Native Grass Strategy for South Australia

Broadacre Adoption and Seed Production of Native Perennial Grasses in Agriculture

Chapter 4 – Native Grasses and Land Management

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4 CHAPTER 4 – NATIVE GRASSES AND LAND MANAGEMENT

4.1 BACKGROUND: MANAGING THE AGRICULTURAL LANDSCAPE

Land management issues have plagued agriculturalists since grazing and cropping began in South Australia in the 1830s (as described in Chapter 1). Issues that may arise directly from agricultural practices include decreased water quality, increased water runoff, dryland and water salinity, water logging, erosion, decline in soil health and fertility. In addition, there is an observed increased relative fire risk due to increased biomass from exotic grasses as opposed to native grasses. Furthermore, there are continuing problems of drought and rainfall irregularity, and the increasingly important and relevant issue involving climate change. This chapter aims to explore the characteristics of native grasses and how they can be used to prevent, manage or ameliorate these land management issues and the impending changes to the landscape as a result of changes to climate and weather irregularities.

The initial wave of agricultural expansion in South Australia began in the 1830-50s, and involved dramatic increases in the number of livestock (Hyde 1995; Whalley 2003). As settlement increased, large-scale clearing began across the arable zones in southern Australia, along with the introduction of fencing and the beginning of set stocking practices (Garden and Bolger 2001; Whalley 2003). Temperate grasslands, particularly in south-eastern Australia, suffered the brunt of the agricultural expansion of grazing, cropping and irrigation (Johnston *et al.* 1999). Over 99% of these grasslands have been destroyed or severely modified (Johnston *et al.* 1999) due to the varying effects of clearing, overgrazing, cultivation, fertiliser use, pasture improvement and invasion by exotic species (Mitchell *et al.* 2001). An extreme drought at the turn of the century (1898-1902) was accompanied by massive erosion of topsoil in pastoral and cropping areas, followed for the first time by the appearance of serious weed problems (Whalley 2003; Whalley 2005).

The advent of such serious land management issues heralded the first application of science and technology to Australian pastoral industries. This approach became widespread by the 1940s and lead to the ‘sub and super’ revolution, aerial agriculture and ‘improved pastures’ (Whalley 2003). The resulting increase in exotic perennials translated into dramatic increases in herbage and farm carrying capacity. This led to the perception that Australian native species were of little value and should be replaced by introduced species to maximise pastoral production (Whalley 2003), and that native grasses were not well adapted to grazing and were inferior to exotic species in grazed pastures (Johnston *et al.* 1999; Whalley 2003).

However, over time few introduced pasture species have been reliably persistent (Lodge 1994; Whalley 2005) as crop and pasture species poorly mimic the growth and water use characteristics of the indigenous vegetation (Mitchell et al. 2001). Fewer still are tolerant of the declining condition of many Australian soils, including soil characteristics related to salinity and acidity (Langford et al. 2004; Whalley 2005). Considerable damage has occurred to large areas of the landscape, leading to dryland salinity, soil erosion, soil acidification and water logging (Lodge 1994; Mitchell *et al.* 2001; Whalley 2005), raising doubts about the long-term sustainability of agricultural practices associated with introduced pastures and cropping (Lodge 1994; Nadolny 1998; Whalley 2003).

Much agricultural and pastoral land in Australia is now considered to be degraded, suffering a myriad of land management issues including poor soil health (salinity, altered fertility, acidity), erosion, reduced resilience and productivity, and specifically poor herbaceous layer diversity including a paucity of native grasses. With increasing land management issues plaguing agricultural areas and the threat of worsening climatic conditions, many agriculturalists and scientists believe that native pastures are vital for the long-term sustainability of agriculture (particularly grazing) in South Australia (Garden *et al.* 1996; McGarva *et al.* 2000; Bolger *et al.* 2005; Davenport 2005; Nicholls 2005). Where stony, shallow, sloped, acid or infertile soils limit the persistence or establishment of current introduced cultivars (such as **Phalaris aquatica* [Phalaris], **Dactylis glomerata* [Cocksfoot], **Lolium* ssp. [Rye Grasses] and **Festuca elatior* var. *erundniacea* [Tall Fescue]), native pastures may prove much more effective (Lodge 1994; Johnston *et al.* 1999; Whalley 2005).

Native grasses evolved in the naturally unpredictable climate and shallow soils of Australia. They are uniquely adapted to a broad range of harsh conditions and sub-standard soils (Lodge 1994; Langford *et al.* 2004). The main advantage of the use of native grasses in relation to land management issues, is their diversity and in particular their varying tolerance to frost, drought, fertility, grazing and adverse soil conditions, which offer opportunities in identifying productive species/ecotypes for a range of soil and climatic conditions (Lodge 1994; McGarva *et al.* 2000; Davies *et al.* 2005; Nicholls 2005). Native grasses are also valued for their low input requirements, ability to produce warm-season green feed, good water use efficiency, persistence, perenniality and provision of permanent ground cover (Lodge 1994; Nadolny 1998; McGarva *et al.* 2000; Langford *et al.* 2004; Davenport 2005).

4.1.1 Aims and objectives

The purpose of this chapter is to review the use of native grasses in relation to past and current significant issues in land management. In particular, this section aims to determine:

The processes involved in the use of Australian native grasses to ameliorate issues of water quality, water runoff, salinity, soil health, erosion control and soil cover, fire as a land management tool, drought tolerance and current and potential impacts of climate change;

The potential of native grasses to provide part or whole land management options to ameliorate issues of water quality, water runoff, salinity, soil health, erosion control and soil cover, fire as a land management tool, drought tolerance and current and potential impacts of climate change; and

To investigate the economic, social and environmental benefits and dilemmas concerning the use of Australian native grasses as land management options to ameliorate issues of water quality, water runoff, salinity, soil health, erosion control and soil cover, fire as a land management tool, drought tolerance and current and potential impacts of climate change.

4.2 LAND MANAGEMENT: WATER RUNOFF, SALINITY AND WATER USE IN NATIVE GRASSES

4.2.1 Problems and causes

The pre-European vegetation of temperate Australia was comprised of species with a capacity for active growth and transpiration during summer (Hatton and Nulsen 1999; Johnston *et al.* 1999; Davies *et al.* 2005). This water use pattern resulted in soil moisture being near capacity in late winter and spring, and exhausted by summer's end. For the high rainfall zone of southern Australia various studies have shown that the exotic annual pastures use less water (25-40%) than native perennial pastures (Murphy 2004). Many native grass species are summer-active C_4 species, and even C_3 species *Austrodanthonia* spp. (Wallaby Grass) and *Microlaena stipoides* (Weeping Grass) can function similarly to summer-active when summer rainfall occurs (Garden *et al.* 2001). Replacement of this mostly perennial vegetation with exotic annual-growing and summer-dormant C_3 species has changed water use pattern so that soils are wetter in autumn than under pre-European vegetation (Lodge 1994; Hatton and Nulsen 1999; Johnston *et al.* 1999; Murphy 2004). This reduces the pre-winter soil moisture deficit so that winter rainfall saturates the soil, causing deep drainage to increase in winter (Johnston *et al.* 1999).

Development of agriculture in south-eastern Australia led to a substantial amount of native perennial vegetation being cleared and replaced with annual crops and pastures (Hatton and Nulsen 1999; Murphy 2004). Farming systems based on annual crop or pasture species are inherently less efficient water-users than perennial species (Garden and Bolger 2001; Murphy 2004). In south-eastern Australia, a significant proportion of the total rainfall is received during the period when annual species are senescent (Johnston *et al.* 1999). This leads to a range of problems, including high runoff rates, water logging and dryland salinity (Lodge 1994). Intense rainfall on soil with low ground cover will generate runoff and has very short duration in the farming system (White 1994; Murphy 2004). This may contribute to topsoil erosion, nutrient loss and results in less efficient use of water across the farm. The perennial nature of natural, native grasslands may help to control water runoff through increased groundcover, rainfall interception and maximising use of rainfall across all seasons.

Dryland salinity is one of the largest challenges facing Australian agriculture (Davies *et al.* 2005; Rogers 2007). Salinity results from excess water draining past the root zone, leading to rising water tables that mobilise ancient stores of salt which then migrate over time, to the surface (Garden and Bolger 2001; Davies *et al.* 2005). Where water tables reach the surface, water logging also becomes a problem (Mitchell and Virgona 2003; Murphy 2004). Areas not productive for cropping or annual pastures, as a consequence of salinity or water logging, can be highly productive by introducing appropriate perennial pastures, a potential often not realised (Davenport 2005; Rogers 2007). Consequently, there is a need to identify native plant species for saline areas that will provide both groundcover and agricultural production (Johnston *et al.* 1999; Rogers 2007).

In annual cropping systems, perennial pasture phases can be used to dry the soil profile and reduce the risk of water logging and salinity effects (Mitchell and Virgona 2003; Murphy 2004). Perennial grain crops, such as native grasses for herbage or seed production, offer an alternate solution to issues associated with broadscale inefficient water use across agricultural landscapes (Davies *et al.* 2005). The integration of ‘pasture cropping’, the direct seeding of an annual C_3 crop, primarily winter cereal crops such as wheat or oats into a permanent, perennial native pasture is another recent innovation suitable for some agricultural areas. Pasture cropping appears to hold promise as a strategy to maximise efficiencies in water use, maintain production and provide opportunity for sustainable farming practices. Pasture cropping is currently being trialled in the mid-north of SA and is discussed in detail in Chapter 3.

In agricultural areas there are areas of prime use and productivity, and there are locations which are less productive for reasons related to soil characteristics, slope, or other factors of landscape character. Indigenous perennial grasses are likely to have adaptive traits favouring their survival on harsh landscapes or hill country, particularly where it is difficult or uneconomic to use high input strategies such as perennial cultivars which require high rates of fertiliser (Johnston *et al.* 1999). Consequently, for landholders with marginal land becoming degraded under traditional grazing and cropping practices may benefit from incorporation of native grasses into the farming system. Native grasses may persist and grow in landscapes which are becoming degraded and are exposed to climatic extremes, more than introduced species which generally require greater inputs and are less well adapted to the harsher environments and climatic extremes.

In summary, as a consequence of the introduction of European farming practices to the Australian agricultural landscape, much of the perennial vegetation, including native grasses, across temperate Australia was replaced with annual introduced species,. Annual introduced pastures and crops use water inefficiently in comparison to native perennials, generally not making use of summer rainfall as efficiently as Australian native grasses. Ultimately this differential in use of water leads to increased deep drainage and disruptions to the soil water balance. Increased deep drainage causes rising water tables and leads to salt deposits deep in the soil to be brought to the surface. Inefficient use of summer rainfall is also linked to high runoff rates, soil erosion, soil acidity and nitrogen leaching. Returning the natural perennial vegetation to the landscape can help reduce and reverse these effects, and native grasses represent an opportunity to address this issue whilst maintaining agricultural productivity.

4.2.2 Maximising use of summer rainfall

Maximising the use of summer rainfall in both cropping and pasture regimes can help prevent or manage many of the issues associated with imbalances in the soil water cycle, including runoff (contributing to soil erosion, nutrient loss and water wastage); deep drainage; water logging and salinity. Native grasses, due to their perennial nature and superior water use efficiency (Johnston *et al.* 1998), offer an opportunity to address imbalances in soil water content by maximising water use when it falls, particularly in the summer months when traditional European crops and pasture species are dormant. Johnston et al*.* (1998), illustrated that increased capacity of vegetation for efficient water use across whole catchments can be achieved using native grasses in pastures and as part of the cropping regime, whilst still maintaining agricultural productivity.

Themeda triandra (Kangaroo Grass) communities were once widespread throughout temperate Australia and found at altitude, in humid and semi arid regions, and occur on both steep and flat land (Dunin 1991). Water use within *Themeda triandra* communities is closely linked to seasonal growth behaviour, where peak rates coincide with the phase of new leaf growth in summer (Dunin 1991). The summer activity of *Themeda triandra* is regulated by temperature cues, where below a threshold of 15°C the sward 'hays off' indicating dormancy in winter and spring (Dunin 1991). Dunin (1991) reports that throughout this period the standing dead material of un-grazed *Themeda triandra* communities inhibits evaporation of soil moisture to less than 50% of that in crops and grazed pastures, conserving high quantities of soil moisture from the cool season rainfall. This soil moisture is subsequently used at the onset of higher summer temperatures as the *Themeda triandra* growth season begins again (Dunin 1991). In this way the soil profile is dried up by the end of the summer period and ready to store cool season rainfall controlling the soil water balance and limiting deep drainage.

Johnston et al. (1999) investigated the potential of native grasses for pasture rehabilitation in the higher rainfall zone of South Australia, particularly the southern section of the Murray-Darling Basin. The investigation formed part of the Low Input Grasses for Limiting Environments (LIGULE) project. They noted that although several methods may be used to treat the problems inherent in C_3-based cool season pastures (such as tree planting to minimise deep drainage, liming to raise pH of acid soils, earthworks to control erosion), no single strategy addresses the cause of the problem, identified as "the loss of plants which provide pastures with a year-long groundcover and a capacity for active growth and transpiration in summer" (Johnston *et al.* 1999). Furthermore, the production and sustainability goals of land and water management plans looking at dryland salinity control, tended to promote the re-introduction of deep-rooted perennial pastures in areas with a high potential for watertable recharge. However, as recharge lands are typically steep non-cultivated catchment uplands with shallow stony soils, the high cost of establishing current perennial pasture cultivars and then maintaining them restrict the success of such measures (Johnston *et al.* 1999). From this, the authors concluded that there is a need to develop more appropriate recommendations for managing and encouraging existing native grass pastures in a productive manner. Johnston et al. (1999) recommend developing a range of better adapted, summer-active perennial grass varieties specifically for sowing on the extensive recharge lands of the Murray-Darling Basin.

Perennial pastures are an important component of farming systems and need to be utilised in such a way that minimises economic penalty, yet maximises control of deep drainage and accumulation of soil nitrogen (Murphy 2004). When cropping paddocks are sown down to the pasture phase, agriculturalists may suffer a decrease in return on that paddock (Murphy 2004). In order to return the paddock to the cropping phase as soon as possible (thus maximising production), Murphy (2004) suggests the aim should be to maximise the production of dry matter and the use of excess soil water during the pasture phase. High levels of plant litter should be maintained to minimise bare soil evaporation, a major source of water loss from pastures (Murphy 2004). Interception of water by the canopy can also play a significant part in maintaining the water balance in pastures (Murphy 2004). For permanent pastures, Murphy (2004) recommends sowing a mix of summer and winter active species in order to maximise water use and herbage mass, litter mass, and ground cover be managed to limit water losses through surface runoff, evaporation and deep drainage. Although these recommendations were made specifically for pastures in northern NSW and did not explicitly mention native grasses, they are broadly applicable to temperate South Australia and can be put into practice by using native grasses as perennial pasture species.

Brennan *et al*. (2005) noted that the drier the soil profile, the larger the likely "buffer" against a recharge event should it occur. The ability of different species to dry the soil profile depends on their green leaf area, rate of transpiration and root distribution, and is usually measured by the soil water content (Brennan *et al.* 2005). The authors investigated soil water content data for a range of perennial pasture species and shrubs to examine any short term seasonal or soil depth differences in the drying of the soil profile. The study included six examined sown pastures (**Medicago sativa* (lucerne); digit grass, **Lolium perenne* (perennial rye grass), *Atriplex nummularia* (old man saltbush), **Phalaris aquatica* (Phalaris), and phalaris/lucerne mix), an adjacent native pasture dominated by *Austrostipa aristiglumis* (Plains Grass) and a fallow plot kept weed-free with regular herbicide applications (Brennan *et al.* 2005). The driest soil profile was the native pasture, with the highest soil water content maintained in the fallow treatment and intermediate soil profile drying was evident in the sown pastures. These results demonstrate the ability of native grasses (in this instance, *Austrostipa aristiglumis* (Plains Grass)) to use water more efficiently than many commonly sown pastures by maximising use of summer rainfall to dry the soil profile, thus reducing deep drainage.

Popular belief is that the reason behind results such as those of Brennan et al. (2005), where native grasses are more efficient at drying the soil profile and thus maximising water use) is because native grasses are deep rooted. Rooting characteristics such as depth and density determine the volume of soil from which a plant can potentially extract water and nutrients (Southwell *et al.* 2005). However, there is very little quantitative data available to support these claims (Bolger *et al.* 2005). Few studies, have examined the variability in rooting characteristics amongst temperate grasses in Australia and even less have examined native temperate grasslands (Southwell *et al.* 2005). Southwell (2005) compared experimentally the rooting depth and distribution of two common native grass species, *Austrodanthonia* spp. (Wallaby Grass) and *Bothriochloa macra* (Red Grass). After one year of growth the roots of *Bothriochloa macra* had exceeded the maximum depth measured (1.5 m) while the *Austrodanthonia* spp. did not have roots extending beyond 1 m. The roots of *Bothriochloa macra* at depth were also denser, leading authors to conclude that *Bothriochloa macra* had access to greater volume of soil water than *Austrodanthonia* spp.. More research is required to evaluate the rooting depths and soil profile drying capabilities of other native species.

A related study by Mitchell (2001) noted that 8 out of 10 grass species tested reached rooting depths of 100 cm within 24 months of planting. The fastest species to reach 100 cm were *Bothriochloa macra* (Red Grass) in 3 months and **Eragrostis curvula* (Lovegrass cv Consol) in 5 months, while both *Austrodanthonia fulva* (Wallaby Grass) and *Digitaria divaricatissima* (Cotton panic) did not reach rooting depths of 100 cm. The remaining species took at least 12 months to reach rooting depths of 100 cm, including *Chloris ventricosa* (Tall Windmill Grass), *Elymus scaber* (Common wheat grass), *Enteropogon acicularis* (Curly Windmill Grass), Weeping grass (*Microlaena stipoides*), Kangaroo grass (*Themeda triandra*) and **Phalaris aquatica* (Phalaris cv Sirosa). Mitchell

(2001) recommends that pastures should be a combination of species with different root architectures to ensure the entire soil water profile is utilised.

Research suggests that it is possible to use perennial native grasses in agricultural systems to maximise water uptake, particularly during the summer period, and still maintain land productivity. Traditional methods of addressing soil water use across catchments include tree planting, which in many cases cannot be undertaken on a large enough scale without significantly reducing the area of land available for production. However, while integrating perennial native pastures into agricultural systems seems to be a potential option, more research is required to determine the actual impact of perennial native grasses on the soil water content, and more particularly at a catchment scale.

4.2.3 Managing salinity and water logging

In 2001, it was estimated that dryland salinity affected approximately 370,000ha of land and wetlands in South Australia, with another 84,000ha of being affected by natural salinity (Green 2001). Virtually all the current species used for revegetating or improving the productivity of saline land are introduced (with the notable exception of the saltbushes) (Bann and Field 2006; Semple *et al.* 2006). As with all introduced species there is a risk that species may become weeds, either at the saline site undergoing remediation or by spreading into neighbouring areas (Semple *et al.* 2004). Examples include **Thinopyrum ponticum* (Tall wheat grass) (Bann and Field 2006) and **Pennisetum clandestinum* (Kikuyu) (Semple *et al.* 2004).

Research into the management of salinity and water logging through the use of native grasses has, effectively, two strategic approaches. The first, is to increase the use of summer rainfall and minimise deep drainage by using perennial summer active species and thus preventing water tables from rising. The second, is the identification of native grasses that will grow on sites already affected by salinity and/or water logging to retain and increase ground cover and/or productivity. Both research approaches warrant review and further investigation in relation to the role of native grasses in the Australian landscape.

Minimising deep drainage

Lodge *et al*. (2004) investigated the establishment year of a range of perennial legumes and grasses sown at recharge sites in northern NSW as part of a national field evaluation program within the CRC for Plant-based Management of Dryland Salinity. The trial was extensive and involved a large number of species/lines representing legumes, perennial grasses and temperate grasses (Lodge *et al.* 2004) with the aim of finding the most productive species for use in reducing deep drainage in soils at risk of becoming saline. In northern NSW the most widely sown perennial legume and temperate perennial grasses are lucerne and phalaris, respectively. These provide the productivity ‘benchmarks’ for the area, thus for a perennial legume or temperate grass to be widely sown in the district it would have to exceed the performance of these species or complement them in a mixture (Lodge *et al.* 2004). The authors found that only **Hedysarum carnosum* (Sulla) (a perennial legume) had higher production and plant frequency than lucerne, while no grass species out-performed phalaris. This indicates that finding a viable and alternative to lucerne and phalaris for controlling water use in northern NSW may be hampered by low productivity of candidate species, a problem that may also affect similar species trials in South Australia.

Similarly, as part of the same CRC, Mitchell and Virgona (2003) looked at the role native grasses might play in the reduction of dryland salinity in the southern half of the Murray-Darling Basin. The project is relevant to most soils in the 550-750mm/yr rainfall zones of south-eastern Australia. In many hilly parts of the landscape native grass pastures offer the only sustainable options for grazing whilst maintaining perennial ground cover as the shallow, infertile soils limit the persistence of introduced perennial cultivars (Mitchell and Virgona 2003). In these parts of the landscape the high cost of sowing introduced perennial cultivars and their low persistence limit the ability of land managers to meet the revegetation targets needed to control dryland salinity (Mitchell and Virgona 2003). The project aimed to examine the capacity for regeneration of native grasses and investigate practical, low cost management strategies to improve the native grass component of pastures (Mitchell and Virgona 2003). The authors found that pastures that contain a high proportion of perennial native grass are stable (i.e. more resistant to pasture decline), especially where the annual grass content is low. Options for increasing the native grass component of a pasture include grazing management (generally the cheapest option), and the use of fire, fertiliser and herbicides (Mitchell and Virgona 2003). However, Mitchell and Virgona (2003) found that in marginal country such as their study sites, techniques used must also be low input in terms of management and cost if they are to be viable. The authors note that although increasing perenniality in pastures may represent only a small change in the water balance of an area, it is still likely to be important over a large catchment area (Mitchell and Virgona 2003).

Identification of native species tolerant to salinity and/or water logging

A review of research into salt tolerance of temperate Australian native grass species conducted by Brown and Rogers (2003), found that 36 native grass species have been investigated for salt tolerance, but only eight of these have been examined more than once (Brown and Rogers 2003). A total of 56 native grass species have been recorded as occurring on saline sites in one study, Cole and Semple (2002) cited in Brown and Rogers (2003), representing a species list for potential further investigation (Brown and Rogers 2003). Brown and Rogers (2003) note that saline sites have high spatial and seasonal variability and the presence of a species on a saline area does not necessarily indicate salt tolerance. With regards to research on salt tolerance in specific species, much of the research has been conducted at the high to extreme range of salt concentrations, and there may be additional species not identified in the literature that are sensitive to high salinity but perform well at slight to moderate salinities (Brown and Rogers 2003). Brown and Rogers (2003) concluded that although there has been a wide range of studies on the salt tolerance of over 30 native species, very little detailed and comprehensive information exists for most of these. Within this research, there is considerable variation in performance, emphasising the need to investigate a wide number of accessions per species under a range of conditions (Brown and Rogers 2003). Since Brown and Rogers' (2003) study, research has continued but their summation of the literature remains broadly applicable.

Semple et al. (2003) undertook a preliminary field assessment of the survival and vigour of accessions of eight native grass species at two saline sites in south-eastern Australia.

The eight accessions were collected from sites in Victoria and New South Wales and compared with the ***Puccinellia* spp. (Saltmarsh-grass) (commonly sown on saline sites). They found that the most successful accessions were *Distichlis distichophylla* (Australian Salt-Grass) and to a lesser degree *Eragrostis dielsii* (Mallee Lovegrass) and *Enteropogon acicularis* (Curly Windmill Grass) (Semple *et al.* 2003). They also found that *Panicum queenslandicum* (Yabila grass) performed well on the less saline but more alkaline site.

Brown (2003) looked at the salt tolerance of one species, *Lachnagrostis filiformis* (Common Blown-grass). This species has a wide distribution across the temperate parts of all Australian states, New Zealand and Polynesia (Brown 2003). It grows in a range of freshwater and brackish habitats, from sub-alpine to arid regions and is often found growing along edges of watercourses, lakes, swamps and wed depressions (Brown 2003). It establishes easily and although it appears to be a short-term annual, the tussocks readily regenerate with a summer shower (Brown 2003). The author found that *L. filiformis* has wide tolerances, and that differences observed in the field have a largely genetic basis rather than a direct environmental one. This indicates that *L. filiformis* has a high level of natural variability which suggests that the potential for selection of useful, salt-tolerant strains is also high (Brown 2003).

Andrew and Brown (2003) took a different approach to identifying salt-tolerant species by identifying all the native grasses in or adjacent to saline areas in the Wimmera. The aim of the project was to identify local native species with salt tolerance that may be used to rehabilitate saline discharge areas following concerns about the invasive potential of the current species used, **Thinopyrum ponticum* (Tall Wheat Grass) (Andrew and Brown 2003). The survey found 14 salt tolerant species: the most widespread were *Eragrostis infecunda* (Southern Cane-grass) and *Distichlis distichophylla* (Australian Salt-grass).

The list of species identified is (Andrew and Brown 2003):

- *Amphibromus spp*. (Swamp Wallaby-grass)
- Austrodanthonia spp. (Wallaby grasses)
- *Austrostipa* spp. (Spear grasses)
- *Cynodon dactylon* (Couch) (native status in dispute)
- Distichlis distichophylla (Australian Salt-grass)
- *Eragrostis dielsii* (Mallee Love-grass)
- *Eragrostis infecunda* (Southern Cane-grass)
- Lachnagrostis billardierei (Coastal Blown-grass)
- Lachnagrostis filiformis (Common Blown-grass)
- Lachnagrostis robusta (Salt Blown-grass)
- Poa labillardieri (Tussock Poa)
- Puccinellia stricta var. perlaxa (Salt-marsh Grass)
- Puccinellia stricta var. stricta (Salt-marsh Grass)
- Sporobolus mitchellii (Rat-tail Couch)

Although these species were identified in western Victoria all are recorded in South Australia. It is likely therefore, that some of these species will prove useful for rehabilitation of saline sites in south-eastern South Australia. Further investigation into their specific characteristics is required before cultivars can be developed for rehabilitation purposes.

Norman *et al*. (2003) looked at species diversity in two saline paddocks in south-western Australia. The authors found that plant diversity (including some native grass species) was much higher in the saline areas than is considered usual for non-saline pastures in this area. The authors postulated that the high levels of plant diversity within saline paddocks are the result of niche differentiation, where no one species can dominate in all of the functional niches within the paddock. They suggested that in light of these results, it is unlikely that monocultures of sown species will be optimal when revegetating saline land (Norman *et al.* 2003).

Bann and Field (2006) investigated potentially useful species for salinity mitigation by identifying which species were able to colonise and persist at sites with increased salinity levels at two sites in the Southern Tablelands of NSW. They noted that tolerance of plants to salinity changes throughout their lifecycle, with juvenile plants generally having more sensitivity to salt levels than adult plants (Bann and Field 2006). They also noted the importance of retaining relatively palatable species on saline land to maintain some productivity (Bann and Field 2006). As a result, they identified eight species of moderately palatable native grasses which persisted through all life cycles in saline conditions, including *Themeda triandra* (Kangaroo Grass), *Austrodanthonia* spp. (Wallaby Grass), *Chloris truncata* (Windmill Grass), *Bothriochloa macra* (Red Grass), *Sporobolus creber* (Slender Rat-tail Grass), **Cynodon dactylon* (Couch), *Elymus scaber* (Common Wheat Grass) and *Dichelachne micrantha* (Short-hair Plume-grass). The authors suggest that trials involving these species should be conducted to evaluate the potential of these native grasses for revegetating saline sites (Bann and Field 2006).

Ten accessions of **Cynodon dactylon* (Couch), along with accessions from *Sporobolus virginicus* (Marine Couch), *Sporobolus mitchellii* (Rats-tail Couch), **Paspalum vaginatum* (Salt-water Couch), **Paspalum distichum* (Water Couch) and one exotic, **Pennisetum clandestinum* (Kikuyu) were evaluated by Semple et al. (2004). The grasses were planted on two scalded saline sites on the inland slopes of NSW (Semple *et al.* 2004). Most species showed some salt tolerance, but **C. dactylon*, *S. virginicus* and **P. vaginatum* outperformed the others. When cutting was used to simulate grazing pressure, *S. virginicus* consistently produced more leaf/seed head dry matter during the growing season, and **P. vaginatum* consistently produced more stolon/rhizome dry matter indicating an ability to maintain groundcover during regular cutting.

Follow up work by Semple et al. (2006) evaluated the most promising accessions from the 2004 study (**Cynodon dactylon* (Couch), *Sporobolus virginicus* (Marine Couch), **Paspalum vaginatum* (Salt-water Couch), and *Sporobolus mitchellii* (Rats-tail Couch)) as well as two new species *Eragrostis dielsii* (Mallee Love-grass) and *Distichlis distichophylla* (Emu-grass) at three new sites. The authors found that *S. virginicus* was the most tolerant of saline conditions and produced high levels of groundcover and biomass, but was sensitive to defoliation. **C. dactylon* and *D. distichophylla* outperformed the other species in alkaline conditions but were considered to have low forage value (Semple *et al.* 2006). **Paspalum vaginatum* performed well at one site with high biomass and moderate forage value but performed poorly at the two other sites (Semple *et al.* 2006). *S. mitchellii* and *E. dielsii* performed well on the less saline sites but produced low biomass but may be useful for environmental plantings where low weed potential is critical (Semple *et al.* 2006).

Rogers (2007) built upon earlier work which produced 2 cultivars of *Austrodanthonia* (*A. richardsonii* (Link) H.P. Linder (syn. Danthonia richardsonii Cashmore) cv. Taranna and *A. bipartita* (Link) H.P. Linder (syn. D. linkii Kunth) cv. Bunderra) for commercial use. The cultivars had been selected from a range of naturally occurring ecotypes primarily due to their relatively high seed production. Both cultivars are suitable for a wide range of rainfall zones (400-800mm) and have been recognised as having some tolerance for adverse conditions, such as low pH. Rogers (2007) expanded this work by looking at the tolerances of these cultivars as well as two other grasses [*]*Lolium perenne* (Perennial Rye Grass) and [*]*Thinopyrum ponticum* (Tall Wheat Grass)) to saline irrigated conditions in northern Victoria. Dry-matter yield and other physiological and agronomical traits were measured (Rogers 2007), and results showed high levels of salt tolerance in the two *Austrodanthonia* cultivars at rates which were comparable to that of perennial ryegrass (a moderately salt tolerant exotic grass). From this research Rogers (2007) suggests that the selected *Austrodanthonia* cultivars are suitable forage grasses for saline conditions that may result from irrigation with saline water or in saline discharge areas. However, further research on establishment methods for saline areas and on the ability to withstand the added pressure of grazing is required if they are to be integrated into current production systems (Rogers 2007).

The research into land management issues and native grasses has focused upon the potential of native grasses for revegetating salinised land with a wide range of salinity levels. The literature suggests that perennial native grasses may have a very useful role to play in managing saline land. Species such as *Lachnagrostis filiformis* (Common Blown Grass), *Eragrostis infecunda* (Southern Cane-grass), *Eragrostis dielsii* (Mallee Lovegrass), *Distichlis distichophylla* (Australian Salt-grass) **Cynodon dactylon* (Common Couch), *Sporobolus virginicus* (Marine Couch) and *Austrodanthonia* spp. (Wallaby Grass) have the potential to be used on saline soils and should be further investigated to establish the value of producing marketable cultivars. Further research is also required to address the most significant gaps in the knowledge of native grasses and saline land, including the use of native grasses on mildly saline sites, the productivity value of salt-tolerant species and managing the long term persistence of native grasses on saline sites.

4.3 LAND MANAGEMENT: SOIL HEALTH

4.3.1 Problems and causes

Australian soils are ancient and for the most part shallow, fragile, and nutrient-poor (Whalley 1990; Harte 1992; White 1994). Most of the nutrients occur close to the surface and nitrogen levels are critically low except where native leguminous vegetation occurs (Harte 1992). Unlike the other continents, Australia has undergone very little soil-building activity such as volcanism or glaciation, with the consequence that the continent is generally flat and the soils are highly weathered (White 1994). The resulting soils are low in essential nutrients and many are also saline (White 1994). These uniquely harsh conditions have helped shape the evolution of native grasses and native grass communities (Whalley 1990).

In South Australia, the soils of the arable farming lands are mainly duplex soils, characterised by a distinct boundary between the A and B horizons (Harte 1992). The B horizon is usually dense and clayey, with restricted hydraulic conductivity which can lead to water logging, especially in winter when evaporation rates are low (Harte 1992). The density of the clay is such that root penetration into the subsoil is often physically impeded, and periodic saturation leads to reduced yield potential. Erosion is also a serious issue with these soils, particularly if they are saline (Bruce *et al.* 2005).

Despite these drawbacks descriptions of the topsoil from the time of settlement often refer to the soft and fluffy nature of the soil surface (Whalley 2003). In his investigation of settlement history and land management in Australia, Whalley (2003) suggests that there is at least some evidence that the topsoil, on some soil types and at some locations, was soft and friable, probably with high organic matter, high water infiltration rates and rapid gas exchange across the soil surface. These conditions would be linked to high rates of nutrient cycling, high levels of available nutrients and high plant growth rates in natural grasslands (Whalley 2003). It seems almost certain that the original vegetation was a highly efficient and productive system even though the dominant soils were shallow and nutrient-poor by world standards.

The traditional view from the 1950s viewed pasture as a crop harvested by grazing animals, and thus any uneaten foliage was seen as a wasted crop (Whalley 2003). This ignored the needs of the soil/air interface and the other living organisms within the pasture and soil, and led to a reduction in biological activity in the soil (Whalley 2003). The formation of surface seals and a reduction in water infiltration rates were also related (Whalley 2003). Whalley (2003) suggests that the original, friable topsoil disappeared rapidly with the introduction of sheep and cattle and was replaced by hard packed topsoil. While this would be partly due to compaction caused by the replacement of soft-footed marsupials with hard-hoofed stock, erosion form loss of vegetation cover and reduced soil biological activity would also have played a part (Whalley 2003). There is some suggestion that the loss of small marsupials that dug up the soil looking for food has also had an impact on soil health (Whalley 2003).

Soil degradation in the forms of erosion, loss of fertility and exotic pasture decline, may be associated with the decline of sown species resulting from droughts or poor grazing management. Whalley (2003) points out that there is little difference between the topsoil characteristics of 'sub and super' pastures treated more than 10 years ago and the topsoil of pastures that have never had this treatment. This strongly suggests that building up soil fertility to support exotic pastures is not a sustainable practice, and that native grasses are better suited to Australian soils than other species. An alternative management strategy currently gaining support, views the main goal of a grazing system as building topsoil fertility, and treats the grazing animals as part of the grassland ecosystem (Whalley 2003).

4.3.2 The role of native grasses in soil health

The condition of the soil/air interface in a native grassland is critical for many ecological processes: infiltration of rainfall into the soil, gas exchange, and incorporation of plant litter into the soil surface so it is accessible to soil animals and micro-organisms (Jones 2000; Whalley 2003). In soils with good topsoil condition the rates and efficiency of these processes will be enhanced (Whalley 2003). There are many interactions between grazing animals, grassland plants and other organisms that need to be managed in order to create healthy topsoil. The interactions and their effects might include defoliation, selective grazing of certain plants, animal impact, soil compaction, incorporation of litter into the soil, and the transport of plant nutrients by grazing animals in spaces and time (Whalley 2003). Native grasslands are thought to benefit soil health in a number of ways, including the provision of groundcover, water use efficiency, and subsoil biomass (Pittaway 2004).

4.3.3 Effects of grazing and cropping in soil health

Wilson and Lernon (2004) investigated the condition of contemporary native grassland soils with that of the relatively undisturbed grassy woodlands from which they are often derived. Their study compared the properties of both native pasture (heavily grazed and native pasture) and cropping soils (cultivated for at least 30 years) with adjacent remnant woodland at four representative sites near Bingara on the North-West Slopes of NSW (Wilson and Lernon 2004). Their results showed that the heavily grazed, unimproved native grassland soils had lower pH, lower organic carbon and lower nutrient contents compared with remnant woodlands, indicating that the contemporary native grassland soils (derived from grassy woodlands) were significantly poorer in nutrients than the woodland soils (Wilson and Lernon 2004). Grazing and clearing activities on the native pasture are thought to have led to the loss of large quantities of organic carbon and nutrients and caused acidification of the soil (Wilson and Lernon 2004). Cropping soils were lower in all of the soil properties measured compared with other management types, except where soils had been recently fertilised, although nutrient content still did not significantly exceed that found in woodland systems (Wilson and Lernon 2004).

Jones (2000) studied grazing management with the intent of improving soil health. The author noted that grazing removes leaf area and results in root pruning, while resting from grazing enables root strengthening. Overgrazing of pastures ultimately results in plant death, but over-resting also results in pasture degeneration (Jones 2000). In medium to low rainfall areas, grasses which are not grazed become senescent and cease to grow productively (Jones 2000). The author suggests that good grazing management should involve intermittent root pruning through pulsed grazing to stimulate the growth of new leaves and to provide sloughed roots (soil organic matter) as food for soil biota. Jones (2000) recommended using grazing cells or shepherding to bunch stock together and move them frequently to improve soil health in low to medium rainfall areas.

Pasture cropping and soil health

Bruce et al. (2005) considered the effect of pasture cropping on biomass production, total cover, soil water and potentially available soil nitrogen compared to more conventional cropping and grazing enterprises. They compared three treatments: C_4-dominant native perennial pastures, winter oats crops, and pasture-cropping (winter oats drilled into C_4-

dominant native perennial pastures). The pasture crop treatment resulted in greater ground cover compared with the cropping treatment; soil water content was reduced in the pasture crop treatment compared to both the crop or pasture treatment; and nitrogen availability was reduced and less variable in the pasture crop treatment (Bruce *et al.* 2005). The authors concluded that the separation of C_3 and C_4 growing periods and the mix of shallow-rooted and deep-rooted plants in the pasture-crop treatment have a number of potential benefits compared to cropping and grazing (Bruce *et al.* 2005). The increased total biomass in the pasture-crop system caused a reduction in both soil water and nitrogen availability, reducing the likelihood of water logging, dryland salinity and nitrogen loss through soil acidification. The increase in total year-round ground cover and increased total biomass would also result in reduced wind and water erosion, weed outbreaks and increased soil organic matter (Bruce *et al.* 2005).

Jones (1999) described the benefits to the soil from using the pasture cropping management technique, as developed by Cluff and Seis. This involves direct drilling of annual C_3 crops such as wheat or oats into permanent perennial native pasture (mostly dormant C_4 grasses) without cultivation or chemical fallow (Jones 1999). This technique resulted in improved percentage ground cover, soil structure, organic matter, water holding capacity, crop health and the biodiversity and biomass of degraded grasslands. During sowing and establishment of the crop, the roots of the dormant C_4 grasses helped to maintain soil structure and reduce erosion risk, while groundcover increased litter, maintained biological activity and reduced weed invasion (Jones 1999). The result was above average grain yields, with very low input costs and a pasture that was available for grazing immediately after the crop harvest (Jones 1999).

Generally, research investigating the effects of native grasses on soil health is limited. There have been a few studies in key areas including soil acidity, soil microbes, erosion, and nutrient content, and as previously discussed, soil salinity. There have also been several studies examining land management practices (such as pasture cropping) that incorporate native grasses and their benefits to aspects of soil health. However, more research is needed in these areas, particularly regarding specific cultivars or species mixes for managing or treating different soil problems such as nitrogen leaching and soil fertility.

4.3.4 Nitrification

One of the key ecosystem functions relating to sustainable land use under threat in southern Australia is nitrogen leaching (Garden and Bolger 2001). Nitrogen leaching, or denitrification, occurs in soils prone to water logging as high water levels dissolve soil nitrogen and wash it below the rooting zone (Pittaway 2004). Pastures which do not grow actively in summer are unable to utilise the products of organic matter mineralisation (nitrogen in the form of ammonia and nitrate) which reach their highest concentrations during the warmer months (Garden and Bolger 2001; Bruce *et al.* 2005). This increases the potential for nitrogen leaching in the later summer and autumn as the soils become waterlogged (Johnston *et al.* 1999).

Active plant growth year round (but particularly in summer) reduces the risk of water logging and increases the use of available nitrogen in the soil profile thus reducing rates of nitrate leaching (Johnston *et al.* 1999; Bruce *et al.* 2005). Garden and Bolger (2001) found that *Themeda triandra* (Kangaroo Grass) grasslands maintain soil nitrogen at low levels, which prevents invasion by high nutrient requiring annual grasses and maintains a high proportion of perennial ground cover, thus stabilising the system. The use of warm-season native grasses in pastures or as part of a pasture cropping system is recommended to reduce the risk of water logging and associated nitrogen leaching in agricultural systems (Garden and Bolger 2001; Bruce *et al.* 2005).

4.3.5 Soil acidity

Surface soil acidity in south-eastern Australia has been linked with nitrogen leaching and water logging (Gutpa *et al.* 2005). The changed patterns of water use following the replacement of perennial vegetation with annual crops and pastures is the most commonly cited cause of soil acidification, as well as being blamed for dryland salinity and water logging (Lodge 1994; Johnston *et al.* 1999). Declines in surface soil pH have also been linked to topsoil rebuilding practices which are exacerbated by decreases in the perennial grass component of pastures (Whalley 2003), although this has been questioned by some researchers (Harris and Duncan 1999). The use of superphosphate fertiliser and annual legumes to boost the nitrogen content of soils has achieved enormous productivity advances in Australian agriculture, but is thought to cause subsurface acidification (Lodge 1994; Johnston *et al.* 1999; Scott *et al.* 2000; Garden and Bolger 2001; Tang 2004).

Remediating acid soils is usually achieved through liming (Vimpany 1979; Scott *et al.* 2000), but this is generally only effective for topsoil acidity and is considered unfeasible for tackling subsoil acidity (Simpson and Kangford 1996; Tang 2004). However, there are many species that will tolerate acid soils, particularly among native grasses (Langford *et al.* 2004). Several researchers have reported that *Austrodanthonia* spp. (Wallaby Grass) and *Microlaena stipoides* (Weeping Grass) are highly tolerant of soil acidity and associated high aluminium (Garden *et al.* 1996; Simpson and Kangford 1996; McGarva *et al.* 2000; Langford *et al.* 2004). Garden et al. (2001) have found similar tolerances in *Eragrostis* spp. (Love Grass), while Langford et al. (2004) listed *Aristida ramosa* (Wire Grass), *Poa* spp. (Meadow Grass) and *Austrostipa* spp. (Spear Grass) as acid tolerant. Simpson (2000) identified *Austrodanthonia* spp., *Themeda triandra* (Kangaroo Grass), *Bothriochloa macra* (Red Grass), *M. stipoides*, *A. ramosa*, *Poa* spp. and *Austrostipa* spp. as tolerant of acid soils, while Simpson and Langford (1996) stated that *T. triandra* and *B. macra* tend not to grow in strongly acid soils. Scott et al. (2000) suggests that soil acidity is likely to decrease under native perennial pastures due to increased summer growth and (usually associated) decreased stocking rates, although the authors concede that this effect has not been measured.

Despite the above discussion, there is generally little quantitative research on the specific acid tolerance levels of different species and few attempts to identify native grass cultivars suitable for use as a pasture on acid soils. Whalley and Huxtable (1993) investigated the potential of *Microlaena stipoides* (Weeping Grass) for domestication for pasture use. They stated that *M. stipoides* is acid tolerant and able to recover well from droughts (Whalley and Huxtable 1993), but did not examine the response of *M. stipoides* to different soil acidity levels in their study. Their investigation did find that there was a wide range of genotypes within populations of *M. stipoides*, making it a suitable candidate for identifying valuable cultivars for future domestication. A number of

Microlaena species are already available and traded as “Griffen”, ‘Shannon”, “Wakefield”, “Ovens” and “Bremmer”. “Ovens” and “Bremmer” originated from the LIGULE project (Johnston *et al.* 2001). Domestication efforts could be aimed at producing cultivars for marginal soils (including acidic soils) and drought prone areas where traditional exotic pastures have failed (Whalley and Huxtable 1993).

4.3.6 Compaction

Compaction of soil can be caused by machinery, deep tillage practices and grazing animals (Harte 1992). The pressures exerted by grazing animals on the soil surface are comparable with those from agricultural machinery (Greenwood *et al.* 1997). Another form of compaction is caused by the loss of organic matter, which can affect soil structure leading to disintegration of aggregates and consolidation during wetting (Harte 1992). Surface sealing often occurs as a related problem, forming a hard surface crust which impedes plant growth, water infiltration and contributes to water run off and erosion (Harte 1992). No information regarding differences between native grasses and other vegetation on the effects or degree of soil compaction were noted in the literature, but there have been some investigations on the effects of native grasses and soil organic matter, which directly relates to soil structure.

4.3.7 Soil organic matter

Soil organic matter (SOM) (also referred to as soil organic carbon) is an important indicator of soil health and agricultural sustainability (Lodge and King 2004; McMullen and Schwenke 2004). It drives vital soil, chemical and biological processes and plays an important role in soil fertility (Harte 1992; McMullen and Schwenke 2004). It is an important source of nutrients for plants, has a very high cation exchange capacity, maintains soil aggregation and is a food source of soil microbes (Harte 1992; McMullen and Schwenke 2004; Baldock *et al.* 2007). SOM is also viewed as a potentially important global carbon sink (McMullen and Schwenke 2004).

Generally, Australian soils are inherently low in SOM (between 0.5 and 4%) (Harte 1992; McMullen and Schwenke 2004). Significant amounts of SOM have been lost through topsoil erosion, while crop production has resulted in the rapid decline of organic fertility as SOM is mineralised to produce essential crop nutrients such as nitrogen, phosphorus and sulphur (Harte 1992; McMullen and Schwenke 2004; Baldock *et al.* 2007). Loss of SOM has resulted in declines in chemical fertility as well as soil structure, increasing the risk of erosion and compaction, and creating problems for plant establishment and growth (McMullen and Schwenke 2004).

The widespread change of native grasslands to agricultural cropping and introduced pastures has significantly altered the SOM cycle, changing the quantity and frequency SOM return to the soil and changing the soil moisture balance (McMullen and Schwenke 2004). Cultivation affects SOM by diluting organic-rich topsoil when it is incorporated with deeper soil material that is naturally lower in SOM, as well as by exposing previously protected SOM to microbial breakdown, causing a loss of SOM (Harte 1992; McMullen and Schwenke 2004). Management strategies used to increase SOM include reduced or no-till cropping and stubble retention (Baldock *et al.* 2007), but one of the most effective ways of improving SOM is through pasture phases in cropping systems (McMullen and Schwenke 2004).

McMullen and Schwenke (2004) investigated no tillage farming in the northern grains areas of NSW. They measured sustainability by calculating the active fraction of SOM and water filtration of soil samples at various sites as compared to a nearby reference site that had not been cropped or heavily stocked (McMullen and Schwenke 2004). The results showed a decline in the soil carbon content at the tillage stage of all of the crop rotation treatments (McMullen and Schwenke 2004). In the crop rotation treatments, there was a small ameliorative effect of no-tillage compared to tilled sites, while water infiltration and soil aggregation was higher in the minimum or no-till sites than in the cultivated sites (McMullen and Schwenke 2004). Large declines in both labile and total carbon were observed between all the continuously-cropped soils when compared to the nearby reference pastures (McMullen and Schwenke 2004). The authors concluded that no till or minimum till farming systems combined with practices that maximise organic returns to the soil, such as pastures, are important factors in maintaining soil fertility.

Native pastures encourage soil organisms by supplying a constant energy source, which improves SOM (McMullen and Schwenke 2004). The use of pastures in cropping systems is one of the most effective means of increasing SOM and sustainability of crop production. No-till systems can slow the rate of SOM decline, but combining this with pasture phases is much more effective (McMullen and Schwenke 2004). Using pastures in the cropping sequence also has other benefits in providing effective weed and disease breaks (McMullen and Schwenke 2004).

Lodge and King (2004) presented preliminary results from a study looking at the microbial biomass carbon and labile carbon content of soils in northern NSW. Sites with contrasting ground cover and herbage mass levels were examined, including native and sown perennial based pastures (Lodge and King 2004). No analysis of the differences between native and sown pastures was presented. However, the authors did find that both microbial biomass carbon and labile carbon were highest in the top 2.5 cm of soil and tended to decline as soil depth increased. Total microbial carbon and labile carbon levels were always lower in the low ground cover/herbage mass sites compared with those of high ground cover/herbage mass (Lodge and King 2004).

The effects of land management practices and the presence of native grasses on SOM are not yet fully understood. However, the beneficial role that SOM plays in the overall health and fertility of soils is well established. Initial investigations strongly suggest that well-managed native grasslands can have a positive effect on SOM content, and it is recommended that these relationships are further examined so that practical guidelines for maximising SOM using native grasses can be developed.

4.3.8 Microbes and other soil organisms

The health and number of organisms in the soil are important for the growth of healthy crops and pastures (King 1997; Pittaway 2004). Microbes and other soil organisms regulate essential soil ecosystem processes required for plant growth, soil health and sustained productivity (Jones 2003; Gutpa *et al.* 2005). Plant nutrients are released and returned to the soil by soil micro-organisms and invertebrates (e.g. nitrogen fixation and mineralisation, phosphorus solubilisation, carbon and sulphur mineralisation), improving the soil structure and fertility (King 1997; Lodge and King 2004; Pittaway 2004; Gutpa *et al.* 2005). Soil structure is also improved by burrowing invertebrates creating networks of tunnels, microbes secreting binding agents and fungal filaments which hold soil particles together, and the passage of soil through earthworms (King 1997; Jones 2003; Pittaway 2004; Gutpa *et al.* 2005). Soil fauna are also thought to affect the plant's ability to withstand disease by preventing aggressive plant pathogens from taking hold, and in agricultural systems are important for the breakdown of chemical herbicides, insecticides and fungicides (Gutpa *et al.* 2005).

Soil fauna rely on organic matter as a food source and for habitat, so activity levels are strongly influenced by the distribution and quantity of organic matter within the soil (King 1997; Pittaway 2004). Moisture, temperature and aeration also have an effect on fauna activity levels (Hutchinson and Roper 1985; King 1997). In Australia, soil biological activity is restricted by an overall lack of soil carbon and other nutrients combined with relatively short periods of optimum moisture conditions, all of which vary significantly over time (Gutpa *et al.* 2005). Soil fauna are concentrated in the top 5-10cm of the soil profile, an area prone to environmental extremes and loss through erosion (Lodge and King 2004; Gutpa *et al.* 2005).

Pastures with a mixed composition can support large, diverse, below-ground faunal communities (Gutpa *et al.* 2005). Pasture systems are highly variable in morphology, space and time providing many micro-niches to support a variety of soil fauna (Gutpa *et al.* 2005). In contrast, annual crops are monocultures and thus do not support the same levels of soil fauna as a pasture system (Gutpa *et al.* 2005). Overgrazing pastures decreases the number of soil organisms in both native and improved pastures by reducing organic matter and compacting the soil (King 1997; Jones 2003). In pasture systems where there is no tillage, carbon inputs from plant roots and litter are the major regulating factor for biological succession in these soils (Gutpa *et al.* 2005). The management of pasture to benefit soil biota, should consider pasture composition and carbon inputs mediated through grazing management in addition to soil organic matter (Gutpa *et al.* 2005).

Lodge (2004) compared earthworm communities underneath native and sown pastures. While earthworms are not essential to all healthy soil systems, their presence is considered an indicator of healthy systems (Lodge 2004). The author found that earthworm numbers were markedly influenced by pasture type, the application of superphosphate and the over-sowing of subterranean clover, but no effect of stocking rate (Lodge 2004). Earthworm numbers at native grass sites were highest where fertiliser and sub- clover were also applied, approximately 5 times that for unfertilised sites (Lodge 2004). The highest earthworm numbers overall were in the phalaris-sub clover pasture. Counts in the top 10cm of soil were around 200 earthworms per square metre (Lodge 2004). The results suggest that earthworms are highly responsive to nutrient availability and do not seem to prefer native vegetation over sown pasture.

Over the last 50 years the structure and fertility of many cropping soils has declined, despite innovations in tillage, fertilisers, pesticides, and the release of improved plant varieties (Pittaway 2004). Cropping can reduce available soil organic matter and disrupts the habitats of soil fauna (King 1997; Jones 2003; Pittaway 2004). After soil has been cultivated it can take up to 2 years for the larger invertebrates such as earthworms to return to pre-cultivation levels (King 1997). Fallowing is another technique that has a detrimental effect on soil fauna, as very few living things inhabit the soil during long fallows, creating ‘biological deserts’ (Jones 1999). Key advantages of pasture phases over annual cropping identified in the study include: a diversity of plant ages and species reducing the risk of pest and disease outbreaks; much less disturbance which protects soil fauna habitats and leaves networks of fungal filament undisturbed; and increased build up of plant litter which is then incorporated into the soil by invertebrates and broken down by microbes.

Pittaway (2004) also noted there are currently many commercial products available that claim to enhance soil microbial activity, while there are many claims that commonly used pesticides and inorganic fertilisers decrease soil fauna. While rates of fertiliser application are usually too low to greatly affect the general microbial population, some forms of mineral fertilisers will kill microbes on contact (including acidifying fertilisers and those with a high electrical conductivity) (Pittaway 2004). Furthermore, commercial products that claim to enhance soil microbial activity (‘soil activator’ products) are usually applied at rates too low to significantly affect the soil fauna (Pittaway 2004). Pittaway (2004) concludes that heavy grazing, stubble burning and excessive tilling do not favour soil fauna, and recommends conservatively grazed pastures and (in cropping systems) regular pasture phases. Furthermore, Pittaway (2004) suggests instead, that the best way to manage soil fauna is to buffer the soil environment with vegetative cover, diversify cropping to diversify the available forms of organic carbon, reduce frequency of tillage and pesticide application and manage stock to reduce the extent of soil disturbance to favour soil animals.

While the importance of soil fauna (including microbes, insects and earthworms) for maintaining healthy and fertile soil has been well established, the links with native grasses are much less well defined. None of the research reviewed in this document directly links native pastures with larger or more diverse soil fauna communities. Rather, the literature suggests that the action of creating and maintaining exotic crops and pastures (through land clearance, grazing, sowing, fertiliser and herbicide application) are the root cause of perceived declines in soil fauna communities (with the exception of Lodge (2004)). It seems that although the healthiest soil fauna may well be found under native grasslands, it is likely that this is more the result of a lack of destructive management techniques than the actual species composition. It is likely that the best method for maintaining a healthy soil fauna is to use low-impact management techniques on the soil and support a diverse range of plant species. However, further research into the interactions between soil fauna and native grasslands may reveal some symbiosis between the two communities, and in any case it is clear that maintaining healthy and diverse native grassland is a good way of supporting a healthy soil fauna.

4.3.9 Erosion and ground cover

Groundcover is generally considered to be the herbaceous component which may include crops, stubble, pasture plants, leaf litter, bark and twigs (Lang and Holmes 1995; Jones 2003). A lack of sufficient groundcover is the primary cause of wind and water erosion (Lang and Holmes 1995), and is considered to be one of the leading land management issues in Australia (Garden and Bolger 2001). Problems with maintaining adequate groundcover are linked to the widespread replacement of native perennial vegetation with annual crop species (Zollinger *et al.* 2005), as well as overstocking, overgrazing and cultivation (Whalley 1970; Jones 2003; Leys 2003).

Topsoil erosion by water causes the loss of valuable soil, water, organic carbon and nutrients (particularly nitrogen and phosphorus) as well as breakdown of soil structure, stream pollution, silting of water storages, damage to roads and other utilities and increased production costs (Johnston 1992; Simpson and Kangford 1996; Jones 2003). Water erosion can occur at very high rates during intense rainfall events, particularly when groundcover is low (Jones 2003). Groundcover plants reduce the impact of raindrops on the soil, slowing down rates of overland water flow, binding soil particles together and reducing the tendency for soil particles to move down hill (Johnston 1992).

Wind erosion can cause problems such as scalds (areas of bare, hard ground that do not soak up water easily), reduced infiltration of water and burying of infrastructure (Leys 2003). Vegetation helps to control erosion by acting as a blanket to prevent wind from picking up soil particles, absorbing the force of the wind and reducing the wind speed at ground level, and trapping eroded soil particles (Leys 2003).

Recommendations for the required percentage of groundcover to prevent/minimise soil erosion range from 50% (Zollinger *et al.* 2005), 70% (Lang and Holmes 1995), 75% (Johnston 1992) 80% (Simpson and Kangford 1996) and 100% (Jones 2003). The benefits of maintaining good ground cover are not only decreased erosion, but also include better use of rainfall from decreased surface runoff, better access to nutrients, higher levels of organic matter and increased soil depth (Lang and Holmes 1995). There has also been some suggestion that good management of ground cover may actually help rebuild topsoil (Jones 2003).

Maintaining permanent ground cover may be achieved using perennial native grasses, many of which are able to grow year-round. Heavily grazed and/or senescent pastures experience higher rates of soil erosion than actively growing pastures which cover the ground (Johnston *et al.* 1999). Ploughing, ripping, grading and other soil disturbances destroy native grasslands and favour the establishment of exotic annual species, exposing the land to further risk of erosion (Lunt 1991). Conservation farming practices that maximise retention of crop residue and reduce tillage are recommended, and during droughts, de-stocking should take place early (Leys 2003).

Once a focal point for soil erosion has established, erosion usually increases in area unless positive steps are taken to allow the regenerative process to commence and repair the damage (Johnston 1992). This process is entirely dependent upon revegetation (Johnston 1992); mechanical and chemical binding methods of controlling erosion are just temporary solutions (Leys 2003). Events that reduce ground cover, such as cultivation or burning should coincide with the season of mildest rainfall (Lang and Holmes 1995). For example spring/summer burns leave the land vulnerable to erosion following intense summer rain events (Lang and Holmes 1995). Other factors affecting vulnerability to erosion include the soil type (poor infiltration and shallow soil) and slope (Lang and Holmes 1995). These areas will need higher levels of ground cover maintained to prevent erosion than flat, well-drained soils (Lang and Holmes 1995).

Zollinger et al. (2005) conducted a study in western Victoria to identify the impact of various grazing strategies on pasture composition, density and erosion risk on steep hill country. The authors found that deferred grazing (withholding defoliation from late spring until the autumn break) increased herbage mass of perennial grasses (mostly natives) and reduced that of annual grasses. The treatment also led to greater ground cover, leading to lowered risk of erosion on hilly country (Zollinger *et al.* 2005). This is one example of how native grasses can be used in conjunction with alternative agricultural practices to improve land management.

The processes and causes of erosion are generally well understood. However, in practice, reversing the effects of erosion can be an expensive and difficult process. The characteristics of native grasses in reducing the risk of erosion include year-round green foliage, good ground coverage and increase herbage mass above and below ground. Although these characteristics are beneficial for reducing erosion risk, it is likely that vegetation management strategies such as reduced grazing pressure (especially during drought) and minimising tillage will have a larger effect on managing erosion risk than species composition. It is likely that a combination of diverse, predominantly perennial, drought resistant pasture and conservation farming techniques will be the best method of managing the risk of erosion.

4.4 LAND MANAGEMENT: WILDFIRE RISK AND NATIVE GRASSES

4.4.1 How can native grasses be used to manage wildfire risk?

Fire can have a devastating effect on communities, agriculture, infrastructure, stock and vegetation. In grasslands as in other vegetation, fire that is too frequent or at the wrong time of year can cause degradation by damaging the vegetation, reducing ground cover and laying bare earth open to wind and water erosion. Fire also affects soil structure and its microbiological and invertebrate faunal content, as well as its organic content (White 1997). Major factors in fire risk of an area include the vegetation type and associated fuel load, both of which can be managed to minimise fire risk, particularly regarding low intensity fires.

There is some suggestion that native grasses can be used to reduce the risk of wildfires. Exotic grasses with high biomass may alter the fuel dynamics and fire behaviour of grassy areas such as road sides (Stoner *et al.* 2004). Stoner et al. (2004) compared the fine fuel biomass of unmanaged, dense *Themeda triandra* (Kangaroo Grass) grasslands with adjacent exotic grasslands dominated by **Phalaris aquatica* (Phalaris). The authors found that fine fuel biomass of the **P. aquatica*-dominated sites ranged from 22.8 t/ha to 31.1 t/ha, while the *T. triandra* sites ranged from 6.9 t/ha to 8.7 t/ha. This translates to a three-fold increase in fine fuel biomass from *T. triandra* to **P. aquatica*, which could have serious implications for fire management (Stoner *et al.* 2004).

Douglas and O'Connor (2004) conducted a study comparing the fine fuel loads of the invasive **Urochloa mutica* (Para Grass) to that of native grasses *Oryza meridionalis* (Wild Rice) and *Hymenachne acutigluma* (Hymenachne) in the Kakadu National Park (NT). The authors found that the fuel loads differed significantly between the **U. mutica* and the two native grasses when sampled towards the end of the dry season (October), when fire risk is highest. The highest fuel load recorded was 49 tonnes/ha in deep **U. mutica*.

In the floodplain margins, **U. mutica* had nearly double the fuel load of *O. meridionalis*, although **U. mutica* had significantly higher fuel load than *H. acutigluma* at only one of the four sites. **U. mutica* was also on average eight times taller than *O. meridionalis* and twice as tall as *H. acutigluma*. The authors concluded that **U. mutica* has very different fuel characteristics to the two native grasses it has displaced, and predicted that fire intensities and flame heights would increase where **U. mutica* has displaced *O. meridionalis*. They noted that although **U. mutica* had similar fuel loads to that of *H. acutigluma*, the former has taller, drier fuel making it more likely to burn. Although this was a study of a tropical savannah, the results indicate that further study should be conducted on the fuel loads of exotic grasses in temperate regions compared with native grasses.

Noble (1991) reported on a wildfire in the western Riverina (NSW) in 1987. The author investigated grass fuel loads in contrasting sites in the vicinity of the fire - one where the dominant species were **Avena* spp. (wild oats) and **Lolium rigidum* (Annual Ryegrass) and the other dominated by *Austrodanthonia caespitosa* (White top) and *Austrostipa nodosa* (Spear Grass). The fuel loads in the wild oats/ryegrass areas averaged 2.1 t//ha, while the Danthonia/Stipa sites averaged only 1.2 t/ha (Noble 1991). However, although the native grassland had significantly lower fuel load, the intensity of the fire was such that both native and exotic dominated grasslands were burnt out in the fire.

There has also been some suggestion that the characteristics of perennial native grasses, particularly the provision of green foliage in summer rather than dead, dry biomass, may also reduce fire risk (Wayne Brown, pers. comm.). However, there has not been any research into whether or not green stands of native grasses burn with less intensity, or catch alight less frequently, than senescent stands of exotic grasses. Further study to determine the ignition point and combustion characteristics of different grass species (native and exotic) in under different management regimes is required to assess the potential for using native grasses to minimise fire risk.

Although managing vegetation type and fuel load can help reduce the risk of low to moderate intensity fires it is unlikely to have much of an effect in the face of a major, high intensity bushfire. Minimising the fuel load build up along roadsides by choosing appropriate grass species may reduce the risk of fires taking hold and developing into larger fires. However, many factors influence the ignition and development of fires and more research is required to assess reduction in fire risk (if any) associated with using native grass species in fire prone areas.

4.5 LAND MANAGEMENT: NATIVE GRASSES, CLIMATE CHANGE AND CARBON

4.5.1 Background – climate change and native grasses

Climate change and its potential effects on agricultural land has become increasingly important for land managers. Associated with these concerns are issues such as increasing severity of drought, increasing variability of rainfall and the effects of elevated CO_2 on plant growth, including introduced pastures and native grasses. Extensive information on current models and predictions regarding climate change, links between greenhouse gasses and global temperature increases has been extensively reported in the literature. However, the role of native grasses in ameliorating some of these climatic effects has yet to be examined in greater detail.

Native grasses evolved in Australia under conditions of severe aridity and cyclical droughts, and there is a suggestion by White (1986), Whalley (1990) and Jones (2003) amongst others that native grasses may provide a more resilient pasture or ground cover than other species in times of drought and climatic extreme. Adapting to climate change requires a detailed and thorough account of the drought tolerance of native grass species suitable for a wide range of environmental conditions and human industry applications. In addition, the potential for grasslands to act as carbon sinks thereby facilitating a reduction in the effects of climate change are only now beginning to be explored.

4.5.2 Adapting to climate change – drought tolerance in native grasses

As climate change threatens, the prospect of increased temperatures and rainfall variability and of more intensive and more frequent droughts is both more probable and more demanding. Research into the characteristics of grasses, particularly native grasses, to cope with these extremes is both imperative and urgent.

The suggestion that native grasses are tolerant of aridity and drought is well supported by their evolutionary history (Jones 2003). Native grasses evolved under the influence of a fluctuating climate with periods of severe aridity and extended drought (Whalley 1990; White 1986). However, although drought tolerance is often quoted as a virtue of native grasses few studies exist that include empirical data on the characteristics of individual species that enable them to survive drought conditions (Simpson and Kangford 1996; Bolger *et al.* 2005). Many native grass species have been identified as having some capacity to tolerate drought, but these are often based on anecdotal information. For example, Simpson (2000) identifies *Themeda triandra*, *Bothriochloa macra*, *Aristida ramosa*, *Austrodanthonia* spp., *Microlaena stipoides*, *Poa* spp. and *Austrostipa* spp. as drought tolerant but does not provide quantitative data to support this assertion .

Garden et al. (1996) suggested a low mortality is characteristic of *Microlaena stipoides* (Weeping Grass) in response to drought. They looked at the response of native and exotic grasses to varying drought intensities. The authors reported high mortality for several exotic species and nil or low mortality for both *M. stipoides* and *Austrodanthonia richardsonii* (Straw Wallaby Grass). Inconsistencies between studies regarding drought tolerance may reflect differences between ecotypes of species from different areas, artefacts of different methodologies or may represent the fact that drought tolerance is a general term that covers a range of plant physiological features and ecophysiological responses, and it is difficult to study all of these factors and their interactions within the scope of a single investigation.

Scholz (1995) refers to three subdivisions of C_4 grasses based on biochemical and physiological variations: NADP-ME, PCK and NAD-ME types. The NADP-ME types dominate in wetter tropical zones (e.g. *Digitaria brownii* [Cotton Panic Grass]), PCK types (e.g. *Eriochloa crebra* [Tall Cup Grass]) tend to be prevalent in regions with approximately 350mm annual rainfall. The two types tend to be largely drought avoiders. NAD-ME types are more dominant in arid environments and are predominantly tolerators or a combination of tolerator/avoiders of drought (e.g. *Eragrostis eriopoda* [Woolybutt]).

'Drought avoiders' also includes those species that employ drought evasion e.g. some 'drought escapers' set large quantities of seed rapidly before the onset of desiccation (Scholz 1995). Species in this group tend to occupy areas which receive water from drainage or in low lying areas that either gather moisture or limit moisture loss, as to withstand dry periods. Drought tolerators tend to use mechanisms or strategies including limiting moisture loss, conserving moisture, and conserving or reducing growth, thereby maximising persistence in a landscape under stress.

An important consideration of drought tolerance is the capacity of an entire plant community to resist drought, rather than a single species. Tilman and Downing (1994) conducted a long term study which showed that primary productivity in more diverse grassland communities is more resistant to, and recovers faster from, a major drought. They found that the most species-rich plots reduced their biomass by about half during the drought, whereas the most species-poor plots reduced their biomass to only one eighth of their pre-drought biomass in the same period (Tilman and Downing 1994). This study suggests that preservation of diversity is essential for maintaining stable productivity in drought-prone ecosystems (Tilman and Downing 1994). The study was based in Minnesota, but the conclusions are relevant to Australian grasslands, particularly for drought resilience in agricultural systems.

Unfortunately, despite assertions that a range of native grasses are drought tolerant, there has been very little quantitative research in this area. More research needs to be undertaken to identify appropriate drought tolerant species for a range of locations, likely to undergo drought and/or climatic stress. Research also needs to include community and catchment-scale effects of drought on native pastures, both their capacity to survive and their productivity.

4.5.3 Native grasses and their soils as carbon sinks

Temperate grasslands contain approximately 304 gigatonnes of carbon, and are the third biggest terrestrial carbon store in the world after boreal forests (559 gigatonnes) and tropical savannahs (330 gigatonnes) (Neilson 2005). The majority of carbon in temperate grasslands is stored as soil carbon rather than above-ground vegetation (Casella and Soussana 1997; Neilson 2005). Agriculture is a significant emitter of carbon dioxide and other greenhouse gasses (Neilson 2005). Groundcover management of temperate grasslands, whether as grazing pastures or pasture phases in cropping systems can determine whether agricultural soils act as a source (net loss) or sink (net gain) of atmospheric carbon (Newton *et al.* 1995; Jones 2007).

Carbon sequestered by plants during photosynthesis, continues through the system in three ways. First, as CO_2 respired back into the atmosphere by animals feeding on plants; second, as methane released by ruminant grazers; and third, as sequestered as a component of the soil (Whalley 2007). One feature of perennial native grasses is that any leaves that are not eaten senesce and are trampled by herbivores into the soil where they are incorporated by soil organisms into soil carbon (Whalley 2007). Furthermore, the fibrous roots of perennial grasses are relatively short-lived (particularly if grazed intermittently) and thus are also incorporated into the soil carbon (Whalley 2007).

'Regenerative management' of grasslands focuses on activities which maximise ground coverage and includes controlled grazing practices, pasture cropping, stubble retention and no-till farming. It aims to maximise retention of plant detritus from native grasses to increase soil carbon while still allowing production through grazing or systems like pasture cropping (Whalley 2007).

Using these techniques, Jones (2007) suggests that a conservative estimate of soil carbon increase of 0.15%, can be achieved every year. This may continue for many years since although the upper limit to soil carbon accumulation will vary, many Australian soils should be able to sequester around 5 times their current level (Jones (2007). Regenerative management to increase soil carbon also leads to a host of other benefits including - improved soil structure, lower bulk density, greater porosity, higher infiltration rates, more effective use of rainfall, enhanced water quality, higher cation exchange capacity, greater sequestration of nitrogen and sulphur, enhanced availability of phosphorus and trace elements, reduced costs, reduced inputs, improved biodiversity and increased productivity (Jones 2007).

Under climate change resulting from elevated atmospheric CO_2, there are several factors which may influence the carbon sequestration potential of grasslands including: a sustained increase in plant productivity resulting from CO_2 enrichment; decreased decomposition of roots and plant residues due to a larger carbon to nitrogen ratio; and increased air temperature resulting in increased soil respiration and soil organic matter decomposition (Casella and Soussana 1997). The first two, increased plant productivity and decreased decomposition of plant residues, will lead to increased carbon sequestration potential of grasslands, while increased soil respiration and organic matter decomposition will result in decreased sequestration potential (Casella and Soussana 1997).

Casella and Soussana (1997) investigated these effects in a perennial ryegrass sward. The authors grew perennial ryegrass in large containers and subjected them to different fertiliser rates, CO_2 levels and temperature, with irrigation adjustment to simulate soil moisture deficit over summer (Casella and Soussana 1997). Elevated CO_2 resulted in increased net carbon assimilation of 29-36%, with the higher values being associated with higher nitrogen supplies (Casella and Soussana 1997). The below ground carbon storage was also increased under elevated CO_2 conditions, 32% at the low nitrogen rate and 96% at the high nitrogen rate. No effect of temperature was recorded (Casella and Soussana 1997). The authors concluded that due to the effects of increased productivity and changes in the carbon to nitrogen ratio, carbon sequestration rates in grasslands can be expected to increase under increased CO_2 concentrations (Casella and Soussana 1997). Furthermore, these effects can be increased by the application of nitrogen fertiliser, at lease in the case of perennial ryegrass (Casella and Soussana 1997).

Soil carbon content was compared between a natural grassland, a long-term maize field and a forest that had been planted on previously cropped land 20 years previously in north-eastern Italy by Del Galdo et al*.* (2003). The authors found that after a century of conventional agricultural use, soil under the maize crop had 48% less carbon in the top 10cm than the natural grassland. They also found that this effect was reversible to some extent after reforestation of cropland, as after 20 years the reforested site had 24% more soil carbon in the top 10cm than the cropped land (Del Galdo *et al.* 2003). This study shows that in a temperate grassland ecosystem, long term cropping (~100 years) significantly reduces soil carbon content, but reforestation can partially remediate the carbon loss. The study did not look at the effects of returning cropped land to natural grassland but confirms the effects of traditional agricultural activities on soil carbon of natural grassland systems.

Williams et al (2004) investigated the carbon sequestration potential of savannas in the Northern Territory. Although tropical savannas differ markedly from temperate grassland systems, the problems inherent in measuring soil carbon sequestration/emission are similar for both types of grasslands. The authors found that previous estimates of carbon sequestration rates in tropical savannas did not include the effect of fire on the grasslands, and thus were inaccurate (Williams *et al.* 2004). They modelled the effect of fire on carbon sequestration in savannas and found that the estimated sequestration rates fell by two thirds compared to the original estimates (Williams *et al.* 2004).The authors point out the differences between measuring Net Ecosystem Productivity (NEP), where carbon sequestration is measured as carbon gained from net primary productivity minus carbon loss from heterotrophic respiration, and the more realistic Net Biome Productivity (NBP), where carbon sequestration is measured as NEP minus carbon losses due to disturbance (such as harvesting, fire, insect plagues). This study highlights the difficulties in modelling and predicting carbon sequestration in grassland ecosystems on a large scale, particularly where environmental variables such as fire are involved.

Comis (2004) reported on recent advances in understanding of soil composition. The discovery of 'glomalin', a previously undescribed soil constituent, has caused a re-examination of the composition of soil organic matter. Glomalin contains 30-40% carbon, and has been shown to account for a significant proportion (27% in one study) of the carbon contained in soil (Comis 2004). This carbon sink had not previously been documented, as glomalin is an unusually strong soil binding material, and traditional methods of ascertaining soil carbon content had not detected it (Comis 2004). Furthermore, glomalin has been shown to last for seven to 42 years, depending on local conditions, so can be considered a viable carbon storage system (Comis 2004). Research has shown that glomalin levels in soil increase under higher atmospheric CO_2 levels, and under no-till farming methods (Comis 2004). Although no research has been conducted on any links between glomalin levels and pasture or crop composition in Australia, the links between glomalin and native grasslands remain to be investigated.

The concept of sequestering carbon in soil through regenerative management of native grasslands is a very new one and as such there has been very little research directed towards carbon sequestration in soils or the role that native grasses may play in this (Jones 2007). Soil carbon has the potential to sequester more carbon for longer time periods than any other terrestrial carbon sink, but it is inherently vulnerable to loss through land use change and soil degradation (Del Galdo *et al.* 2003). However, measuring and modelling the actual carbon sequestered in temperate grassland soils is surprisingly difficult (Williams *et al.* 2004). Increased net primary productivity as a result of CO_2 enrichment may lead to higher rates of carbon sequestration in grasslands, but differences in the response between natural and exotic pastures have not been adequately investigated. Grazing pressure may increase carbon sequestration but this is yet to be fully explored. Furthermore, there is currently little or no financial incentives for landholders to manage the land for soil improvement (Whalley 2007), despite recent a recent pilot program in soil carbon accreditation (ASCAS). More research on the soil carbon system is urgently required to address all of these issues.

4.5.4 The Australian Soil Carbon Accreditation Scheme (ASCAS)

The ASCAS is a private carbon accreditation scheme set up as a 3-year trial in Western Australia, launched in Katanning in March 2007. The project was founded by soil biochemist and Carbon for Life CEO Dr Christine Jones. Funding is being provided by Rio Tinto who will purchase the carbon credits at a fixed price of $25/tonne of atmospheric CO_2 (Jones 2007). The project is also being run in conjunction with state departments (e.g. West Australian Department of Agriculture) and local NRM Boards. With a compulsory federal domestic carbon trading scheme scheduled for introduction in 2010 (as announced by Prime Minister Rudd in 2008) the carbon trading market in Australia is set to take off, and this trial will assist the development of technologies to support the industry. The project also aims to reverse land degradation associated with declines in soil fertility and erosion and give farmers an alternative and reliable income from the land (Jones 2007).

The carbon credits must be certified through an Accredited Certificate Provider (ASCAS) in order to qualify for the scheme. Defined Sequestration Areas (DSAs) are set up when a participant enters the scheme, and involve soil sampling and analysis to set the baseline values of soil carbon in each DSA and determine the magnitude of any subsequent increases (Whalley 2007). ASCAS sampling protocols for determining soil carbon will follow the National Carbon Accounting System (Neilson 2005; Jones 2007). Soil Carbon Credits will be paid annually and retrospectively at one hundredth the 100-year rate of $25 per tonne of CO_2 equivalent in each DSA. This is similar to being paid ‘on delivery’ for livestock or grain and eliminates financial risk (Jones 2007).Conservative estimates of increases in soil organic carbon of 0.15% per will translate to a return of about $127/ha over the project’s three year period (Jones 2007).

The concept of using soil as a carbon sink is yet to be accepted at an official level but the ASCAS trial, if successful, is expected to change that. However, a significant issue is that traditional forms of agriculture often result in loss of soil carbon, so it would seem that accepting soil carbon sequestration as a carbon offset would mean accepting soil degradation as a carbon emission. This would have significant impacts for agriculture and carbon trading in Australia.

4.6 THE USE OF NATIVE GRASSES IN AMELIORATING LAND MANAGEMENT ISSUES

The social, economic and environmental benefits of using native grasses in place of an introduced grass species in ameliorating land management issues appear to be numerous but have not as yet, been thoroughly investigated, particularly in South Australia. Benefits may include: the use of species with negligible or lower weed potential; the restoration of degraded land to more productive land; creating a positive and increasingly healthy environment in which to work and live, as opposed to one in decline; and perhaps in creating an environment of hope through adaptive change. With adaptive change comes confidence, confidence to apply new strategies and techniques to survive and prosper in broadscale agriculture.

Adaptive change brings ‘ecological release’ where traditional methods of agriculture and species selection may be substituted with or transition towards the application of local and appropriate knowledge of native grasses, of species selection, of grassland ecology and management, of seed characteristics and grass species reproductive and adaptive behaviour.

Ultimately, adaptive change brings a myriad of benefits - social, economic and environmental - with respect to managing, surviving and sustaining within landscapes of degradation and change - landscapes which will continue to change into the future.

Social benefits and gaps

Clearly, social benefits to farmers and farming communities accrue if the problems related to salinity and waterlogging can be managed effectively, not least of which is a viable and healthy rural environment both at the landscape and local level. These social benefits apply also to the wider community, as salinity is a catchment-scale problem which affects most, if not all, inhabitants of a region, i.e. flora, fauna and human. Improving soil health in conjunction with improvements to salinity and waterlogging provides economic benefits largely through increased production, greater viable land area, less disease and unhealthy environments, thereby influencing positively both individual land managers and whole communities. Better land management to improve soil health provides land managers with greater opportunity to control their circumstances and ultimately greater confidence about their future.

Any social benefit accrued from having a sustainable resource is dependent upon whether that resource is resistant to climatic variability, not just climate change. Whilst there may be tangible benefits in the short and long term from planting, enhancement or improved management of native grasses, resulting in a reduction of CO_2 concentrations through carbon sequestration, long term changes to native grasses in the landscape as a consequence of climate change, are difficult to predict even given climate change modelling and current predictive research studies. Nevertheless, social research into human adaptability in response to landscape and climate change is only now commencing, and is probably only able to realise realistic results after one or two generations of monitoring. More immediate mechanisms and strategies for assessment of change in relation to land management and land remediation in relation to native grasses are warranted.

Economic benefits and gaps

The development of native grasses in SA may potentially increase the natural and agricultural availability of land previously affected by land management problems. More likely is the potential of native grasses to prevent further salinisation of land as has been occurring over the past century in Australian landscapes, thereby maintaining both the conserved and agricultural land available in this State. Clearly, there are enormous potential benefits of using productive native grasses suitable for marginal saline or other degraded lands. Moreover, native grasses suitable for remediation of such lands provide the opportunity for restoration or limited production capacity and contribute to improved land management. Unfortunately, we do not as yet know enough about Australian native grasses in relation to landscape remediation.

New industry opportunities appear to be evident for species identification and selection or development of cultivars aimed at steep, shallow, saline soils. The use of native grasses for land management purposes presents a market opportunity for native grass seed growers through the species selection and or the development of cultivars used for ameliorative purposes. Furthermore, there is potential reduction in costs associated with the use of native grasses in land management amelioration techniques e.g., with fire control of grasses as opposed to spraying controls; with native pastures or selected native species with improved longevities relative to short lived exotic annual grasses with limited persistence; with the reduction in erosion, water run-off in native pasture compared to exotic pasture; and finally with the resilience of native grasses in times of climatic and other environmental stress relative to exotic species.

The economic benefits of adapting to climate change through greater use of native grasses will ensure the agricultural industry is sustainable long term, and is better able to resist the effects of drought or other extreme climatic events. Clearly, there are immediate and future economic benefits of trading carbon credits from sequestering carbon in native grasslands. There is however, much to be learned about climate change, native grasses and the impact of change on native grasses, native grass pastures and landscape resilience and productivity.

Regenerative land management also results in increased biodiversity and increased conservation value of the land, and a reversal of land degradation will likely increase property value and productivity (Whalley 2007). Are the predicted changes in photosynthetic rates resulting from CO_2 and temperature increases due to climate change likely to affect the chemistry and nutrition of native grasses, native grass seeds, land containing native grass and ultimately the biodiversity of the flora and fauna comprising these grassy ecosystems? These are difficult questions to answer and present serious challenges for researchers and communities.

Environmental benefits and gaps

Salinity, soil and water management issues identified at the farm scale are frequently catchment scale concerns. Any successful attempt reversing the effects of these environmental issues is likely to have numerous flow-on benefits to the remainder of the catchment not just the farm, particularly in relation to water quality, soil health, species survival, pasture productivity and ultimately biodiversity.

Increased sustainability of farming practices through use of native grasses provides the potential for less landscape degradation and more efficient water use. The encouragement of native plant species in addition to native grasses, will increase a region's biodiversity. Increased uptake of native grasses on a large scale will provide a variety of environmental benefits including: reduction of water run-off by allowing good penetration of rainfall into soils; better weed management by competition for space; possible remediation of saline and other contaminated soils; potential reduction in soil erosion; potential improvement in both floral and faunal biodiversity, and a reduction in CO_2 concentrations through carbon sequestration. Regenerative land management also results in the increased biodiversity providing increased conservation value of the land (Whalley 2007).

However, significant gaps in our knowledge of land management issues with respect of the perceived benefits and or responses of native grasses remain. In particular, little is understood of the fire fuel loads of a range of grass species; the responses of growth under changing soil and rainfall conditions; the responses of native grasses to a range of other environmental stresses, particularly nutritive stresses; and the time taken for native grasses to remediate degraded landscapes of varying types.

Nevertheless, more research into the benefits of sowing native grasses/allowing re-establishment of native pastures for soil health is required. Moreover, quality research into the use of native grasses for specific ameliorative purposes should be initiated in the short and long term. More research is required into drought tolerance and rooting depth of native grass species, since very little of the past research has been conducted over sufficient periods of time. Future research is required on the potential effects of increased CO_2 and warming on the productivity of native grass species. More local and specific research on the predicted effects of climate change for South Australia is also necessary, although there is clearly opportunity and potential to include native pastures in the Federal Governments Carbon Emissions Trading Scheme (Carbon Pollution Reduction Scheme).

4.7 SUMMARY: KEY ISSUES - LAND MANAGEMENT AND NATIVE GRASSES

Much of the research reviewed in this chapter advocates the use of native grasses in preventing, ameliorating or reversing land management issues, and also changes to traditional land management practices and principles. This includes changing land management practices to improve soil health, regenerative management to increase groundcover in pastures, pasture cropping in suitable areas, use of ‘no-till’ farming, reductions in the use of fertiliser and herbicides, and increased use of pasture phases in cropping areas. Combined with increased use and understanding of native grasses in agricultural systems, such changes in agricultural practices will assist the development of a new generation of sustainable and productive farming systems. The key issues to emerge from reviewing this area include:

4.7.1 Issue: water quality, runoff management and salinity

- Water moves through the landscape in three dimensions (air, surface, ground) and although one dimensional studies provide some insights, there is a pressing need to develop a greater understanding of these processes in a catchment context;
- Native grasses have a diversity of characteristics and tolerances applicable to marginal and degraded landscapes. Further investigation of these characteristics and tolerances needs to be undertaken and further applied to land management;
- Further research on establishment methods of native grasses for saline areas is required and field trials implemented to test remediation capabilities;
- Cost-benefit analysis studies, including field trials need to be undertaken to determine the cost of native grass perennial establishment relative to exotic grass establishment, in relation to short – term land management amelioration, for both prime and low grade or degraded land; and

- Cost-benefit analysis studies, including field trials need to be undertaken to determine the cost of native grass perennial pasture establishment versus exotic grass pasture establishment, in relation to long-term land management amelioration, for both prime and low grade or degraded land.

4.7.2 Issue: soil health

- Methods such as use of a pasture phase to spell the soil from cropping rotations need to be encouraged;
- Pasture cropping to increase diversity of microhabitats and soil cover should be encouraged where possible;
- Stocking rates to increase soil cover and pasture diversity need to be assessed.
- Issue: land management and wildfire
- Regional trials using native grasses in open areas such as roadsides to assess fire prevention potential, fire fuel loads and fire risk in high fire risk areas should be implemented;
- Landholders replace exotic grasses in high fire risk areas where biomass is high, during high fire risk seasons, with low biomass native species, as deemed appropriate for the property - such as along property tracks, or in areas where fire breaks are maintained.

4.7.3 Issue: climate changes

- More research is required on the use of native grasses as buffers and as resilient species, to tolerate extreme conditions as a consequence of climate change;
- Native perennial pastures should be included in the Federal Government Carbon Emissions Trading Scheme to facilitate the development of carbon sequestration technologies such as soil carbon from native grass pastures;
- State regional predictions regarding climate change should be established and disseminated to farmers to allow them to plan for future pasture and remediation plantings using native grass species;
- Landholders implement drought resilience innovations across the property such as increasing the use of perennial native grasses to improve soil quality, soil health, water efficiency and enhance the biodiversity of pastures;
- Investigate and work towards sustainability of marginal agricultural areas through diversification with native grasses and other appropriate native plant species.

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