum phylliraeoides / Acacia salicina / Acacia stenophylla*	
Wire	*Unknown*	
Roly Poly	*Sclerolaena spp.*	
Wattle	*Acacia spp.*	

Appendix 6. Semi-arid woodland associations

From fusion analysis of 259 perennial quadrats.

Group 1 (consisting of 5 small groups)

Species rich *Casuarina pauper* woodland

Group 1 data	
Mean sp. richness	17.40
St dev	3.76
No. quadrats	25
Total no. species	58

Species	%Freq	Species	%Freq
Casuarina pauper	100.00	*Sclerolaena obliquicuspis*	16.00
Sclerolaena diacantha	96.00	*Zygophyllum apiculatum*	16.00
Olearia pimeleoides	88.00	*Acacia nyssophylla*	12.00
Scaevola spinescens	88.00	*Beyeria lechenaultii*	12.00
Chenopodium curvispicatum	84.00	*Eriochiton sclerolaenoides*	12.00
Enchylaena tomentosa var. tomentosa	80.00	*Hakea leucoptera* ssp. *leucoptera*	12.00
Alectryon oleifolius ssp. *canescens*	76.00	*Maireana brevifolia*	12.00
Olearia muelleri	68.00	*Maireana erioclada*	12.00
Exocarpos aphyllus	64.00	*Myoporum platycarpum* ssp. *platycarpum*	12.00
Chenopodium desertorum	56.00	*Olearia magniflora*	12.00
Eremophila glabra	56.00	*Senna artemisioides*	12.00
Senna artemisioides ssp. *X coriacea*	56.00	*Zygophyllum eremaeum*	12.00
Pimelea microcephala ssp. *microcephala*	52.00	*Acacia hakeoides*	8.00
Senna artemisioides ssp. *petiolaris*	48.00	*Allocasuarina luehmannii*	8.00
Eremophila oppositifolia ssp. *oppositifolia*	44.00	*Atriplex stipitate*	8.00
Callitris gracilis ssp. *murrayensis*	40.00	*Sclerolaena parviflora*	8.00
Rhagodia spinescens	40.00	*Acacia ligulata*	4.00
Maireana pentatropis	36.00	*Amyema linophylla* ssp. *orientale*	4.00
Pittosporum phylliraeoides	36.00	*Arabidella trisecta*	4.00
Senna artemisioides ssp. *filifolia*	36.00	*Callitris verrucosa*	4.00
Beyeria opaca	32.00	*Grevillea huegelii*	4.00
Acacia oswaldii	28.00	*Jasminum didymum* ssp. *lineare*	4.00
Dodonaea viscosa ssp. *angustissima*	28.00	*Maireana georgei*	4.00
Dodonaea viscosa	24.00	*Olearia decurrens*	4.00
Zygophyllum aurantiacum ssp. *aurantiacum*	24.00	*Rhagodia crassifolia*	4.00
Acacia colletioides	20.00	*Santalum murrayanum*	4.00
Atriplex vesicaria	20.00	*Maireana georgei*	4.00
Maireana trichoptera	20.00	*Olearia decurrens*	4.00
Santalum acuminatum	20.00	*Rhagodia crassifolia*	4.00
Templetonia egena	20.00	*Santalum murrayanum*	4.00
Westringia rigida	20.00		
Maireana radiata	16.00		

Group 2

Species rich *Callitris gracilis* woodland

Group 2 data	
Mean sp. richness	12.53
St dev	3.52
No. quadrats	17
Total no. species	54

Species	**%Freq**
Callitris gracilis ssp. *murrayensis*	100.00
Enchylaena tomentosa var. tomentosa	94.12
Olearia pimeleoides	70.59
Sclerolaena diacantha	70.59
Acacia oswaldii	64.71
Pimelea microcephala ssp. *microcephala*	58.82
Pittosporum phylliraeoides	58.82
Chenopodium curvispicatum	47.06
Dodonaea viscosa ssp. *angustissima*	47.06
Dodonaea viscosa	35.29
Einadia nutans ssp. *nutans*	29.41
Eremophila glabra	29.41
Hakea leucoptera ssp. *leucoptera*	29.41
Senna artemisioides ssp. *petiolaris*	29.41
Acacia ligulata	23.53
Alectryon oleifolius ssp. *canescens*	23.53
Allocasuarina luehmannii	23.53
Beyeria lechenaultii	23.53
Beyeria opaca	23.53
Chenopodium desertorum	23.53
Senna artemisioides ssp. *filifolia*	23.53
Acacia hakeoides	17.65
Atriplex stipitata	17.65
Maireana brevifolia	17.65
Maireana erioclada	17.65
Rhagodia spinescens	17.65
Sclerolaena obliquicuspis	17.65
Senna artemisioides ssp. *X coriacea*	17.65
Acacia lineata	11.76
Casuarina pauper	11.76
Eremophila longifolia	11.76
Exocarpos aphyllus	11.76
Maireana pentatropis	11.76
Senna artemisioides ssp.	11.76
Westringia rigida	11.76
Zygophyllum eremaeum	11.76
Acacia acinacea s.l.	5.88
Acacia brachybotrya	5.88
Atriplex vesicaria	5.88
Dodonaea bursariifolia	5.88
Dodonaea viscosa ssp. *cuneata*	5.88
Eremophila deserti	5.88
Grevillea huegelii	5.88
Hakea tephrosperma	5.88
Jasminum didymum ssp. *lineare*	5.88
Maireana trichoptera	5.88
Maireana turbinata	5.88
Marrubium vulgare	5.88
Myoporum platycarpum ssp. *platycarpum*	5.88
Olearia muelleri	5.88
Santalum acuminatum	5.88
Sclerolaena parviflora	5.88
Senna artemisioides	5.88
Zygophyllum aurantiacum ssp. *aurantiacum*	5.88

Group 3

Depleted *Callitris gracilis* woodland

Group 3 data	
Mean sp. richness	6.84
St dev	1.92
No. quadrats	19
Total no. species	27

Species	**%Freq**
Callitris gracilis ssp. *murrayensis*	100.00
Maireana brevifolia	94.74
Enchylaena tomentosa var. tomentosa	94.74
Sclerolaena obliquicuspis	63.16
Atriplex vesicaria	63.16
Atriplex stipitate	47.37
Myoporum platycarpum	31.58
Dodonaea viscosa	31.58
Sclerolaena diacantha	15.79
Maireana pentatropis	15.79
Chenopodium desertorum	15.79
Zygophyllum aurantiacum ssp. *aurantiacum*	10.53
Marrubium vulgare	10.53
Maireana trichoptera	10.53
Chenopodium curvispicatum	10.53
Acacia colletioides	10.53
Senna artemisioides ssp. *X coriacea*	5.26
Senna artemisioides ssp. *filifolia*	5.26
Scaevola spinescens	5.26
Rhagodia spinescens	5.26
Maireana georgei	5.26
Maireana pyramidata	5.26
Dodonaea viscosa ssp. *angustissima*	5.26
Acacia wilhelmiana	5.26
Acacia oswaldii	5.26
Acacia montana	5.26
Acacia ligulata	5.26

Group 4

Depleted *Alectryon oleifolius - Casuarina pauper* woodland

Group 4 data	
Mean sp. richness	8.64
St dev	3.16
No. quadrats	45
Total no. species	41

Species	**%Freq**	**Species**	**%Freq**
Enchylaena tomentosa var. tomentosa	97.78	*Allocasuarina luehmannii*	2.22
Alectryon oleifolius ssp. *canescens*	88.89	*Amyema miraculosa* ssp. *boormanii*	2.22
Atriplex vesicaria	77.78	*Dissocarpus paradoxus*	2.22
Casuarina pauper	62.22	*Dodonaea viscosa* ssp. *angustissima*	2.22
Maireana brevifolia	57.78	*Eremophila glabra*	2.22
Sclerolaena diacantha	57.78	*Hakea tephrosperma*	2.22
Sclerolaena obliquicuspis	44.44	*Maireana erioclada*	2.22
Atriplex stipitate	42.22	*Maireana pyramidata*	2.22
Maireana pentatropis	42.22	*Maireana triptera*	2.22
Callitris gracilis ssp. *murrayensis*	37.78	*Olearia pimeleoides*	2.22
Myoporum platycarpum	24.44	*Pimelea microcephala* ssp. *microcephala*	2.22
Zygophyllum apiculatum	24.44	*Rhagodia spinescens*	2.22
Chenopodium desertorum	22.22	*Westringia rigida*	2.22
Pittosporum phylliraeoides	20.00	*Zygophyllum eremaeum*	2.22
Zygophyllum aurantiacum ssp. *aurantiacum*	20.00		
Chenopodium curvispicatum	17.78		
Eriochiton sclerolaenoides	13.33		
Maireana appressa	13.33		
Einadia nutans ssp. *nutans*	11.11		
Acacia colletioides	8.89		
Dodonaea viscosa	8.89		
Exocarpos aphyllus	8.89		
Senna artemisioides ssp. *X coriacea*	8.89		
Olearia muelleri	6.67		
Senna artemisioides ssp. *petiolaris*	6.67		
Hakea leucoptera ssp. *leucoptera*	4.44		
Senna artemisioides ssp. *filifolia*	4.44		

Group 5

Depleted *Callitris gracilis* - *Allocasuarina luehmannii* woodland

Group 5 data	
Mean sp. richness	7.02
St dev	2.33
No. quadrats	59
Total no. species	45

Species	**%Freq**
Sclerolaena diacantha	94.92
Enchylaena tomentosa var. tomentosa	91.53
Callitris gracilis ssp. *murrayensis*	67.80
Allocasuarina luehmannii	57.63
Atriplex vesicaria	38.98
Sclerolaena obliquicuspis	35.59
Chenopodium desertorum	33.90
Chenopodium curvispicatum	32.20
Alectryon oleifolius ssp. *canescens*	30.51
Pittosporum phylliraeoides	20.34
Maireana brevifolia	15.25
Atriplex stipitata	13.56
Hakea leucoptera ssp. *leucoptera*	11.86
Maireana pentatropis	11.86
Acacia oswaldii	10.17
Dodonaea viscosa	10.17
Myoporum platycarpum	10.17
Acacia wilhelmiana	8.47
Dodonaea viscosa ssp. *angustissima*	8.47
Einadia nutans ssp. *nutans*	8.47
Rhagodia spinescens	8.47
Hakea tephrosperma	6.78
Olearia pimeleoides	6.78
Amyema linophylla ssp. *orientale*	5.08
Eremophila glabra	5.08
Exocarpos aphyllus	5.08
Maireana erioclada	5.08
Maireana georgei	5.08
Zygophyllum apiculatum	5.08
Casuarina pauper	3.39
Marrubium vulgare	3.39
Osteocarpum salsuginosum	3.39
Rhagodia crassifolia	3.39
Zygophyllum aurantiacum ssp. *aurantiacum*	3.39
Zygophyllum eremaeum	3.39
Acacia colletioides	1.69
Acacia lineata	1.69
Acacia nyssophylla	1.69
Beyeria opaca	1.69
Callitris glaucophylla	1.69
Eremophila longifolia	1.69
Maireana radiata	1.69
Maireana triptera	1.69
Psilocaulon granulicaule	1.69
Santalum acuminatum	1.69

Group 6

Depleted *Myoporum platycarpum - Alectryon oleifolius - Callitris gracilis* woodland

Group 6 data	
Mean sp. richness	6.29
St dev	1.72
No. quadrats	58
Total no. species	39

Species	%Freq
Sclerolaena obliquicuspis	93.10
Myoporum platycarpum	91.38
Enchylaena tomentosa var. tomentosa	87.93
Atriplex vesicaria	74.14
Alectryon oleifolius ssp. *canescens*	44.83
Maireana brevifolia	32.76
Chenopodium curvispicatum	27.59
Maireana pentatropis	27.59
Maireana pyramidata	22.41
Atriplex stipitata	10.34
Rhagodia spinescens	10.34
Sclerolaena diacantha	10.34
Callitris gracilis ssp. *murrayensis*	8.62
Eriochiton sclerolaenoides	8.62
Maireana sedifolia	8.62
Hakea leucoptera ssp. *leucoptera*	6.90
Maireana radiata	6.90
Zygophyllum aurantiacum ssp. *aurantiacum*	6.90
Nitraria billardierei	5.17
Dissocarpus paradoxus	3.45
Lycium australe	3.45
Maireana trichoptera	3.45
Psilocaulon granulicaule	3.45
Sclerolaena patenticuspis	3.45
Zygophyllum spp.	3.45
Acacia oswaldii	1.72
Amyema miraculosa ssp. *boormanii*	1.72
Casuarina pauper	1.72
Chenopodium desertorum	1.72
Dodonaea viscosa	1.72
Exocarpos aphyllus	1.72
Frankenia foliosa	1.72
Grevillea huegelii	1.72
Hakea tephrosperma	1.72
Lawrencia squamata	1.72
Osteocarpum salsuginosum	1.72
Pittosporum phylliraeoides	1.72
Zygophyllum apiculatum	1.72
Zygophyllum eremaeum	1.72

Group 7

Depleted *Myoporum platycarpum, Alectryon oleifolius, Callitris gracilis* and *Casuarina pauper* woodland.

Group 6 data	
Mean sp. richness	4.96
St dev	3.06
No. quadrats	27
Total no. species	31

Species	%Freq
Myoporum platycarpum ssp. *platycarpum*	92.59
Sclerolaena obliquicuspis	59.26
Alectryon oleifolius ssp. *canescens*	40.74
Callitris gracilis ssp. *murrayensis*	37.04
Zygophyllum apiculatum	25.93
Casuarina pauper	22.22
Psilocaulon granulicaule	22.22
Maireana pyramidata	18.52
Dissocarpus paradoxus	14.81
Dodonaea viscosa ssp. *angustissima*	14.81
Maireana brevifolia	14.81
Rhagodia spinescens	14.81
Atriplex vesicaria	11.11
Enchylaena tomentosa var. tomentosa	11.11
Nitraria billardierei	11.11
Senna artemisioides ssp. *petiolaris*	11.11
Dodonaea viscosa	7.41
Exocarpos aphyllus	7.41
Maireana pentatropis	7.41
Senna artemisioides	7.41
Zygophyllum aurantiacum ssp. *aurantiacum*	7.41
Acacia ligulata	3.70
Atriplex stipitata	3.70
Grevillea huegelii	3.70
Hakea tephrosperma	3.70
Maireana radiata	3.70
Maireana sedifolia	3.70
Olearia pimeleoides	3.70
Sclerolaena diacantha	3.70
Sclerolaena patenticuspis	3.70
Senna artemisioides ssp. *X coriacea*	3.70

Appendix 7. Predicted group membership of recruitment classes

Predicted group membership of *Callitris gracilis* recruitment classes (97.2 % of cases correctly classified into *C. gracilis* recruitment classes).

	Predicted Group Membership		
Callitris gracilis recruitment class	**0**	**1**	**2**
0	97.23	0	2.75
1	0	100	0
2	0	0	100

Predicted group membership of *Myoporum platycarpum* recruitment classes (85.6 % of cases correctly classified into *M. platycarpum* recruitment classes).

	Predicted Group Membership		
Myoporum platycarpum recruitment class	**0**	**1**	**2**
0	88.12	6.93	4.95
1	25.00	75.00	0.00
2	11.11	22.22	66.67

Predicted group membership of *Alectryon oleifolius* recruitment classes (82.2 % of cases correctly classified into *A. oleifolius* recruitment classes).

	Predicted Group Membership		
Alectryon oleifolius recruitment class	**0**	**1**	**2**
0	84.62	12.09	3.30
1	11.11	77.78	11.11
2	11.11	16.67	72.22

Appendix 8. Species list for grazing exclosure

GYMNOSPERMS

CUPRESSACEAE

Callitris gracilis ssp. *murrayensis*

MONOCOTYLEDONS

POACEAE

Austrodanthonia sp.
Austrostipa scabra ssp. *falcata*
* *Bromus rubens*
* *Critesion* sp.
* *Lolium* sp.
* *Pentaschistis airoides* ssp. *airoides*
* *Vulpia* sp.

DICOTYLEDONS

AMARANTHACEAE

Ptilotus seminudus
Ptilotus exaltatus var *exaltatus*

APIACEAE

Daucus glochidiatus

ASTERACEAE

Actinobole uliginosum
Angianthus tomentosus
Brachyscome debilis
Brachyscome lineariloba
Calotis hispidula
Euchiton sphaericus
Hyalosperma semisterile
Hypochoeris radicata
* *Isoetopsis graminifolia*
Millotia myosotidifolia
Olearia pimeleoides
Podolepis ciliaris
Podolepis tepperi
Pogonolepis muelleriana
Reichardia tingitana
* *Rhodanthe pygmaea*
Senecio glossanthus
Senecio gregorii
* *Sonchus oleraceus*
Vittadinia cuneata var *cuneata*
Vittadinia dissecta var *hirta*

BORAGINACEAE

Omphalolappula concava

BRASSICACEAE

Alyssum linifolium
Brassica tournefortii
* *Carrichtera annua*
* *Sisymbrium erysimoides*
Stenopetalum lineare

CAESALPINACEAE

Senna artemisioides ssp. *petiolaris*

CAMPANULACEAE

Wahlenbergia sp.

CARYOPHYLLACEAE

* *Silene nocturna*

CASUARINACEAE

Casuarina pauper

CHENOPODIACEAE

Atriplex stipitata
Atriplex vesicaria
Chenopodium curvispicatum
Chenopodium desertorum
Einadia nutans
Enchylaena tomentosa var *tomentosa*
Eriochiton sclerolaenoides
Maireana appressa
Maireana pentatropis
Maireana radiata
* *Salsola kali*
Sclerolaena diacantha
Sclerolaena obliquicuspis
Sclerolaena parviflora

CRASSULACEAE

Crassula colorata

FABACEAE

* *Medicago minima*

GERANIACEAE

Erodium crinitum

MIMOSACEAE

Acacia colletioides
Acacia oswaldii

MYOPORACEAE

Myoporum platycarpum ssp. *platycarpum*

MYRTACEAE

Eucalyptus leptophylla

PITTOSPORACEAE

Pittosporum phylliraeoides

PLANTAGINACEAE

Plantago turrifera

PORTULACACEAE

Calandrinia eremaea
Calandrinia granulifera

PROTEACEAE

Grevillea huegelii

SANTALACEAE

Dodonaea viscosa
Exocarpos aphyllus
Santalum acuminatum

SAPINDACEAE

Alectryon oleifolius ssp. *canescens*

SOLANACEAE

* *Solanum nigrum*

ZYGOPHYLLACEAE

Zygophyllum apiculatum
Zygophyllum aurantiacum ssp. *aurantiacum*

Appendix 9. Soil gravimetric water content

Soil gravimetric water content of unwatered experimental plots (mean value of two samples).

	No soil disturbance		Scarified	
Date	A1	B21	A1	B21
October 2000	1.57	-	3.57	-
November 2000	1.74	-	1.69	-
January 2001	1.54	3.33	1.56	3.42
February 2001	1.76	2.37	1.68	2.28
April 2001	2.32	2.80	2.18	2.77
June 2001	4.71	3.67	4.51	3.54
September 2001	4.63	5.09	4.53	5.42
January 2002	0.83	1.29	0.84	1.39
February 2002	0.85	1.27	0.83	1.72
July 2002	4.92	4.00	4.59	4.31

Soil gravimetric water content on experimental plots without soil disturbance before and after watering treatment.

	Pre-watering		Post-watering	
Date	A1	B21	A1	B21
October 2000	-	-	9.47	-
November 2000	-	-	6.46	-
January 2001	1.45	3.50	9.86	7.37
February 2001	1.52	2.40	12.94	13.10
April 2001	2.19	2.64	12.56	6.20
June 2001	4.57	3.77	12.52	12.37
September 2001	4.69	5.57	12.38	11.34
January 2002	0.99	1.21	10.06	9.20
February 2002	0.95	6.34	10.27	7.40
July 2002	4.61	4.72	9.29	8.44

Soil gravimetric water content of experimental plots with soil disturbance - before and after watering treatment (mean value of two samples).

	Pre-watering		Post-watering	
Date	A1	B21	A1	B21
October 2000	-	-	11.47	-
November 2000	1.75	-	14.78	-
January 2001	2.09	4.11	9.86	7.32
February 2001	1.74	2.51	13.12	13.07
April 2001	2.36	2.85	12.47	6.16
June 2001	4.65	3.55	12.61	12.35
September 2001	4.66	5.01	12.41	11.35
January 2002	0.90	1.28	10.17	9.17
February 2002	0.86	6.65	10.23	7.48
July 2002	4.22	4.56	13.05	11.71

Appendix 10. Germinants and seed abundance (m^{-2}) from soil seedbank

Family	Scientific name	Grazing Exclosure	Settlement Road	Mallanbool FFR
ASTERACEAE				
	Actinobole uliginosum	280.7	217.0	91.5
	Angianthus tomentosus		1012.7	
	Brachyscome debilis			6.3
	Brachyscome lineariloba	46.3	31.8	104.2
	Calotis hispidula	49.2	23.2	101.0
	Euchiton sphaericus	2.9	5.8	9.5
	Hyalosperma semisterile		2.9	
	Isoetopsis graminifolia		28.9	9.5
	Millotia myosotidifolia	11.6	60.8	249.4
	Podolepis ciliaris			3.2
	Podolepis tepperi	2.9	31.8	211.5
	Pogonolepis muelleriana		8.7	
	Rhodanthe pygmaea			3.2
	Senecio glossanthus	8.7		
	Vittadinia cuneata var *cuneata*			3.2
	Vittadinia dissecta var *hirta*	2.9		
	**Reichardia tingitana*	2.9	2.9	
	unidentified		2.9	
BORAGINACEAE				
	Omphalolappula concava	2.9		
BRASSICACEAE				
	Stenopetalum lineare	86.8	40.5	28.4
	**Brassica tournefortii*	5.8	2.9	6.3
	**Carrichtera annua*			3.2
	**Sisymbrium erysimoides*	141.8	2.9	151.5
	**Sisymbrium orientale*			3.2
	unidentified	20.3	17.4	63.1
CAMPANULACEAE	*Wahlenbergia gracilenta*	14.5	26.0	12.6
CARYOPHYLLACEAE	*Spergularia* sp. 3	5.8	28.9	
	**Herniaria cinerea*		2.9	
	**Silene nocturna*	205.4	2.9	716.5
	**Spergularia diandra*	5.8		151.5
CASUARINACEAE				
	Casuarina pauper	399.3	11.6	344.1
CHENOPODIACEAE				
	Atriplex stipitata		75.2	
	Chenopodium cristatum	2.9		
	Chenopodium curvispicatum			22.1
	Chenopodium desertorum		11.6	56.8
	Enchylaena tomentosa			3.2
	Sclerolaena diacantha	8.7	451.4	15.8
	Sclerolaena obliquicuspis	5.8	2.9	
	Sclerolaena parviflora			3.2
	Sclerolaena sp.	2.9		
	unidentified		5.8	
CRASSULACEAE				

Family	**Scientific name**	**Grazing Exclosure**	**Settlement Road**	**Mallanbool FFR**
	Crassula sp.	3,425.9	8,020.8	5,640.7
CYPERACEACE				
	Isolepis marginata	8.7	5.8	6.3
FABACEAE				
	**Medicago minima*	11.6	222.8	3.2
JUNCAGINACEAE				
	Triglochin calcitrapum			3.2
MESEMBRYANTHEMUM				
	**Mesembryanthemum nodiflorum*		214.1	486.1
MYOPORACEAE				
	Myoporum platycarpum	5.8		
PLANTAGINACEAE				
	Plantago turrifera		8.7	
POACEAE				
	Agrostis avenacea			3.2
	Austrodanthonia sp.	2.9	5.8	12.6
	Austrostipa elegantissima			3.2
	Austrostipa sp.	75.2	57.9	148.4
	**Briza minor*		2.9	
	**Bromus rubens*	2.9		6.3
	**Critesion* sp.			3.2
	**Lolium* sp.	2.9	2.9	
	**Schismus barbatus*	28.9	75.2	170.5
	unidentified	2.9	2.9	37.9
	**Vulpia* sp.		17.4	151.5
PORTULACACEAE				
	Calandrinia eremaea	376.2	141.8	280.9
	Calandrinia granulifera	185.2		6.3
SOLANACEAE				
	**Solanum nigrum*		8.7	
ZYGOPHYLLACEAE				
	Zygophyllum eremaeum		11.6	3.2
UNIDENTIFIED		57.9	86.8	167.3

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Appendix 11. Exclusion Plot Murray-Sunset National Park